PART 3

ANIMALS

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16. Contamination of animals

16.1 Important radionuclides

The radionuclides of most concern with respect to contamination of animals after a nuclear accident are radioiodine, radiocaesium and radiostrontium (ICRP 30, 1979). Of the other significant anthropogenic radionuclides likely to be released in most accidents, only small proportions of that ingested will be absorbed in an animals gut, and the main animal products, milk and meat, will not normally be contaminated to a significant extent. Animal products will mostly be contaminated as a result of ingestion of contaminated feed and possibly, but to a much lesser extent, from inhalation (for radioiodine only). Direct external contamination of animals is of little or no consequence in human food production. Radioiodine and radiostrontium are important with respect to contamination of milk; radiocaesium contaminates both milk and meat. Over the mid to long term perspective of this document, short-lived radioiodine isotopes such as ^{131}I are not relevant, and hence will not be considered.

The physical and chemical form of a radionuclide can influence its absorption in the animal gut. For example, following the Chernobyl accident radiocaesium incorporated into vegetation by root uptake was more readily absorbed than that associated with the original deposit (e.g. Beresford *et al.,* 1989; Ward *et al.,* 1989; Hansen and Hove, 1991; Mayes *et al.,* 1996).

16.2 Transfer of radionuclides to animals

Transfer of radionuclides to animals may be described using different transfer parameters, which assume linearity between fallout deposition rates (Bq m^{-2}) and activity concentrations in plants and animal products (Bq kg^{-1} or L^{-1}). Conventional transfer coefficients, F_m or F_f, are defined as the equilibrium ratio between the activity concentration in milk or meat respectively divided by daily intake. Consequently, F_m will be expressed in units of days per litre (d L^{-1}) and F_f in days per kilogram (d kg^{-1}), i.e.:

$$F_m \text{ or } F_f = \frac{\text{activity concentration in food product (Bq L}^{-1}\text{ or Bq kg}^{-1}\text{)}}{\text{activity in animal feed (Bq d}^{-1}\text{)}}$$

Transfer coefficients are generally higher for young animals than for adults (Howard, 1989). Whilst in common usage, it should be remembered that single recommended values for transfer coefficients do not take account of the effect of interactions between radionuclides and stable analogues, or of homeostatic mechanisms or the effects of physiological status, all of which can significantly affect transfer for some radionuclides. This problem, and approaches to deal with such variation, will be discussed later with respect to radiostrontium.

The calculation of transfer coefficients requires information on both the daily intake of feed (in k d^{-1}) and the contamination level of the feed (Bq kg^{-1}). Estimates of these values can easily be obtained for housed farm animals fed under controlled conditions. However, in many semi-natural or natural environments animals graze in areas with highly variable soil and vegetation types and with characteristically highly variable transfer of radionuclides to plants. In addition, it is difficult to estimate the daily herbage intake of animals grazing semi-natural environments. For these reasons *aggregated transfer coefficients*, T_{ag}, have often been used, which integrate all processes from soil deposition, via plant uptake to animal products:

$$T_{ag} = \frac{\text{activity concentration in the food product (Bq kg}^{-1}\text{ or Bq L}^{-1}\text{)}}{\text{activity of deposit per unit area (Bq m}^{-2}\text{)}}$$

Aggregated transfer coefficients refer to the relationship between ground deposit and contamination of plants or animals when direct contamination is negligible, and consequently *they should not be used under conditions of continuous fallout.*

Aggregated transfer coefficients are easy to derive compared to the conventionally used transfer coefficients. However, in common with other transfer parameters the use of aggregated transfer coefficients requires specific knowledge of the environments in which they were obtained. Because of the complexity of semi-natural and natural environments and lack of knowledge, there are currently few alternatives to the use of T_{ag} values for making mid to long-term predictions in many types of semi-natural environments. However, to make mid to long-term predictions T_{ag} values need to be combined with estimates of effective ecological half-lives (i.e. the time required to reduce the activity concentration to one half the original level, including physical decay), to allow for changes in radionuclide uptake with time.

In this chapter the transfer of radiocaesium and radiostrontium to animals will be presented both as transfer coefficients and aggregated transfer coefficients. As mentioned, *aggregated* transfer coefficients should not be used in the initial period after fallout has been deposited when there is direct contamination of animal feed, and therefore transfer is affected by interception and weathering. In such circumstances the transfer coefficients should be used, taking due account of the bioavailability of the deposit. Transfer coefficients are most useful when there is good information on an animals daily intake of radionuclides (i.e. for farm animals where feeding schedules are well defined).

16.2.1 Farm animals

For most animal meat products, only radiocaesium is important as other radionuclides do not significantly contaminate muscle. Farm animal products are the most important foodstuff determining radiocaesium intake by the average consumer in the Nordic countries. The major potential source of radioiodine and radiostrontium to humans is milk and milk products. Of the different species, the smaller animals have the highest transfer of radiocaesium from fodder to meat and milk.

When considering long-term consequences of radioactive fallout special attention should be paid to animals grazing unimproved pasture and woodland, since the transfer of radiocaesium is higher in such environments compared to cultivated areas (Hove and Strand, 1990; Howard *et al.*, 1991). However, in such ecosystems the variation in soil types, vegetation species and transfer of radionuclides to the different species will make prediction of radionuclide intake by the animals difficult. In the Nordic countries, sheep and goats, and to some extent also cattle, commonly graze unimproved pastures during summer.

In addition to the higher transfer on unimproved pastures and woodlands, the production rate and subsequent ingestion of mushrooms will significantly influence radiocaesium activity concentrations in animal products from these areas. Fungal fruiting bodies appear in the autumn, and will cause elevated radiocaesium activity concentrations in animals which consume them because of the higher uptake of radiocaesium in many species of fungi compared to most vegetation species. Even if mushrooms are eaten in only small amounts they often contain high radiocaesium activity concentrations and therefore can influence radiocaesium intake to a considerable extent.

16.2.1.1 Sheep

When fed under controlled conditions, a transfer coefficient of 0.49 d kg^{-1} has been recommended to estimate the radiocaesium activity concentrations in lamb muscle (IAEA, 1994a). Since 1991, a Nordic study has been ongoing with the aim of quantifying transfer to sheep grazing a range of unimproved areas and the aggregated transfer coefficients obtained in this study are summarised in Table 16.1. Hove *et al.* (1994) concluded that the T_{ag}'s in the Nordic countries clearly fall within two groups: low values of 0.0005-0.003 m^2 kg^{-1} were observed in Denmark, Faeroe Islands and Finland, whilst higher values of between 0.015-0.047 m^2 kg^{-1} were observed in Iceland, Norway and Sweden. Because the grass-to-meat concentration ratios were of similar magnitudes in most of the study locations, the higher radiological vulnerability to radiocaesium observed in Finland (forest pasture), Iceland, Norway and Sweden must relate to differences between soil types and uptake of radiocaesium into herbage. In addition, higher aggregated transfer coefficients from organic soils may be accentuated by the ingestion of a range of more highly contaminated vegetation species by sheep grazing in forested or mountainous areas.

Table 16.1 *Aggregated transfer coefficients for radiocaesium from soil to lamb meat at 6 different sites in the Nordic countries in 1990-93 (from Hove et al., 1994).*

T_{ag} (m^2 kg^{-1} fw) Mean ± SD	Country
0.00055 ± 0.00064	Denmark
0.0030 ± 0.0017	Faeroe Island
0.00083 ± 0.00054	Finland
0.0148 ± 0.0006	Iceland
0.0390 ± 0.0037	Norway
0.047 ± 0.012	Sweden

In their study of nuclear weapons fallout data, Hove and Strand (1990) calculated T_{ag} values for transfer of ^{137}Cs from soil to lamb meat on unimproved pastures of 0.013-0.093 m^2 kg^{-1} in the period 1966-1972, and 0.07-0.10 m^2 kg^{-1} in 1986-1988. Similar T_{ag} values of 0.024-0.136 m^2 kg^{-1} were also observed for Chernobyl ^{137}Cs in 1986-1988; the highest value was obtained in 1988 when mushrooms were abundant. This demonstrates that potential intake of radiocaesium via fungal fruiting bodies needs to be taken into account when assessing the consequences of radiocaesium fallout. An assessment was made by Mehli (1996) of the importance of fruiting bodies compared to vegetation as sources of radiocaesium to sheep. The results showed that in years when fruiting bodies were abundant, 70-80% of the radiocaesium in the animals may be due to ingested mushrooms. In certain areas, the selective intake by sheep of highly contaminated ericaceous species will also increase radiocaesium activity concentrations in the meat.

16.2.1.2 Goats

Goats are kept for both milk and meat production. Norway has the largest number of goats of the Nordic countries, and large volumes of goat milk are used in whey cheese production.

i) Meat

For transfer from feed to goat meat, Hansen and Hove (1991) reported the transfer coefficient, F_f, for ionic radiocaesium to be 0.23 d kg^{-1}.

ii) Milk

Radiocaesium

The transfer of radiocaesium from different feeds to goat milk has been investigated by Hansen and Hove (1991). For hay harvested in the period 1986-1989, they found lower F_m values in 1986 of 0.042 ± 0.007 d L^{-1}, compared with those from 1987-89 which ranged from 0.091 ± 0.018 d L^{-1} to 0.124 ± 0.023 d L^{-1} . The increase in the transfer coefficients were interpreted as a consequence of higher bioavailability of radiocaesium in the later years. Lower F_m values were observed for contaminated willow bark, and soil fed to goats.

In a summary of aggregated transfer coefficients to goat milk in Norway, Strand (1994) reported T_{ag} values ranging from 0.008-0.03 m^2 kg^{-1}.

To demonstrate the effect of differences in pasture type on the radiocaesium levels in goat milk, Garmo and Hansen (1993) grazed goats on meadow and willow pastures. The aggregated transfer coefficients were 0.0002 and 0.001 m^2 kg^{-1} respectively. The higher value on willow pasture was due to both higher transfer of radiocaesium from soil to plants, and the presence of plant species with an higher uptake of radiocaesium on this pasture compared to the meadow. Data on milk from goat grazing natural mountain pastures in Northern Norway (Troms county) in 1987-88 gave T_{ag} values of 0.002-0.004 m^2 kg^{-1} for a period when ingestion of mushrooms was negligible (Hove and Strand, 1990).

When mushrooms are abundant the T_{ag} may increase 2-4 fold (Hove and Strand, 1990; Hove *et al.*, 1990). This is one probable reason for the higher T_{ag} values of 0.011 and 0.014 m^2 kg^{-1} found by Strand and Hove (1996) for the years 1993-94 in two mountain areas in Southern Norway.

Radiostrontium

There is much less information available on radiostrontium transfer to goat milk than there is for radiocaesium, with Coughtrey (1990) giving an expected model prediction value of 0.056 d L^{-1} after 100 d, based on the small amount of available information. Recent studies have provided more relevant data (Howard *et al.*, 1995a; Beresford *et al.*, 1997; Crout *et al.*, in press). A model of radiostrontium transfer in dairy goats was developed which was based on calcium metabolism, because radiostrontium behaviour in animals is dependent on calcium. Assuming a bodyweight of 55kg, a calcium intake of 10 g d^{-1} (about twice requirement) and a milk yield of 1.5 L d^{-1} , the model predicts an F_m value of 0.024 d L^{-1}. Individual measurements of F_m for $^{85}SrCl$ administered to lactating goats ranged from 0.0038 to 0.033 d L^{-1}, but the higher values were obtained from goats in negative calcium balance. Later studies with goats receiving 12 g d^{-1} Ca measured F_m values of 0.02 d L^{-1} (Beresford *et al.*, 1997). Further consideration of the relationship between calcium intake and radiostrontium transfer to milk is given in section 16.2.1.3.

16.2.1.3 Cattle

Transfer of radiocaesium to cattle milk and meat is lower than that to sheep and goat products. For instance, Strand and Hove (1996) found that the aggregated transfer coefficient to goat milk was 2-4 fold higher than that for cow milk.

i) Meat

There have been few studies on aggregated transfer coefficients to beef, reflecting the generally low rate of beef production from unimproved grazing. In a study in Valdres, Norway, radiocaesium activity concentrations in both milk and meat were measured in an experimental herd grazing unimproved mountain pastures. The average T_{ag} value for beef from these cattle were estimated to be 0.006 $m^2\ kg^{-1}$.

ii) Milk

Radiocaesium

In the above study in Norway, the range of T_{ag} values for milk was 0.003-0.0045 $m^2\ kg^{-1}$. The difference between the T_{ag} values for milk and beef are smaller than what would be expected from the transfer coefficient values of 0.0079 d kg^{-1} and 0.051 d kg^{-1} for cow milk and meat (beef) respectively, given by the IAEA (1994a).

The vulnerability of cow milk production in the different Nordic countries to ^{137}Cs fallout has been studied by Hansen and Andersson (1994). Their results are summarised below.

The mean transfer coefficients ranged from 0.0045 to 0.0285 d L^{-1}, with an overall mean of 0.0094 ± 0.0132 d L^{-1}. These values are in agreement with those reported in the literature (Hansen and Andersson, 1994). The transfer of ^{137}Cs from vegetation to cow milk did not increase significantly from 1986 to 1992, although data from some individual farms indicated a tendency of increasing values. This is in contrast to other studies showing increased transfer coefficients to lamb meat and goat milk after the first harvest following the deposition of fallout (Howard *et al.,* 1989; Hansen and Hove, 1991).

When estimating mean T_{ag} values for whole countries, Hansen and Andersson (1994) found that T_{ag}'s decreased with time after 1986 for all countries. The T_{ag} was lowest for Denmark, Finland and Sweden and about 2-10 times higher for the Faeroe Islands, Iceland and Norway. In 1987, the T_{ag} values were about 0.0005, 0.0011 and 0.0006 $m^2\ L^{-1}$ for Denmark, Finland and Sweden, respectively, whilst those for the Faeroe Islands, Iceland and Norway were about 0.0043, 0.0016 and 0.0023 $m^2\ L^{-1}$ respectively. By 1989, the values of the aggregated transfer had decrease to about 0.00025 $m^2\ L^{-1}$ for Denmark, Finland and Sweden, whereas the values for the Faeroe Islands, Iceland and Norway were about 0.0014 $m^2\ L^{-1}$. This indicates that cows' milk production in the Faeroe Islands, Iceland and Norway is considerably more sensitive to ^{137}Cs fallout than the other Nordic countries, which is consistent with the findings of Hove *et al.* (1994) for lamb meat discussed earlier. The only discrepancy between the two products, of a low T_{ag} value for cow milk compared to a high T_{ag} for lamb meat in Sweden, is because the particular flock of lambs studied grazed an uncultivated mountain pasture where high radiocaesium transfer values occurred. The reason for the other discrepancy in the Faeroe Islands is unclear, but is possibly due to interactions between soil types and the different production systems for cow milk and lamb meat (Hansen and Andersson, 1994).

Data on milk from some individual farms in Norway using unimproved pastures was also studied by Hansen and Andersson (1994). At these farms the aggregated transfer values were about 0.004 $m^2 L^{-1}$ which is higher than the dairy bulk milk discussed above. Less than 5% of cow milk in Norway is produced on uncultivated pastures; dairy milk is mainly produced on farms with intensive production of high quality roughage and concentrates. Assuming constant F_m values, the results indicated considerably higher transfer from soil to vegetation on uncultivated pastures than on cultivated pastures (Hansen and Andersson, 1994).

Radiostrontium

Few studies on radiostrontium transfer to animal products have been performed in the environment following the Chernobyl accident. This is probably due to two main reasons:

1) the low quantity of ^{90}Sr in the fallout (in Sweden the level of ^{90}Sr was < 1% of that of ^{137}Cs (Suomela and Melin, 1992)), and it's concomitantly low contribution to the total dose, and
2) the laborious and costly analysis needed for ^{90}Sr, which is a beta emitter (compared to the gamma emitting radionuclides such as ^{131}I, ^{134}Cs and ^{137}Cs).

Information is, however, available from studies in the USSR following the Kyshtym accident, the USA in the period of atmospheric nuclear weapons fallout and from recent studies within EC programmes. A range of transfer coefficients for radiostrontium transfer to cow milk have been reported, with 0.0028 predicted after 100 d of continuous feeding, using model calculations, by Coughtrey (1990), the IAEA (1994a) gave a reported range of 0.001 - 0.003 around this value. These values are largely based on pre-Chernobyl studies on radiostrontium transfer to milk.

However, transfer coefficients for particular radionuclides are not necessarily constant, and can vary due to a number of factors. Indeed, for radiostrontium, which has a stable analogue, calcium, which is an essential nutrient under homeostatic control, the recommendation of a single value of transfer coefficient is not valid. For radiostrontium, the absorption of its stable analogue, calcium, varies inversely with dietary calcium intake at a given calcium requirement. It has therefore been proposed that under normal ranges of calcium intake the transfer of radiostrontium to milk is likely to be inversely proportional to dietary calcium intake (Comar *et al.*, 1966). Using experimental data, Sirotkin (1978) reported a double-exponential relationship between dietary calcium intake and the transfer of radiostrontium to cows milk. Therefore the commonly quoted single value transfer coefficients for radiostrontium transfer to milk are not applicable across a wider range of dietary calcium intakes.

Recently, Howard *et al.* (1997) have derived a simple relationship for ruminants between the radiostrontium transfer coefficient (F_mSr) for cow milk and the calcium intake (I_{Ca}) where:

$$F_m Sr = \frac{0.11 x [Ca] milk}{I_{Ca}}$$

Using this relationship the transfer coefficient for radiostrontium can be predicted on the basis of the calcium intake of dairy animals.

To test the validity of the relationship Howard *et al.* (1997) collated available literature on radiostrontium transfer to cow milk and compared it to calcium intake (Fig. 16.1). This showed the expected relationship between F_mSr and calcium intake, assuming a [Ca]milk of 1.15 g kg^{-1}, and was similar to the trend reported independently by Sirotkin (1978).

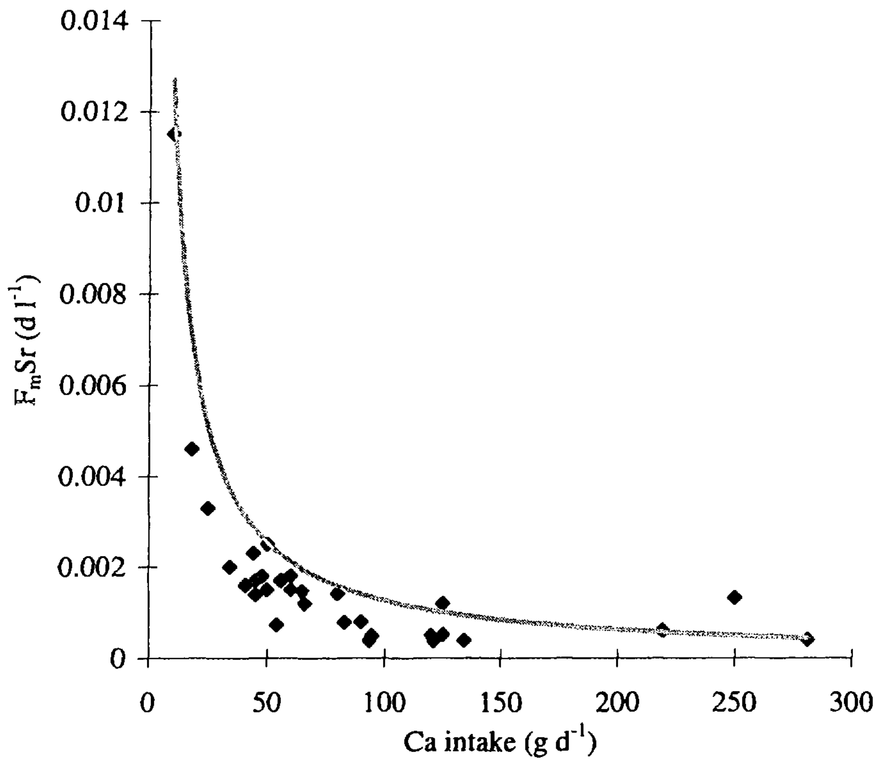

Figure 16.1 *Relationship between the observed (point data) and predicted transfer coefficient for cow milk and the daily calcium intake using the relationship derived by (Howard et al., 1997).*

16.2.1.4 Pigs

Following the Chernobyl accident fewer studies have been performed on the transfer of radionuclides to pigs than to most other farm animals. This is probably partly due to the feeding practice; most pigs are usually intensively managed, and their radionuclide intake is normally low and can be easily controlled..

In a post-Chernobyl, German study, pigs were fed contaminated whey (Voigt*et al.,* 1989). The F_f values derived for pork were in the range 0.35-0.45 d kg^{-1}, which was similar to that as for veal and sheep meat in the same study. No significant influence of different feeding vegetation strategies was found on the transfer of radiocaesium to pigs, with transfer coefficients to pork in the ranges 0.18-0.26 for potato-feeding and 0.17-0.33 d kg^{-1} for grain-feeding, which were both lower than those for whey-feeding (Voigt *et al.,* 1988). These values were in good agreement with earlier data in the literature, and were used as the basis of the F_f value recommended by the IAEA (1994a) of 0.24 d kg^{-1}.

16.2.2 Wild animals

Although products from farm animals are the most important pathway for radiocaesium intake by the average consumer, meat from animals such as moose, roe deer and reindeer may be important sources of radiocaesium for special groups of the population. Other deer species, such as red and white-tailed deer, are not considered in this report since their consumption rate in Nordic countries is low and few studies on transfer to these animals in the Nordic countries have been performed. Since the number of red deer killed has recently been increasing (Statistics Norway, 1996) attention should be paid to people with a high intake of meat from these animals in the case of a future contamination event.

16.2.2.1 Moose

In some areas within the Nordic countries moose constitute an important part of some peoples diet, and in Sweden moose hunting constitutes about 5-10% of the total meat consumed (Bergman *et al.*, 1991). Studies on transfer of radiocaesium to moose meat have mainly been conducted in Sweden and Finland.

The application of an aggregated transfer coefficients assumes a positive correlation between deposition and activity concentrations in the product for a given ecosystem and soil type. In areas where the deposition varies little, the ecosystem and soil type can have a considerable effect, as has been noted for moose (Nelin, 1995). Moose with access to farmland can have lower contamination levels than those without such access (Johanson and Bergström, 1994). In contrast, moose with access to wet bogs may become more highly contaminated (Nelin, 1995).

In summer, moose in Sweden were found to graze mainly on fireweed and birch, both of which are species with low levels of radiocaesium, but in autumn they change their diet to include more contaminated species such as bilberry (Johanson *et al.*, 1994). In the autumn, consumption of fungal fruiting bodies will also increase the radiocaesium intake by moose, as for other free ranging animals. Although Johanson *et al.* (1994) found on average only small amounts of mushrooms in rumen samples (1.3-1.6%), in one animal mushrooms constituted 20% of the rumen contents, and 1% of mushrooms in rumen contents was estimated to contribute about 25% of total radiocaesium intake. The contribution of mushrooms to radiocaesium intake will be higher in years when mushrooms are particularly abundant. Thus, whether hunting is conducted before or after mushrooms appear will have an important influence on the potential contribution of moose meat to radiocaesium transfer to humans.

Aggregated transfer coefficients for moose meat are summarised in Table 16.2. Few sources give individual data. To give an idea about the variation between individual moose from the same area, values from an area of Sweden where the mean radiocaesium activity concentration in moose meat was 750 Bq kg^{-1} (fw) and ranged from 100 to 3,000 Bq kg^{-1}. As demonstrated in the Table, calves have consistently higher mean radiocaesium activity concentrations than adults.

Table 16.2 Aggregated transfer coefficients for radiocaesium to moose meat.

T_{ag} (m^2 kg^{-1} fw)	Year	Reference
0.011-0.026 (all ages)	1979	Rantavaara (1982)
0.013	1985	Nylén (1996)
0.015 (calves)	1985	Bergman *et al.* (1991)
0.010 (adults)	1985	Bergman *et al.* (1991)
0.014 (calves)	1986*-91	Rantavaara (priv. comm.)
0.010 (adults)	1986*-91	Rantavaara (priv. comm.)
0.009-0.032 (calves)	1986-90	Bergman *et al.* (1991)
0.006-0.017 (adults)	1986-90	Bergman *et al.* (1991)
0.02	1986-88	Von Bothmer *et al.* (1990)
0.018-0.024	1986-91	Johanson and Bergström (1994)
0.009-0.019	1989	Johanson *et al.* (1994)
0.009-0.087 mean 0.032	1991	Nelin (1995)
0.006-0.02	1985-1996	Nylén (1996)

* 1986 data given in Rantavaara *et al.* (1987).

From the values presented in Table 16.2 a mean value of 0.02 m^2 kg^{-1} for transfer of radiocaesium to moose meat is recommended. Lower values close to 0.01 m^2 kg^{-1} were observed by Nylén (1996) in northerly counties in Sweden, but in these areas values closer to 0.02 m^2 kg^{-1} were also recorded in some years. The difference might be explained by the availability of mushrooms.

Johanson *et al.* (1991; 1994) also estimated transfer coefficients, F_f, for moose and found values ranging from 0.04-0.29 d kg^{-1}. These values may be applied instead of T_{ag} values when surface contamination of the vegetation is dominating.

16.2.2.2 Roe deer

Transfer of radiocaesium to roe deer is very variable both between and within countries, and, as for moose, access to farmland is one of several important factors influencing contamination levels of the animals. Roe deer have a variable diet, and consequent large seasonal variations in radiocaesium intake rates. As for many other free ranging animals, the highest radiocaesium activity concentrations occur in August-September when fungi are most abundant (e.g. Karlén *et al.,* 1991; Strandberg and Knudsen, 1994). Maximum values recorded by Karlén *et al.* (1991) in August-September were 3-4 times higher than those of the average spring and summer values. Dwarf-shrubs (heather, cowberry and bilberry) are also important contributors to roe deer radiocaesium intake in autumn (Karlén *et al.,* 1991).

Table 16.3 presents some aggregated transfer coefficient values from the literature. The values are largely derived from data where roe-deer have some access to farmland, and the results are therefore most relevant for such environments. The variability in activity concentrations is large, and the calculation of meaningful aggregated transfer coefficients is difficult. Compared to the data in Table 16.2, the data in Table 16.3 indicates that there is a higher transfer of radiocaesium to roe deer than to moose.

Table 16.3 Aggregated transfer coefficients for radiocaesium to roe deer.

T_{ag} (m^2 kg^{-1} fw)	Year	Reference
0.15 (autumn)	1988-89	Karlén *et al.*(1991)
0.04 (rest of year)	1988-91	Johanson and Bergström (1994)
0.05 (annual mean)	1988-91	Johanson and Bergström (1994)
0.005-0.1	1987-94	Kiefer *et al.* (1996)

16.2.2.3 Reindeer

Reindeer is an important food source in the Nordic countries, especially for the Sami population. A comprehensive review, combining information on reindeer nutritional ecology and radiocaesium transfer is given by Gaare and Staaland (1994) which also discusses factors such as seasonal migration patterns.

Since the atmospheric nuclear weapons tests mostly in the 1950-60's it has been known that reindeer meat is especially sensitive to radiocaesium fallout, because reindeer ingest lichens as a major food source. During winter lichens may constitute 70-80% of the food intake, even during summer lichens constitute 10-20% of the intake (Gaare and Staaland, 1994). Lichens have a high ability to accumulate radiocaesium directly from precipitation; they do not absorb many nutrients, or radionuclides via root uptake.

Due to the importance of lichens in the diet, radiocaesium concentrations in reindeer will change according to changes in intake of lichens. There is therefore an increase in radiocaesium activity concentrations in reindeer in the autumn from late August/early September to November. During winter (November - early May) radiocaesium activity concentrations remain relatively constant, although there is a slow decrease due to reduced food intake. Then there is a faster decrease as they change to summer grazing (from May to end June), where they reach a stable low plateau which lasts to late August. The start of the increase in radiocaesium activity concentrations will depend on the time when mushrooms, which are also eaten by reindeer, appear.

The difference between maximum winter radiocaesium concentrations and minimum summer concentrations in reindeer varies from 2-4 fold (Strand and Hove, 1996) to 20 fold (Åhman and Åhman, 1994). For instance, Åhman and Åhman (1994) estimated T_{ag} values ranging from 0.6-1.1 m^2 kg^{-1} during the winter of 1986-87, with minimum values of about 0.025 m^2 kg^{-1} during summer. Other transfer values are given in Table 16.4.

Table 16.4 Aggregated transfer coefficients for radiocaesium transfer to reindeer.

T_{ag} (range or mean±SE)	Year, month	Reference
1-2	1987 Winter	Strand (1994)
0.12 ± 0.01	1986 September	Åhman and Åhman (1994)
0.44 ± 0.02	1986 Nov - Dec	Åhman and Åhman (1994)
0.76 ± 0.03	1987 Jan - Apr	Åhman and Åhman (1994)

Transfer of radiocaesium to reindeer meat is not uniformly high in all Nordic countries. Pálsson *et al.* (1994) measured the transfer of radiocaesium to reindeer in Iceland and found low ^{137}Cs activity concentrations in meat compared to that from other Nordic countries. The data presented by Pálsson *et al.* for 1990-91 corresponds to an aggregated transfer coefficient of 5.6 x 10^{-3} m^2 kg^{-1}. The difference from the values in other Nordic countries (e.g. those presented in Table 16.4) was attributed to a diet dominated by a particular lichen species which is much less highly contaminated than the reindeer lichen commonly consumed elsewhere.

In cases where the reindeer diet is known and can be quantified, transfer coefficients, F_f can be used to estimate the radiocaesium activity concentrations in animals. F_f values of 0.20-0.50 d kg^{-1} have been calculated from feeding experiments (Strand and Hove, 1996; Åhman, 1990).

16.3 Long term behaviour of radionuclide contamination

When radioactive contamination of the environment occurs it is important to know for how long foodstuffs will remain contaminated. This is determined by the physical decay rate of each radionuclides, and by various natural physical, chemical and biological processes which influence accumulation and elimination of radionuclides in the environment. One example is the rate of fixation in soil, which influences bioavailability and hence uptake by vegetation.

The effective ecological half-life (T_{eff}) is a parameter used to describe the long term decline in contamination levels in the environment. This is defined as the time over which the activity concentration in the product falls to one half of the initial value when no measures are taken to reduce the contamination levels. It is expressed as:

$$\frac{1}{T_{eff}} = \frac{1}{T_{phys}} + \frac{1}{T_{eco}}$$

where T_{phys} represents the physical half-life and T_{eco} the ecological half-life of the radionuclide being considered.

For short lived radionuclides the effective ecological half-life is essentially the same as the physical half-life. The ecological half-life will be more important for long lived radionuclides.

16.3.1 Farm animals

Long term behaviour of radionuclides and their decline in farm animals is of most concern when the products (e.g. milk and meat) are obtained from animals on unimproved pastures, since most animals are given feed from cultivated areas when they are housed. Fodder from cultivated areas will, in the long term, have generally low contamination levels (see Part 2), and consequently radionuclide activity concentrations in animals given these feedstuffs will also be low.

As earlier mentioned (section 16.2.1) variation in soil and vegetation types across unimproved pastures may be considerable, giving rise to differences in radionuclide intake by individual animals. Diet selection will also influence the long term behaviour of radionuclides in animals, since long term behaviour of radiocaesium differs between vegetation species (e.g. Haugen and Uhlen, 1992). Thus, changes with time might be observed in the relative radiocaesium activity concentrations in animal products from different grazed ecosystems. Additionally, appearance and abundance of mushrooms will influence the long term behaviour of radiocaesium in animal products, since many mushroom species have longer T_{eff} than those of vegetation species. Quantifying these effects is further complicated by the variable rate of production of fruiting bodies from year to year.

16.3.1.1 Sheep

Sheep are mainly slaughtered in the autumn, following summer grazing on mainly unimproved pastures. Due to differences in ecological half-lives between vegetation species due to species differences or underlying soil types, long term behaviour of radiocaesium in sheep may differ between grazing areas. In the autumn, appearance of mushrooms will strongly influence radiocaesium activity concentrations in sheep, and also contribute to differences between areas. The first estimates of effective ecological half-lives for sheep meat were reported by Hove and Strand (1990). On the basis of nuclear weapons tests fallout in Norway they calculated values of between 22-27 years (Table 16.5). In agreement with this long half-life, the Nordic studies during 1990-1993, compiled by Hove *et al.* (1994), only found significant decreases in lamb meat in Denmark from 1990 to 1992 (but there was no significant difference between 1992 and 1993). Within the other Nordic countries there was no significant decrease and the within year variation in most locations was much greater than that from year to year (Hove *et al.*, 1994). The authors also stressed that observations from only four years are of limited use for effective ecological half-life calculations.

***Table 16.5** Summary of effective ecological half-lives for ^{137}Cs in sheep.*

T_{eff}, years (range)	Time period	Reference
22-27	1959-1988	Hove and Strand (1990)
3.4	1990-1993	Rosén *et al.* (1995)
2.2 [a] -8.5	1988-1995	Strand and Hove (1996)
2.4-7.8	1987-1995	Mehli (1996) [b]

[a] The lower value indicate estimates without the influence from ingested mushrooms.
[b] Range of half-lives without influence from ingested mushrooms.

Ingestion of contaminated mushrooms by animals will influence long term contamination in animal products, since the estimated T_{eff} will be affected by years with unusually high or low mushroom abundance. At one of their study sites, Strand and Hove (1996) found that the calculated half-life increased from 2.2 to 7 years when using sheep radiocaesium activity concentrations measured before and after mushrooms appeared respectively. Furthermore, Mehli (1996) found that radiocaesium from ingested mushrooms could contribute roughly 70-80% to the radiocaesium body burden of sheep, and that declines in sheep radiocaesium concentrations due to that in grazed vegetation should be about 7.8 years. In addition, some longer term data suggest that the T_{eff} is not constant but increases with time (see for instance Part 2) due to the influence of more than one environmental factor in determining decreases, and half life estimates will therefore be dependent on the number of years being considered.

16.3.1.2 Goats

Goats are milked throughout the grazing season, and the T_{eff} will be affected by the time needed for radiocaesium activity concentration in goat milk to become equilibrated with radiocaesium intake rates once the goats are released outdoors. Garmo (1996) reported that a T_{eff} based on milk radiocaesium activity concentration values measured at the beginning of the grazing season was lower than for values based on data from later in the grazing season. Such changes with time could be due to equilibration, or changes in diet or radiocaesium uptake by vegetation over the grazing period. Furthermore, the T_{eff} increased as additional data for extra years was incorporated into the calculation. Some of Garmo's results are shown in Table 16.6, together with other reported values. The estimates vary considerably, therefore assuming one mean T_{eff} value will a have limited validity.

Table 16.6 *Summary of effective ecological half-lives for ^{137}Cs in goat milk.*

Range of T_{eff}, years	Time period	Reference
15-25	1963-1988	Hove and Strand (1990)
3-14.5[a]	1986-1995	Garmo (1996)
5.7-30[b]	1986-1995	Strand and Hove (1996)

[a] The T_{eff} is calculated from the mean goat milk radiocaesium activity concentrations during the grazing period. It increases with time after the Chernobyl fallout.

[b] Data from some study sites show no decline. The value of 5.7 is based on data which are influenced by ingested mushrooms, thus also even shorter T_{eff} values may be obtained.

16.3.1.3 Cattle

Radiocaesium

The review by Hansen and Andersson (1994) summarised the Nordic studies on effective half-lives in cows milk (Table 16.7). The estimated T_{eff} for the transfer of ^{137}Cs to cows milk estimated, on the basis of whole countries, ranged from 1 to 2 years. The exception was Iceland, where the T_{eff} was estimated for global fallout (not Chernobyl fallout) and was much longer at 18.4 years. The mean T_{eff} values for individual farms in Sweden were similar to that of the whole country, whilst for the individual farms in Norway (where cattle graze unimproved pastures) longer half-lives were experienced (Hansen and Andersson, 1994; NRPA, unpublished).

The data for Chernobyl fallout from Norway indicate that T_{eff} in milk are considerably longer from cattle grazing on unimproved pastures compared to milk from cows grazing on cultivated pastures. The even longer T_{eff} observed for radiocaesium from nuclear weapons tests (Hove and Strand, 1990) suggests that there will also be an increased T_{eff} with time for the Chernobyl fallout (which also corresponds with the longer T_{eff} in Iceland).

Table 16.7 Estimated effective half-lives for ^{137}Cs in cows milk sampled from dairies and dry milk factories, or individual farms.

T_{eff} (±SD, range), years	Country (area)	Years	Reference
Milk from dairies or dry-milk factories			
1.6	Denmark	1987-1991	Hansen and Anderson (1994)
1.6	Faeroe Islands	1987-1991	"
1.4	Finland	1986-1991	"
18.4[a]	Iceland	1986-1992	"
2.0	Norway	1986-1991	"
2.3	Norway	1987-1991	"
1.0	Sweden	1986-1990	"
3[b]	Sweden	1986-1990	Suomela and Melin (1992)
Milk from individual farms			
1.0±1.4 (0.2-3.7)	Sweden	1986-1992	Hansen and Andersson (1994)
6.8, 7.7, 24	Norway[c]	1988-1995	NRPA, unpublished
5±3 (2.6-8.5)	Norway[d]	1988-1995	

[a] Global fallout (all other data are for Chernobyl fallout)
[b] Data from the diaries Tärnaby and Vittangi.
[c] Data from three individual farms in Valdres using unimproved pastures.
[d] Data from three individual farms at Brønnøysund using unimproved pastures.

No studies are known to the authors specifically studying long term decline in radiocaesium levels in cattle meat, but similar general declines as those reported in milk from diaries/dry-milk factories would be expected (see Table 16.7). Also in the limited cases of production on unimproved pastures the decline is expected to follow the T_{eff} in milk.

Radiostrontium

Few data are available on the effective ecological half-life of radiostrontium. Suomela and Melin (1992) observed a T_{eff} of about 3 years in milk from Swedish diaries in the first years after the Chernobyl accident. However, Suomela and Melin stated that the value was rather uncertain due to the low levels of ^{90}Sr.

16.3.2 Wild animals

16.3.2.1 Moose

The data on moose meat from Sweden and Finland show similar aggregated transfer values both before and after the Chernobyl accident, indicating that there is no significant decrease in radiocaesium activity concentrations in moose (Bergman *et al.*, 1991; Rantavaara *et al.*, 1987). One explanation for this is that no significant changes occurred in mean radiocaesium concentrations in fireweed and birch, which were two of the main constituents of moose diet in summer (Bergman *et al.*, 1991). Furthermore, more recent studies have also failed to detect any significant decrease in moose radiocaesium activity concentrations (Johanson and Bergström, 1994; Nylén, 1996). Since moose are hunted in the autumn, the year to year variation in mushroom abundance will also make predictions of the long term behaviour of radiocaesium difficult (see also section 16.3.1.1). Thus, currently available information indicates that the physical half-life of ^{137}Cs should be applied as an estimate of the T_{eff} of radiocaesium in moose meat.

16.3.2.2 Roe-deer

As for the other free ranging animals considered in this report, radiocaesium activity concentrations in roe deer exhibit large seasonal variations (e.g. Kiefer *et al.*, 1996), and occurrence of mushrooms in the autumn will influence the estimates of the T_{eff}. Karlén *et al.* (1991) have concluded that there had been no decrease in ^{137}Cs activity concentrations in roe-deer in central Sweden since 1986. In accordance with this, Johanson and Bergström (1994) found that the decline of the ^{137}Cs level in Swedish roe deer is determined by the physical half-life (see also section 16.3.2.1). In contrast to this, Kiefer *et al.* (1996) found that roe deer radiocaesium activity concentrations in Germany declined during 1986-1991 with a effective ecological half-life of about 3 years, but from 1991 to 1994 there was no further decrease.

16.3.2.3 Reindeer

Lichens dilute radiocaesium activity concentrations by new growth, but since the growth rate is low the biological half-life of radiocaesium in lichens is relatively long compared with other grazed vegetation. Consequently, since lichens are major feeds for reindeer, reindeer meat will have similarly long effective ecological half-lives.

Table 16.8 summarises some of the reported values of T_{eff}. Generally, the winter values are between 3-4 years, while the decrease is a little faster during summer due to generally shorter half-lives in grazed summer vegetation compared to lichens. Appearance of mushrooms may affect the long term decline of the reindeer radiocaesium concentration during late summer.

Table 16.8 Estimated effective ecological half-lives of ^{137}Cs in reindeer meat.

T_{eff}±SE	Year, season	Country	Reference
1.6	1987-1992, Aug.	Norway	Pedersen *et al.* (1993)
3.5	1987-1992, Nov.	Norway	Pedersen *et al.* (1993)
3.3	1987-1992, Apr.	Norway	Pedersen *et al.* (1993)
4-5	1987-1992	Finland	Rissanen and Rahola (1993)
3 2 ± 0.3	1986-92, Sep.	Sweden	Åhman and Åhman (1994)
3.2 ± 0.2	1986-92, Nov.-Dec.	Sweden	Åhman and Åhman (1994)
4.2 ± 0.4	1987-92, Jan.-Apr.	Sweden	Åhman and Åhman (1994)
3-5	1987-1994	Norway	Amundsen (1995)
2.8	1987-1995, summer	Norway	Strand and Hove (1996)
3.2	1987-1995, winter	Norway	Strand and Hove (1996)

The T_{eff} values reported in Table 16.8 are somewhat shorter than those observed after the nuclear bomb test fallout prior to the Chernobyl accident (e.g. Westerlund *et al.*, 1987). The difference is mainly due to continuous fallout which occurred over at least a decade after atmospheric bomb testing, compared to the single contamination event from Chernobyl (Strand, 1994).

17. Countermeasures

A wide variety of countermeasures are available for reducing intake of radionuclides by the population via animal products. They may be classified into three main groups:

I Restriction of foodstuffs from human consumption
- a) banning contaminated foodstuffs
- b) dietary advice

II Additives given to animals to reduce gut absorption of radionuclides
- a) given daily with concentrates
- b) within boli (radiocaesium only)
- c) within salt licks (radiocaesium)[1]
- d) stable analogue supplementation (radiostrontium only)

III Animal management
- a) providing clean fodder, or removing animals
- b) changing slaughter time
- c) changing from milk to meat production, or changing animal species

[1] Could also be used for ^{90}Sr by incorporating calcium, but this is not tested.

17.1 Restriction of foodstuffs from human consumption

17.1.1 Banning contaminated foodstuffs

Banning contaminated foodstuffs is a drastic countermeasure and is often the most costly alternative. However, it may, in some instances, be the only available measure, particularly in the early phase of an accident where full response procedures have not yet been developed. In addition, there may be inadequate information about the actual activity concentrations to make alternative decisions.

17.1.1.1 Effectiveness

This countermeasure is 100% effective in removing artificial radioactivity from the food chain. However, it also removes the product.

17.1.1.2 Practical aspects

The routine slaughtering system could be readily adapted to handle banned meat. The main difficulties in banning food may be the social and political consequences for an area which relies on these products and the subsequent economical problems of loss of public confidence in other products from the affected area. This may be important if the countermeasure needs to be applied over a longer period than only the early phase. The banning of food may be followed by restricted food production in the contaminated area, and this might have serious psychological and social effects in the community.

17.1.1.3 Cost

The cost of this countermeasures would comprise the value of the product which is being banned, and, if necessary, the provision of an alternative food. In Norway, condemnation of sheep and reindeer meat after the Chernobyl accident was estimated to cost 125,000 and 42,500 ECU per person-Sv respectively, and was the most expensive countermeasure applied (Brynildsen *et al.*, 1996).

A possible variation of this countermeasures is to divert animal products from human to animal feeding since contaminated animal products could be used as feed for other, possibly non-food producing animals. The use of animals products as animal fodder has previously been well established in countries where production in the agricultural system exceeds demand, but has been banned in some countries due to BSE. A moderate use of this countermeasures will probably be easy to implement, but limited in its capacity. The cost of this countermeasures is mainly the difference in value between animal products and animal fodder.

17.1.2 Dietary advice

Dietary advice would be in the form of guidance on upper limits for consumption of certain foodstuffs in a given period of time. The use of dietary advice may also be considered as a countermeasure for reclaiming abandoned areas. Application of this countermeasure, together with advice on food preparation in the household, can give significant reduction in individual doses, especially for critical groups. In the long term, particular attention should be focused on people with a high consumption rate of wild animals, as some of these products will have persistently high radiocaesium activity concentrations.

17.1.2.1 Effectiveness, practical aspects and costs.

It is difficult to assess the effectiveness, practical aspects and cost of this countermeasure. However dietary advice can be important in helping people to maintain their way of life and their work. This was observed in Norway after the Chernobyl accident for the reindeer breeding Sami people. By using dietary intake with food preparation advice, a yearly radiocaesium dose reduction of 50-85% was achieved in Norway (Strand *et al.*, 1992). Based on the experiences after the Chernobyl fallout, the cost effectiveness of this countermeasure in Norway has been estimated to be 5 ECU per person-Sv (Strand *et al.*, 1990).

17.2 Additives given to animals to reduce gut absorption of radionuclides

17.2.1 Radiocaesium binders

The addition of chemical binders to animal feedstuffs to make radiocaesium unavailable for gut uptake is a well-established method of preventing radiocaesium from contaminating the meat and milk of farm animals. The binders may be administered by mixing as a powder into concentrate feed, placing as a bolus in the stomach of the animal where it resides for a period of several weeks, or incorporating into salt licks for use with animals which graze in large open areas.

17.2.1.1 Effectiveness

Several clay minerals are known to be effective in reducing radiocaesium activity concentrations; bentonite is the most effective in reducing radiocaesium uptake in the gastro-intestinal tract of ruminants and non-ruminants in relation to the daily dosage (Voigt, 1993). A reduction of 50-60% (sometimes more) in the transfer of radiocaesium to the cows milk and meat, goat milk, and sheep and reindeer meat can be achieved with bentonite administration rates of about 0.5 g d^{-1} per kg of body weight (Hove and Ekern, 1988; Unsworth *et al.*, 1989; Hove *et al.*, 1991; Åhman, 1996). Higher reductions can be achieved by applying higher bentonite administration rates, but this is not recommended since the animals water requirements may increase (Åhman, 1996).

Hexacyanoferrate derivatives are more effective than bentonite in preventing radiocaesium uptake (Giese, 1989). Of these, the derivative ammonium-ferric (III)-cyano-ferrate (II) (AFCF or Giese salt) has been widely used. The reduction factors achieved using AFCF for transfer of radiocaesium to animal products in the literature, reviewed by Voigt (1993), are given in Table 17.1

Table 17.1 *Effectiveness of AFCF in reducing transfer of radiocaesium to different animal products (dosage 3 g d^{-1} for cows, 1-2 g d^{-1} for other animals (after Voigt, 1993).*

Animal	Product	Reduction factor
Cows	Milk	5-10
	Meat	4
Calves	Meat	11-12
Sheep	Meat	4-8
Lamb	Meat	2

The AFCF dose levels recommended for ruminants are generally 3 g d^{-1} to cows (6 mg kg^{-1} body weight) and 1-2 g d^{-1} (10-40 mg kg^{-1} body weight) for smaller ruminants. At these administration rates, reductions of tenfold or more can be achieved for both milk and meat (Hove, 1993), but it has been observed that doses as low as 1 mg kg^{-1} body weight can achieve a 50% reduction in radiocaesium uptake by reindeer feeding on contaminated lichens (Hove *et al.*, 1991).

The use of AFCF as a feed additive has recently been approved by the EC (EC, 1996).

17.2.1.2 Practical aspects

Bentonite or AFCF are fed to animals either as powder or included in concentrate mixtures; AFCF can also be included in a dilute form in salt licks or rumen boli. For housed animals, or for animals who are being kept under conditions where feeding is practical (i.e. for dairy animals which are milked daily), the easiest method of application is to incorporate the binder into concentrates. The binder can be incorporated into the concentrate without any significant change in the normal method of concentrate manufacturing. AFCF concentrations of 1 g kg^{-1} concentrate have been used in Norway. In addition, at a practical level on farms, it is not necessary to give any instruction to the farmer on how to use the binder since it is already in the concentrate and the normal use of concentrates is maintained.

The addition of bentonite or AFCF to concentrate is not always possible, for instance if concentrate is not available, or if it is not practical to administer because animals are freely grazing large areas over long periods of time. In such situations, sustained release boli or salt-licks may be used (Hove *et al.*, 1990; Hansen *et al.*, 1996). Of these two methods, the boli is the most efficient and control is achieved for all animals treated. After treating the animals, boli without a protective wax surface coating will last 4-8 weeks, whilst boli with a wax coating will last 10-12 weeks (Hansen *et al.*, 1996). The effectiveness of the two types of boli are similar; with 43-75% and 48-65% reduction in meat radiocaesium activity concentrations after 4-8 and 9-11 weeks respectively. Therefore the time of boli administration relative to the time of slaughter will determine which type of boli to use. Treatment may be done by the farmers, and it is recommended to administer 3 AFCF boli per animal to secure sufficient release rates of AFCF and to minimise the effect of losses caused through regurgitation. After administration, no further treatment is necessary.

Use of salt licks (with 25 g AFCF kg^{-1}) achieves less control of the effect on all animals since it is impossible to ensure that each individual animal uses the salt-lick sufficiently frequently and thereby ingests adequate amounts of AFCF. This causes a greater variation in the radiocaesium contamination of animal populations, but the average radiocaesium activity concentration in an flock can be reduced by about 50% (Hove *et al.*, 1990).

17.2.1.3 Costs

In Sweden, bentonite has been preferred as a feed additive as it is readily available and cheap (Åhman, 1996), whilst, due to the effectiveness, AFCF has been preferred in Norway since 1989. The costs associated with the use of AFCF in concentrates and salt licks are mostly incurred in the initial purchase, in Norway this has been estimated to be about 75 ECU per kg. One rumen bolus costs a maximum of 0.25 ECU (Hansen *et al.*, 1996). In addition, there is the cost of administration to animals, with the highest costs associated with the individual treatment of animals with boli. The compensation to the farmers for this work has been 5.6 ECU per animal (Brynildsen *et al.*, 1996). Overall, use of boli as a countermeasure for sheep was estimated to be 2.5 times as cost effective as special feeding (see next section). For reindeer, a compensation of 19 ECU was paid per animal.

Based on the price and the reduction effect, Brynildsen *et al.* (1996) estimated the cost per averted dose to the population. They found that the price per person-Sv was about 125 ECU for AFCF in dairy concentrates, 2,000-4,690 ECU for salt licks, and 12,500 ECU for boli in lambs.

A series of other hexacyanoferrate derivatives are also effective in reducing radiocaesium uptake in animals. AFCF is about twice as effective as these other chemicals, but is also considerably more expensive. The alternatives should be considered by relevant authorities for future use (Hove, 1993).

17.2.2 Radiostrontium analogue

In contrast to radiocaesium, currently no highly effective radiostrontium binder has been identified which can reduce radiostrontium uptake in the gastro-intestinal tract, although experiments in Norway have achieved a reduction of about 40% using Zeolite A (IAEA, 1994b). Furthermore, experiments by Hove *et al.* (in Howard *et al.*, 1995b) have shown that radiostrontium transfer to milk was not affected by administration of stable strontium.

Currently, calcium is one of the few practical dietary supplements which might be used as a countermeasure to reduce radiostrontium transfer to milk (Voigt, 1993; Howard *et al.*, 1995a). Published reduction factors, which predict the effect of giving additional calcium (reviewed by Voigt, 1993) are not generally applicable because the reduction achieved will depend on the current calcium intake and status of the animal. Howard *et al.* (1997) predicted the reduction factors which might be achieved in radiostrontium transfer to milk if different supplementary levels of calcium are added to diets of cows with different calcium intake rates. In the example shown in Figure 17.1, they estimated the fractional reduction in F_m for a cow with an unsupplemented dietary calcium intake of between 40 and 140g d^{-1}. This range of dietary calcium intake is representative of those to be expected in both Western and Eastern Europe.

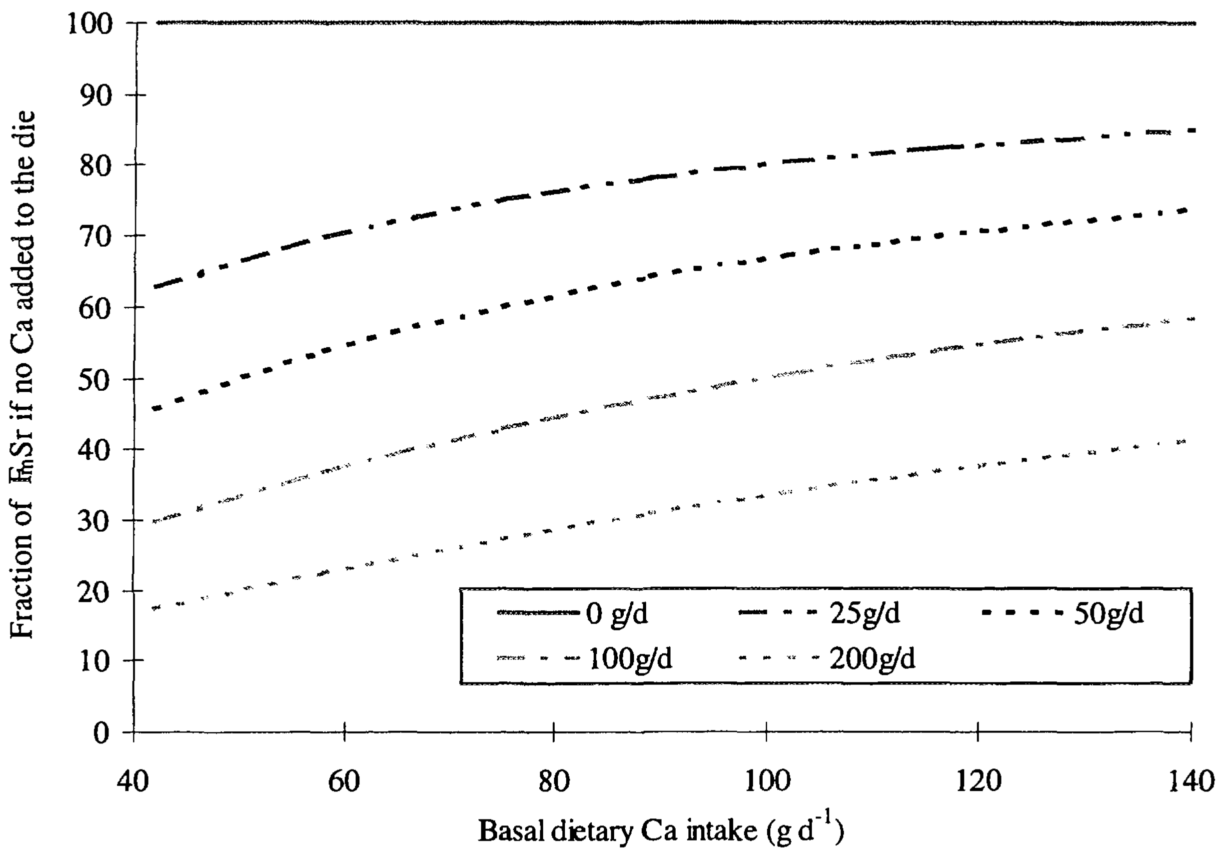

Figure 17.1 *Effect of varying supplementary calcium intake on Fm for radiostrontium for dairy cattle receiving different basal dietary calcium intakes (Howard et al., 1997).*

Supplementation rates of 100 to 200 g Ca d^{-1} are predicted to effectively reduce the transfer of radiostrontium to milk by at least 40-60 % for the range of dietary calcium intake rates considered. Advised maximum intake rates of calcium for ruminants are 1-2 % of the dry matter dietary intake; intake rates greater than this may reduce dry matter intake. A supplementation rate of 200 g for dairy cows would be at the upper limit of this range and would not therefore be advisable for prolonged periods. Maximum supplementation rates would obviously depend upon intakes from the basal diet. The availability or metabolism of other essential trace elements (eg. Cu, Zn and P) may be affected by too high a calcium supplementation.

17.3 Change in animal management

17.3.1 Providing clean fodder or removal of animals

Where animals become contaminated through grazing contaminated pastures, an effective countermeasure for all radionuclides is to provide uncontaminated ("clean") pasture or feedstuffs until the activity concentration of the radionuclides in herbage in contaminated areas has fallen to sufficiently low levels that milk and meat will not become too highly contaminated. The need for providing clean feeding will therefore be more of a long term problem in areas where agricultural practices involves the utilisation of unimproved pastures, since the decline in herbage contamination levels on these pastures is generally slower than in cultivated areas (Hove and Strand, 1990; Howard *et al.*, 1991).

17.3.1.1 Effectiveness

Providing clean fodder for stalled animals gives an almost 100% reduction in contamination levels. Also for grazing animals, such as free ranging sheep and reindeer, it is possible to achieve a reduction of almost 100% if clean feeding is continued for long enough after the animals are removed from the contaminated pasture. In this case the main factor determining the effectiveness of this measure will be the biological half-life of each radionuclide in the animals. For radiocaesium contamination the biological half-life when moving to non-contaminated pasture is about 3 weeks for both sheep and reindeer (Hove *et al.*, 1990; Åhman, 1996). In general, biological half times are longer for larger ruminants, such as cattle which have a biological half-life of roughly 30 days in meat (IAEA, 1994a). Thus, the reduction in contamination levels achieved depends on the time over which the clean fodder is used, and the species of animal.

Adding caesium binders (section 17.2.1) will enhance excretion of radiocaesium from the animals, thereby reducing the biological half-life. For reindeer adding binder reduced the half-life by about one week (Åhman, 1996).

17.3.1.2 Practical aspects

There are some limitations to this countermeasure. For instance, sufficient uncontaminated fodder may not be available in the event of widespread contamination. In addition, the provision of clean food may involve transport of food or animals and also management of the animals during the feeding period. There may also be limitations due to the climate (e.g. snow) and the lack of available housing (e.g. for sheep in late autumn/winter) or alternative pastures.

17.3.1.3 Costs

The duration of the clean feeding period is an important factor influencing the cost of this countermeasure. For grazing animals, two factors comprise the cost of this countermeasure; the original contamination level prior to clean feeding, and the biological half life for each animal species. In the event of widespread contamination clean feed may also cost.

In Norway a special feeding programme was established after the Chernobyl accident (Strand *et al.*, 1990; Brynildsen *et al.*, 1996). The compensation rates and arrangements for the farmers have changed over the years. Initially, they received 0.5 ECU d^{1} per sheep. From 1994, they received no compensation for the first week of special feeding of sheep and goats, then 0.63 ECU d^{-1} per animal from week 2 to 5, and 1 ECU d^{-1} per animal from week 6 to 8. The compensation is intended to cover extra expenditures connected to the feeding program, mainly additional work because the animals could not be slaughtered at the usual time. Extra compensation is paid to farmers with special difficulties in applying the feeding program, for instance, when alternative pastures are lacking or transport of animals is required to other areas. For reindeer, being fenced and fed, the compensation varied from 44 to 94 ECU per animal for the whole period, depending on the level of radiocaesium in the meat. For cattle 1 ECU d^{-1} per animal has been paid to carry out the feeding programme.

In terms of cost effectiveness in reducing dose to man, special feeding of sheep has been estimated to cost 35,000-41,000 ECU per person-Sv in Norway (Brynildsen *et al.*, 1996).

17.3.2 Changing slaughter time

In certain animal species radiocaesium contamination levels vary greatly with season. Therefore changing the normal time of slaughtering may have a pronounced effect on intake of radiocaesium by humans. The effectiveness and feasibility of this approach as a countermeasure is highly dependent on the environment and animal being considered.

One variant of this countermeasure is to collect the animals earlier than usual from unimproved pastures; this has been done with sheep in Norway in years when there has been a high abundance of mushrooms (Brynildsen *et al.*, 1996).

17.3.2.1 Effectiveness

The effectiveness of this countermeasure will depend on the animal species and season, and also on the time difference between the selected slaughter time and that of the normal slaughter. Hence, it is not possible to assign one general reduction factor for this countermeasure. In autumn, radiocaesium activity concentrations in reindeer can be expected to be 2-4 times lower than during winter (see section 16.2). In Sweden, a hunting season for roe deer buck in the spring was introduced which considerably reduced the transfer of ^{137}Cs to man. In 1990, the ^{137}Cs activity concentrations were about 5 times lower during the spring compared to the period for normal roe-deer hunting of August- September (Johanson *et al.*, 1991).

17.3.2.2 Practical aspects

For animals such as reindeer, it may be necessary to develop a new infrastructure when changing slaughter time, because of differences in arrangements for gathering and slaughtering animals during summertime compared with wintertime. Earlier collection of sheep from mountain and woodland pastures may also requires changes in management practices. Changing hunting seasons will have to take animal protection and hunting traditions into account.

17.3.2.3 Cost

There would be an income shortfall during the transition period. For herded animals, it may be necessary to compensate for the loss of increased body weight in the animals between summer and winter. For reindeer, there may be a roughly 10 kg difference between late summer weight for calf compared with winter weight, and in Norway early slaughter of reindeer calves was encouraged by paying 6.25 or 12.5 ECU per animal, depending on the time of slaughter (Brynildsen *et al.*, 1996). In addition, there will be the cost of providing the infrastructure. However this is a one time expense, which does not reoccur.

Normally, it should not be necessary to pay compensation for changing hunting seasons of wild animals.

17.3.3 Changing from milk to meat production, or changing animal species

The rate of transfer of radiocaesium from fodder to meat is generally 4-6 fold higher than that to milk (see chapter 16.2) so changing production strategies from meat to milk production could be considered. For radiostrontium, the transfer to meat is negligible compared to that for milk. Since transfer of radionuclides differs to different animal species (see chapter 16.2) altering the animal species, generally from small to larger ruminants, is also a method of reducing the transfer of radioactivity from plants to animals.

17.3.3.1 Effectiveness

The effect of changing from milk to meat production is a reduction in contamination of the end-product of more than 90% in the case of radiostrontium and up to 35% for radiocaesium.

17.3.3.2 Practical Aspects and Costs

Providing cattle or buying beef producing cattle cannot be accomplished rapidly, and requires significant changes in management practices. There will also be a difference between income between milk producing and meat producing system. Investment in the infrastructure needed for milk production is also lost. This countermeasure would not be feasible in all ecosystems.

18. An outline strategy for reduction of dose via animals.

Reducing dose via animals involves countermeasures that must be applied every year, because none of the measures described in Chapter 17 have any long term effects. The countermeasures involving reductions in contamination of vegetation which may be fed to animals, which often do have a long-term effect have been considered in Part 2. The different countermeasures and their effectiveness and cost are summarised in Table 18.1.

Table 18.1 *Effectiveness (% reduction) of countermeasures against radiocaesium contamination and cost (ECU/kg) for different animal products. The cost of banning is equal the monetary value of the product and assumes that there are other similar products easily available. The costs are mainly based on values given in Chapter 17, but some values have also been estimated from other Norwegian experiences with countermeasures after the Chernobyl fallout.*

Food product	**Cattle milk**		**Cattle meat (beef)**		**Sheep meat**		**Goat milk**		**Moose meat**		**Roe deer meat**		**Reindeer meat**	
	Reduction %	**Cost ECU/kg**	**Reduction %**	**Cost ECU/kg**	**Reduction %**	**Cost ECU/kg**	**Reduction %**	**Cost ECU/kg**	**Reduction %**	**Cost ECU/kg**	**Reduction %**	**Cost ECU/kg**	**Reduction %**	**Cost ECU/kg**
Banning:	100	0.63	100	6.3	100	6.3	100	0.63	100	6.3	100	6.3	100	6.3
Special feeding:	100	0.059	50	0.12	50	0.66	100	0.10	-	-	-	-	50	6.0
Change slaughter/ hunting time	-	-	-	-	-	-	-	-	75	-	75	-	70	0.60
Binder:														
AFCF as powder or in concentrate	90	0.022	80	0.14	-	-	90	0.008	-	-	-	-	-	-
Bentonite as powder or in concentrate	50	0.004	45	0.024	-	-	50	0.001	-	-	-	-	-	-
AFCF in salt lick	-	-	-	-	50	0.076	50	0.004	50	0.081	50	0.078	50	0.077
AFCF in boli	70	0.030	70	0.22	50	0.54	50	0.031	-	-	-	-	50	0.49

Monetary cost of the countermeasures and the reduction achieved in radiological risk are important, but are not the only factors determining whether countermeasures should be applied. For monetary cost and the radiological risk reduction, Crick (1992) developed a theory showing that countermeasures should be applied for contamination activity concentrations in the range:

$$\left\langle \frac{P}{B} \cdot \frac{f}{f-1} \cdot \frac{B}{\alpha \bullet H}, f \cdot \left(1 - \frac{P}{B}\right) \cdot \frac{B}{\alpha \bullet H} \right\rangle$$

where P is the cost of the countermeasure per unit mass of the animal product (ECU kg^{-1})
B is the cost of providing alternative food when the product is banned (ECU kg^{-1})
f is the effectiveness of the countermeasure (the ratio of the dose without countermeasure to that with the measure)
α is the cost assigned to saving unit collective dose (in the Nordic countries the value of 100,000 USD per person-Sv is accepted (SSI, 1991)), and
H is the dose per unit intake (for ^{137}Cs - $1.3x10^{-8}$ Sv/Bq)

This may be used as a guide to when countermeasures could be applied. Using B values of 0.63 and 6.3 for milk and meat respectively (Table 18.1) it is possible to estimate the range of contamination levels where countermeasure use is justified. The range of contamination levels depending on the costs of the countermeasures are shown for countermeasures with differences in effectiveness in Fig. 18.1. For cases where all the contamination is removed (e.g. banning or special feeding for dairy animals) the countermeasure will be justified as long as the contamination levels in the products are above the value of P/(α H).

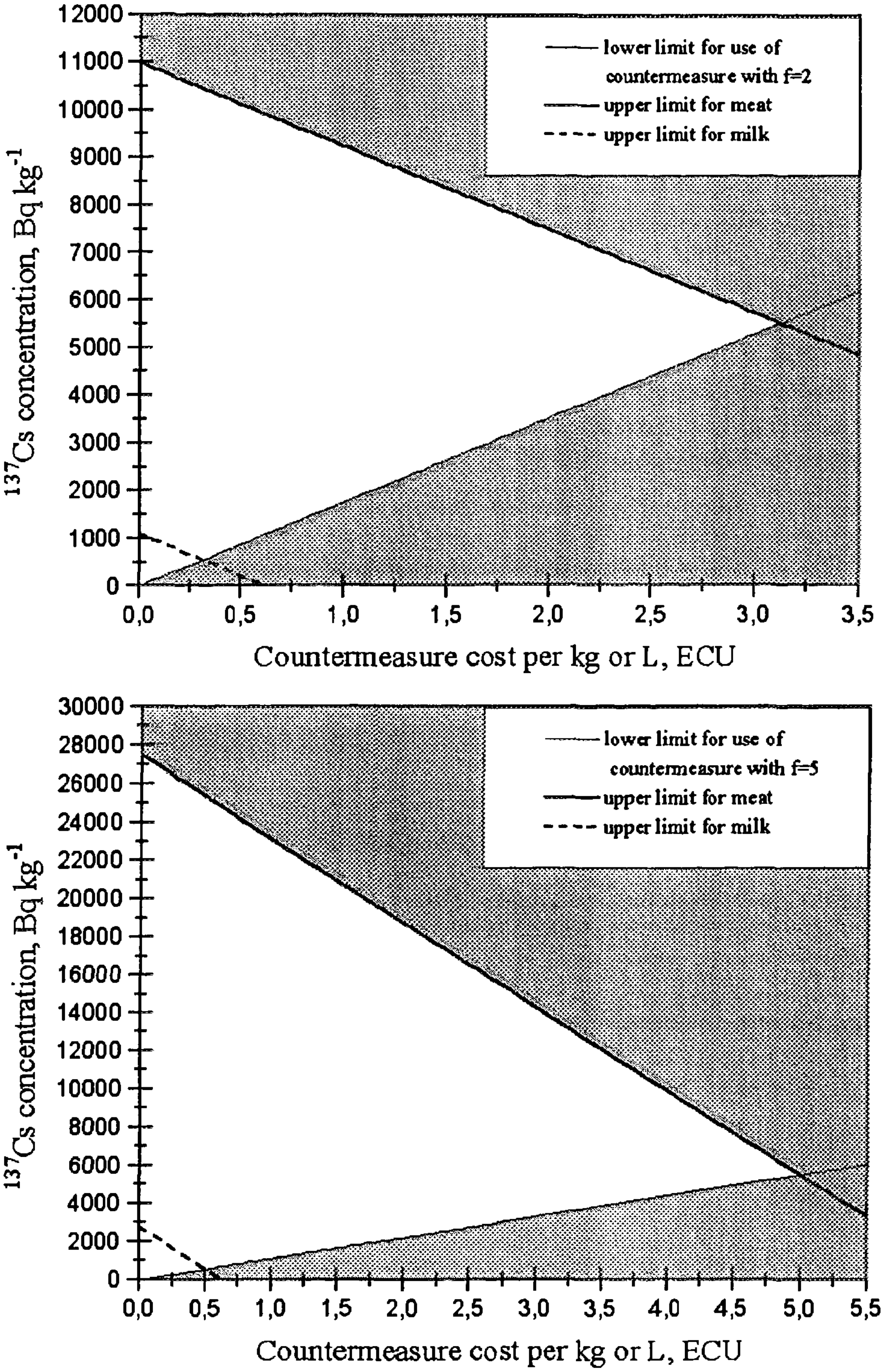

Figure 18.1 *Range of ^{137}Cs contamination levels in animal products and countermeasure costs (Table 18.1) where application of countermeasures are justified (white area) on the basis of monetary cost and achieved radiological risk reduction. The upper figure shows the situation for countermeasures with f=2 (i.e. 50% reduction, see Table 18.1), the lower figure for f=5 (i.e. 80% reduction). For meat ^{137}Cs concentrations and countermeasure costs within the whole white area are justified, while the values for milk should lie between the dashed line and the shaded area below.*

Table 18.3 Contamination activity concentrations (Bq kg^{-1} or L^{-1}) in animal products the first year (year 1) after deposition of 1 MBq/m^2. Also given are the achieved radiocaesium activity concentrations following application of different countermeasures, and the cost of the measure per reduced dose (ECU/person-Sv) obtained using the values in Table 18.1.

Food product	**Cattle milk**		**Cattle meat (beef)**		**Sheep meat**		**Goat milk**		**Moose meat**		**Roe deer meat**		**Reindeer meat**	
^{137}Cs (Bq kg^{-1} or L^{-1})	**4000**		**24000**		**50000**		**10000**		**20000**		**50000**		**800000**	
Countermeasure:	^{137}Cs level	Cost per person-Sv	^{137}Cs level	Cost per person-Sv	^{137}Cs level	Cost per person-Sv	^{137}Cs level	Cost per person-Sv	^{137}Cs level	Cost per person-Sv	^{137}Cs level	Cost per person-Sv	^{137}Cs level	Cost per person-Sv
Banning:	0	12000	0	20000	0	9600	0	4800	0	24000	0	9600	0	600
Special feeding:	0	1100	12000	740	25000	2000	0	770	-	-	-	-	400000	1150
Change slaughter/ hunting time	-	-	-	-	-	-	-	-	5000	-	12500	-	240000	82
Binder:	-	-	-	-	-	-	-	-	-	-	-	-	-	-
AFCF as powder or in concentrate	400	470	4800	550	-	-	1000	64	-	-	-	-	-	-
Bentonite as powder or in concentrate	2000	140	13200	170	-	-	5000	19	-	-	-	-	-	-
AFCF in salt lick	-	-	-	-	25000	230	5000	58	10000	620	25000	240	400000	15
AFCF in boli	1200	820	7200	1000	25000	1600	5000	480	-	-	-	-	400000	95

From the table, a priority order can be made for countermeasures for different products:

1. Applying bentonite as powder or as concentrate is the most cost effective countermeasure for daily «hand-fed» animals. However, with high radiocaesium activity concentrations bentonite may not be sufficiently effective. In such cases, AFCF as powder or as concentrate should be considered before special feeding.
2. For free ranging animals, salt licks with AFCF is most cost effective. When this is not applicable AFCF in boli should be considered before special feeding.

The cost effectiveness of a countermeasure varies between products, and a countermeasure strategy will also need to take contamination levels into account. It might, for instance, be necessary to combine different countermeasures to achieve contamination levels below intervention limits, for instance special feeding after animals have been grazing with boli or salt licks with AFCF.

As indicated in Table 18.3, radiocaesium activity concentrations can still persist above intervention levels after application of different countermeasures. Then dietary advice may be needed. Changing methods of preparing food (e.g. cooking, preserving) can significantly reduce radiocaesium concentrations, and reducing the intake of specific food products (like game, wild mushrooms) will also give lower radiocaesium intake by the population. These countermeasures have been shown to be very cost effective, but they also have larger impacts on peoples lifestyle.

Banning is the most expensive countermeasure, and would only be justified when contamination levels are so high that other countermeasures together with dietary advice cannot sufficiently reduce the radiocaesium intake in the population. And when you are not yet ready with other measures.

Since transfer and long term behaviour of radionuclides, as well as costs and effectiveness of countermeasures against the radionuclide contamination differs between animal products, the cost-effectiveness in terms of cost per saved population dose will vary with the passage of time. The cost per person-Sv will follow the equation:

$$R_{manSv} = \frac{P}{H \cdot E \cdot C_0} \cdot e^{\frac{\ln 2}{T_{eff}} \cdot (t-1)}$$

Here P is the cost of the countermeasure per unit mass of the animal product (ECU kg^{-1})
H is the dose per unit intake (for ^{137}Cs it is 1.3×10^{-8} Sv/Bq)
E is the effectiveness of the countermeasure (% reduction in radionuclide activity concentrations)
C_0 is the radionuclide activity concentration the first year after the fallout (year 1)
T_{eff} is the effective ecological half-life, and
t is years since the fallout occurred.

In the fallout scenario given here the 17th year after the accident will be the last year when countermeasures against caesium contamination of milk from unimproved areas will be needed. Two values can illustrate the changed cost effectiveness during these 16 years; compared to the values in Table 18.3 applying AFCF in concentrates for dairy cattle is 10 times more expensive, while the cost per person-Sv of salt licks for sheep is 4-5 times more expensive.

Wild animals
Radiocaesium contamination of wild animals can be a significant long term problem, and give higher doses to special groups in the population. Countermeasures appropriate for game have been discussed in the text earlier, and are also indicated in Table 18.1 and 18.2. In addition to the countermeasures shown in these tables, changing slaughter time and imposing restrictions on consumption of game animal products (including dietary advice and culinary preparation methods) are countermeasures that should be assessed.

19. References

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