DRAFT

# TRANSFER OF RADIONUCLIDES BY TERRESTRIAL FOOD PRODUCTS FROM SEMI-NATURAL ECOSYSTEMS

**VAMP TERRESTRIAL WORKING GROUP**

Part of the IAEA/CEC Co-ordinated Research Programme on the Validation of Environmental Model Predictions (VAMP)

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# TRANSFER OF RADIONUCLIDES BY TERRESTRIAL FOOD PRODUCTS FROM SEMI-NATURAL ECOSYSTEMS TO MAN

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## 1. INTRODUCTION

The Chernobyl accident focussed attention on the transfer of radionuclides, and in particular radiocaesium transfer through natural and semi-natural ecosystems to man. Within the VAMP programme attention has been drawn to the need to consider the potential significance of foodstuffs from these ecosystems (hereafter termed "natural food products") in radiological assessments.

Natural and semi-natural ecosystems, as discussed in this review, include non-intensively managed areas such as forests, heathlands, uplands, mountain pastures, mediterranean dry-shrub areas, marshlands and tundra (Desmet & Myttenaere 1988). In such ecosystems high and, in some cases, prolonged, bioavailabilities of some important radionuclides have been noted. Major foodstuffs taken from these ecosystems include fungi, berries, honey, meat from game animals and meat and milk from domestic ruminants. In these ecosystems, species diversity is much more pronounced than in agricultural systems, and many species are known to accumulate comparatively high levels of radiocaesium. Even within the same species the variation in radiocaesium levels can be considerable.

Models developed before the Chernobyl accident were usually limited to considering agricultural food production, and they predicted radionuclide levels in such foodstuffs reasonably well. However, when these models were used they failed to predict radionuclide levels in natural food products from semi-natural ecosystems with an adequate degree of precision. Semi-natural ecosystems were previously considered only to a limited extent (with the notable exception of reindeer), because the quantities of food consumed from these systems were assumed to be comparatively small and usually by only a minor proportion of most countries populations. They were therefore usually not considered in dose assessments.

Although semi-natural ecosystems provide comparatively small quantities of food, the long-term dose commitment from these ecosystems to humans can be significant, particularly for radiocaesium. This is primarily due to the high levels of $^{137}Cs$ found in many natural food products and the much longer effective half-lives of $^{137}Cs$ observed in these ecosystems compared with agricultural systems (Hove & Strand 1990; Bergman & Johansson 1989; Bergman et al. 1991; Howard et al. 1991). Intake of food from semi-natural ecosystems is not evenly distributed in the population, and certain critical groups, such as hunters, sheep and reindeer breeders and berry and fungi pickers can have particularly large intakes of natural food products (Strand et al. 1989; Strand et al. 1990; Johanson & Bergström in press a). Consumption rates and the size of the critical groups which eat natural food products from semi-natural ecosystems varies greatly both between and within countries. Whilst high radiocaesium levels in natural food products often persist, those from agricultural areas generally decline quickly so that with each year after the deposition of fallout the comparative importance of transfer of radiocaesium by natural food products from semi-natural ecosystems to man increases.

The aim of this report is to provide a short summary of the most important factors affecting radionuclide transfer in semi-natural ecosystems, and to attempt to derive empirical relationships which can be used for modelling purposes.

For agricultural systems models have been developed which describe the transfer of radionuclides to food products with a reasonable degree of accuracy. However, modelling such transfers to food and finally to man is much more difficult for semi-natural ecosystems due to a lack of understanding of the processes involved in controlling the fate of radionuclides in these complex ecosystems. There is a pronounced heterogeneity in soil properties in these ecosystems, and furthermore, a much higher variability in dietary selection of food producing animals compared with agriculturally improved areas.

The transfer to plants and animals is often expressed using concentration ratios, or transfer coefficients for animals, defined as the equilibrium ratio between the activity concentration in milk or meat divided by the daily intake ($F_m$ and $F_f$ respectively). However it is often difficult to get reliable estimates of these parameters to use to estimate transfer ratios or coefficients in semi-natural ecosystems. For instance, the bulk density of soils in semi-natural ecosystems can vary markedly giving rise to very different transfer values to plants. For animals, the estimation of daily herbage intake is particularly difficult in semi-natural ecosystems. Therefore, it is questionable if it is appropriate to use such transfer parameters in these circumstances.

Therefore in this report we have attempted to provide easily derived, empirical transfer coefficients, termed aggregated transfer coefficients ($T_{ag}$) which can be used in predictive models instead of the commonly used transfer parameters. $T_{ag}$'s are calculated using the expression:

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

and therefore are expressed as $m^2\ kg^{-1}$ or $m^2\ L^{-1}$. Activity levels are commonly expressed as fresh weight (fw), and whilst this is most appropriate for animal products more consistent values will be obtained for plant products by using dry weights (dw).

$T_{ag}$ values are relatively easy to calculate although, in common with other transfer coefficients, they require specific knowledge of the ecosystems for which they are being used (Howard et al. 1991). Given the complexity of the ecosystems and the lack of detailed knowledge there seems, at present, to be few practical, generally applicable, alternatives to the use of $T_{ag}$ values for making long-term predictions in many types of semi-natural ecosystem.

Transfer parameters are of limited usefulness when trying to predict radiological impact, unless they are combined with estimates of time-dependent changes in transfer. This is of particular importance in semi-natural ecosystems where high radiocaesium levels can persist for many years. Reductions in activity levels with time are usually expressed using ecological ($T_{ec}$) or effective ($T_{ef}$) half-lives.

The effective half-life describes the time required for the activity concentration of a radionuclide in a food product to be reduced to one half of the original concentration in a specific system. Therefore the $T_{ef}$ incorporates physical decay. In this report the $T_{ef}$ takes into account all biological, chemical and physical processes within the ecosystem which influence the transfer of a radionuclide to a food product.

The ecological half-life does not take physical decay into account, and therefore can be adapted for different isotopes of the same element. For example, the $T_{ef}$ of $^{134}Cs$ and $^{137}Cs$ will differ because of the differences in physical half-lives, while the $T_{ec}$ would be identical. The relationship between $T_{ef}$ and $T_{ec}$ for a radionuclide with a physical half-life ($T_{phys}$) will be:

$$1/T_{ef} = 1/T_{phys} + 1/T_{ec}$$

The $T_{ef}$ and $T_{ec}$ values considered here describe only the long-term losses. They do not include the short-term reductions in activity concentrations which often occur in the first few months after deposition.

The discussion will primarily consider radiocaesium; the few data which are available on the transfer of other radionuclides to foodstuffs will be included where possible.

## 2. IMPORTANT PARAMETERS GOVERNING RADIONUCLIDE BEHAVIOUR IN SEMI-NATURAL ECOSYSTEMS.

### 2.1 Interception and retention on plant surfaces

The rate of interception is highly variable in these ecosystems, and will depend on factors such as vegetation biomass, and the presence of coniferous or deciduous tree stands. Interception of radiocaesium can therefore range from below 5% of the total deposited on heavily grazed pastures in mountain areas with low vegetation biomass to >50% in semi-natural pasture (Livens et al. 1992) and 60-100% in dense coniferous forest (e.g. Tikhomirov & Shcheglov 1991). In thick lichen mats interception is nearly complete and the $T_{ec}$ of retention of the original deposit is often in the order of years. Although it may take weeks or even months before all intercepted activity is transferred from the surface of many plant species to the soil, this intercepted component usually will not greatly influence the overall dose commitment to man from consumption of natural food products because of the high root uptake and long $T_{ec}$ of radiocaesium in these ecosystems.

The physico-chemical form of the intercepted deposition may be important when plants which are externally contaminated are eaten by animals, as demonstrated by the relatively low bioavailabilty of Chernobyl fallout radiocaesium in 1986 compared with later years (Ward et al. 1989; Beresford et al. 1989; Hansen & Hove 1991). However, the effect will be limited to the period during which the initial deposit is retained on the plant surfaces. Furthermore, the physico-chemical form of the deposit, and how this changes with time, will also influence the rate of root uptake.

## 2.2 Soil properties

One of the most important factors responsible for the high radiocaesium levels in food products from semi-natural ecosystems is root (or hyphae) uptake from the soil. This was not extensively studied before the Chernobyl accident possibly because the fallout from atmospheric weapons testing was deposited continuously on to plant surfaces over a long period, thereby masking the importance of root uptake. Nevertheless the high uptake of $^{137}Cs$ from soils with a high organic matter content in the upper layers was noted (Barber 1964). The low capacity of semi-natural ecosystems to immobilize radiocaesium is one of the main factors responsible for the continuing high radiocaesium levels in plants and animals. A soil in which a high rate of root uptake of radiocaesium occurs has been characterized in N. Europe as typically having a low clay and potassium content, a low pH, and a high organic matter content (Livens & Loveland 1988). Radiocaesium is present in a bioavailable state in these soils, not strongly bound to soil components, and consequently available for uptake from the soil solution (review: Desmet et al. 1991). Since radiocaesium usually migrates slowly down the soil profile it remains for long periods in the upper soil layers where root biomass is greatest.

In soils of many semi-natural ecosystems the fungal hyphae are usually confined to the upper organic horizons of the soil. Assessments of fungal biomass and measurements of radiocaesium uptake by hyphae suggest that a high proportion of deposited radiocaesium may be present in the fungal mycelium (Olsen et al. 1990; Dighton et al. 1991).

Radiocaesium uptake by vegetation is also affected by the rate at which it moves down the soil profile. For example, Beresford et al. (1992) found that $T_{ag}$ values for Chernobyl $^{137}Cs$ in vegetation in upland ecosystems in the UK were higher than those for pre-Chernobyl $^{137}Cs$ ("aged deposits"). They attributed this to the greater depth of the "aged deposit", over 50% of which was below the 0-4 cm layer down the soil profile and therefore out of the rooting zone of the vegetation. Such distribution in the soil profile will obviously also affect $T_{ag}$ estimates (for comparison <25% of Chernobyl $^{137}Cs$ was below 4 cm).

## 2.3 Radiocaesium uptake into vegetation

The proportion of radiocaesium that resides in vegetation of semi-natural ecosystems is often much greater than that which occurs in agricultural ecosystems. In upland ecosystems of the UK between 0.5 - 5.5% of the total deposition in December 1986 was found to be associated with above-ground vegetation (Coughtrey et al. 1989) although some of this may have been residual surface deposition from the Chernobyl accident. In Norway 0.1 - 0.5% of the total deposit has been transferred yearly from soil to above ground vegetation in the period 1986-1990 in mountain pastures (Garmo pers comm). In comparison, for agricultural crops values would usually be in the range 0.01% per year for cereals and root crops.

The observed high radiocaesium activity concentrations in mosses and lichen are primarily due to efficient interception and retention of the original deposit. However, fungi have the highest radiocaesium activity concentrations among all the species in semi-natural ecosystems. Radiocaesium levels in vascular plants can be ranked in the following order: dwarf shrubs > herbs > grasses > shrubs and trees (e.g. Garmo et al. 1990, Horrill et al. 1990), however, there are many exceptions.

## 2.4 Animal feeding habits

Herbivores vary widely in their feeding habits when grazing in semi-natural ecosystems. In addition to differences in feed selection between animal species, seasonal differences in the available forage plants contribute markedly to variation in radiocaesium activity concentrations recorded in animal food products. High radiocaesium activity concentrations have been reported for animal products from many semi-natural ecosystems. This is due to the high radiocaesium levels in the vegetation species eaten by animals combined with the comparatively high transfer of radiocaesium to small ruminant species such as sheep, goats and roe-deer which predominate in these ecosystems (review: Howard et al. 1991).

# 3. TRANSFER OF RADIOCAESIUM TO FOOD PRODUCTS FROM SEMI-NATURAL ECOSYSTEMS

A brief description of the characteristics of radiocaesium accumulation in a range of natural food products is given below, together with $T_{ag}$ values. The review does not include freshwater fish, which can be a significant source of radiocaesium in the diet (e.g. Mattsson & Moberg 1991).

## 3.1 Fungi

High radiocaesium activities have been reported in the fruiting bodies (mushrooms) of a number of fungal species both before and after the Chernobyl accident (e.g. Grüter 1967; Dighton & Horrill 1988; Horyna & Randa 1988; Bakken & Olsen 1990a). However, radiocaesium levels are highly variable among fungal species (e.g. Roemmelt et al. 1990) and within species variation can be considerable (Mascanzoni 1990; Guillite et al. 1990). Radiocaesium activity concentrations in the fruiting bodies of some species (e.g. *Cantharellus cibarius, Leccinum testaceoscabrum*) are comparable with higher plants growing in the same area. In contrast, in other species radiocaesium activity concentrations which are 10 - 100 times higher than those of higher plants can be found (e.g. *Rozites caperata, Suillus variegatus, Lactarius rufus*) (Bakken & Olsen 1990a; Dighton & Horrill 1988; Haselwandter et al. 1988). The various reasons suggested as causes of the variation include differences in soil properties, hyphal distribution in the soil profile, mycorhizzal physiology and biochemical transportation properties (Roemmelt et al. 1990; Randa et al. 1990; Guillitte et al. 1990; Bakken & Olsen 1990b). Taking the depth of the mycelium and precipitation during the 10 days following the Chernobyl accident into consideration, Nimis et al. (1986) were able to subdivide radiocaesium contamination of macrofungi into several ecological categories (parasites, symbionts with deciduous plants, symbionts with coniferous trees and saprophytes). The most highly contaminated species have been ranked into ecological categories by Guillitte (1992) and are summarised in Table I.

Table I. Examples of fungal species in different ecological categories which are generally most highly contaminated with radiocaesium.

| Ecological category | Genera | Other species |
|---|---|---|
| Mycorrhizal | *Cortinarius*<br>*Dermocybe*<br>*Inocybe*<br>*Rozites*<br>*Xerocomus*<br>*Suillus*<br>*Tyolopilus*<br>*Paxillus*<br>*Hygrophorus*<br>*Laccaria* | (thought to be facultative mycorrhiza species)<br>*Lactarius rufus*<br>*L. theiogalus*<br>*L. camphoratus*<br>*Cantharellus tubaeformis*<br>*Tricholoma album*<br>*Amantopsis* (sub-genera of *Amanita*) |
| Saprophytic | *Clitocybe*<br>*Lepista*<br>*Cantharellula* | especially *C. clavipes* |
| Parasitic | *Armarillaria* | *Pholiota squarrosa* |

Changes in fungal radiocaesium levels with time since the Chernobyl accident generally depend on the depth of the mycelium. It has been suggested that the gradual migration of radiocaesium into deep soil layers will increase the uptake of radiocaesium by fungal species which have mycelium at lower depths (Byrne 1988; Horyna & Randa 1988; Guillite et al. 1990; Mascanzoni 1990; Randa et al. 1990; Borio et al. 1991). Radiocaesium levels in fungi with deeper mycelium, such as *Boletus edulis*, had comparatively low levels soon after the Chernobyl accident (Elstner et al. 1989; Fraiture et al. 1990; Randa et al. 1990) and also a low ratio of $^{134}Cs$:$^{137}Cs$ which indicates an uptake of nuclear weapons test fallout. In the first year after deposition of Chernobyl fallout, Wirth et al. (1992) found that saprophytic species, which have a superficial mycelium in the upper litter horizon (L-horizon), were the most highly contaminated. After the first year following deposition the pattern changed and the mycorrhizal (symbionts) and parasitic fungi had higher radiocaesium activities than the saprophytes (Wirth et al. 1992).

Few data in the literature are presented in an appropriate form, giving activity concentrations in fungi and ground deposition in soil which allow $T_{ag}$ estimations to be made. Where possible $T_{ag}$ for different species have been estimated and are presented in Table II. Dry weight conversions have been carried out where necessary assuming 8% dry matter in fungi. Changes in mean activity concentrations with time in a few species from the most comprehensive data set available from Finland are also shown in Fig. 1 (Rantavaara 1990 and priv. comm.).

Table II. Aggregated transfer coefficients for $^{137}Cs$ in various fungal species calculated for different years after the Chernobyl accident.

| Species | | Year | $T_{ag}$ ($m^2$ $kg^{-1}$ dw) | | | |
|---|---|---|---|---|---|---|
| | | | Mean | Median | Min | Max |
| *Cantharellus cibarius* | (a) | 1984 | 0.28 | 0.33 | 0.039 | 0.42 |
| | | 1986 | 0.12 | 0.091 | 0.016 | 0.54 |
| | | 1987 | 0.081 | 0.035 | 0.018 | 0.36 |
| | | 1988 | 0.15 | 0.091 | 0.016 | 0.43 |
| | | 1989 | 0.23 | 0.29 | 0.034 | 0.38 |
| | | 1990 | 0.26 | 0.18 | 0.021 | 0.66 |
| | (b) | 1991 | 0.29 | 0.20 | 0.010 | 0.96 |
| | (c) | 1990 | 0.02 | | | |
| | | 1988-91 | 0.34 | | | |
| | (d) | 1987-91 | 0.12 | 0.11 | 0.05 | 0.20 |
| *Cantharellus tubaeformis* | (a) | 1986 | 0.55 | 0.39 | 0.28 | 1.5 |
| | | 1988 | 0.98 | 1.08 | 0.44 | 1.61 |
| | | 1989 | 1.18 | 0.90 | 0.46 | 2.47 |
| | | 1990 | 0.86 | 0.97 | 0.40 | 1.20 |
| | | 1991 | 0.73 | 0.73 | 0.18 | 1.52 |
| | (d) | 1987-91 | 0.63 | 0.68 | 0.14 | 1.17 |
| *Lactarius trivialis* | (a) | 1984 | 1.13 | 1.12 | 0.92 | 1.37 |
| | | 1986 | 0.89 | 1.03 | 0.02 | 3.00 |
| | | 1987 | 2.51 | 2.01 | 1.65 | 4.50 |
| | | 1988 | 2.66 | 2.63 | 0.85 | 5.15 |
| | | 1989 | 1.37 | 1.02 | 0.07 | 3.47 |
| | | 1990 | 1.92 | 1.91 | 0.69 | 4.14 |
| | | 1991 | 2.15 | 2.21 | 1.17 | 3.07 |
| *Leccinum versipelle* | (a) | 1984 | 0.096 | | | |
| | | 1986 | 0.05 | 0.03 | 0.005 | 0.14 |
| | | 1987 | 0.14 | 0.057 | 0.014 | 0.57 |
| | | 1988 | 0.11 | 0.046 | 0.016 | 0.74 |
| | | 1989 | 0.058 | 0.045 | 0.020 | 0.11 |
| | | 1990 | 0.065 | | 0.020 | 0.11 |
| *Leccinium scabrum* | (e) | 1991 | 0.050 | | | |
| | | 1969 | 0.023 | | | |
| *Boletus edulis* | (a) | 1984 | 0.061 | | 0.030 | 0.092 |
| | (f) | 1986 | 0.25 | 0.02 | 0.003 | 1.7 |
| | (a) | 1986 | 0.063 | | | |
| | | 1987 | 0.008 | 0.009 | 0 | 0.016 |
| | | 1988 | 0.052 | 0.041 | 0.003 | 0.19 |
| | | 1989 | 0.13 | 0.053 | 0.016 | 0.32 |
| | | 1990 | 0.20 | | 0.013 | 0.38 |
| | (b) | 1991 | 0.057 | | | |
| | (c) | 1990 | 0.039 | | | |
| | | 1988-91 | 0.057 | | 0.013 | 0.026 |
| | (d) | 1987-91 | 0.07 | 0.07 | 0.05 | 0.10 |

Table II. Aggregated transfer coefficients for $^{137}Cs$ in various fungal species calculated for different years after the Chernobyl accident (continued).

| Species | | Year | $T_{ag}$ ($m^2$ $kg^{-1}$ dw) | | | |
|---|---|---|---|---|---|---|
| | | | Mean | Median | Min | Max |
| *Suillus granulatus* | (e) | 1969 | 0.12 | | | |
| | (d) | 1987-91 | 0.08 | 0.08 | 0.01 | 0.14 |
| *Rozites caperata* | (g) | 1988 | | | 1.88 | 3.13 |
| | (a) | 1990 | 2.25 | 2.15 | 1.89 | 2.70 |
| | | 1991 | 1.54 | | 0.78 | 2.30 |
| *Xerocomus badius* | (a) | 1986 | 1.7 | 1.44 | 0.29 | 4.8 |
| | (h) | 1987 | 1.25 | 0.84 | 0.58 | 3.9 |
| | | 1988 | 3.22 | 2.3 | 0.44 | 7.1 |
| | | 1990 | 0.31 | | | |
| | (b) | 1988-91 | 2.19 | | 0.43 | 2.8 |
| | (c) | 1986 | 1.36 | 0.9 | 0.25 | 2.8 |
| | (d) | 1987-91 | 0.55 | 0.45 | 0.19 | 1.37 |
| *Xerocomus chrysentheron* | (g) | 1987 | 1.29 | 0.76 | 0.23 | 5.2 |
| | | 1988 | 1.63 | 1.38 | 0.64 | 2.5 |
| | (d) | 1989 | 0.03 | | 0.010 | 2.3 |
| | (c) | 1988-91 | 0.90 | | | |

(a) Rantavaara 1990 and priv. comm. (b) Tikhomirov et al. in press, (c) Henrich, priv. comm., (d) Kammerer & Wirth priv. comm., (e) Trabalka, priv. comm., (f) Mascanzoni 1990, (g) Hove & Strand 1990, (h) Horyna priv. comm.

Even an intake of comparatively small quantities of fungi may transfer substantial activities of $^{137}Cs$ due to the high activity concentrations found in some fungi. In Sweden for example, about 24 million kg fw of mushrooms are picked each year, corresponding to about 3 kg fw per person (about 0.25 kg dw) (Kardell 1985). A mean $T_{ag}$ based on dry weights for all fungal species picked within one area in Sweden during 1988 to 1991 was 1.25 $m^2$ $kg^{-1}$ (Johanson priv. comm.). This implies that the mean $^{137}Cs$ activity concentration for mushrooms in Sweden, where the mean deposition was 10 kBq $m^{-2}$, will be 12.5 kBq $kg^{-1}$ dw, and the mean annual intake of $^{137}Cs$ from mushroom could be 3 kBq.

Similarly a small proportion of the population in Czechoslovakia, which were assumed to have an annual consumption of about 5 kg fw of mushrooms, were estimated to have a potential radiocaesium intake from fungi of 1.5 kBq $y^{-1}$ (Horyna 1991). In contrast, other foodstuffs from agricultural systems contributed 4-5 kBq during the first year after deposition, but only 0.5 kBq $y^{-1}$ thereafter.

When using $T_{ag}$'s for modelling transfer to humans it is important to note that we have not considered the reduction in $^{137}Cs$ activity concentrations that occur during food processing which can exceed 90% in mushrooms (Noordijk & Quinault 1992). If people eat species with low transfers of radiocaesium and/or cook them using a method which reduces the radiocaesium content they will consume much lower amounts of radiocaesium.

Information given to the public after a nuclear accident may change food consumption habits markedly. In many countries affected by high deposition after the Chernobyl accident people were advised to reduce their consumption of species with particularly high radiocaesium content. In a household survey in Finland in 1990 the consumption of fungi was found to be reduced by 60% compared with a similar survey in 1985, while the consumption of wild berries was reduced by 30% (Rantavaara priv. comm.).

Since there are large variations both between and within different species, doses based on the mean values given in Table 1 are likely to be uncertain. Knowledge of $T_{ag}$'s for fungi is, at present, fragmentary. Because of the high transfers of radiocaesium to fungi, and the potential importance for radiocaesium transfer to humans, it is clear that systematic collection of dry weight activity concentrations in fungi and deposition values for the soil in which they are growing are needed. An alternative approach to using $T_{ag}$'s has been adopted by Wirth et al. (1992) who have calculated transfer factors for soil to fungi based on the radiocaesium activity concentration in only the organic horizon of the soil. This method was suggested because fungal mycelium are preferentially found in organic horizons and are unlikely to absorb much radiocaesium from lower mineral horizons, where it will be strongly fixed.

Levels of $^{90}Sr$ in fungi are low compared with those of higher plants and correspond to the similarly low levels of its analogue calcium (Roemmelt et al. 1990; Wirth et al. 1992). Accumulation of $^{110m}Ag$ from the Chernobyl fallout in fungi was noted at activity concentrations of up to 500 Bq $kg^{-1}$ dw in Slovenia (Byrne 1988).

3.2 Berries

Ericoid dwarf shrubs, such as bilberry (*Vaccinium myrtillus*) and lingonberry (*Vaccinium vitis-idaea*) accumulate comparatively high levels of radiocaesium (Bunzl & Kracke 1986, Johanson et al. 1991). In northern parts of Europe cranberries (*Vaccinium oxycoccus*) and cloudberries (*Rubus chameomorus*), which grow on peaty, nutrient poor bogs, also have high levels of radiocaesium (Johanson et al. 1991). Cloudberries growing in an area with a deposition of 35 kBq $m^{-2}$ had radiocaesium activity concentrations of up to 8 kBq $kg^{-1}$ dw.

Estimates of $T_{ag}$'s for commonly consumed wild berries are listed in Table III. In comparison with many fungal species, transfer from soils to berries is low. Berries will therefore make only a small contribution to population dose, even when the average annual consumption of berries is high such as occurs in Nordic countries (e.g. 4-8 kg of fresh berries, Hultman 1983, Finnish household survey 1990). However, as for fungi, there may be small, critical groups who collect and eat large quantities of berries. It is possible that members of the critical group for fungi will also be in the critical group for berries.

Table III. Aggregated transfer coefficients for radiocaesium transfer to berries used for human consumption.

| Species | $T_{ag}$ ($m^2$ $kg^{-1}$ dw) |
|---|---|
| Cloudberry | 0.002-0.23[a]; 0.13[b] |
| Lingonberry | 0.04[a]; 0.032[b] |
| Bilberry | 0.017[c]; 0.04[a]; 0.041[b] |

(a) Rantavaara 1990 and priv. comm., (b) Johanson et al. 1991, (c) Block & Pimpl 1990.

Data is also available from Ukraine which gives $T_{ag}$ values for "fresh berries" of 0.001 - 0.004 $m^2$ $kg^{-1}$, which can be converted to dry weight using a dry matter content of 15 % to 0.006 - 0.03 $m^2$ $kg^{-1}$ (Tikhomirov & Shcheglov 1991). Equivalent values for $^{90}Sr$ based on dry weight from this source are 0.0015 - 0.002 $m^2$ $kg^{-1}$.

3.3 Honey

The comparatively high levels of radiocaesium which are found in heather (*Calluna vulgaris*) result in high levels of radiocaesium in heather honey. Few data for transfer to heather honey are available, although in Germany heather honey was found to be nearly 100 times as highly contaminated as flower honey and levels of $^{239/240}Pu$ and $^{90}Sr$ were also higher in heather honey (Bunzl & Kracke 1981).

In Sweden a mean $T_{ag}$ for $^{137}Cs$ for all types of honey was estimated to be about 0.005 $m^2$ $kg^{-1}$ (Stark priv. comm.). Since the mean $^{137}Cs$ deposition in Sweden is about 10 kBq $m^{-2}$ the estimated mean activity concentration in honey will thus be 50 Bq $kg^{-1}$. The consumption of honey in Sweden is about 4 million kg or about 0.5 kg per person. Honey consumption by the critical group of bee keepers in Sweden is probable 10 times higher than the mean.

In Finland from 1977 and 1978 flower honey had a mean $T_{ag}$ of 0.003 with a range from 0.00006 - 0.023 with an annual median of 0.0014. Pollen from ericaceous species dominated in honey samples with the highest $T_{ag}$'s. From 1986 to 1990 the annual median $T_{ag}$ value was 0.0010 with a range in $T_{ag}$'s of 0.00004 - 0.01 (Rantavaara pers. comm).

Few relevant data have been found on consumption of honey from semi-natural ecosystems. In Norway the average consumption of heather honey is 0.1 - 0.2 kg $y^{-1}$. However, the variation in intake is large, as it is for all foods from these ecosystems. The maximum concentration recorded in Norway is 2.6 kBq $kg^{-1}$ (Hagen priv. comm.). Consumption by critical groups, estimated to consume 2-5 kg $y^{-1}$, would result in a maximum intake of 13 kBq $y^{-1}$.

3.4 Animals

1) GAME ANIMALS

*(i) Moose*

Moose (*Alces alces*) graze mainly on herbs and leaves from shrubs in summer and increase their intake of more contaminated species such as bilberry and heather in the autumn (Cederlund et al. 1981). However radiocaesium intake does not necessarily increase because there is a concommitant reduction in overall herbage intake (Schwartz et al. 1987).

Estimated $T_{ag}$'s for moose are presented in Table IV. Mean $T_{ag}$ values for calves have been found to be consistently higher than for adults (Bergman et al. 1991). The values quoted are usually mean values for $T_{ag}$'s from within one area, but considerable temporal and spatial variation exists in some areas. For instance, in an area of Sweden where the mean value is 750 Bq $kg^{-1}$ fw a variation from 100 Bq $kg^{-1}$ to 3 kBq $kg^{-1}$ was found (Johanson priv. comm.). Since the above $T_{ag}$'s have been calculated from data from an area of Europe where most moose are found they would appear to be the most appropriate values to describe the transfer of radiocaesium for this particular foodchain from the forest ecosystem to man.

Table IV. Aggregated transfer coefficients for moose.

| Year | $T_{ag}$ ($m^2$ $kg^{-1}$ fw) | Reference |
|---|---|---|
| 1979 | 0.011-0.026 (all ages) | Rantaavara (1982) |
| 1985 | 0.015 (calves) | Bergman et al. (1991) |
| 1985 | 0.010 (adults) | Bergman et al. (1991) |
| 1986*-91 | 0.014 (calves) | Rantavaara (priv. comm.) |
| 1986*-91 | 0.010 (adults) | Rantavaara (priv. comm.) |
| 1986-90 | 0.009-0.032 (calves) | Bergman et al. (1991) |
| 1986-90 | 0.006-0.017 (adults) | Bergman et al. (1991) |
| 1986-88 | 0.02 | von Bothmer et al. (1990) |
| 1989 | 0.009-0.019 | Johanson et al. (in press b) |

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

Consumption of moose meat is only important in a few Nordic countries and in parts of the CIS and North America. In Sweden, for example, 135,000 moose were harvested during 1988 corresponding to 12 million kg of meat. This is equivalent to 1.5 kg per person, but the distribution of consumption is highly skewed and many of the critical group of hunters consume around 50 kg $y^{-1}$. In the Gävle district, the most contaminated area in Sweden, 938 moose were harvested in 1989 and the mean $^{137}Cs$ activity concentration was 1.3 kBq $kg^{-1}$ fw. Each of the critical group of approximately 2000 hunters obtained meat containing 60 kBq (Fig. 2: Johanson & Bergström in press a). During the period after the Chernobyl accident some yearly variation in $^{137}Cs$ levels has been noted, but no significant decreases in $^{137}Cs$ activity concentrations have been detected in moose meat (Johanson et al. 1991). Similarly both Bergman et al. (1991) and Rantavaara (1987) have found that $T_{ag}$ values for $^{137}Cs$ were similar before and after the Chernobyl accident, indicating that the physical half-life of $^{137}Cs$ will determine the effective half-life.

*(ii) Roe-deer*

Roe-deer (*Capreolus capreolus*) are the most common small wild deer in the areas which were most highly contaminated by Chernobyl deposition, and the small ruminant species for which we have the most comprehensive understanding of radiocaesium accumulation. It is therefore used as an example of the small deer for the present discussion. Unlike moose, roe-deer eat a wide variety of herbs, grasses and also fungi in large quantities when they are available.

Roe-deer have a pronounced annual variation in their $^{137}Cs$ activity concentrations with a peak in August-September when fungi are abundant. There is a considerable variation in the transfer of radiocaesium to roe-deer both between countries and even between areas within countries. Schönhofer & Tataruch (1988) found high variation in small forested areas where roe-deer have access to agricultural land and suggested that this situation prevents the calculation of meaningful transfer coefficients.

The $T_{ag}$ values presented in Table V are largely derived from, and therefore most relevant for, roe deer inhabiting Nordic forested areas with some access to farmland. In areas where farmland dominates less transfer of radiocaesium to roe-deer seems to occur. The $T_{ag}$ value from the Kyshtym area in Table V is for roe deer harvested in the autumn, when they would be expected to be comparatively highly contaminated. However, in common with other $T_{ag}$ values for the Kyshtym area they are among the lowest recorded (Trabalka priv. comm.).

Table V. Aggregated transfer parameters for roe deer.

| Year | $T_{ag}$ ($m^2$ $kg^{-1}$ fw) | Reference |
|---|---|---|
| 1968 | 0.0009* | Trabalka (pers. comm.) |
| 1988-89 | 0.15 (autumn) | Karlen et al. (1991) |
| 1988-91 | 0.04 (rest of year) | Johanson & Bergström (in press a) |
| 1988-91 | 0.05 (annual mean) | Johanson & Bergström (in press a) |
| 1991 | 0.01-0.20 | Lindner et al. (in press) |

* Kyshtym value, see text.

A roe-deer carcass provides about 10 kg of meat (Johanson & Bergström in press a). So, for example, 200,000 roe-deer harvested annually in Sweden provide a total of about 2 million kg of roe-deer meat, equivalent to 0.25 kg per person. However, this average figure is misleading since most of the meat is consumed within the critical group, the hunters and their families.

Reliable estimates of the $T_{ef}$ for radiocaesium in roe deer have not, as yet, been possible in the Nordic countries due to the large seasonal variations in radiocaesium concentrations in roe deer. There is little evidence of a decline from 1988-1991 (Johanson & Bergström in press a). However, Lindner et al. (in press) have calculated a $T_{ef}$ of 2.6 y in Germany. In Scotland there has been a decrease from a typical value of 500 Bq $kg^{-1}$ fw $^{137}Cs$ during 1986 to 170 in 1989 (Department of Agriculture and Fisheries for Scotland 1990).

*(iii) Other hunted animals*

Other ruminants which are hunted such as red deer (*Cervus elaphus*), fallow deer (*Dama dama*) chamois (*Rubicapra rubicapra*), white-tailed deer (*Odecoileus virginianus*) have not been found to have particularly high levels of radiocaesium. Few data are available from which we can calculate $T_{ag}$'s. Data from Scotland taken in 1989 gives approximate values for red deer of

0.02-0.04 $m^2$ $kg^{-1}$ (Howard priv. comm.). Considering the small volumes of meat from these animals consumed in most countries the contribution to collective dose is likely to be small. In Norway, where levels of radiocaesium in 1986 were comparable to those in moose, about 10,000 red deer are harvested each year corresponding to about 700,000 kg meat.

$T_{ag}$ values for white-tailed deer of 0.017 in 1986 and 0.03 in 1990 were estimated in Finland (Rantavaara priv. comm.). These values are slightly higher than those for moose in Finland. About 6,000-8,000 individual deer are hunted annually with an annual per capita meat consumption of 0.1 kg or less.

The few data available for Arctic hare (*Lepus timidus*) and brown hare (*Lepus capensis*) from Finland quoted in Rantaavara (1990) and from the Kyshtym area are shown in Table VI.

Table VI. Aggregated transfer coefficients in some species of hare.

| Species | n | Year | $T_{ag}$ ($m^2$ $kg^{-1}$ fw) | | | |
|---|---|---|---|---|---|---|
| | | | Mean | Median | Min | Max |
| Arctic hare | 7 | 1967-68[a] | 0.0009 | | | |
| | 75 | 1988-90[b] | 0.038 | 0.027 | 0.006 | 0.104 |
| | 8 | 1986-89[c] | 0.03 | | 0.025 | 0.13 |
| Brown hare | 8 | 1967-68[a] | 0.0021 | | | |
| | 11 | 1987-89[b] | 0.008 | - | 0.0005 | 0.053 |
| | 11 | 1986-89[c] | 0.003 | - | 0.00018 | 0.023 |

(a) Trabalka, priv. comm., (b) Rantaavara, 1990 and priv. comm., (c) Johanson et al. 1990.

*(iv) Wild boar*

Wild boar (*Sus scrofa*) is hunted in many central and eastern European countries. Data from Austria (Tataruch et al. 1990) indicate that a wide range of values may be found even within the same forest area, possibly due to the large home range of wild boar. They reported typical $^{137}Cs$ activity concentrations of 37 Bq $kg^{-1}$ fw for a forest contaminated with about 50 kBq $m^{-2}$ in 1986, with levels apparently rising in 1988 to a maximum of 17.6 kBq $kg^{-1}$ in February 1988. The calculation of $T_{ag}$'s must await further data collection.

*(v) Wildfowl and game birds*

Some high levels of radiocaesium were reported in wildfowl during the first hunting season after the Chernobyl accident (Mascanzoni 1987). In Finland mean values of $T_{ag}$ for both inland waterfowl and terrestrial game birds were 0.01 (Rantavaara, priv. comm.). Lowe & Horrill (1991) reported that $^{137}Cs$ levels in woodcock (*Scolopax rusticola*) muscle have been low when compared with ruminants. In willow grouse (*Lagopus mutus*) radiocaesium levels were much lower than in sheep grazing in the same area. Values for $T_{ag}$ of 0.01-0.02 $m^2$ $kg^{-1}$ can be calculated from the work of Pedersen (1989). In Norway there has been no noticeable decrease in $^{137}Cs$ levels in grouse. In the Kyshtym area a $T_{ag}$ value estimated for black grouse was comparatively low at 0.0012 in 1967-68 (Trabalka, priv. comm.). Radiocaesium levels in red grouse (*Lapogus lapogus*) and black grouse (*Tetrao tetrix*) are higher in Scotland than those recorded in Nordic countries, probably because these species of grouse eat large quantities of heather, whereas heather does not form an important part of the diet of willow grouse in the hunting season. Hence although the low levels in grouse meat from Nordic countries and the small quantities consumed implies that game birds are not contributing significantly to the doses from semi-natural ecosystems, the discrepancy between Scotland and Nordic Countries clearly shows that such conclusions would not necessarily be applicable for other ecosystems. It also demonstrates that species selected in the diet is important in determining radiocaesium levels in birds as well as other animals inhabiting semi-natural ecosystems.

*(vi) Reindeer*

The transfer of $^{137}Cs$ through the food chain from lichens to reindeer to humans was studied in great detail during the period of the above-ground nuclear weapons tests. Effective half-lives of between 5 and 10 years were estimated from measurement of both reindeer meat and from whole body monitoring of humans with high intakes of reindeer meat. However whole body counting of Finnish Lapps showed a decrease in the body burden of $^{137}Cs$ from 4,100 Bq in 1976 to 2,850 in 1986 (a decrease of only 30%) (Rahola & Suomela 1987). In Norwegian reindeer herders (Lapps) the decrease in whole body $^{137}Cs$ content from 1975-1984 was also less (42%) than expected from data on half-lives during the 1960s (Westerlund 1985). Since lichens are directly contaminated, estimations of the bio-availability of radiocaesium in the lichen is more important than in green vegetation where contamination after the initial deposition occurs mainly by root uptake.

Values of $T_{ag}$ estimated during the first winter after the deposition of Chernobyl fallout were 0.6-1.1 $m^2\ kg^{-1}$. When radiocaesium levels in reindeer were at their lowest (late July and early August) the estimated $T_{ag}$ in Sweden was about 0.025 (Ahman & Ahman 1990). A suitable $T_{ag}$ for reindeer meat is difficult to define because lichens which are initially highly contaminated dilute the radiocaesium activity concentration by new growth. Nevertheless, the total deposition (Bq $m^{-2}$) remains the same and therefore it is particularly important to incorporate an estimate of $T_{ef}$. Due to grazing of contaminated biomass and dilution by new growth the radiocaesium content in lichen declined with a $T_{ef}$ of 3-4 years in Norway (Staaland et al. 1990) and around 7-10 years for both pre- and post-Chernobyl $^{137}Cs$ in reindeer lichen in Sweden (Eriksson et al. 1991). Other post-Chernobyl studies in Sweden show a $T_{ef}$ for $^{137}Cs$ in reindeer meat in the first few years after the accident (1986-1991) of between 3 and 4 years during the winter period (Åhman 1992). Since lichen is the main feed source for reindeer (>75%) (Rissanen & Rahola 1989), use of the ratio between radiocaesium levels in lichen and reindeer meat during the winter season are probably a more appropriate and accurate measure than aggregated transfer coefficients. Data collected during field studies in Norway in 1987-1989 in November after the Chernobyl accident give a range of 1.0-1.2 $kg^{-1}$ for meat per $kg^{-1}$ dw in pooled samples of lichens collected from different locations within the grazing range of the studied herd. Therefore a meat to lichen ratio of 1.0 Bq $kg^{-1}$ of meat per Bq $kg^{-1}$ dw of lichen can be recommended.

The $^{137}Cs$ activity concentrations in reindeer during summer and early autumn have not been considered to be as important as those during the winter because they are generally about 10 to 20% of the latter. After the Chernobyl accident a recommendation to change the slaughter times in some Nordic countries was made in order to take advantage of the low summer values, thus decreasing the dose commitment to man. Reindeer generally graze the same pastures as sheep during summer, and the $T_{ef}$ for lambs in these ecosystems has been much longer than for the lichen-reindeer foodchain. Differences between winter and summer activity concentrations in reindeer meat may therefore become less pronounced as time passes in areas where substantial root uptake is taking place. Finnish studies from 1983 of the $^{137}Cs$ levels in various fodder plants for reindeer found that lichens still had higher $^{137}Cs$ activity concentrations than most other fodder plants, but the differences were small (Rissanen et al. 1987). During the period 1987-1992 in Norway both winter and summer radiocaesium activity concentrations in reindeer meat have decayed with a $T_{ef}$ of 3.5 y. Radiocaesium activity concentrations in green vegetation six years after the Chernobyl accident were still below 10% of those in lichens. In this situation a lichen intake during the summer season of only 10-15% dw will dominate radiocaesium levels in meat to the extent that the $T_{ef}$ will still approach that of lichens.

Radiocaesium levels in a substantial fraction of reindeer meat in Norway and Sweden have been above the intervention levels (6 and 1.5 kBq $kg^{-1}$ fw respectively) at the normal time of slaughter since the Chernobyl accident. Considerable efforts have been made to reduce the radiocaesium levels in reindeer. As for other game products, consumption of reindeer meat is not evenly distributed in the population. Average values for meat consumption of 0.3-0.5 kg $y^{-1}$ in Scandinavian countries will contribute a maximum of 3 kBq $y^{-1}$ of $^{137}Cs$ when eating meat with an activity concentration at the intervention level. In the critical group of hunters and reindeer herders much higher intakes have been noted; a maximum annual dose of 13 mSv was calculated based on whole body counting in Norway (Strand et al. 1990).

2) DOMESTICATED RUMINANTS

*(i) Sheep*

In many countries sheep meat is the most important food product obtained from semi-natural ecosystems. Before the Chernobyl accident little emphasis was placed on measurement of radiocaesium levels in sheep from semi-natural compared with agricultural ecosystems, although there was some indication, based on limited data, that radiocaesium levels in upland sheep were elevated (ARCRL 1966). After the Chernobyl accident, high levels were observed in sheep from semi-natural ecosystems in many affected countries. The values observed exceeded intervention limits in the United Kingdom, Ireland, Norway, Sweden and Austria. For sheep grazing in semi-natural ecosystems the major factors responsible for the persistently high radiocaesium activities in meat are the comparatively high root uptake of grazed vegetation and, in certain areas, the selective intake by sheep of highly contaminated fungi and ericaceous species. Post-Chernobyl measurements of the transfer of radiocaesium to lamb has shown that the previously assumed transfer coefficient ($F_f$) for lamb was too low.

Major efforts have been undertaken to study the transfer of radiocaesium to sheep in semi-natural ecosystems because of the importance of sheep production in affected countries. It has become clear that there are considerable difficulties in determining accurate values for various transfer parameters for sheep grazing in semi-natural ecosystems. It is particularly difficult to estimate radiocaesium intake in freely grazing sheep. Therefore more effort has been devoted to trying to calculate $T_{ag}$'s, particularly in the Nordic countries, where a collaborative study is in progress. Estimated pre- and post-Chernobyl values for $T_{ag}$'s for sheep meat are given in Table VII.

Table VII. Aggregated transfer coefficients for sheep meat.

| Radiocaesium Source | Year | $T_{ag}$ ($m^2$ $kg^{-1}$ fw) | | Comments, Site |
|---|---|---|---|---|
| | | *Large area | **Pasture | |
| Nuclear weapons test | 1965 | 0.020-0.050 | | Norway |
| | 1991 | 0.011 | | Ewes: Cumbria, UK |
| Chernobyl | 1988-90 | | 0.020-0.030 | Coastal, Norway |
| | 1988 | | 0.05 | mountain pasture, Central Norway |
| | 1988 | | 0.043 | lambs: Cumbria, UK |
| | 1988 | | 0.025 | ewes: Cumbria, UK |
| | 1989 | | 0.030-0.050$^{s}$ | ewes: Scotland |
| | 1990 | | 0.01-0.074 | Nordic study (range) |
| | 1991 | 0.036 | | ewes: Cumbria, UK |
| | | 0.09$^{w}$ | | ewes: heather moor, Ireland |
| | | 0.15$^{s}$ (0.03-0.29) | | ewes: heather moor, Ireland |

* Large area: Deposition calculated from a large area (several $km^2$)
** Pasture: Deposition measured in the specific pasture in which the sheep were grazing
w Winter
s Summer

All $T_{ag}$ values fall within the range 0.01 to 0.29. The highest values were obtained in Ireland for ewes grazing on a montane blanket bog covered with shrub vegetation dominated by *C. vulgaris* in Ireland (calculated from data given in McGee et al. in press). $T_{ag}$ values of sheep in agricultural ecosystems are much lower. For instance ten-fold lower values have been found for agricultural ecosystems compared with semi-natural ecosystems in the same district in Norway.

In some intensively studied areas in Norway where estimates of $T_{ag}$ are available for consecutive years after the Chernobyl accident, the maximum variation has been 50% prior to the autumn period (Strand & Hove priv. comm.). In some of the contaminated pastures in Norway fungi occasionally produce a substantial crop of fruiting bodies. When this happened in 1988, consumption of highly contaminated mushrooms by sheep led to a 3-4 fold increase in $T_{ag}$ values (Hove et al. 1990).

Although radiocaesium levels have declined in several areas, sheep from semi-natural ecosystems in the Nordic countries, Ireland and the United Kingdom still exceed intervention limits 6 years after the Chernobyl accident. In some closely monitored flocks in Ireland negligible declines in $^{137}Cs$

activity concentrations have taken place in the last few years (McGee et al. in press).

This parallels the long $T_{ef}$ values which have been calculated for above ground nuclear weapons test $^{137}Cs$ in Norway (Hove & Strand 1990). Similarly observations in the UK show that four years after deposition Chernobyl radiocaesium is being taken up by plants to a similar extent as aged deposits of $^{137}Cs$, suggesting that future reductions in radiocaesium levels in vegetation, and therefore sheep, will be slow and predominantly due to physical decay and migration of radiocaesium down the soil profile (Beresford et al. 1992). $T_{ag}$ values for ewe muscle for $^{137}Cs$ from "aged deposits" in this area in December 1991 of 0.011 were lower than those for Chernobyl $^{137}Cs$ of 0.036 (Beresford priv. comm.). Because conventional transfer coefficients (ie $F_m$ and $F_f$) incorporate total daily radionuclide intake, they give higher values for radiocaesium transfer to animal products such as milk and meat from smaller compared with larger ruminants, because food intake and also, therefore, radionuclide intake is proportional to metabolic body size. $T_{ag}$'s remove this effect, nevertheless higher transfer of many radionuclides, including radiocaesium, occurs to sheep and goat milk compared with cow milk (Coughtrey 1990). Consequently comparatively high levels of radiocaesium (and radioiodine) in sheep (and goat) milk will be found in contaminated areas and this is particularly important in those countries where milk is obtained from sheep grazing semi-natural ecosystems.

For many countries it is not possible to distinguish consumption of sheep meat from semi-natural ecosystems from that produced in agricultural ecosystems. However, in Norway the great majority of sheep are slaughtered when returning from summer grazing in semi-natural ecosystems.

*(ii) Goats*

Dairy goats graze in semi-natural ecosystems in many parts of the world, and as for sheep, transfers of radiocaesium to goat milk is high compared with cows milk (Hansen & Hove 1991). Long effective half-lives of approximately 20 y have also been observed in semi-natural ecosystems for $^{137}Cs$ in goat milk products (Hove & Strand 1990). $T_{ag}$'s for goat milk of 0.002-0.004 were calculated from Norwegian data for the period when ingestion of fungi was negligible, $T_{ag}$ values increased 2-4 fold in years with a comparatively high fungal abundance (Hove & Strand 1990; Hove et al. 1990; Garmo et al. 1991). It is common practice in Norway to feed 10-25% of the dry matter intake as concentrates, depending on pasture conditions. This feed usually has very low radiocaesium activity concentrations, which should be taken into account as a source of variation when estimating $T_{ag}$'s for goat milk products.

In Europe, goat milk is generally used for cheese production and by certain groups such as those with allergies to cow milk. For the average population intakes are small. However, in Norway the more commonly consumed whey cheeses (Norwegian brown cheese) can have comparatively high radiocaesium levels because manufacturing involves evaporation of the whey to dryness which increases the radiocaesium activity concentration by a factor of about 10.

*(iii) Cattle*

In large ruminants conventionally used transfer coefficients for radiocaesium to meat and milk are about an order of magnitude lower than in sheep. Since food intake per kg of body weight is lower for large animals than for small animals, deposition levels must therefore be somewhat higher for cattle than for sheep or goats before intervention limits are exceeded.

Comparatively high radiocaesium contamination of bovine dairy products from areas where the organic matter content in soil was high, and especially from cattle grazing in forests, was noted for nuclear weapons fallout in Sweden (Mattsson & Moberg 1991).

There are few data for cattle grazing in semi-natural ecosystems. In Valdres, Norway where the average deposition of Chernobyl radiocaesium was 150 kBq $m^{-2}$ $T_{ag}$ values calculated for the summer of 1986 were 0.002-0.003 $m^2$ $L^{-1}$. From 1987 onwards data are difficult to interpret because of the variable use of caesium binders. In an experimental herd, the average $T_{ag}$ values for 5-10 cattle were 0.001-0.002 $m^2$ $kg^{-1}$ during the years 1989-1991. Milk activity concentrations were 10 fold lower than those in lamb meat from the same area (ie 600 Bq $L^{-1}$ compared with 6 kBq $kg^{-1}$, Strand & Hove priv. comm.). Based on live monitoring, $T_{ag}$ values for beef from these cattle averaged 0.006 $m^2$ $kg^{-1}$. However it should be pointed out that some of the food consumed by the cattle comes from uncontaminated concentrates, thus underestimating the $T_{ag}$ value.

In Austria $T_{ag}$'s ($m^2$ $L^{-1}$ cow milk) in the range 0.00004 to 0.00049 were found in agricultural ecosystems in 1988 compared with 0.0018 to 0.018 in a semi-natural alpine pasture (Mück et al. 1990). For an alpine pasture near Salzburg an unusually high $T_{ag}$ value for 1988 of 0.0176 was observed; the following years, 1990 and 1991, the $T_{ag}$ values for this pasture were 0.01 $m^2$ $L^{-1}$ (Gerzabek priv. comm.).

## 4. SUMMARY OF AGGREGATED TRANSFER COEFFICIENTS ($T_{ag}$'s)

A summary of the $T_{ag}$ values presented in the preceeding sections is given in Table VIII. They are presented as a range between the maximum and minimum observed values, together with a "best estimate" which has been reached using informed judgement based on critically assessing the range of data available.

Table VIII. Summary of the range of aggregated transfer coefficients ($T_{ag}$) for radiocaesium transfer to foodstuffs from semi-natural ecosystems.

| Food product | $T_{ag}$ ($m^2\ kg^{-1}$) | |
|---|---|---|
| | Range | Best estimate |
| Fungi (dw) | | |
| *Cantharellus cibarius* | 0.014 - 0.54 | 0.3 |
| *Cantharellus tubaeformis* | 0.14 - 1.5 | 0.9 |
| *Lactarius trivialis* | 0.02 - 4.7 | 2.2 |
| *Leccinum versipelle* | 0.005 - 0.56 | 0.1 |
| *Leccinum scabrum* | 0.023 | 0.02 |
| *Boletus edulis* | 0.003 - 0.18 | 0.06 |
| *Rozites caperata* | 1.5 - 3.1 | 2.0 |
| *Xerocomus badius* | 0.19 - 7.1 | 2.0 |
| *Xerocomus chrysenteron* | 0.03 - 5.2 | 1.0 |
| Berries (dw) | | |
| Cloudberry | 0.002 - 0.23 | 0.1 |
| Lingonberry | 0.032 - 0.04 | 0.04 |
| Bilberry | 0.04 - 0.041 | 0.04 |
| Honey (fw) | 0.00004 - 0.023 | 0.003 |
| Game animals (fw) | | |
| Moose meat | 0.001 - 0.032 | 0.02 |
| Roedeer meat | 0.0009 - 0.2 | 0.05 |
| Red deer meat | 0.02 - 0.04 | 0.03 |
| Arctic hare | 0.0009 - 0.13 | 0.03 |
| Brown hare | 0.00018- 0.053 | 0.004 |
| Domesticated animals (fw) | | |
| Sheep meat | 0.01 - 0.074 | 0.04 |
| Goat milk | 0.002 - 0.015 | 0.004 |
| Cow milk | 0.001 - 0.018 | 0.002 |
| Beef | 0.006 | 0.006 |
| Reindeer (august) | 0.025 - 0.1 | *0.1 |
| (winter) | 0.6 - 1.1 | 0.8 |

* See discussion on reindeer in Section 3.4

The range of variation in $T_{ag}$ values observed for species other than fungi in Table VIII is about one order of magnitude. For fungi the spread of values is much greater at up to three orders of magnitude within the same species of fungi.

# 5. APPLICATION OF AGGREGATED TRANSFER COEFFICIENTS

## 5.1 Potential usage of $T_{ag}$'s

The main use anticipated for the $T_{ag}$ data is in the prediction of the consequences of accidental releases of radionuclides to the environment. In an emergency situation, planners and decision makers need to have an appreciation of which groups in the population are likely to be the highest exposed so that resources allocated to countermeasures can be appropriately prioritized. The $T_{ag}$ data, together with information on critical group dietary intake rates and on the average ground deposit of the radionuclide, can be used to provide rapid, order of magnitude, estimates of the highest doses likely to result from the intake of foodstuffs derived from semi-natural ecosystems.

The $T_{ag}$ data may also be used in impact assessment studies, to evaluate the collective doses associated with contamination of semi-natural ecosystems. In this case average consumption rates for the exposed population should be used in the dose calculation.

In both cases it may be necessary to extend the prediction of dose into the future and therefore, information on the time dependence of the $T_{ag}$'s needs to be provided.

## 5.2 Time dependence of $T_{ag}$'s

The data on the time dependence of $T_{ag}$'s is limited and in the absence of data on changes with time it will be necessary to adopt the conservative or default assumption that the associated ecological half-life ($T_{ec}$) is infinite and that the effective half-life ($T_{ef}$) is governed solely by the physical half-life ($T_{phys}$) of the radionuclide. Therefore, in the case of $^{137}Cs$, the $T_{ef}$ would be 30 years.

The $T_{ec}$ of radiocaesium in food products from many semi-natural ecosystems is likely to be site-specific, and to depend to a large extent on the rate of migration of radiocaesium out of the rooting zone or into highly mineralised horizons of the soil. Thus the ability of plant roots or fungal hyphae to absorb radiocaesium will be determined by soil characteristics and the rooting depth of each plant. It is therefore difficult to generalize about $T_{ec}$, or to apply data derived in one particular area to another. An exception is the reindeer, where the consumption of aerially contaminated lichen is important.

Despite these difficulties some general trends are evident. Fungi, which have hyphae in the litter zone would be expected to be more contaminated in the first few years after deposition, whilst those with hyphae in the organic horizons will gradually become more contaminated as the radiocaesium becomes incorporated into these layers of the soil (Wirth et al. 1992). There is little evidence for a decline in radiocaesium in ruminants such as moose, but for semi-domesticated sheep and roe deer gradual declines have been found in some areas (e.g. UK - Howard & Beresford in press; Germany - Lindner et al. in press), but not others (e.g. Ireland - McGee et al in press; Sweden - Johanson priv. comm.). It is evident that ecological half lives for many species will be site-dependent and generalisations are difficult to make.

## 5.3 Critical groups

A critical group is a group of individuals in the population, who, due to their location, consumption or other habits may be expected to receive the highest radiation doses. They are usually identified by some type of local survey of habits and consumption rates. Consumption rates observed for some of the critical groups associated with semi-natural ecosystems have been reported in the preceding sections.

It is often possible to distingush two types of critical group in the case of food products coming from these ecosystems. The most extreme critical groups are those in which the food product is collected, or hunted, and then consumed by the individuals themselves. For example, the hunter who freezes and consumes the meat from highly contaminated game animals at a high rate over the whole year. The more moderate type of critical group consists of high rate consumers who obtain their food from a distributor, for example a butcher, thus allowing for the possibility of dilution with less contaminated equivalent food from various sources.

The importance of such critical groups in semi-natural ecosystems has been clearly demonstrated since the Chernobyl accident. For instance, in Sweden the highest radiation doses are received by people living in high deposition areas who consume large amounts of meat from game animals, lake fish and reindeer (Mattsson & Moberg 1991). The incidence of individuals belonging to more than one critical group is likely to be higher in these circumstances than for critical groups receiving their food only from agriculturally improved areas.

In performing critical group dose assessments, care has to be taken in the selection of appropriate consumption rates and $T_{ag}$'s. It is conceivable, but unlikely, that an individual could consume a food product at the highest rate observed and at the highest $T_{ag}$ value observed. This can be allowed for in a probabilistic dose assessment provided sufficient data exist, but in a deterministic assessment a suitably conservative but realistic choice of values for consumption rates and $T_{ag}$ values has to be made. Judgement has to be exercised in the choice of the values from Table VIII and, for example, it will usually be overly pessimistic to use the highest values reported.

## 5.4 An example of the application of $T_{ag}$'s for dose assessment

Although natural food products contribute to the total food consumption for the general public to only a small extent in most countries, they may contribute a considerable proportion of the long-term ingestion dose due to $^{137}$Cs. In Table IX the potential lifetime ingestion dose (committed effective dose) to adults is assessed assuming a deposition of 10,000 Bq m$^{2}$ of $^{137}$Cs and a normalized consumption rate of 1 kg y$^{-1}$ (fw) for each foodstuff considered. In this calculation only the contribution of long-term root uptake is included (the contamination of food products due to direct deposition of radionuclides is not considered). The $T_{ag}$'s as given in Table XIII are applied to calculate contamination of natural food products. A best-estimate of the $T_{ag}$ is used; in addition values are calculated for the extremes of the range.

Table IX. Estimated lifetime doses due to the consumption of natural food products, assuming an initial soil deposition of 10,000 Bq $m^{-2}$ and a consumption rate of 1 kg $y^{-1}$.

| Natural food product | $T_{ag}$ ($m^2$ $kg^{-1}$) best estimate (range) | $T_{ef}$ (y) | Committed effective dose# mSv |
|---|---|---|---|
| Fungi | 1.0 (0.003-7.1) | 10<br>20<br>30 | 0.16 (0.00047-1.1)<br>0.27 (0.0008-1.9)<br>0.33 (0.001-2.3) |
| Berries | 0.06 (0.002-0.23) | 10<br>20<br>30 | 0.018 (0.00059-0.068)<br>0.030 (0.001-0.11)<br>0.037 (0.012-0.14) |
| Roe deer | 0.05 (0.0009-0.2) | 10<br>20<br>30 | 0.098 (0.0018-0.39)<br>0.17 (0.003-0.67)<br>0.21 (0.0037-0.83) |
| Red deer | 0.03 (0.02-0.04) | 10<br>20<br>30 | 0.059 (0.039-0.078)<br>0.10 (0.066-0.13)<br>0.12 (0.083-0.17) |
| Sheep meat | 0.04 (0.01-0.074) | 10<br>20<br>30 | 0.078 (0.020-0.15<br>0.13 (0.033-0.25)<br>0.17 (0.041-0.31) |
| Reindeer | 0.8 (0.6 -1.1)<br>(winter)<br>0.1<br>(summer) | 4<br>4<br>10 | 0.70 (0.53 -0.97)<br>0.09<br>0.20 |

# Ingestion dose factor: 1.4 x $10^{-8}$ Sv $Bq^{-1}$

The doses arising from the ingestion of natural food products should be compared with the dose due to the ingestion of food product contaminated by root uptake and resuspension which are produced on intensively managed, well fertilized farmland. Using the radioecological model ECOSYS-87 (Müller and Pröhl in press) for this assessment, the consumption of an average German food basket after a single deposition of 10,000 Bq $m^{-2}$ of $^{137}Cs$ gives rise to a committed dose for an adult of approximately 0.035 mSv. Comparing this value with the doses calculated in Table IX, it is obvious that natural food products may contribute significantly to, or even dominate, the long-term ingestion dose due to $^{137}Cs$. Although a general assessment is difficult, because the consumption rates for natural food products are very site-specific, it can be concluded that in certain population groups which consume enhanced amounts of mushrooms, berries, game, sheep, etc. the long-term ingestion dose due to $^{137}Cs$ is caused mainly by these foodstuffs.

Similar conclusions are reported by Bergman et al. (1991), Aarkrog (1992) and Mattsson & Moberg (1991). For example, Bergman et al. compared the doses after the Chernobyl accident in the Swedish population due to consumption of agricultural products (meat, milk, and milk products) with the doses arising from the consumption of berries and moose meat. They concluded that the doses accumulated over 20 years following the accident via agricultural products and natural products are about the same. However, in Bergman's assessment the contamination of vegetation due to direct deposition in the first year was included. Without this contribution, and considering only the contribution of root uptake, Bergman's results indicate that the natural food products produce the greater contribution to the ingestion dose. An alternative approach adapted by Aarkrog (1992) has been to compare the $^{137}Cs$ intake due to consumption of game, freshwater fish and mushrooms from semi-natural ecosystems with that of food from agricultural ecosystems using a transfer-factor defined as Bq $^{137}Cs$ $kg^{-1}$ y divided by Bq $^{137}Cs$ $m^{-2}$. Preliminary conclusions for a number of European countries indicate that semi-natural ecosystems may contribute about half the dietary intake in Sweden, 10% in Denmark and less than 1% in Ireland and the UK. Aarkrog also noted that the relative contribution from semi-natural ecosystems was anticipated to be higher for Chernobyl radiocaesium compared with that in nuclear weapons fallout due to the low contribution of Chernobyl fallout from agricultural ecosystems because of the season in which the deposition occurred. The spring-time deposition of Chernobyl fallout would have had a less marked effect on food products from semi-natural ecosystems because of the long effective half-lives.

## 6. DISCUSSION

Since the Chernobyl accident there has been a considerable improvement in our understanding of the behaviour of radiocaesium, and of the rates at which it is transferred to food products in semi-natural ecosystems and hence to humans. Although there was some appreciation during the above-ground weapons testing era of the high transfer to reindeer in semi-natural ecosystems there was less attention given to the transfer of radiocaesium to other food products, particularily to fungi and to animals other than reindeer. Neither was there much data available on the long effective half-lives which occur for $^{137}Cs$ in many of the food products from these ecosystems. Hence these factors were often not considered in assessing radiation doses to humans.

The long effective half-life for radiocaesium in many semi-natural ecosystems is of central importance in modelling and dose assessment. For a significant proportion of the various natural food products there seems to have been little, if any, decrease with time, particularly after the first year following the deposition of Chernobyl fallout. Further comparisons of the behaviour of "aged deposits" of $^{137}Cs$ with that from Chernobyl would be valuable for such assessments.

Much of the variation in $T_{ag}$ values within species is due to site-specific variables, particularly in the plant availability of radiocaesium in different soils. Generally it seems reasonable to expect that the highest $T_{ag}$ values would be from areas with soils of low nutrient status, high organic matter content (e.g. > 40%) and low clay mineral content. For animals dietary selection is also an important variable which must be taken into account.

Further consideration needs to be given to the validity and usefulness of aggregated transfer coefficients. One particular advantage of $T_{ag}$'s is that they integrate all the transfers in ecosystems between deposition and the natural food product itself. They can therefore help to rapidly identify "vulnerable" pathways which may need some form of intervention. Although they are useful for such rapid radiological assessment they do not give any information about mechanisms and are not a substitute for detailed studies on the various food webs. They are consequently of limited usefulness in helping to devise countermeasures. Other disadvantages of using $T_{ag}$'s are that they are ecosystem-specific and can vary to a significant extent with season, particularly for animal derived food products.

The use of $T_{ag}$ values will obviously depend on the availability of data on deposition. Caution must be exercised when applying $T_{ag}$ values since ground deposition in semi-natural ecosystems can be very heterogenous, particularly in forests where it is affected by stem flow and variations in the canopy. In these circumstances particular care must be taken to sample adequately when measuring ground deposition.

Currently it is difficult to assess the general applicability of the $T_{ag}$'s presented in this report. Particular caution should be used when applying the $T_{ag}$'s presented here in assessments for regions other than Europe. To make more precise assessments of the radiation dose arising from semi-natural ecosystems using $T_{ag}$'s we need:

*(i)* More information on relevant $T_{ag}$'s for the most important natural food products from other areas of the world, with different semi-natural ecosystems to those in the northern European countries from which most of the data in this report is derived.
*(ii)* More information on ecological half-lives for radionuclides in natural food products from semi-natural ecosystems.
*(iii)* Better, more precise evaluation of the consumption rates of natural food products of both critical groups and the general population, particularly for fungi.

## 7. SUMMARY

In semi-natural ecosystems the modelling of radionuclide transfer to man presents difficulties because of the complexity of the ecosystems. The difficulties are further increased in the case of transfer to animals because the animals may cover large areas over which the deposition may be very heterogenous.

The $T_{ag}$'s for animal food products effectively integrate transfer over the areas grazed by the animals. The use of empirical aggregated transfer coefficients ($T_{ag}$'s), which directly relate observed activity concentrations of radionuclides in natural food products to the measured average ground deposit, gives the possibility of making the order of magnitude dose estimates needed in emergency situations.

$T_{ag}$ values have been derived from reported measurements in several countries although the majority have been obtained from Northern Europe. They are given for various fungi, berries, honey, game animals and domesticated animals in semi-natural ecosystems.

Little precise information exists on the time dependence of the $T_{ag}$'s but there is good evidence that their decline with time will be slow in many semi-natural ecosystems and may be determined largely by the physical half-life in the case of radiocaesium. It is recommended that in the absence of specific information, the effective half-life is taken to be equal to the physical half-life.

Generic calculations using $T_{ag}$'s of the lifetime dose due to the consumption of modest amounts of individual natural food products from semi-natural ecosystems show the dose to be comparable to that from the consumption of a normal diet of food derived from agriculturally improved ecosystems contaminated to a similar level. If the higher reported values for $T_{ag}$'s and of critical group consumption rates are used, the dose commitments from semi-natural food products are substantially higher than those due to normal dietary intakes of agriculturally produced foodstuffs.

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