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Investigation of selected natural and anthropogenic radionuclides in reindeer *(Rangifer tarandus tarandus)* and lynx *(Lynx lynx)*

Doctoral thesis
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Fannrem, May 2005

List of papers

The thesis is based on the following six papers, which will be referred to by their Roman numerals:

I. Skuterud L, Pedersen Ø, Staaland H, Røed KH, Salbu B, Liken A, Hove K (2004). Absorption, retention and tissue distribution of radiocaesium in reindeer: effects of diet and radiocaesium source. Radiation and Environmental Biophysics 43: 293-301. With errata on p. 313 of the same issue

II. Skuterud L, Gaare E, Steinnes E, Hove K (2005). Physiological parameters that affect the transfer of radiocaesium to ruminants. Radiation and Environmental Biophysics 44: 11-15

III. Skuterud L, Gjøstein H, Holand Ø, Salbu B, Steinnes E, Hove K. Transfer of ^{85}Sr and ^{134}Cs from diet to reindeer foetuses and milk. Radiation and Environmental Biophysics, in press

IV. Skuterud L, Gaare E, Eikelmann IM, Hove K, Steinnes E (2005). Chernobyl radioactivity persists in reindeer. Journal of Environmental Radioactivity 83: 231-252

V. Skuterud L, Gwynn JP, Gaare E, Steinnes E, Hove K. ^{90}Sr, ^{210}Po and ^{210}Pb in lichen and reindeer in Norway. Journal of Environmental Radioactivity, in press

VI. Skuterud L, Gaare E, Kvam T, Hove K, Steinnes E (2005). Concentrations of ^{137}Cs in lynx (*Lynx lynx*) in relation to prey choice. Journal of Environmental Radioactivity 80: 125-138

Summary

The present thesis investigates a range of dietary, physiological and environmental factors influencing transfer and long-term behaviour of radionuclides in reindeer and lynx. Feeding experiments were designed to provide details on factors related to absorption, retention and secretion of radionuclides in reindeer, while concentrations and time trends of radionuclides in the environment were studied in two reindeer herding districts during the period 2000 – 2003. Data on ^{137}Cs in reindeer from 1986 onwards and samples of lynx were obtained from national archives.

In the first feeding experiment, reindeer calves were offered either combined diets of fallout contaminated lichens and pelleted concentrates applicable for clean feeding, or a pure lichen diet. Constant daily radiocaesium (^{134}Cs and ^{137}Cs) administration rates were used throughout the experiments. Due to lower dietary potassium and crude fibre intake, reindeer on the pure lichen diet had significantly lower excretion of radiocaesium and a 40 % longer biological half-time for radiocaesium in red blood cells than reindeer fed a diet with a higher mineral element content. The study showed that the bioavailability of Chernobyl radiocaesium in lichen was 35 % of that of ^{134}Cs as chloride in aqueous solution. The results were in agreement with previous studies demonstrating the effect of seasonal variations in potassium intake on radiocaesium retention in reindeer. A literature review of the many factors influencing transfer of radiocaesium to ruminants suggested that additional seasonally affected factors need to be taken into account in studies of reindeer, particularly the effects of digestibility and metabolic rates on absorption and endogenous faecal excretion of radiocaesium.

In the second feeding experiment, four pregnant reindeer on a concentrate diet were given daily constant quantities of ^{134}Cs and ^{85}Sr as chlorides in aqueous solution during the last part of gestation. The experiment showed that similar fractions of the administered activities of ^{134}Cs and ^{85}Sr were transferred to the foetus, and 1.4 – 2.5 % of the total administered activities were deposited in the calves at birth. The distribution of the nuclides in different tissues of newborn calves was comparable to that reported in older calves. The transfer coefficients (F_m) for ^{134}Cs and ^{85}Sr from diet to reindeer milk were as expected from extrapolated F_m values observed in other ruminants, with F_m for ^{134}Cs 8.5 times higher that of ^{85}Sr. The influence of the mineral element intake, particularly K and Ca, on absorption of Cs and Sr in reindeer, suggests that transfer of the nuclides to foetus and milk of free-ranging reindeer may be considerable higher than observed in this experiment. Furthermore, secretion of Sr accumulated prior to the lactation period will probably cause higher Sr concentration in milk of free-ranging reindeer than indicated by the F_m estimated in this experiment. No differences in half-times for ^{134}Cs and ^{85}Sr secretion in milk were observed, with both nuclides secreted with short- and long-term half-times of 1 – 2 and 12 – 19 days, respectively.

The study of ^{137}Cs, ^{90}Sr, ^{210}Po and ^{210}Pb in the environment included sampling of soil, vegetation and reindeer tissues in two climatically different reindeer herding districts, Østre Namdal and Vågå. These districts were among the areas in Norway most affected by fallout from the Chernobyl accident. The study showed that ^{137}Cs concentrations in reindeer from both areas declined by effective ecological half-times (T_{ecol}) of 3 – 5 years in autumn and winter up to the mid and late 1990s, when the rates of decline decreased possibly due to a reduced role of lichens as sources of absorbed ^{137}Cs in reindeer. Future time-trends may well be governed simply by physical decay,

although incidences of high abundances of fungi may potentially cause elevated ^{137}Cs concentrations in reindeer in autumn for many years to come.

Higher ^{137}Cs concentrations in vascular plants, lichens and reindeer in Østre Namdal compared to Vågå (relative to the ^{137}Cs deposition density), suggested that climatic influences on soil properties that influence the availability of ^{137}Cs for plant uptake and on lichen growth and abundance may have a larger impact on long-term transfer of radiocaesium in the soil-plant/lichen-reindeer food chain than has been previously observed. Furthermore, the results of this thesis suggest that various proportions of lichens and vascular plants in the diet may cause appreciable differences in transfer of radiocaesium to reindeer across Norway due to differences in radiocaesium, potassium and crude fibre intake.

The samples of bone and antlers of reindeer of different ages showed that ^{90}Sr concentrations in bone of older females were 40 % higher than in calves due to higher dietary ^{90}Sr intake during their periods of growth (< 2 year) and continuous ^{90}Sr intake thereafter. Combined with previously reported data, a T_{ecol} of 9.03 ± 0.06 years was estimated for ^{90}Sr concentrations in antlers of reindeer calves in Vågå for the period 1988 – 2002. The study supports the use of antlers as monitors of ^{90}Sr concentrations instead of bone, since concentrations in bone appear to be significantly influenced by age and constant bone renewal. Concentrations of ^{90}Sr were 50 – 80 % higher in reindeer from Vågå compared to those from Østre Namdal.

Age did not appear to have a major effect on muscle and liver tissue ^{210}Po and ^{210}Pb concentrations in reindeer from Østre Namdal and Vågå. Concentrations of ^{210}Po and ^{210}Pb were similar in the two districts and to previously reported values from other Nordic areas. Thus, climatic differences did not have noticeable effects on ^{210}Po and ^{210}Pb concentrations in reindeer in this study.

The study of ^{137}Cs in muscle samples of 747 lynxes killed in Norway from the 1986 Chernobyl accident up to the year 2001 showed that a model with ^{137}Cs deposition density, the year lynxes were killed, age, and extent of reindeer grazing area in the lynxes' home ranges could account for 50 % of the variability in observed ^{137}Cs concentrations. The analyses were equivocal regarding the lynxes' specialization in prey species. Further work on the possible use of radiocaesium as a tracer of reindeer predation by lynxes require experimental data on Cs retention in lynx and better estimates of deposition density in the lynxes' home ranges.

The calculated absorbed doses to reindeer from both anthropogenic and natural nuclides suggested that some of the most exposed individuals received dose rates approaching 1 mGy d^{-1} after the Chernobyl fallout. No effects on morbidity, mortality or reproductive capacity of reindeer would be expected from these doses. Only the lynxes with the highest radiocaesium concentrations received doses comparable to those received by reindeer.

The assessment of doses to humans from the studied radionuclides showed that ^{137}Cs continues to be the most important contributor to ingestion doses by South Saamis. The time trend in ^{137}Cs concentrations in the studied reindeer herding districts suggests that ingestion doses by persons with average Saami consumption rates of reindeer meat will continue to exceed the 1990 recommendation by the International Commission on Radiological Protection for many years to come, if countermeasures are not applied.

Contents

1 Introduction

1.1 Why study radionuclides in reindeer and lynx?

There are several reasons why studies of radionuclides in reindeer and lynx are of interest:

Firstly, environmental adaptations of reindeer, including the selection of lichens as important dietary components and the ability to regulate their metabolism, results in higher tissue concentrations of the anthropogenic radionuclides caesium-137 (^{137}Cs) and strontium-90 (^{90}Sr) and the natural radionuclides lead-210 (^{210}Pb) and polonium-210 (^{210}Po) in reindeer than in many other animals (see, e.g. Paakkola and Miettinen, 1963; Holtzman, 1966; Hill, 1967; Kauranen and Miettinen, 1967; Persson, 1971). Information on the transfer of these nuclides to reindeer, developing foetuses and milk, and data on actual activity concentrations in free-ranging reindeer is required for accurate assessments of potential biological consequences resulting from body burdens of these radionuclides.

Secondly, the 1986 Chernobyl accident had significant consequences for reindeer herding in Scandinavia. In Norway, radiocaesium activity concentrations in reindeer meat in the most affected areas reached 150 000 Bq kg^{-1} (fresh mass, FM; Strand et al., 1992), i.e. 250 times the intervention limit of 600 Bq kg^{-1} for basic foodstuffs, resulting in the condemning of 545 tons of reindeer meat during 1986 (Brynildsen et al., 1996). Thereafter practices of clean feeding and live monitoring of ^{137}Cs in animals has successfully reduced the annual amount of meat condemned (Brynildsen et al., 1996; Tveten et al., 1998). During the period 1987 – 1991 a total of 424 tons of reindeer meat was condemned, which was reduced still further to only a few tons per year from 1992 onwards (Tveten et al., 1998). Presently however, reindeer are still being condemned due to ^{137}Cs concentrations exceeding the current intervention limit for reindeer meat of 3000 Bq kg^{-1} in Norway (Norwegian Reindeer Herding Administration, unpublished data). It was hypothesised that the long-term rates of decline in ^{137}Cs concentrations in reindeer would slow down after some 15 – 20 years (Gaare and Staaland, 1994), while studies in the late 1990s indicated that rates of decline in ^{137}Cs concentrations in other components of the ecosystem were slower than previously reported (Jonsson et al., 1999; Smith et al., 2000). Reduced decline in ^{137}Cs concentrations may reduce the seasonal differences in ^{137}Cs concentrations in reindeer and can have consequences for the contemporary application of countermeasures against ^{137}Cs contamination, as well as for nuclear emergency preparedness planning. Detailed field studies are needed to elucidate the timing and significance in these changes in ^{137}Cs concentrations in reindeer.

Thirdly, animals predating upon reindeer may contain elevated concentrations of the nuclides (Mohn and Teige, 1968; Holleman and Stephenson, 1981; Thomas et al., 1994). Indeed, due to the significantly higher ^{137}Cs concentrations in reindeer than most other prey animals, ^{137}Cs concentrations in lynx could potentially be applied as a tracer of reindeer predation.

Finally, population groups like reindeer herding Saamis that consume large quantities of reindeer meat may receive appreciably higher doses from these nuclides than the average population (Lidén, 1961; AMAP, 1998; Mehli et al., 2000). In particular, data on the concentrations of natural radionuclides in reindeer in Norway are

required for more accurate estimates of the total absorbed doses received by these population groups.

1.2 Nuclides investigated

1.2.1 Isotopes of caesium (Cs)

Caesium is an alkali metal, together with for example sodium (Na) and potassium (K), and occurs naturally as the stable isotope ^{133}Cs. The radioactive isotope ^{137}Cs is an important fission product due to its high yield and relatively long physical half-life of 30.07 years, while ^{134}Cs is principally produced by neutron activation, and has a half-life of 2.065 years. Both isotopes are beta-emitters, but ^{134}Cs and ^{137}Cs are commonly more easily identified by the gamma-radiation spontaneously emitted by their metastable barium progeny.

The total releases of ^{137}Cs from atmospheric nuclear weapons testing during the 1950s and 1960s have been estimated to be 948 PBq (1 PBq = 10^{15} Bq) (UNSCEAR, 2000a), and resulted in average municipality deposition densities in Norway up to 10 kBq m^{-2} (as observed in 1986; Backe et al., 1986). Additionally, approximately 85 PBq ^{137}Cs and 54 PBq ^{134}Cs were released into the atmosphere from the Chernobyl accident (UNSCEAR, 2000b). Central Norway and mountainous parts of southern Norway were among the most contaminated areas outside Ukraine, Belarus and Russia (CEC, 1998), with average municipality deposition densities up to 150 kBq m^{-2} (sum of ^{134}Cs and ^{137}Cs; Backe et al., 1986).

Caesium behaves in many respects similarly to K, which in biological systems is generally found in large quantities as free ions inside cells. Soluble compounds of Cs are almost completely absorbed from the gastrointestinal (GI) tract (ICRP, 1990), but low bioavailability of radiocaesium in animal forage can appreciably reduce absorption (Hansen and Hove, 1991; Mayes et al., 1996; Beresford et al., 2000). Caesium is relatively uniformly distributed in soft tissues (ICRP, 1990; Paper I; Paper III) but some differences in accumulation between tissues exists probably due to different affinities for K and Cs in cell-membrane transport mechanisms (Burkardt and Wirth, 1986; Oughton and Salbu, 1992).

In soil, Cs is bound to mineral particles, and is therefore less available for uptake by plants growing on mineral soils compared to organic soils (Kruyts and Delvaux, 2002; Staunton et al., 2002; Rigol et al., 2002). Furthermore, ^{137}Cs uptake is low in plants growing on nutrient-rich soils compared to nutrient-poor soils (Varskog et al., 1994; Kruyts and Delvaux, 2002). In particular, uptake of Cs is high in K deficit organic soils (Kruyts and Delvaux, 2002; Staunton et al., 2002; Rigol et al., 2002). However, trace amounts of the clay minerals illite or vermiculite in an organic soil can potentially absorb all radiocaesium present (Rigol et al., 2002).

Fixation of Cs to the soil matrix is complete 3 – 7 years after fallout, and thereafter there is little change in transfer rates to plants (Ehlken and Kirchner, 1996, 2002). However, in peat soils there are reports of a lack of apparent ageing processes for Cs (Ehlken and Kirchner, 1996; Rigol et al., 2002). Consequently, variable long-term trends in ^{137}Cs concentrations in plants in natural pastures have been reported (Gaare et al., 2000; Andersson et al., 2001; Strebl et al., 2002).

Vertical distribution of Cs relative to the roots will influence Cs uptake (Varskog et al., 1994; Belli et al., 1996; Ehlken and Kirchner, 2002), as root densities of many plant species, including repeatedly defoliated perennial grasses, follow an exponential decline with depth (Ehlken and Kirchner, 1996). However, vertical migration is slow, and more than 30 years after the nuclear weapons testing, practically no ^{137}Cs from these tests was found below a depth of 10 cm in German and Italian alpine pastures (Bunzl et al., 2000). Grazing or cropping of plants increases plant Cs uptake (Ehlken and Kirchner, 1996), whilst seasonal changes in concentrations in plants may be due to different growth stages and climatic conditions (Albers et al., 2000).

Several mycorrhizal fungi are able to accumulate metals in high concentrations (see, e.g. Collin-Hansen, 2004), and fungi may therefore be an important source of radiocaesium to consumers (e.g. Hove et al., 1990; Skuterud et al., 1997; Mehli and Skuterud, 1998). The roots of most plants are associated with mycorrhizal fungi. Although mycorrhizal associations often enhance the plant's nutrient – and Cs – uptake, the effect cannot be generalized (Ehlken and Kirchner, 2002).

Gastberger et al. (2000) summarized that high transfer of radiocaesium in semi-natural habitats cannot be explained by a single factor but by the combination of several, such as the lack of dilution due to slow plant growth, climatic factors like extended snow cover in combination with frozen soil profiles, runoff effects, water-logging and biological, physical and chemical soil characteristics. In addition, radiocaesium fixation in the biomass and cycling between living and dead parts of the vegetation are likely to be further reasons for persistent and high plant availability (Gastberger et al., 2000).

The mobility of Cs deposited onto lichens is generally low (Nevstrueva et al., 1967), and removal of Cs by grazing and continuous dilution by lichen growth are considered to be more important factors than internal movement and washout in determining the long-term decline in Cs concentrations (Mattsson, 1975). Atmospheric fallout causes higher concentrations of several contaminants in lichens than vascular plants since lichens are slow growing perennials with high interception potentials for aerosols and particles in air and normal and indirect precipitation (Conti and Cecchetti, 2001), e.g., the absorption surface of *Cladonia stellaris* is roughly 10 times that of annual grass crops per unit dry mass (Nevstrueva et al., 1967). In general, three mechanisms influence the absorption of metals in lichens: 1) intracellular absorption through an exchange process; 2) intracellular accumulation; and 3) entrapment of particles that contain metals (Conti and Cecchetti, 2001).

1.2.2 Isotopes of strontium (Sr)

Strontium is one of the alkaline earth elements together with for example calcium (Ca), and occurs naturally as the stable isotopes ^{84}Sr, ^{86}Sr, ^{87}Sr and ^{88}Sr. The radioactive isotope ^{90}Sr is the most important Sr isotope from nuclear fission due to its high yield and relatively long physical half-life of 28.79 years. The radioisotope ^{85}Sr (used in Paper III) is produced by neutron activation, and has a half-life of 64.84 d. Sr-90 is a beta-emitter commonly determined via the relatively high-energy beta-particles emitted by the yttrium progeny, while ^{85}Sr decays by electron capture and is determined by the gamma-radiation emitted by the rubidium progeny.

The total releases of ^{90}Sr from atmospheric nuclear weapons testing have been estimated to be 622 PBq, approximately 66 % of the ^{137}Cs activity (UNSCEAR, 2000a). Although total ^{90}Sr releases from the Chernobyl accident were 11 – 12 % of that of ^{137}Cs (UNSCEAR, 2000b), ^{90}Sr deposition in Norway was only 1 – 3 % of that of ^{137}Cs (Bjørnstad and Salbu, 1992) due to faster deposition of ^{90}Sr from the atmosphere compared to ^{137}Cs (CEC, 1998). Nevertheless, the most contaminated areas in Norway received ^{90}Sr deposition densities following the Chernobyl accident comparable to the integrated nuclear weapons tests fallout (Bjørnstad et al., 1990).

Sr is discriminated in preference of Ca in transfer across biological membranes, and is therefore less effectively absorbed from the GI tract, excreted in milk and transferred to the foetus, and more effectively excreted by the kidneys (Comar, 1966; ICRP, 1994, 2001; Beresford et al., 1998b). A number of factors have been found to increase absorption of Sr from the GI tract, including milk diets, fasting, animal growth, the presence of vitamin D and low dietary contents of Ca, magnesium (Mg) and phosphorus (P) (see e.g. Howard et al. (1997) and references in ICRP (1994)). Strontium accumulates in bone in a similar fashion as Ca (as phosphates), but is perhaps less readily incorporated into new bone than Ca (ICRP, 1994). There appear to be no noticeable differences in ^{90}Sr concentrations between larger bones in reindeer (Staaland et al., 1991; Hognestad and Lie, 1998), and concentrations are similar in both bone and antlers of calves (Hognestad and Lie, 1998). Sr-90 in antlers reflects the dietary intake during the period of antler growth (Strandberg and Strandgaard, 1995; Hognestad and Lie, 1998). Persson (1971) observed lower ^{90}Sr/Ca-ratios in reindeer meat during summers when summer dietary ^{90}Sr/Ca-ratios were lower than in winter, but the seasonal differences in ^{90}Sr concentrations were small compared to differences in concentrations of ^{137}Cs (e.g. Paper IV), ^{210}Pb and ^{210}Po (Kauranen and Miettinen, 1967). Furthermore, Persson (1971) suggested that there would be seasonal differences in ^{90}Sr/Ca-ratios in bone, but such differences were not observed by Staaland et al. (1991).

In soil, a relatively large proportion of ^{90}Sr is available for plant uptake (see, e.g. Coughtrey and Thorne, 1983), and root uptake of ^{90}Sr in alpine pasture plants has been found to be comparable to or higher than that of ^{137}Cs (Bjørnstad et al., 1990; Gastberger et al., 2000; Schimmack et al., 2003). Strontium is exchanged in preference of Ca in minerals but the preference is reversed in organic matter (Ehlken and Kirchner, 2002) and plant uptake of ^{90}Sr is therefore normally reduced when the organic matter content is increased (Gastberger et al., 2000; Schimmack et al., 2003).

Sr-90 is not bound as strongly in lichens as radiocaesium (Nevstrueva et al., 1967) and correspondingly, shorter effective ecological half-times (T_{ecol}) for ^{90}Sr compared to ^{137}Cs have been observed in lichens (Persson, 1971; Heinrich et al., 1999).

1.2.3 Isotopes of lead (Pb) and polonium (Po)

Lead-210 and ^{210}Po are both metals and occur naturally as products in the ^{238}U decay series released into the atmosphere via the decay of radon-222 (^{222}Rn). Rn-222 is emanated at an average rate of 1.5 mBq m^{-2} s^{-1} from continental lands free from glaciers and permafrost (El-Daoushy, 1988). Pb-210 has a mean atmospheric residence time of about 29 d (Jaworowski, 1969; Salmon et al., 1998b) and is continuously deposited from the atmosphere in association with aerosols at a rate of approximately 55 Bq m^{-2} year^{-1} over Scandinavia (El-Daoushy, 1988). Generally, atmospheric ^{210}Pb

concentrations are related to the size of the underlying landmasses, and oceanic areas including islands, ice and snow covered lands have reduced atmospheric ^{210}Pb concentrations (El-Daoushy, 1988). However, deposition increases with increasing precipitation (Hill, 1960).

Whereas four stable Pb isotopes are found naturally, Po has no stable isotopes. In total, seven radioisotopes of Po occur as members of the natural radioactive decay series from ^{232}Th (thorium), ^{235}U (uranium) and ^{238}U. Po-210 (physical half-life 138.4 d) is the "grand-daughter" of ^{210}Pb (physical half-life 22.3 years), and is the only one of the seven Po isotopes with half-lives exceeding a few seconds or minutes. Pb-210 is a beta-emitter, while ^{210}Po emits high-energy alpha-particles. Po-210 therefore gives a relatively high absorbed dose in surrounding tissues, and is a substantial contributor to the natural radiation exposure of humans (UNSCEAR, 2000a).

In animals, Pb accumulates preferentially in certain soft tissues and bone (Salmon et al., 1998a). Polonium behaves in a broadly similar fashion to sulphur (Leggett and Eckerman, 2001) with an affinity to certain amino acids and proteins, e.g. in red blood cells, plasma protein and liver metallothionein (Aposhian and Bruce, 1991; Bulman et al., 1995).

Absorption of Pb from the GI tract varies widely because of the age, sex, and diet of the animal (Eisler, 1988; ICRP, 1994). Diet can influence the amount of Pb absorbed more than Pb intake per se, with a lack of vitamins and mineral elements (e.g., vitamin E, thiamin, ascorbic acid, Ca, zinc, iron, copper, Mg, P) or nutrients (e.g., fat, protein) in the diet resulting in an increase of Pb absorption (see references in Eisler (1988)). Furthermore, physical and chemical characteristics affect absorption, including the size of Pb particle, type of Pb compound ingested, presence of other compounds (acting synergistically or antagonistically), and dosage (Eisler, 1988; Beresford et al., 1998a). Beresford et al. (1998a) estimated an apparent absorption coefficient of 0.25 for sheep, but found this value relatively high compared to previously reported values of 0.01 – 0.18 for adult domestic animals.

Less is known about factors influencing absorption of Po, but chemical form is important (Naylor et al., 1993). Only one study attempting to characterize the forms of ^{210}Po in the foodstuffs ingested by man or animals appears to have been conducted, indicating that ^{210}Po is available in various forms in mussel flesh, brown crabmeat, lamb's liver and pig's kidney (Bulman et al., 1995). About 50 % of Po ingested in organic form is generally assumed to be absorbed by humans whereas absorption of inorganic compounds is assumed to be less (Eckerman et al., 1998). However, humans absorb 60 – 94 % of Po in crab meat (Hunt and Allington, 1993), and 35 – 80 % of Po in caribou meat (Tracy et al., 1999).

Contamination of vegetation by ^{210}Pb and ^{210}Po occurs predominantly by direct deposition (Ewers et al., 2003). For the reasons mentioned in section 1.2.1, lichens contain appreciably higher ^{210}Pb concentrations than vascular plants (Holtzman, 1966; Kauranen and Miettinen, 1969; Jaworowski, 1969) and fungi (Skwarzec and Jakusik, 2003). The ^{210}Po/^{210}Pb activity ratio in lichen is typically ~1 as ^{210}Po approaches secular equilibrium with ^{210}Pb (Kauranen and Miettinen, 1969; Mattsson and Persson, 1972; Thomas et al., 1994).

1.3 The reindeer (Rangifer tarandus tarandus)

The genus *Rangifer* (reindeer and caribou) belongs to the family *Cervidae*. The genus includes only one species, *Rangifer tarandus*, but there are several subspecies which in the Northern Hemisphere live between 50° - 82° N. The Norwegian reindeer belongs to the subspecies *Rangifer tarandus tarandus*. Approximately 58 % of Norwegian territory (not including Svalbard) is used as pasture by reindeer, a major part of which is used for reindeer herding (Norwegian Reindeer Husbandry Administration, 2004; Danielsen, J. Directorate for Nature Management, Trondheim. Personal communication). The semi-domesticated reindeer in Norway are free-ranging and graze mainly on natural pastures all year round. Generally there is a seasonal migration from summer grazing at the coast to winter areas where the ambient temperatures are lower but stable, with less likelihood of the snow covering the pastures freezing than occurs at the coast. However, in areas typically experiencing mild winters, winter grazing occurs at the coast.

The reindeer is adapted to seasonal variations in the quantity and quality of the diet. Its reduced feed intake in winter (Fig. 1) is a result of a complex and dynamic regulation system, where factors such as reduced metabolism and energy requirements (Nilssen et al., 1984), lower production of thyroid and growth hormones (Ryg and Jacobsen, 1982a,b), appetite (Tyler et al., 1999), forage availability and the physical characteristics of the forage plants act in concert (Storeheier et al., 2003). Indeed, much of the reduction in live body mass of Norwegian reindeer during winter is merely a result of reduced content of the GI tract (Tyler et al., 1999).

The diet of many reindeer populations consists of lichens, mosses and a wide variety of vascular plants throughout the year. Reindeer utilize fibrous grasses poorly (see, e.g. Aagnes et al., 1996), and on natural pasture they therefore select a variety of more digestible plant species. However, availability and nutrient content are also

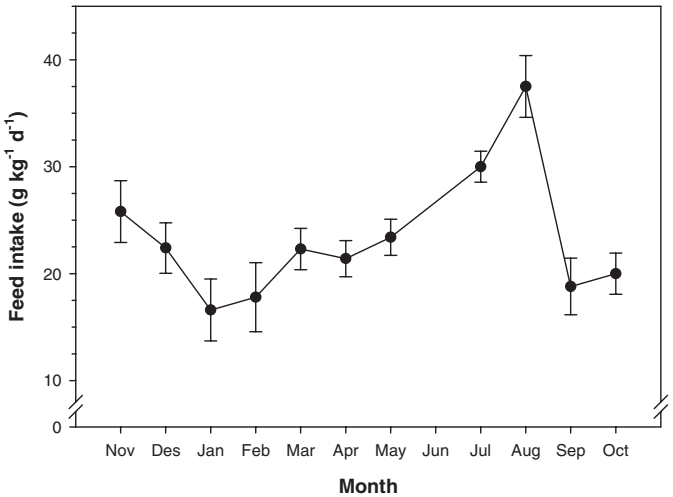

Fig. 1 Seasonal changes in voluntary feed intake (g kg^{-1} body mass d^{-1}) of adult female Norwegian reindeer fed a commercial reindeer feed (RF-71) ad libitum. The symbols represent mean values (± SD) of three animals obtained during the middle ten days of each month (figure adapted from Larsen et al. (1985)).

important factors determining the total diet (Gaare and Skogland, 1975; White and Trudell, 1980; Côté, 1998). During the summer, the diet of reindeer is dominated by graminoids (e.g. *Deschampsia flexuosa*, *Carex* spp.), leaves (e.g. *Vaccinium* and *Salix* spp.) and herbs (Gaare and Skogland, 1975; Staaland et al., 1995). The proportion of lichens (mostly *Cladonia* and *Cetraria* spp.) in the diet is often increased in winter as compared to summer, but a common winter diet is a mixture of lichens, shrubs, grasses and sedges (Gaare and Skogland, 1975; Mathiesen et al., 2000; Storeheier et al., 2003). The amount of lichen in the diet may vary substantially, but under relatively good winter pasture conditions, lichens are considered to form 60 – 85 % of the diet (Gaare and Staaland, 1994).

Lichen is a symbiosis of fungi and algae, with structure and chemical composition which differs considerably from those of vascular plants (Garmo, 1986; Svihus and Holand, 2000; Storeheier et al., 2002a,b). The utilization of lichens by ruminants requires a special adaptation of the rumen ecology, and the extent to which reindeer are able to utilize lichens depends both on the species that are selected (Storeheier et al., 2002b) and on what the animals have been eating recently (Storeheier et al., 2002a). Different experiments have shown that digestibility of *Cladonia* spp. can reach about 80 % (Storeheier, 2003).

Lichens are energy-rich and highly palatable feed for reindeer (Storeheier et al., 2002a), but not essential if there is other suitable food available. Lichens rely on the absorption of nutrients from the atmosphere and precipitation and consequently have a low nitrogen and mineral element (including Na, K and Ca) content (Garmo, 1986; Staaland and Sæbø, 1993). As a result, a lichen only diet is insufficient for reindeer over long periods (Staaland and Sletten, 1991), and a mixed winter diet is required in order to obtain a well-balanced intake of nutrients (Storeheier, 2003).

Reindeer milk is high in fat (10-15%) and protein (about 10%), and moderate to high in mineral elements (1-1.5%) compared to milk of domestic ruminants (Holand et al., 2002). The potential daily milk production during peak lactation is estimated to be only about 1.5 l d^{-1}, and the production declines rapidly to about 10 % of peak production after 20 weeks (Holand et al., 2002). Reindeer milk was traditionally part of the Saami diet, and continued locally in Fennoscandia until it eventually stopped in the early 1960s. Today, milking is practised only in the border area between Russia, China and Mongolia, although recently there has been a renewed interest in reindeer milk as a niche product in Fennoscandia (Holand et al., 2002).

1.4 The lynx (Lynx lynx)

The genus *Lynx* belongs to the family *Felidae*, and includes four species. The species Eurasian lynx (*Lynx lynx*) lives principally in the coniferous forest zone between the Atlantic Ocean and the Pacific Ocean and the Bering Strait. Eurasian lynx is the most numerous large carnivore mammal in Norway with a population of 300 – 350 individuals in 2002 (Andersen et al., 2003), and is found in most parts of the country.

The lynx is a solitary predator, preferring rugged forested terrain, with an average range in Norway of 250 to 1500 km^2 depending on prey density (Kvam and Jonsson, 1998). The population density of lynxes is only 3 – 10 animals per 1000 km^2, as there is little territorial overlap between individuals of the same sex (Solberg et al., 2003). Home ranges of male individuals are generally larger than that of females; the largest

recorded male home range in Norway reaching 3100 km^2 (Kvam and Jonsson, 1998). Cubs are born in May – June, and follow their mother for nearly a year (till February – May) before establishing separate home ranges. Six out of 18 tracked lynxes in Hedmark county established home ranges more than 150 km from their mother's (Andersen et al., 2000), while another sub-adult migrated about 400 km from Sarek (northern Sweden) to Steinkjer (central Norway). Average daily roaming distances, occuring mostly during the night, range from about 2 km for adult females to 5.3 km for adult males, with recorded extremes of 23.5 and 45 km. The latter was a male searching a female in the mating season (Andersen et al., 2000).

The size of lynx prey ranges from small rodents to larger cervids such as reindeer and moose. Stomach content analyses of 441 Norwegian lynxes showed that 67 % had eaten cervids (mainly roe deer and reindeer), 25 % small game (such as hare, capercaillie and grouse) and 8 % other species (such as fox and rodents) (Sunde and Kvam, 1997). Lynxes predate more on cervids during winter than summer (Kvam and Jonsson, 1998; Andersen et al., 2000), with males predating slightly more on cervids than females (Sunde and Kvam, 1997). For females, Sunde and Kvam (1997) found a tendency towards a body mass effect on food choice, indicating that the smallest individuals may have problems handling the largest prey species. However, another study found no difference in prey choice due to gender and age groups (Andersen et al., 2000). When cervids are the only prey, lynxes kill on average one animal every 5 d (Kvam and Jonsson, 1998).

2 Aims and hypotheses of the study

The main objectives of this study were to study absorption and retention of some important anthropogenic and natural radionuclides in reindeer, to describe time trends in anthropogenic radionuclides in reindeer, and study the transfer of radionuclides through the food chain to reindeer meat consumers. The answers to the following questions were important to achieve these objectives:

1) To what extent will different combinations of lichens and pelleted concentrates in the diet of reindeer affect radiocaesium absorption and retention? What is the bioavailability of Chernobyl radiocaesium in lichen?
2) How much is known about effects of general physiological factors on radiocaesium absorption and retention in animals, particularly ruminants?
3) How much dietary Sr and Cs is transferred to reindeer foetuses, and secreted in reindeer milk? And at what rates are Sr and Cs secreted in milk?
4) Are radiocaesium concentrations in free-ranging reindeer still decreasing at the same rate as in the first decade after the Chernobyl fallout, and are lichens still the predominant sources of radiocaesium absorbed by reindeer?
5) What is the time trend of ^{90}Sr concentrations in free-ranging reindeer? To what extent do concentrations of ^{90}Sr, ^{210}Pb and ^{210}Po in reindeer increase with the age of reindeer? Does higher precipitation in Østre Namdal result in higher concentrations of ^{210}Pb and ^{210}Po in reindeer in this area compared to the more continental Vågå area?
6) Can ^{137}Cs in lynxes be used as a tracer of reindeer predation, and reveal if individual lynxes specialize in prey species?
7) What are the absorbed doses in reindeer and reindeer meat consumers from the ionizing radiation emitted by the studied anthropogenic and natural radionuclides?

3 Summary of individual papers

3.1 Paper I: Absorption, retention and tissue distribution of radiocaesium in reindeer: effects of diet and radiocaesium source

In this study, different combined diets of fallout contaminated lichens and pelleted concentrates applicable for clean feeding of reindeer were offered to two groups of three calves, while 8 calves were offered a pure lichen diet and ^{134}CsCl in different quantities. The calves were aged 6 – 8 months at the beginning of the experiment. Constant daily radiocaesium administration rates were used throughout the experiments, which lasted 113 and 171 days respectively. Samples of blood, urine and faeces where collected at regular intervals for studies of absorption and retention of radiocaesium. In addition, radiocaesium concentrations were determined in tissues sampled from calves sacrificed at the end of the experiments.

Although differences in the offered combined diets of lichens and concentrates were appreciable (0.7 kg of lichen dry matter (DM) and 0.6 kg of concentrates, and 1.4 kg of lichen DM and 0.3 kg of concentrates daily, respectively), the consumed quantities of concentrates were not significantly different. Consequently there were no significant differences in absorption and retention of radiocaesium in these reindeer. However, the pure lichen diet resulted in significantly higher Cs retention: The biological half-time (T_{biol}) for radiocaesium in red blood cells (RBC) in animals fed with a pure lichen diet exceeded that in animals fed with a combined diet of lichen and concentrates by 40 % (17.8 ± 0.7 d vs. 12.7 ± 0.4 d). Corresponding differences in the T_{biol} values for urinary and faecal excretion were approximately 60 % and 40 %, respectively. The transfer coefficients (F_f) to reindeer meat were estimated to be 0.25 ± 0.01 d kg^{-1} for fallout radiocaesium and 1.04 ± 0.03 d kg^{-1} for ^{134}CsCl, reflecting differences in both radiocaesium bioavailability and retention.

The bioavailability of the Chernobyl radiocaesium in lichen in 1988 was estimated at ca. 35 % compared to ^{134}CsCl. Information about the bioavailability of radiocaesium was obtained from the F_f estimates, corrected for the differences in radiocaesium retention in the two experiments. The majority of the 4-fold difference in F_f and transfer coefficients to red blood cells was accounted for by the 2.9 fold difference in radiocaesium bioavailability, while differences in T_{biol} values (1.4 fold difference) accounted for the remainder. Estimates of F are commonly not accompanied by separation and quantification of the components absorption and retention. This study indicated that such quantification may be useful in studying reasons for variability in transfer coefficient estimates.

3.2 Paper II: Physiological parameters that affect the transfer of radiocaesium to ruminants

This paper is a review of different factors that affect absorption and retention of radiocaesium in animals, with particular reference to ruminants. The review was initiated by the study reported in Paper I, which demonstrated how differences in transfer coefficient values could be explained by differences in absorption and retention of radiocaesium in reindeer.

Uptake of radiocaesium and other radionuclides by animals from feed has been reported to vary according to species, size, age, growth rate, milk production and to the digestibility of the feed. In particular, allometric relationships between body mass and radiocaesium retention or *F* have been subjected to several studies. However, among animals of similar size, body mass may not explain much of the variability in radiocaesium transfer, and in all cases, considerable variability in radiocaesium transfer will be caused by the underlying biological and physiological factors determining an animal's feed intake and radiocaesium absorption.

Our understanding of the importance of source-dependent bioavailability on absorption of radiocaesium from the GI tract has undergone considerable improvements during the last decade. In light of this, the review indicates how a better understanding of the biological and physiological parameters affecting radiocaesium absorption and retention (e.g., feed intake, feed digestibility, milk production and physical activity) may help to reduce within-species variability in *F* values for radiocaesium. Particularly the effects of feed digestibility and physiological factors on absorption and endogenous faecal excretion of radiocaesium require further studies.

3.3 Paper III: Transfer of ^{85}Sr and ^{134}Cs from diet to reindeer foetuses and milk

This paper presents a feeding experiment with four pregnant reindeer on a concentrate diet given $^{85}SrCl_2$ and $^{134}CsCl$ daily during the last part of gestation. The aim of the experiment was to obtain data on the transfer of the nuclides from does to calves, and on the secretion of the nuclides in reindeer milk. Radionuclide concentrations were therefore determined in calves sacrificed at birth, and secretion of the nuclides was measured in milk up to 92 days postpartum. The moderate to high mineral element content of reindeer milk compared to milk of domestic ruminants suggests that reindeer milk will contain relatively high concentrations of these nuclides.

The experiment showed that similar fractions of the administered activities of ^{85}Sr and ^{134}Cs were transferred to the foetuses, i.e. 1.4 – 1.6 and 1.5 – 2.5 % of the total administered activities of ^{85}Sr and ^{134}Cs respectively were deposited in the calves at birth. The does absorbed only small fractions of ^{85}Sr available in the GI tract, probably since the concentrate diet satisfied the Ca requirements, and the transfer of absorbed ^{85}Sr from the mother to the foetus was therefore high compared to ^{134}Cs. The distribution of ^{85}Sr and ^{134}Cs in different tissues of the calves was similar to that reported from feeding experiments with about 1 year old reindeer calves.

Transfer coefficients (F_m) for ^{85}Sr and ^{134}Cs from feed to milk were estimated at 0.0218 ± 0.0026 and 0.185 ± 0.025 d kg^{-1}, respectively, and the observed ratio ($OR_{milk\text{-}diet}$) for ^{85}Sr was 0.124 ± 0.037. The F_m values are in agreement with estimates based on transfer of the nuclides in other ruminants. A reported general relationship between Ca intake and $F_{m,Sr}$ for milk of ruminants suggests that transfer of radiostrontium to foetuses and milk of free-ranging reindeer can be considerably higher than observed in this experiment (due to lower Ca intake with natural forage), but the transfer of Sr to milk will not be as high as that of ionic Cs. The concentrations of ^{85}Sr in milk suggested that the does mobilized skeletal stores of Ca and ^{85}Sr for milk production, although the diet appeared to satisfy the Ca requirements. In free-ranging reindeer with

radiostrontium intake during the whole year, radiostrontium concentrations in milk will therefore be higher than indicated by the F_m value observed in our study.

No differences in T_{biol} values for ^{85}Sr and ^{134}Cs secretion in milk were detected. Both nuclides were secreted with short and long-term half-times of 1 – 2 and 12 – 19 d, respectively, but the fraction of the nuclides being secreted with the different half-times varied. For ^{85}Sr, 80 – 90 % of the activity was excreted with the short T_{biol}, whereas the corresponding figure for ^{134}Cs was 30 – 50 %.

3.4 Paper IV: Chernobyl radioactivity persists in reindeer

The present study was initiated to obtain new data and more detailed knowledge about the long-term transfer of ^{137}Cs in the soil-plant/lichen-reindeer food chain that could help explain the persistent elevated ^{137}Cs concentrations in reindeer. The study was conducted during the period 2000 – 2003 through sampling of soil, vegetation and reindeer tissues in two climatically different areas, Østre Namdal and Vågå, representing areas in Norway most affected by fallout from the Chernobyl accident. Supplementary data on ^{137}Cs in reindeer from 1986 onwards were obtained from the Norwegian Reindeer Husbandry Administration.

No decline in ^{137}Cs concentrations was detected in reindeer from Vågå and Østre Namdal slaughtered in autumn since 1995, nor in reindeer slaughtered in winter since 1998 - 1999. In Vågå, autumn ^{137}Cs concentrations in reindeer declined with a T_{ecol} of 3.49 ± 0.44 years until 1995, while winter concentrations declined with a T_{ecol} of 3.24 ± 0.12 years until 1998. In Østre Namdal activity concentrations in autumn were lower and declined continuously with a T_{ecol} of 5.0 ± 1.2 years until 1995, while concentrations in winter declined until 1999 with a T_{ecol} of 3.63 ± 0.21 years. Seasonal differences in ^{137}Cs concentrations in reindeer have been less pronounced in recent years than during the first years after the Chernobyl accident, with ^{137}Cs concentrations occasionally higher in autumn than in winter.

The transfer of ^{137}Cs through the food chain from soil to reindeer was higher in Østre Namdal than Vågå, especially in winter. Higher soil-to-plant transfer was observed in 7 of 12 vascular plants sampled in Østre Namdal, while 2 of 3 lichen species in Østre Namdal contained relatively higher ^{137}Cs concentrations than in Vågå (compared to the ^{137}Cs deposition density). The study suggested that climatic influences on lichen growth and abundance, and on soil properties that influence the availability of ^{137}Cs for plant uptake, have a larger impact on long-term transfer of radiocaesium in the soil-plant/lichen-reindeer food chain than has been previously observed.

3.5 Paper V: ^{90}Sr, ^{210}Po and ^{210}Pb in lichen and reindeer in Norway

The field study described in Paper IV was designed with the aim of determining concentrations of ^{90}Sr, ^{210}Po and ^{210}Pb in lichens and reindeer tissues from Østre Namdal and Vågå. In addition to documenting current concentrations of these nuclides in reindeer in Norway, the study focussed on potential differences in concentrations of these nuclides in reindeer of different ages (adults and calves), since ^{90}Sr and ^{210}Pb accumulate in bone, and ^{210}Pb and ^{210}Po accumulate in liver. An additional aim of the study was to reveal whether higher precipitation rates in Østre Namdal result in higher

concentrations of ^{210}Pb and ^{210}Po in reindeer in this area compared to Vågå and other Nordic areas.

Age was an important parameter in determining concentrations of ^{90}Sr in reindeer. Sr-90 concentrations were approximately 40 % higher in bones of ~10 year old adult females compared to calves' antlers (^{90}Sr concentrations in antlers and bones of calves are similar). The difference with age did not appear to reflect the differences in intake during their periods of growth (< 2 year). Combined with previously reported data from Vågå this study suggested that ^{90}Sr concentrations in reindeer calves declined with a T_{ecol} of 9.03 ± 0.06 years during 1988 – 2002. Concentrations of ^{90}Sr were 50 – 80 % higher in bone and antlers of reindeer of a similar age from Vågå compared to those from Østre Namdal.

The analyses regarding the effect of animal age on muscle and liver tissue ^{210}Po and ^{210}Pb concentrations in reindeer gave equivocal results, and we concluded that age did not have a major effect on concentrations of these nuclides in this material. Concentrations of ^{210}Po and ^{210}Pb in muscle and liver tissues were comparable in the two study areas, and comparable to those reported for reindeer in other Nordic areas. Thus, differences in climate do not appear to have noticeable effects on soft tissue activity concentrations of ^{210}Po and ^{210}Pb in reindeer in Vågå and Østre Namdal.

3.6 Paper VI: Concentrations of ^{137}Cs in lynx (*Lynx lynx*) in relation to prey choice

This paper presents ^{137}Cs concentrations determined in muscle samples of 747 lynxes killed in Norway from the 1986 Chernobyl accident up to the year 2001. Through knowledge of reindeer grazing areas and the results of stomach analyses of killed lynxes, we attempted to quantify the effect of prey selection and in particular reindeer predation, on ^{137}Cs concentrations in lynxes.

Highly variable ^{137}Cs concentrations and aggregated transfer coefficients (T_{ag}) values were observed, probably caused by variable ^{137}Cs concentrations in prey and the lynxes' extensive home ranges and roaming distances. Concentrations of ^{137}Cs in lynxes were highest in 1988 – 1989, and decreased thereafter with average T_{ecol} in the range 6.9 – 8.9 years. Adult lynxes had higher ^{137}Cs concentrations than sub-adults, and lynxes killed in regions with extensive reindeer grazing areas were more contaminated than others. There were no systematic differences in ^{137}Cs concentrations in female and male lynxes.

A regression model with ^{137}Cs deposition density, the year lynxes were killed, age, and extent of reindeer grazing area could account for 50 % of the variability in observed ^{137}Cs concentrations. The analyses were equivocal regarding the influence of stomach content on ^{137}Cs concentrations in lynx muscle, i.e., on the lynxes' specialization in prey species.

4 Results and discussion

4.1 Uptake and excretion of Cs and Sr by reindeer

The seasonal variations in diet and metabolism of reindeer represent larger challenges for management of radiocaesium contamination in reindeer than in other domesticated ruminants. The major implications of these seasonal changes are:

- Diet: Increased intake of lichens during winter will result in higher radiocaesium intake during this season as long as lichens remain the most contaminated vegetation species. Furthermore, lower Cs excretion following the low potassium intake associated with a lichen diet (Holleman et al., 1971), and the relatively low crude fibre content of lichens (Garmo, 1986; Svihus and Holand, 2000), will contribute to increased radiocaesium concentrations in reindeer (Paper II). However, the relatively low bioavailability of fallout radiocaesium in lichen (Paper I) may limit the increase in radiocaesium concentrations in free-ranging reindeer suggested by these factors.
- Metabolism: Reduced metabolism, lower thyroxin levels and reduced physical activity during winter probably increase Cs retention in reindeer at this time of the year (Paper II). The accompanying reduced feed intake (see for example Fig. 1) will limit the radiocaesium intake, but probably also reduce the endogenous faecal excretion of radiocaesium (Paper II).

Seasonal changes in radiocaesium concentrations in reindeer, i.e. the combined effect of all of the above factors, have been described in several studies during the last four decades (see e.g., Åhman and Åhman, 1994). In the Vågå herd, up to 4-fold differences in radiocaesium concentrations between seasons were observed in the immediate years following the Chernobyl accident (Pedersen et al., 1993). However, only the effect of K (and Na) intake on Cs retention has been attempted to be quantified in isolation (Holleman and Luick, 1975b; Birke et al., 1995). Adding different quantities of K to a pure lichen diet, Holleman and Luick (1975b) reduced the T_{biol} for radiocaesium in individual reindeer from approximately 22 to 11 d, whereas average T_{biol} in free-ranging reindeer ranged 16 – 18 d during winter and 6 – 8 d during summer (Holleman and Luick, 1975a). In free-ranging reindeer the effect of the K intake will be confounded in time with the other seasonal factors mentioned above, and the quantitative effect of differences in K intake may therefore not be the same in free-ranging reindeer as in K dosing experiments. Furthermore, Valtonen (1979) observed an increased retention of potassium in the autumn in both male and female reindeer, presumably resulting from increased sex steroid secretion during the breeding season. It is not known if this has implications for Cs retention.

The difference in diets consumed by the two groups of reindeer fed both lichens and concentrates in Paper I (experiment 1) was not large enough to cause differences in T_{biol} between the groups. However, the difference in diets in the two experiments in Paper I, including differences in K and fibre content, resulted in differences in T_{biol} (Table 1). Both the average values and the difference between them corresponded to those in Holleman and Luick (1975a,b). Due to a shorter experimental period, the T_{biol} estimated for ^{134}Cs in pregnant reindeer on a concentrate diet (Paper III) were more uncertain than those in Paper I, but the T_{biol} appeared longer than that in reindeer calves fed concentrates and lichens (Table 1). In humans, Cs is excreted more rapidly during

Table 1 Summary of apparent absorption coefficients (A_a) and fractional absorption (a), biological half-times (T_{biol}) and transfer coefficients (F_{RBC}) for radiocaesium into red blood cells (RBC) of reindeer from Paper I and III. The unit of a is l^{-1} (Paper I) or kg^{-1} (Paper III), correspondingly the unit for F_{RBC} is d l^{-1} or d kg^{-1}.

Animal and experiment	A_a	a, 10^{-3}	T_{biol} (d)	F_{RBC}[1], 10^{-2}
Calves, exp. 1, Paper I	0.17	2.52 ± 0.06	12.7 ± 0.4	4.62 ± 0.26
Calves, exp. 2, Paper I	0.65 - 0.67	7.2 ± 0.2	17.8 ± 0.7	18.5 ± 1.3
Does[2], Paper III	0.52	4.5 ± 0.2	15.3 ± 1.1	9.9 ± 1.2

[1] Calculated as the product of a and T_{biol} divided by ln 2, cf. Eq. 4 in Paper I.

[2] The A_a for does was estimated from average faecal excretion values from day 17 onwards in Fig.1, Paper III. Values of a and T_{biol} are averages of doe A and B, which received ^{134}Cs the longest periods (the a value given in Paper III was not normalised to the daily ^{134}Cs intake).

gestation possibly due to higher glomerular filtration rate and lower reabsorption of Cs by the nephrons (ICRP, 2001). This effect was not evident for reindeer.

The average fractional absorption (a) of ^{134}Cs in RBC of does in Paper III was 37 % lower than in the reindeer calves given ^{134}CsCl added to the pure lichen diet in Paper I (Table 1). The reduction in a was probably due to the reduced GI absorption caused by the higher dietary K and fibre content, the reduced endogenous faecal excretion due to transfer of absorbed ^{134}Cs to the foetus, and probably a reduced endogenous faecal excretion due to higher metabolism (Paper II).

The mathematical description of the physiological processes influencing transfer of radiocaesium to ruminants was reviewed on a general level in Paper II. The transfer coefficient concept was introduced in the 1960s as a measure of the transfer of radiocaesium to cow's milk as this parameter significantly reduced the variation between individual animals, as compared to expressing total milk ^{137}Cs activity as a percentage of intake (Ward and Johnson, 1986). However, variability in all the factors referred to above (e.g., feed intake, digestibility, metabolic rates and physical activity) will influence the magnitude of the transfer coefficient. Paper II therefore suggests that improved understanding of variability in F values requires more detailed analyses of the underlying physiology. In particular, feed digestibility and physiological factors influencing absorption and endogenous faecal excretion are suggested as factors that deserve priority in future research. For instance, a two fold reduction in $F_{m,Cs}$ to cow's milk due to higher fibre intake was reported by Stewart et al. (1965). Furthermore, the majority of the 4-fold difference in F_{RBC} in Paper I (Table 1) could be accounted for by the 2.9 fold difference in radiocaesium bioavailability, while differences in T_{biol} values accounted for the remainder. In Paper III pregnant reindeer were given ^{134}CsCl, as per the calves in experiment 2 in Paper I, and the T_{biol} for Cs in pregnant reindeer was not noticeably shorter than in the reindeer calves in Paper I. Nevertheless, the transfer coefficient to RBC (Table 1) was 46 % lower than that of the reindeer calves in Paper I. The reasons for the difference are given in relation to the fraction absorption in the preceding paragraph. For the same reasons, a 22 % lower A_a for ^{134}Cs was estimated in the pregnant reindeer compared to the reindeer calves in Paper I.

The analyses of experiment 2 in Paper I suggested that for animals on a similar diet, individual variability in radiocaesium retention will affect F estimates more than differences in absorption. However, as in other experiments where accumulation of radionuclides in animals is studied by feeding constant daily activities, the estimated fractional absorption (or excretion) and T_{biol} values were correlated, and could therefore

mask an underlying individual variability in absorption. Furthermore, variability in absorption may be a more significant factor in free-ranging reindeer where diets will differ between individuals.

Rates of absorption and excretion of radiocaesium vary in different tissues in the body, and radiocaesium concentrations in reindeer's muscle may change slower than in other soft tissues when intake changes (see, e.g. Åhman, 1994). Differences in rates of absorption and excretion between tissues are likely reasons for the lack of correlation between ^{137}Cs concentrations in muscle, kidney and liver tissue in free-ranging reindeer in Paper IV. Furthermore, differences in absorption and excretion rates may explain why concentrations of radiocaesium in tissues of reindeer may differ both more (Åhman, 1994; Paper IV) and less (Rissanen et al., 1990) than that reported from the continuous feeding experiment in Paper I. However, the general pattern remains the same, with high concentrations in kidney and low in liver relative to muscle. This is in agreement with the relative Cs/K-ratios in the corresponding tissues in sheep (Oughton and Salbu, 1992).

The results of the current study suggest that pregnant reindeer feeding on concentrates have a limited uptake of Sr from the diet. Based on the estimated daily excretion of Sr, virtually no Sr was absorbed (Paper III). Daily excretions of more than 100 % of the daily administered activity may be observed when small amounts of radionuclides are absorbed, partly due to endogenous excretion of radionuclides to faeces (Howard et al., 2001). Variations in Sr absorption appear to be more dependent on the Ca status of the animals than on differences in the bioavailability of the Sr source (Beresford et al., 2000). Apparently the concentrate diet in the current experiment provided a sufficient Ca intake, which in addition to the discrimination of Sr in favour of Ca resulted in the limited uptake. Free-ranging reindeer will have a considerable lower Ca intake during winter (Staaland and Hove, 2000), and correspondingly higher Sr absorption. Significantly lower Ca concentrations in serum of reindeer calves during winter have been reported (Ringberg Lund-Larsen et al., 1978), and Persson (1971) related lower ^{90}Sr/Ca-ratios in reindeer meat during summer to corresponding ^{90}Sr/Ca-ratios in the diet. For example, with a Ca intake of 5.2 g d^{-1} (Staaland and Hove, 2000) the relationship between Ca intake and F_m in Howard et al. (1997) suggests that the absorption of Sr would be 3.5 times higher compared to that in the current experiments. Furthermore, the low Mg and P content of lichens (Garmo, 1986; Staaland and Hove, 2000) and the reduced feed intake during winter may potentially enhance Sr absorption in reindeer more than suggested by the relationship with Ca intake alone.

Persson (1971) concluded that a three-compartment model for ^{90}Sr in reindeer based on a similar model for humans, where 64 % of ^{90}Sr was excreted by an effective half-time (T_{eff}) of 2 – 3 d, 18 % was excreted with a T_{eff} of 20 d, and 18 % with a T_{eff} of 250 – 750 d depending on age, gave a satisfactory reproduction of observed ^{90}Sr concentrations in free-ranging reindeer. The two shortest half-times correspond well with those observed for Sr excreted in reindeer milk (Paper III; section 4.2), while a longer experimental period would be required to identify a possible third component. The variable Sr uptake, continuous and relatively slow accumulation (see, e.g. Farris et al., 1967) and continuous mobilization of skeletal Sr are challenging factors in studies of uptake and retention of Sr in free-ranging animals (section 4.3; Paper V).

4.2 Transfer of Cs and Sr to reindeer foetuses and milk

The experiment with pregnant reindeer fed concentrates showed that the transfer of ^{134}Cs from diet to foetus was comparable to that of ^{85}Sr, i.e. 1.5 – 2.5 and 1.4 – 1.6 % of the total administered activities of ^{134}Cs and ^{85}Sr respectively, were deposited in the calves at birth. Absorption and retention of Cs will probably be higher in free-ranging reindeer during winter due to the lower K and crude fibre intake and reduced metabolic rates. For example, the 60 % higher *a* value for reindeer calves on a pure lichen diet in Paper I compared to the does in the current experiment (Table 1) suggests a correspondingly higher absorption and subsequent accumulation in foetuses in animals on a pure lichen diet. Furthermore, as mentioned in section 4.1, absorption of Sr in free-ranging reindeer will probably be enhanced due to reduced Ca, Mg, P and feed intake during winter and spring until green vegetation is plentiful. Based on the reported relationship between Ca intake and the F_m for Sr (Howard et al., 1997) it was estimated that Sr absorption could be 3.5 times higher during winter, with Sr accumulation in calves correspondingly higher than in the current study.

The estimated F_m values for ^{134}Cs and ^{85}Sr from feed to reindeer milk were as expected on the basis of reported relationships for other ruminants, and confirmed that transfer of radiocaesium to milk of reindeer is higher compared to domestic ruminants. The F_m for Sr to reindeer milk appears comparable to reported values in sheep and goats (Beresford et al., 1998b). Higher transfer of ^{134}Cs and ^{85}Sr than that observed in the current experiment may be expected from a diet low in mineral elements, i.e. early in the lactation period (Paper III and discussion above). Furthermore, the study suggests that the F_m for Sr will be applicable only for fallout of radiostrontium occurring during the lactation period, since secretion of skeletal Sr accumulated before the lactation period will result in higher Sr concentrations in milk than indicated by the $F_{m,Sr}$ value. In addition, reindeer with suckling calves may mobilize more skeletal Sr than those in the current experiment due to higher milk production, if the does are not able to absorb enough Ca from the diet (Braithwaite, 1983).

The secretion of ^{134}Cs and ^{85}Sr in milk by three of the does could be reproduced by double exponential models with short- and long-term half-times of 1 – 2 and 12 – 19 d, respectively. However, a larger fraction of ^{134}Cs compared to ^{85}Sr was secreted with the longer T_{biol} (50 – 70 % vs. 13 – 18 %), causing ^{134}Cs concentrations to remain relatively high for a longer time period. Probably the short T_{biol} represents clearance of the nuclides from the plasma pool, and the longer T_{biol} secretion of ^{134}Cs being released from intracellular pools and mobilized ^{85}Sr, respectively. This was supported by the average long-term T_{biol} for ^{134}Cs in milk (15.0 ± 1.8 d) being similar to that in RBC (Table 1). The ^{85}Sr T_{biol} values were comparable to those in Persson (1971), while the percentage of Sr excreted with the different half-times differed (see section 4.1). Furthermore, doe D had least time for ^{85}Sr accretion in bone and had the smallest fraction of ^{85}Sr being secreted with the longer T_{biol} (13 %), while doe A received radionuclides for the longest period and had the largest fraction of ^{85}Sr secreted with the longer T_{biol} (18 %). In addition, doe A was able to retain more skeletal ^{85}Sr during the period of low milk production in early lactation which was thus available for mobilization later. These observations led to the above conclusion about the importance of secreted skeletal Sr in relation to the proportion directly transferred from the diet (expressed by $F_{m,Sr}$) in determining radiostrontium concentrations in milk. Furthermore, the conclusion was supported by studies showing that ewes are unable to absorb enough

dietary Ca in late pregnancy and early lactation to meet the high demands, irrespective of their Ca intake, and that they therefore mobilize skeletal stores of Ca (Braithwaite, 1983). The current experiment provided no direct evidence for Ca deficiency and increased resorption in the does, but the initial time trend in ^{85}Sr concentrations in milk of doe A (during the period of low milk production) compared to the other does supported this explanation. However, the inferred secretion of skeletal ^{85}Sr in milk might merely reflect the continuous dynamic processes in bone accretion and resorption, irrespective of Ca demands.

4.3 ^{137}Cs, ^{90}Sr, ^{210}Pb and ^{210}Po in free-ranging reindeer

The time series data on ^{137}Cs in reindeer indicated reduced seasonal differences and slower decline in ^{137}Cs concentrations in reindeer from Vågå and Østre Namdal from the end of the 1990s, and the field study found few noticeable differences in ^{137}Cs concentrations in lichens and vascular plants during 2001 – 2003 (Paper IV). Furthermore, the lower bioavailability of ^{137}Cs in lichens (i.e., approximately 35 % compared to ionic Cs; Paper I) compared to vascular plants (Beresford et al., 2000) will reduce the relative importance of ingested lichens as sources of ^{137}Cs in reindeer meat. The combination of these observations suggested that ingested lichens were no longer the predominant sources of ^{137}Cs in reindeer meat from Vågå and Østre Namdal from the late 1990s. Previously, ingested lichens were estimated to be main source of ^{137}Cs in reindeer from Vågå during winter as well as summer (Staaland et al., 1995). However, the reduced importance of lichens as sources of ^{137}Cs to reindeer was hypothesised by Gaare and Staaland (1994) due to observations of slower rates of decline in radiocaesium concentrations in vascular plants compared to lichens.

The point in time at which seasonal differences in radiocaesium concentrations in reindeer would become reduced would be dependent on the species composition of the reindeer diet, and might vary between grazing areas. The higher soil-to-plant transfer observed for 7 of 12 vascular plants in Østre Namdal than in Vågå (Paper IV) would indicate that the seasonal differences in concentrations in reindeer would be reduced earlier in Østre Namdal. However, at the same time 2 of 3 lichen species in Østre Namdal contained relatively higher ^{137}Cs concentrations than in Vågå (compared to the ^{137}Cs deposition density), and this may explain why seasonal differences apparently were reduced at similar points in time. The most plausible explanation for the generally higher aggregated transfer coefficients (T_{ag}) to plants in Østre Namdal is a likely combination of lower clay mineral contents, higher organic matter content and more nutrient-poor soils (Paper IV). The higher ^{137}Cs concentrations in lichens relative to the deposition density in Østre Namdal may be due to lower grazing pressure, resulting in a higher proportion of the original Chernobyl fallout still remaining within the thalli (Paper IV). Furthermore, leached ^{137}Cs from taller plants and trapped litterfall may have been absorbed by the lichen thalli, resulting in a slower decline in ^{137}Cs concentrations (Synnott et al., 2000).

Up to the end of the 1990s, the estimated T_{ecol} for ^{137}Cs in reindeer from Vågå and Østre Namdal were in the range of 3 – 5 years as reported from other Chernobyl affected areas in central and southern Scandinavia (Åhman and Åhman, 1994; Amundsen, 1995; Gaare et al., 2000; Åhman et al., 2001). No significant decline in ^{137}Cs concentrations was detected in Vågå and Østre Namdal since 1998 – 1999 (Paper

IV). From the suggested increased importance of vascular plants as sources of ^{137}Cs to reindeer, the slow decline in ^{137}Cs concentrations reported in vascular plants some years after the contamination occurred (Ehlken and Kirchner, 1996; Rigol et al., 2002), as well as the variable autumn ^{137}Cs concentrations in Vågå and Østre Namdal reindeer that have not declined significantly since the mid 1990s (Paper IV), it can be inferred that future declines in ^{137}Cs concentrations will occur slowly and may be governed by physical decay. In addition, variable abundances of fungi will probably have a significant impact on future time trends in autumn ^{137}Cs concentrations in reindeer, as demonstrated in 2003 when the average ^{137}Cs concentration in reindeer from Østre Namdal was the highest in 11 years, and comparable to the average concentration back in 1989.

The longer T_{ecol} values of 6 – 9 years observed in northern Fennoscandia during winter (Skuterud et al., 1999; Åhman et al., 2001; Rissanen et al., 2003) were hypothesized to be due to a combination of slower lichen growth rates and a more dominant role of lichens in the reindeer's diet in the more continental climate in the winter grazing areas in the north (Paper IV). Furthermore, climatic influences on the frequency and nutritional quality of soils (Moen, 1999), as well as the growth and abundance of lichens, were plausible explanations for the higher transfer of ^{137}Cs through the food chain from soil to reindeer in Østre Namdal compared to Vågå (Paper IV). If this hypothesis proves to be correct, it will have considerable influence on assessments of long-term consequences of radiocaesium fallout (such as in Howard et al. (2004)) in large areas of Norway. Another implication is that future climatic changes may alter today's transfer rates and long-term behaviour of radiocaesium in reindeer grazing areas, if the climate becomes more oceanic.

The available data for assessment of long-term trends in ^{90}Sr concentrations in reindeer were scarce compared to ^{137}Cs. In the most studied area in Norway, Vågå, concentrations of ^{90}Sr have been determined in only four different years after the Chernobyl accident. Time trends in ^{90}Sr concentrations were attempted and assessed, comparing concentrations in bone or antlers of adult animals and calves in 1988 – 1989 (Staaland et al., 1991), 1996 (Hognestad and Lie, 1998) and 2000 (Paper V).

Concentrations of ^{90}Sr in reindeer were approximately 50 – 80 % higher in Vågå than Østre Namdal, similar to the difference indicated by the average ^{90}Sr concentrations in lichens from the two districts (Paper V). Compared with the concentrations determined by Staaland et al. (1991) and Hognestad and Lie (1998) the concentrations determined in Vågå in 2000 and 2002 suggested that ^{90}Sr concentrations in reindeer calves declined with a T_{ecol} of 9.03 ± 0.06 years between 1988 and 2002 (Paper V). This T_{ecol} implies a larger decline in concentrations in animals than indicated by the 40 % difference in concentrations in the bones of ~10 year old females and antlers of calves in 2000 (Paper V). Hognestad and Lie (1998) could not identify any increase in ^{90}Sr concentrations with age in bone of animals born after 1986 and slaughtered in 1996.

The knowledge of ^{90}Sr uptake and excretion in reindeer is not satisfactory for detailed assessments of how much of the ^{90}Sr in adult animals is accumulated during their periods of rapid growth (during their first 2 years), and how much is due to subsequent continuous accumulation. In agreement with the T_{eff} values for ^{90}Sr in reindeer bone suggested by Persson (1971) (see section 4.1), the differences in concentrations from calves to older ages in the current study showed that ^{90}Sr in bones

is not a reliable record of exposure during the years of rapid growth (as suggested by Hognestad and Lie (1998)). These results support the use of antlers as monitors of environmental ^{90}Sr levels (Strandberg and Strandgaard, 1995; Schönhofer et al., 1994; Tiller and Poston, 2000), although ^{90}Sr concentrations in calves will not reflect ^{90}Sr in vegetation alone since some of the ^{90}Sr intake by calves (i.e., via milk) would have been originally accumulated by their mothers. Potentially bone remodelling is of less importance in male animals since they do not annually mobilize skeletal resources of Ca, and by association Sr, during pregnancy and lactation (Braithwaite, 1983; Paper III).

The comparable concentrations of ^{210}Po and ^{210}Pb in muscle and liver tissues of reindeer from Vågå and Østre Namdal indicated that the more oceanic climate in Østre Namdal, with potentially higher ^{210}Pb deposition than in Vågå, did not cause noticeably higher concentrations of these nuclides in reindeer tissues in Østre Namdal. Furthermore, the concentrations of these natural nuclides in Vågå and Østre Namdal were comparable to those reported for reindeer in other Nordic areas (Kauranen and Miettinen, 1967, 1969; Persson, 1974), while concentrations of ^{210}Pb and ^{210}Po in lichen and reindeer in the Nordic countries are generally lower than those in lichen and caribou in Canada (Tracy, 1993; Thomas et al., 1994; Macdonald et al., 1996). Although no differences between Vågå and Østre Namdal could be identified, reindeer herding in Norway occurs across such climatically different areas that differences in ^{210}Pb and ^{210}Po due to differences in deposition or intake of lichens might be appreciable elsewhere.

The analyses regarding potential accumulation of ^{210}Po and ^{210}Pb in reindeer from Vågå and Østre Namdal with age gave equivocal results. There were no significant differences in mean concentrations in reindeer calves and adult females, but correlation coefficients suggested that age accounted for 20 – 28 % of the variability in concentrations in half the tissue – district combinations. In conclusion, age did not appear to have a major effect on concentrations of these nuclides in this material.

Using the observed average ^{210}Po concentrations in lichen and reindeer muscle and an estimated lichen intake during winter of approximately 1.2 kg DM d^{-1} (Gaare and Staaland, 1994), the transfer coefficient (F_f) for ^{210}Po to reindeer meat was estimated at 0.04 – 0.06 d kg^{-1} (FM; Paper V). Correspondingly an F for ^{210}Po to reindeer liver of approximately 1 d kg^{-1} (FM) can be estimated. These estimates are a factor of 10 higher than the corresponding values for cattle (IAEA, 1994; Ewers et al., 2003). Estimates of transfer coefficients require an assumption of equilibrium radionuclide concentrations in the tissues. The limited information available on uptake and excretion of ^{210}Po in ruminants suggests that most of the ingested activity is excreted with a relatively short T_{biol} of 1 – 4 d (Sejkora, 1982; Watters et al., 1971). Together with the observation of no noticeable increase in ^{210}Po concentrations in reindeer from December onwards by Persson (1972), this suggested that the assumptions of equilibrium concentrations in this study were reasonable. On the contrary, Kauranen et al. (1971) observed increasing ^{210}Po concentrations until late winter. The cited observations by Persson (1972) and Kauranen et al. (1971) may be influenced by changing dietary intake of lichens. Furthermore, since ^{210}Po is associated with proteins (see section 1.2.3) the lower metabolism in reindeer during winter may result in slower excretion of ^{210}Po in reindeer during winter than in the studies cited above. Slower ^{210}Po excretion would limit the validity of the equilibrium assumption, in which case the estimated F values would be underestimates.

4.4 Radioactive caesium as tracer for reindeer predation by lynx

Reindeer generally contain higher concentrations of radiocaesium than other animals, especially during winter, and radiocaesium concentrations will therefore be higher in lynxes predating upon reindeer than in lynxes predating upon other animals. Mohn and Teige (1968) inferred that most of the ^{137}Cs contamination in lynxes in Norway was traceable to reindeer, and Åhman et al. (2002) found higher ^{137}Cs concentrations in lynxes from reindeer grazing areas than in lynxes from areas without reindeer. One of the aims of the study of radiocaesium in lynxes (Paper VI) was therefore to see if radiocaesium concentrations in lynxes could be employed as tracers of reindeer predation and to test the hypothesis that individual lynxes specialize in prey species (e.g., some predate only roe deer, others only reindeer).

Radiocaesium concentrations were highest in lynxes killed in municipalities with extensive reindeer grazing areas, and a simple modelling approach indicated that ^{137}Cs in the most contaminated lynxes in Nord-Trøndelag county could be mainly due to reindeer predation (Paper VI). Nevertheless, the analyses of the influence of the lynxes' stomach content on T_{ag} for ^{137}Cs to lynx and ^{137}Cs concentrations in lynx muscle did not give simple answers: One-way analysis of variance showed that stomach content was a significant factor, while two-way analysis of variance did not identify stomach content as significant (Paper VI). The analyses did therefore not provide unequivocal support for the hypothesis that lynxes specialize in prey species. However, variability in ^{137}Cs concentrations in lynxes resulting from spatially and temporally variable ^{137}Cs concentrations in all prey species, variable and uncertain ^{137}Cs deposition density estimates applied in calculation of T_{ag} values, as well as the extensive home ranges and mobility of particularly sub-adult lynxes and males during the mating season (which coincides with the hunting season), possibly prevented statistical identification of the significant role of stomach content on ^{137}Cs concentrations in lynxes.

Average T_{ecol} values for ^{137}Cs in lynxes in Norway were estimated to be 6.9 – 8.9 years, and no systematic differences in T_{ecol} due to counties or grazing area categories could be detected. The estimated T_{ecol} in lynxes appeared long compared to reported T_{ecol} in reindeer during the first decade after the Chernobyl accident (section 4.3), but the slower decrease in concentrations in reindeer from the mid 1990s onwards (Paper IV) may be part of the explanation. Furthermore and as for reindeer, concentrations in other prey species are probably decreasing slower than in the first years after the Chernobyl accident, e.g. a T_{ecol} approaching the physical half-life of 30 years was estimated for roe deer (Johanson and Bergström, 1994).

4.5 Absorbed doses by reindeer and reindeer meat consumers

Although the physiological behaviour of the studied radionuclides and the use of them as tracers of ecological processes is interesting per se, the main reason why they are studied is generally that they emit ionizing radiation that potentially induce negative biological effects either in the organisms themselves, or in organisms higher in the trophic chain. Recently, greater attention has been paid to the protection of the environment against ionizing radiation, for example through the EC Inco-Copernicus project "Environmental protection from ionising contaminants in the Arctic" ("EPIC"; Hosseini et al., 2005).

4.5.1 Doses to reindeer and lynx

The calculations of absorbed doses to animals in this study are based on the radionuclide specific dose conversion coefficients (DCC) provided by Hosseini et al. (2005). These DCCs have been derived assuming uniform distribution of the radionuclides in ellipsoids representing the various geometric forms of animals. Given that activity concentrations are known, the DCCs provide absorbed dose rates. The relative biological effectiveness (RBE) and radiation weighting factors for wild organisms are areas of much uncertainty (Garnier-Laplace et al., 2004), and EPIC has suggested the use of a provisional weighting factor of 10 for alpha-particles (e.g., for ^{210}Po; Hosseini et al., 2005). The weighted DCCs for internally incorporated ^{134}Cs and ^{137}Cs (including their Ba progeny), ^{90}Sr (including the progeny ^{90}Y), ^{210}Po and ^{210}Pb (including the progeny ^{210}Bi) for reindeer are, respectively, 3.54×10^{-6}, 2.25×10^{-6}, 5.69×10^{-6}, 2.73×10^{-4} and 2.17×10^{-6} Gy year^{-1} per Bq kg^{-1}. Contrary to the dose conversion coefficients applied for humans (see section 4.5.2), the DCCs do not include tissue weighting factors since little of the information required for such weighting is available (Hosseini et al., 2005).

This study calculated absorbed doses from estimated annual average concentrations of radiocaesium (i.e., ^{134}Cs and ^{137}Cs) and ^{90}Sr in reindeer during the first year after the Chernobyl fallout, and integrated doses during 10 years after 1986 (since most animals are slaughtered before or when they are 10 years old). Annual absorbed doses from ^{210}Po and ^{210}Pb to reindeer were calculated, and some indications of doses to lynxes from the studied radionuclides are discussed. As for all living organisms, reindeer and lynxes are subjected to ionizing radiation from several natural radionuclides (see, e.g. Hosseini et al., 2005). The absorbed dose in caribou from ^{40}K is for instance 0.3 – 0.4 mGy year^{-1} (Macdonald et al., 1996). Furthermore, a range of other mostly short-lived radionuclides in the Chernobyl fallout could be added to the doses received during the first year after the accident.

The calculation of doses from ^{134}Cs and ^{137}Cs assumed that the concentrations of the nuclides by mid autumn corresponded to the annual average concentrations, and the average body concentration was estimated to be 0.57 times the concentration in muscle (Åhman, 1994). The ^{137}Cs concentration was thus given by the average of autumn and winter concentrations in Paper IV, although this approach probably overestimates the annual average concentrations in years when fungal fruit bodies were abundant. Radiocaesium concentrations in reindeer have been reported to increase during winter in some areas (see, e.g. Åhman and Åhman, 1994), but the Vågå and Østre Namdal herds migrate to pastures with lower Chernobyl deposition from mid winter. Therefore no increase in concentrations occurs after December, as demonstrated for Vågå by Pedersen et al. (1993). The live monitoring of reindeer in Norway also considered the ^{134}Cs isotope, and ^{134}Cs concentrations were obtained from the same archives as the ^{137}Cs concentrations*. The resulting estimated average absorbed doses are given in Table 2. Average dose rates in reindeer during 2002 – 2004 were approximately 2.5 and 2.0 mGy year^{-1} in Vågå and Østre Namdal, respectively.

* The archives of the Norwegian Reindeer Husbandry Administration gave concentrations as the sum of ^{134}Cs and ^{137}Cs, according to Brynildsen and Strand (1994). This sum was split into ^{134}Cs and ^{137}Cs using a ^{134}Cs/^{137}Cs ratio in the Chernobyl fallout of 0.5 per 1 July 1986 (Backe et al., 1986) which was decay corrected to each monitoring occasion.

Table 2 Estimated approximate average whole body doses to reindeer in Vågå and Østre Namdal reindeer herding districts from ^{134}Cs and ^{137}Cs during 1986 – 1987 and 1986 – 1996 (mGy).

Reindeer herding district	Nuclide	Absorbed dose first year	Integrated absorbed dose during 10 years
Vågå	^{134}Cs	25	58
	^{137}Cs	34	160
Østre Namdal	^{134}Cs	19	40
	^{137}Cs	26	100

Calculations of absorbed doses from ^{90}Sr required an estimate of concentrations during the first year after the Chernobyl fallout. An estimate of ^{90}Sr concentrations in reindeer bone in Vågå in 1986 – 1987 was obtained by using the concentration of 1814 ± 270 Bq kg^{-1} DM in 1988 - 1989 (Staaland et al., 1991) corrected for a T_{ecol} of 9.03 ± 0.60 years back to 1986. This gave a value of approximately 2100 Bq kg^{-1} DM. The concentration was then assumed to decrease exponentially to the value of 1500 Bq kg^{-1} DM by 1996, which was the concentration observed in animals born during 1984 – 1986 and slaughtered in 1996 (Hognestad and Lie, 1998). From Hognestad and Lie (1998) a concentration ratio between bone (FM) and the average whole body of approximately 6.7 was estimated after 8 weeks of feeding with ^{85}Sr. Concentrations in muscle relative to bone in these animals were similar to values reported from field studies, i.e. about a factor 1000 lower (Paakkola and Miettinen, 1963; Hanson, 1966; Persson, 1971). Although Persson (1971) observed seasonal variations in ^{90}Sr in reindeer meat, these variations were less pronounced than those seen in ^{137}Cs concentrations (e.g., Paper IV). Furthermore, since seasonal variations in ^{90}Sr concentrations in bone were not observed in Vågå (Staaland et al., 1991), the current dose estimates did not include such variations. Thus, with a DM/FM ratio in bone of 0.83 (Hognestad and Lie, 1998), the average body ^{90}Sr concentrations in reindeer born in Vågå in 1986 were assumed to decrease exponentially from 260 Bq kg^{-1} in 1986 to 190 Bq kg^{-1} in 1996. These concentrations correspond to absorbed doses of approximately 1.5 mGy during the first year after the Chernobyl fallout, and an integrated dose of 14 mGy during 10 years for animals born in 1986. Absorbed doses in bone were about seven times these values. Based on the observed differences in ^{90}Sr concentrations in bone from Vågå and Østre Namdal during 2000 – 2002 (Paper V), the absorbed doses from ^{90}Sr have presumably been lower in Østre Namdal.

The data from the current study were not sufficient for estimates of doses from ^{210}Po and ^{210}Pb due to both the limited number of tissues and seasons studied. The dose estimates therefore to a large extent rely on the data in Kauranen and Miettinen (1969, 1971) and Persson (1972), giving the values presented in Table 3 as estimates of annual average concentrations in different tissues. The estimated values were slightly lower than those observed during winter in the present study (Paper V), as expected from the seasonal variation. The concentration in "other" tissues in Table 3 was estimated from the average concentrations in kidney, spleen, lung, ovary, testes, adrenals and pancreas relative to muscle in Kauranen and Miettinen (1967), i.e. a ^{210}Po concentration 6 times that in muscle and a ^{210}Pb concentration equal to 1/20th of this. Use of the relative organ masses in reindeer given in Gaare and Staaland (1994) gave estimated annual average whole body concentrations of 31 Bq ^{210}Po kg^{-1} (FM) and 20 Bq ^{210}Pb kg^{-1} (FM). These average concentrations correspond to absorbed dose rates of approximately 8.4 and

Table 3 Estimated annual average concentrations of ^{210}Po and ^{210}Pb in reindeer (Bq kg^{-1} FM). Estimates are based on data by Kauranen and Miettinen (1967, 1969, 1971) and Persson (1972).

Tissue	^{210}Po	^{210}Pb
Bone	74	170
Muscle	7.4	0.22
Liver	130	30
Blood	11	2.2
Other	40	2

0.044 mGy year^{-1} respectively. Doses to the organs of highest concentrations (Table 3) were correspondingly higher (e.g., 35 mGy year^{-1} from ^{210}Po in liver). Up to nearly ten times higher annual doses from these nuclides were estimated for some caribou herds in the Canadian Arctic by Macdonald et al. (1996), due to higher radionuclide concentrations and omission of seasonal variation in concentrations in the estimates.

Comparing observed radiocaesium concentrations in lynxes (Fig. 3a in Paper VI) with those in reindeer in Vågå and Østre Namdal (Fig. 5 in Paper IV) suggests that the highest concentrations in lynxes were similar to the highest concentrations in reindeer. Consequently, since DCCs for ^{134}Cs and ^{137}Cs are lower for lynxes than reindeer due to the lynxes' smaller body size, the absorbed doses from radiocaesium in lynxes have probably not exceeded those in reindeer. No data on ^{90}Sr in lynx in Norway are available, but concentrations were presumably significantly lower than in reindeer due to low ^{90}Sr intake with meat. Consequently, absorbed doses from ^{90}Sr in lynx were negligible. Nor are data on ^{210}Po and ^{210}Pb in lynxes in Norway available. However, according to Kauranen et al. (1971) concentrations of ^{210}Po in lynxes are 15 – 20 % of those in reindeer, and doses are correspondingly lower. From this discussion it follows that lynxes predating reindeer after the Chernobyl accident received their highest absorbed doses from radiocaesium, and that the doses in lynx in most cases did not exceed those in reindeer.

The calculations in this study suggested that Chernobyl radiocaesium appreciably increased the absorbed dose to reindeer in the most contaminated areas of Norway. Without a radiation weighting factor of 10 for alpha-radiation, the absorbed dose from the natural nuclides ^{210}Po and ^{210}Pb would be about an order of magnitude lower than that from radiocaesium during the first years after the Chernobyl accident. Regardless of weighting factors, the calculated doses from both anthropogenic and natural nuclides suggested that some of the most exposed animals received dose rates approaching 1 mGy d^{-1}. Although dose rates of this size may induce some biological effects (Sazykina et al., 2003), no effects on morbidity, mortality or reproductive capacity would be expected (Real et al., 2004). Furthermore, the doses from the natural radionuclides may also provide cytogenetic adaptive and protective effects (Ulsh et al., 2004). In the feeding experiment reported in Paper I, where reindeer calves were administered different quantities of ^{134}Cs for 171 days, linear relationships were found between absorbed doses and number of chromosomal aberrations, and between absorbed doses and number of mitoses with chromosomal aberrations in lymphocytes (Røed, 2004). However, the number of aberrations or mitoses with aberrations were significantly higher than in the control calves only in the animals receiving the highest dose rates of approximately 3 Gy year^{-1} (during the latter half of the study). Corresponding numbers

in the calves receiving dose rates ranging 140 to 870 mGy $year^{-1}$ were not significantly higher than in the controls (Røed, 2004).

4.5.2 Doses to humans

The calculations of absorbed doses to humans in this study are based on ingestion dose coefficients from ICRP (1994). These coefficients give effective dose equivalents (Sv) for adults during 50 years per Bq ingested. The effective dose is weighted for the probability of stochastic effects in different tissues (using tissue weighting factors), and also incorporates different radiation weighting factors (e.g., a factor of 20 for alpha-particles) (ICRP, 1991). The ingestion dose coefficients for ^{134}Cs, ^{137}Cs, ^{90}Sr, ^{210}Po and ^{210}Pb in adults are, respectively, $1.9x10^{-8}$, $1.4x10^{-8}$, $2.6x10^{-9}$, $1.2x10^{-6}$ and $7.0x10^{-7}$ Sv Bq^{-1} (ICRP, 1994).

Current annual reindeer meat consumption rates among South Saamis range up to 140 – 150 kg, with an estimated average of 58 kg (Thørring et al., 2004a). Most animals selected by the South Saamis for their own consumption are slaughtered during winter (Thørring et al., 2004a), and the observed radionuclide concentrations in reindeer meat in early winter (Paper IV and V) were therefore assumed representative of those in ingested meat. There has been a decline in the traditional consumption of entrails, and no estimates of doses from consumption of liver or kidney were therefore made.

Calculated radiocaesium doses to humans based on the observed concentrations during the first years after the Chernobyl fallout (Paper IV) are only of hypothetical interest, as countermeasures (e.g., clean feeding and selection of reindeer with low concentrations) have substantially reduced the doses to the Saami population in central Norway (Strand et al., 1992; Mehli et al., 2000). A person consuming 58 kg meat from Østre Namdal with the average concentrations of 11000 Bq ^{134}Cs kg^{-1} and 26000 Bq ^{137}Cs kg^{-1} during winter 1986 – 1987 (Paper IV) would receive a total effective dose of approximately 33 mSv. Only one of the persons whole body monitored during late winters of 1987 – 1989 received an effective dose exceeding 5 mSv $year^{-1}$, i.e., 12 – 13 mSv $year^{-1}$ (Strand et al., 1992). Doses from radiocaesium to the South Saamis in subsequent years are given in Mehli et al. (2000) and Thørring et al. (2004b). Even in 2002 – 2004 consuming 58 kg meat with average ^{137}Cs concentrations from Vågå or Østre Namdal in December (Paper IV) would give effective doses exceeding 1 mSv $year^{-1}$, the dose limit for the public recommended by ICRP (1991).

An estimate of effective dose due to ^{90}Sr can be obtained from the concentrations in bone, assuming that concentrations in muscle are a factor of 1000 lower than in bone (Paakkola and Miettinen, 1963; Hanson, 1966; Persson, 1971; Hognestad and Lie, 1998). The ^{90}Sr concentrations in bone in Vågå were estimated using the same approach as in section 4.5.1, giving concentrations in meat declining exponentially from 1.7 Bq kg^{-1} during 1986 – 1987 to 1.2 Bq kg^{-1} by 1996. Ingestion of 58 kg meat with these concentrations would give effective doses the first and tenth year of approximately 0.26 and 0.19 μSv, respectively. Doses to humans from ^{90}Sr in reindeer meat from Østre Namdal have probably been even lower (cf. the difference in concentrations in bone in 2000 – 2002, Paper V).

The average ^{210}Po concentration in meat of adult animals in Vågå and Østre Namdal in December was approximately 9.3 Bq kg^{-1} FM (obtained from the corresponding DM value in Paper V using a DM/FM ratio of 4). Consuming 58 kg meat

with this concentration gives an effective dose of approximately 0.64 mSv. Since storage will appreciably influence ^{210}Po concentrations in meat due to the relatively short physical half-life of 138.4 d, this dose is probably an overestimate. The lower dose coefficient for ^{210}Pb compared to ^{210}Po, as well as the lower ^{210}Pb concentrations in edible tissues, makes the dose from this nuclide in humans negligible compared to ^{210}Po. An exemption occurs if products are stored for long time periods. Then concentrations of unsupported ^{210}Po will decline, while the relative importance of ^{210}Pb increases.

The countermeasures applied after the Chernobyl accident have reduced the effective dose from radiocaesium to the South Saami population during the last decade (see Thørring et al., 2004b) to doses comparable to that from ^{210}Po. However, the time trend in ^{137}Cs concentrations in Paper IV suggests that countermeasures will continue to be important to keep the dose from ^{137}Cs at this low level in the reindeer herding districts of Norway with the highest ^{137}Cs deposition.

5 Conclusions

- Experiments performed during winter showed that a pure lichen diet of low potassium content significantly reduced radiocaesium excretion in reindeer compared to feeding diets of lichens and concentrates. The T_{biol} for radiocaesium in RBC was 40 % longer with a pure lichen diet. The bioavailability of Chernobyl radiocaesium in lichen was ca. 35 % compared to ^{134}CsCl in aqueous solution.
- Knowledge of dietary and physiological factors influencing radiocaesium absorption and retention in other animals suggests that dietary potassium level is not the only factor appreciably influencing transfer of radiocaesium to reindeer. Further studies on absorption and retention of radiocaesium in reindeer should consider other seasonally varying factors, particularly the effects of digestibility and metabolic rates on absorption and endogenous faecal excretion of radiocaesium.
- Similar proportions of dietary ^{134}Cs and ^{85}Sr were transferred to the foetuses of reindeer on a diet rich in mineral elements, while the F_m for ^{134}Cs was a factor of 8.5 higher compared to ^{85}Sr. Transfer of the nuclides to foetuses and milk of free-ranging reindeer may be considerable higher than observed in this experiment, due to lower K and Ca intakes in winter and spring. Furthermore, secretion of Sr accumulated prior to the lactation period and higher milk production in free-ranging reindeer will probably cause higher Sr concentrations than indicated by the F_m in this experiment.
- Effective ecological half-times for ^{137}Cs in reindeer from Vågå and Østre Namdal were 3 – 5 years up to the late 1990s. Thereafter there has been no significant decline, probably due to a reduced role of ingested lichens as ^{137}Cs sources. The transfer of ^{137}Cs through the food chain from soil to reindeer appeared appreciably higher in Østre Namdal than Vågå, especially in winter. Higher soil-to-plant transfer was demonstrated for 7 of 12 vascular plants sampled in Østre Namdal, and 2 of 3 lichen species in Østre Namdal contained relatively higher ^{137}Cs concentrations than in Vågå (compared to the ^{137}Cs deposition density).
- Combined with previously reported data from Vågå this study suggested that ^{90}Sr concentrations in antlers of reindeer calves decreased with a T_{ecol} of 9.03 ± 0.06 years during 1988 – 2002. This decrease was larger than indicated by the 40 % difference in ^{90}Sr concentrations in calves and ~10 year old adult reindeer observed in 2000. Concentrations of ^{90}Sr were 50 – 80 % higher in reindeer from Vågå compared to those from Østre Namdal, while there were no significant differences in ^{210}Po and ^{210}Pb concentrations between the districts. Age did not appear to have a major effect on ^{210}Po and ^{210}Pb concentrations in muscle and liver tissues of reindeer.
- Deposition density of ^{137}Cs, the year lynxes were killed, age, and extent of reindeer grazing area could account for approximately 50 % of the variability in observed ^{137}Cs concentrations in lynx. The results were ambiguous regarding the lynxes' specialisation in prey species, and suggested that further work on the possible use of radiocaesium as a tracer of reindeer predation by lynxes will require experimental data on Cs retention in lynxes and better estimates of deposition density in the lynxes' home ranges.

- The calculated absorbed doses to reindeer from both anthropogenic and natural nuclides indicated that some of the most exposed individuals received dose rates approaching 1 mGy d^{-1} after the Chernobyl fallout. Only lynxes with the highest radiocaesium concentrations received doses comparable to those received by reindeer.
- The assessment of doses to humans from the studied radionuclides showed that ^{137}Cs continues to be the most important contributor to ingested doses by South Saamis. The time trend in ^{137}Cs concentrations in the studied reindeer herding districts suggests that ingestion doses by persons with average South Saami consumption rates of reindeer meat will continue to exceed the 1990 recommendation by the International Commission on Radiological Protection, if countermeasures are not applied.

6 Further investigations

Following on from the conclusions of this study, the following two areas should deserve further investigations:

- The significance of seasonal differences in feed digestibility and various physiological parameters on absorption and endogenous faecal excretion of radiocaesium. In experiments with potassium-rich diets in winter, neither Holleman and Luick (1975b) nor our experiment (Paper I), reproduced the shortest half-times observed by Holleman and Luick (1975a) during summer. This suggests that other seasonal factors than K intake may be of significance in determining radiocaesium retention in reindeer.

- Long-term transfer of ^{137}Cs to free-ranging reindeer. Although transfer of ^{137}Cs in the food chain leading to reindeer has been considered relatively well known after 40 years of studies, the slower rates of decline and regional differences in transfer demonstrated in this study, highlight the need for continuing monitoring and research in this field of radioecology. It is hypothesized that climate can cause appreciable regional differences in long-term transfer of radiocaesium in the soil-plant/lichen-reindeer food chain. Without relatively elevated intervention limits for ^{137}Cs concentrations in reindeer meat, reindeer herding in the contaminated areas of Norway is likely to suffer practical consequences of the Chernobyl fallout (i.e., countermeasure application) for many years to come.

Since approximately 58 % of Norwegian territory is used as pasture by reindeer, a major part of which is used for reindeer herding, the probability that any future ^{137}Cs contamination event in Norway will affect reindeer grazing areas is great. A thorough understanding of reindeer radioecology is therefore essential in Norwegian nuclear emergency preparedness.

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Glossary and abbreviations

Absorbed fraction is the fraction of the ingested intake by an animal that is transferred to a specified receptor tissue

Activity concentration is the activity of a radionuclide (in Bq) per unit mass (kg) of a specified substance (e.g. meat)

Ad libitum is a Latin based term that means at one's pleasure

Aggregated transfer coefficient (T_{ag}) is the activity concentration (Bq kg^{-1}) in a specified object per unit deposition density (Bq m^{-2}) in the soil. Unit m^2 kg^{-1}.

Allometry is the study of size and its consequences. There are for instance relationships between body size and metabolic rates, ingestion rates, inhalation rates, lifespan, home range.

Anthropogenic means caused or produced by humans

Apparent absorption (A_a) is the difference between dietary radionuclide intake and faecal excretion, while *true absorption* is corrected for endogenous faecal excretion of the radionuclide (i.e., direct transfer from blood to the intestine of radionuclides initially absorbed from the GI tract)

Bioavailability is the availability of an element for plant root uptake or uptake through the gastrointestinal tract of an animal

Biological half-time (T_{biol}) is the time required for the activity of a radionuclide in an organism to be reduced to half of the original activity as a consequence of biological processes, not accounting for radioactive decay

Contaminant is a substance that is found in a place where it should not be. This does not necessarily mean that it is harmful, but depending on what it is and the amount that is present, it may be.

Countermeasure is action to reduce radionuclide transfer in food chains or to reduce environmental contamination

d Time, in days

DCC Dose conversion coefficient

DM Dry mass; mass of a sample after it has been dried at a given temperature to constant mass

Effective ecological half-time (T_{ecol}) is the time required for the activity of a radionuclide in an environmental compartment (e.g. organism) to be reduced to half of the original activity as a consequence of radioactive decay, biological and ecological processes

Effective half-time (T_{eff}) is the *biological half-time* corrected for radioactive decay, i.e., the time required for the activity of a radionuclide in an organism to be reduced to half of the original activity as a consequence of biological processes as well as radioactive decay.

F See *transfer coefficient*

Fallout is atmospheric deposition of particles resulting from a nuclear explosion or accidental release

FM Fresh mass; mass of a sample intended to represent the mass of a living tissue

Fractional absorption (a) is the fraction of the ingested activity that is transferred to a specified receptor tissue

GI Gastrointestinal

Gy Gray; absorbed dose

In utero is a Latin based term that means within the womb

Isotope is any of two or more forms of a chemical element having the same atomic number but different atomic masses. For example, different isotopes of carbon and nitrogen can be found in nature.

l Volume, in litre

Metabolism is the sum of the physical and chemical processes in an organism by which its material substance is produced, maintained, and destroyed, and by which energy is made available

Mycorrhiza is a symbiotic association of the mycelium of a fungus with the roots of plants, in which the hyphae form a closely woven mass around the rootlets or penetrate the cells of the root

Observed ratio refers to the quantity of a given radionuclide per g of the analogous element (e.g., Bq ^{85}Sr/g Ca) in a medium divided by the same ratio in a precursor medium (e.g., milk/feed) in order to measure discrimination of the radionuclide relative to the analogous element (*OR*)

Physical half-life is the time required for the activity of a radionuclide to be reduced to half of the original activity as a consequence of radioactive decay

Postpartum is a Latin based term that means after parturition

Radionuclide is a radioactive isotope. May be naturally present or human-made (anthropogenic)

Radiocaesium is abbreviation of radioactive caesium

SD Standard deviation

Secular equilibrium is equilibrium attained between a long-lived parent radionuclide and short-lived progeny in a decay scheme

Semi-natural refers to extensively (as opposed to intensively) used land (Norwegian: utmark)

spp. Abbreviation of species

Stochastic effects are effects for which the probability of their occurring, rather than their severity, is a function of radiation dose without threshold (stochastic means random)

Sv Sievert; effective dose equivalent

T_{ag} See *aggregated transfer coefficient*

T_{biol} See *biological half-time*

T_{ecol} See *effective ecological half-time*

T_{eff} See *effective half-time*

Transfer coefficient (*F*) is the mass or volumetric activity concentration in the receptor tissue or product of an animal (Bq kg^{-1} fresh mass or Bq l^{-1}) divided by the rate of the radionuclide into the mouth by ingestion (Bq d^{-1}), i.e., it is a measure of the amount of the animal's daily radionuclide intake that is transferred to 1 kg of the animal product at equilibrium. *F* takes the subscript *f* or *m* when referring to meat (flesh) or milk, respectively.

Paper I

Absorption, retention and tissue distribution of radiocaesium in reindeer: effects of diet and radiocaesium source

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ORIGINAL PAPER

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Absorption, retention and tissue distribution of radiocaesium in reindeer: effects of diet and radiocaesium source

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Abstract Radiocaesium absorption and retention in reindeer (*Rangifer tarandus*) calves was compared in groups fed diets containing different proportions of lichen and concentrates, and different chemical forms of radiocaesium (^{134}CsCl in solution or fallout from the Chernobyl accident). Daily intakes of fallout radiocaesium were 15–23 kBq, while daily intakes of ^{134}CsCl ranged from 70 kBq to 1,160 kBq. The half-life for radiocaesium in red blood cells (RBC) in animals fed with a pure lichen diet exceeded that in animals fed with a combined diet of lichen and concentrates by 40% (17.8±0.7 days vs. 12.7±0.4 days). Corresponding differences in the half-lives for urinary and faecal excretion were about 60% and 40%, respectively. Transfer coefficients (F_f) to reindeer meat were estimated to be 0.25±0.01 days kg^{-1} for fallout radiocaesium and 1.04±0.03 days kg^{-1} for ^{134}CsCl, reflecting differences in both radiocaesium bioavailability and retention. The bioavailability of the Chernobyl radiocaesium in lichen in 1988 was estimated at ca. 35% compared to ^{134}CsCl.

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Introduction

Several studies have been conducted on the transport of radioactive caesium along the food chain lichen-reindeer-man (see e.g. [1, 2, 3, 4, 5, 6]), following the observation of elevated concentrations of radiocaesium from atmospheric nuclear weapon testing in humans using reindeer for food [7]. In 1986, the Chernobyl accident caused significant radiocaesium deposition on mountain and forest areas in Scandinavia. As a result, in the early 2000s the radiocaesium content in reindeer meat in some areas still exceeded the national intervention levels of 3,000 or 1,500 Bq kg^{-1} in Norway and Sweden, respectively [8, 9]. The significant levels of contamination that followed the Chernobyl accident, with individual reindeer meat values reaching 150 kBq radiocaesium kg^{-1}, gave rise to concern about possible biological consequences in reindeer [10, 11, 12]. A feeding experiment was launched using relatively high radiocaesium administration rates to establish possible dose-response relationships regarding biological effects of ionising radiation in reindeer. Results on radiocaesium absorption and excretion in this experiment are presented here, while some preliminary results concerning biological endpoints have been presented by Røed [13].

Elevated radiocaesium concentrations in free-ranging reindeer are due to the natural dietary intake of contaminated lichens and other vegetation (see e.g. [14, 15]). Contaminated lichens are generally considered to be the main source of radiocaesium intake in reindeer during winter, whereas fungi can be an important source during autumn [16]. The particularly high concentrations in winter resulting from an increased lichen and therefore increased radiocaesium intake (e.g. [17, 18]), are possibly enhanced by a slower radiocaesium excretion rate in winter, when potassium intake is lower (e.g. [19]). In addition to variations in intake and retention, the bio-

availability of radiocaesium in forage (i.e., the degree of absorption across the gastrointestinal tract [20, 21, 22]) will determine the contamination levels in reindeer.

This paper presents results of two experiments[1] designed to obtain detailed knowledge on radiocaesium absorption and retention in reindeer fed different combinations of lichen and concentrates, and radiocaesium from different sources. Furthermore the accumulation of ^{134}Cs in tissues of animals on a range of daily ^{134}Cs administration rates was studied. Daily intakes ranged from those encountered on contaminated pastures (about 15 kBq) to quantities expected to induce biological effects (1,160 kBq).

Materials and methods

Two feeding experiments were carried out during November 1988–March 1989 and October 1990–April 1991 on 6 and 12 male reindeer calves (aged 6–8 months), respectively, obtained from commercial herds in southern Norway. All animals were housed in metabolism cages for the duration of the experiments. Prior to the start of the experiments, the animals were given a 30-day preparatory period to allow the calves to become accustomed to the experimental conditions. Radiocaesium was fed to the animals in two forms:

– Experiment 1: Lichen contaminated by Chernobyl fallout
– Experiment 2: ^{134}CsCl in aqueous solution sprayed onto lichen.

During the 30-day preparatory period, each animal was given ammonium-iron-hexacyanoferrate at a rate of 300 mg day^{-1} to purge existing radiocaesium from the body [23]. During the course of the experiment all animals had free access to water. At the end of experiment 1, one animal from each group was sacrificed, while all animals were sacrificed at the end of experiment 2.

Experiment 1: concentrates and lichen contaminated by Chernobyl fallout

In experiment 1, 2 groups of 3 animals (groups A and B) were offered the following daily diets for 113 days:

a. 0.7 kg lichen dry matter (DM) and 600 g concentrates (pelleted, barley-based; 13% crude protein content).
b. 1.4 kg lichen DM and 300 g concentrates (18% crude protein content).

The lichen used in the experiment was collected by hand in September 1988 from an area of southern Norway which had received the highest radiocaesium deposition from the 1986 Chernobyl accident. The collected lichen was thoroughly mixed, and contaminating soil and vegetation removed, before being sub-divided into sufficient daily portions for a period of ca. 2 months and stored in plastic bags. During the experiment, 21 daily portions were randomly selected for ^{137}Cs determination, giving a mean concentration (±SD) of 21.3±2.1 kBq ^{137}Cs kg^{-1} DM. The collected lichen (350 g kg^{-1} DM) consisted of approximately 60% *Cladonia stellaris*, 30% *Cetraria nivalis*, and 10% combined *Cladonia arbuscula*, *Cladonia rangiferina* and *Cetraria islandica*.

Concentrates with different crude protein contents were used in order that both groups of animals were provided with a similar overall nitrogen intake.

Experiment 2: ^{134}CsCl sprayed on lichen

In experiment 2, each individual in 4 groups of 2 animals was given fixed, daily quantities of ionic ^{134}Cs (in 0.5 ml of ^{134}CsCl solution) sprayed individually onto approximately 5 g of dry lichen for 171 days. The contaminated lichen was given to the animals in the morning after cleaning the feed troughs, and consumed immediately. Daily intakes of ^{134}Cs were 70, 140, 420 or 1,160 kBq in groups 1–4, respectively. A fifth group of 4 animals served as a control. All 12 animals were fed on a diet consisting only of lichen (fed ad libitum). High administration rates were given to enable a dose-response test of possible effects of radiation from the internal radiocaesium contamination on chromosomal aberrations [13].

Sample preparation and analysis

The design of the metabolism cages (rough grated floors overlying tilted finer gratings) allowed for the separate collection of urine and faeces during the experiment. Faeces, urine and blood samples were collected twice a week for the first 3 weeks of experiment 1 and 8 weeks of experiment 2, and once a week thereafter (2–3 week intervals for blood samples in experiment 2). Feed remains were collected on a daily basis (experiment 1), and body masses were measured every 2–4 weeks. Blood was taken from the jugular vein using sodium heparin as an anticoagulant. Blood samples were centrifuged for 20 min at 3,000 rpm, and a 5 ml portion of the red blood cell (RBC) fraction was used for radiocaesium determination. Radiocaesium concentrations were corrected for plasma dilution using the haematocrit of the RBC fraction (generally about 90%). In experiment 1 it was assumed that ^{137}Cs concentrations in plasma were one-sixth of the corresponding concentrations in RBC (see Results section, experiment 2). Plasma was collected after centrifugation of blood samples (experiment 2 only). The urine produced individually during each sampling interval was weighed and sub-divided into two 5 ml samples for analysis. Radiocaesium concentrations in blood and urine were determined using an LKB Wallace 1280 UltraGamma counter with a NaI(Tl) detector, regularly calibrated using standard solutions of ^{134}Cs and ^{137}Cs. The detection limit was about 50 Bq l^{-1} with the counting times used.

The faeces produced during each sampling interval were mixed and weighed. A 500 g sub-sample from each individual was dried for 2 days at 105°C, homogenised and sub-divided into two 12 g samples for analysis. In experiment 1, any remaining feed at the end of each day was collected, weighed, dried and homogenised. All tissue samples from the sacrificed animals were analysed as fresh weight. Determination of radiocaesium in faeces, feed remains and tissue samples was performed on a Minaxi 5000 Auto-Gamma counter with a NaI(Tl) detector. The detection limit was ca. 200 Bq kg^{-1} with the counting times used.

Statistical analyses

Analysis of variance and two-sample *t*-tests were used to compare body mass, feed intake and urine and faeces produced by each group of animals in each experiment. Two-way analysis of variance was used to compare activities in RBC, urine and faeces between groups A and B in experiment 1, while a non-linear regression analysis was used to compare the results of experiments 1 and 2, due to differences in experimental design (e.g., sampling scheme, length of project period and number of animals). Before any statistical analyses, all radiocaesium activity concentrations and daily excreted activities were normalised to the mean daily radiocaesium intake during the experimental period. Logarithmic transformation was used to harmonise variances of the normalised activity levels.

The non-linear regression model fitted to the activity concentration and daily excreted activity data was derived from the assumption that, following termination of radiocaesium administra-

[1] The experimental work reported here was carried out as part of the late Øyvind Pedersen's PhD work.

Table 1 Overview of feeding experiments, consumed feed and radiocaesium administration rates

Exp.	Experimental period (days)	Group	Number of animals	Diet (kg day^{-1}) (mean±SD)		Administration rate (kBq day^{-1}) (mean±SD)
				Lichen (DM[a])	Concentrates	
1	113	A	3	0.7	0.34±0.14	15.0±1.0
		B	3	1.1±0.1	0.30±0.03	23.2±1.9
2	171	Control	4	*ad libitum*	-	<1
		1	2			70
		2	2			140
		3	2			420
		4	2			1160

[a] Dry matter

tion, the activity concentration of RBC will decrease exponentially with time (t) according to:

$$R(t) = y_0 \cdot e^{-\frac{\ln 2}{T_{1/2}} t} \quad (1)$$

where y_0 is the activity concentration (Bq l^{-1}) when radiocaesium administration is terminated. Similar equations were used for radiocaesium excretion (Bq day^{-1}) with urine and faeces. Hence, the increase in radiocaesium activity concentrations in RBC, urine and faeces with time in non-contaminated animals (initial state) that are administered radiocaesium can be described by:

$$A(t) = \int_0^t I \cdot a \cdot e^{-\frac{\ln 2}{T_{1/2}} t} dt \quad (2)$$

where I is the intake, and a is a constant factor describing the fractional absorption of radiocaesium in the RBC fraction, or fractional excretion with urine or faeces (a is also called the assimilation fraction [24]). $T_{1/2}$ is the biological half-life. With a constant daily intake (I), integration of Eq. 2 and normalisation to the daily intake gives:

$$\frac{A(t)}{I} = \frac{a \cdot T_{1/2}}{\ln 2} \cdot \left(1 - e^{-\frac{\ln 2}{T_{1/2}} t}\right) \quad (3)$$

This model, which is a non-linear model of the general form $y = \beta_0\left(1 - e^{-\beta_1 \cdot t}\right)$, was log transformed and used to estimate a and $T_{1/2}$ for the log transformed normalised radiocaesium activity concentrations and daily excreted activities in experiments 1 and 2. The estimated parameters were used to compare absorption and retention in the two experiments. Parameters were considered significantly different if their asymptotic 95% confidence intervals did not overlap.

Transfer coefficients (i.e., the amount of one animal's daily intake of a radionuclide that is transferred to 1 kg of the animal product at equilibrium; unit day kg^{-1}) are commonly used to estimate activity concentrations occurring in milk, meat and other tissues of animals fed contaminated fodder [25]. Equation 3 shows that the transfer coefficient, F, is given by:

$$F = \frac{a \cdot T_{1/2}}{\ln 2} \quad (4)$$

i.e., the transfer coefficient is determined by the fractional absorption and the biological half-life of the radionuclide in the compartment studied.

Statistical analyses were carried out using SPSS for Windows (release 11). All data are presented as mean ±SD, unless otherwise specified. When P-values are not given, a significance level of 0.05 was used.

Table 2 Body mass (BM) and faeces and urine excretion during the experimental periods (mean±SD of all individual values in each group)

Experiment	Group	BM	Faeces	Urine
		(kg)	(kg DM day^{-1})	(kg day^{-1})
1	A	49.5±2.1	0.298±0.029	1.14±0.41
	B	51.6±2.7	0.350±0.080	0.80±0.31
2	Control	52.2±4.0	n.a	1.41±0.55
	1	51.3±3.4	0.193±0.027	1.52±0.56
	2	58.4±5.3	0.209±0.031	2.16±0.82
	3	56.4±2.0	0.200±0.021	2.03±0.86
	4	57.0±3.6	0.206±0.032	1.54±0.43

n.a Data not available.

Results

Experiment 1

Table 1 gives an overview of both feeding experiments and the amount of fodder consumed in experiment 1. In experiment 1, group A animals consumed all offered lichen and although group B animals refused on average 0.3 kg DM lichen day^{-1} they had a significantly greater lichen intake than group A (P=0.04). Due to refusals of concentrate by group A there was no significant difference in consumption of concentrate between the groups. Daily ^{137}Cs intake in group B was on average 55% higher than that in group A. The body mass (Table 2) of group A animals increased on average by 3.2±3.2 kg, while group B animals increased by 6.2±1.0 kg. Neither average body mass, nor body mass increases differed between the groups. Urine production was largest in group A, while group B animals produced most faeces (P<0.01).

Figure 1 shows ^{137}Cs activity concentrations in RBC and daily excretion of ^{137}Cs with urine and faeces, normalised to the average daily ^{137}Cs intake in each group. There was no significant difference in normalised activity concentrations in RBC between animals of group A and B (two-way analysis of variance, P=0.38). Both the excretion of ^{137}Cs in urine and urine production was highest in group A (P<0.01; cf. the higher excretion during the initial period of the experiment, Fig. 1b). This may be due to a higher mineral intake increasing both urine production and also urinary caesium excretion [26]. In contrast, the higher faecal production in group B (probably due to

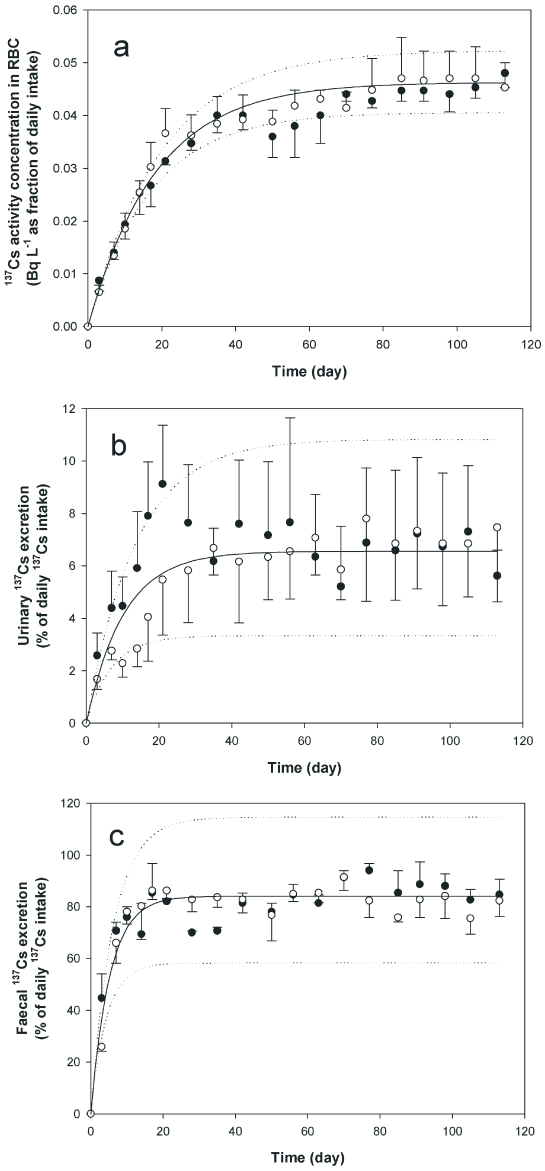

Fig. 1 Activity concentrations of ^{137}Cs in **a** RBC (Bq l^{-1}), **b** daily urinary and **c** fecal ^{137}Cs excretion (Bq day^{-1}) in group A (*filled circles*) and B (*open circles*) animals of experiment 1, normalised to the average ^{137}Cs administration rate (Bq day^{-1}). *Dots* represent observed values. One-sided standard deviations are shown. *Curves* represent fits by the model to the average of all animals (with 95% confidence intervals)

higher feed intake) did not increase the faecal ^{137}Cs excretion (P=0.12).

The estimated parameters a and $T_{1/2}$ (Eq. 3) from the non-linear regression analyses are given in Table 3. Since no difference in normalised ^{137}Cs absorption in RBC in groups A and B was observed, nor in faecal excretion, the results for all six animals were treated as one group in the RBC and faeces regression analyses. In agreement with the results of the analysis of variance, the regression analysis gave significantly different values for the parameter a for ^{137}Cs excretion in urine in the two groups of animals, while there was no significant difference in estimated $T_{1/2}$-values.

Table 3 Estimated fractional absorption of radiocaesium in RBC or excretion with urine and faeces (a), and biological half-lives ($T_{1/2}$) in the model for ^{137}Cs absorption and retention in reindeer offered different mixed diets (Eq. 3). Mean±SE, and model R^2. For RBC and faeces, results are given for regression analyses using the average of all animals only

Compartment	Group	a, 10^{-3}	$T_{1/2}$ (days)	R^2
RBC	A+B	2.52±0.06	12.7±0.4	0.996
Urine	A	10.0±1.5	4.7±0.9	0.81
	B	4.1±0.5	11.1±2.0	0.86
	A+B	6.4±0.7	7.1±1.0	0.89
Faeces	A+B	160±10	3.6±0.3	0.95

Although the statistical analyses revealed significant differences in the urinary excretion of ^{137}Cs between animals in groups A and B, visual inspection of Fig. 1b shows small differences between the groups of animals. It was therefore concluded that this experimental set-up could not reveal significant differences in absorption and excretion of ^{137}Cs in reindeer with the two diets used.

The average apparent absorption (A_a; i.e., the difference between daily radionuclide intake and faecal excretion relative to daily intake [21]) was estimated to be 0.17 for fallout ^{137}Cs in lichen using the model parameters for daily faecal excretion.

Due to an error with the feeding practices, the daily ^{137}Cs intake was inadvertently lowered by some 12% from day 28 to day 65 of the experiment through feeding of lichen with lower ^{137}Cs concentration. Neither intake values in Table 1 nor normalised activity concentration nor excretion values were corrected for this lower concentration. The non-linear regression analyses (Eq. 3) gave correspondingly significant negative residuals due to the lower activity concentrations and excretions in this period (Fig. 1). The parameter values in Table 3 were therefore calculated excluding data from day 28 to day 77.

Concentrations of ^{137}Cs in thigh muscle (m. semitendinosus and m. semimembranosus) samples from slaughtered animals in group A and group B were 3.6 and 6.3 kBq kg^{-1}, respectively. Normalised to their daily ^{137}Cs intake this gives transfer coefficient values of 0.24 and 0.26 days kg^{-1}. The RBC activity concentrations in these animals were 0.64 and 1.19 kBq l^{-1} (average of last 5 observations), i.e. 0.18 and 0.19 of the concentrations in meat.

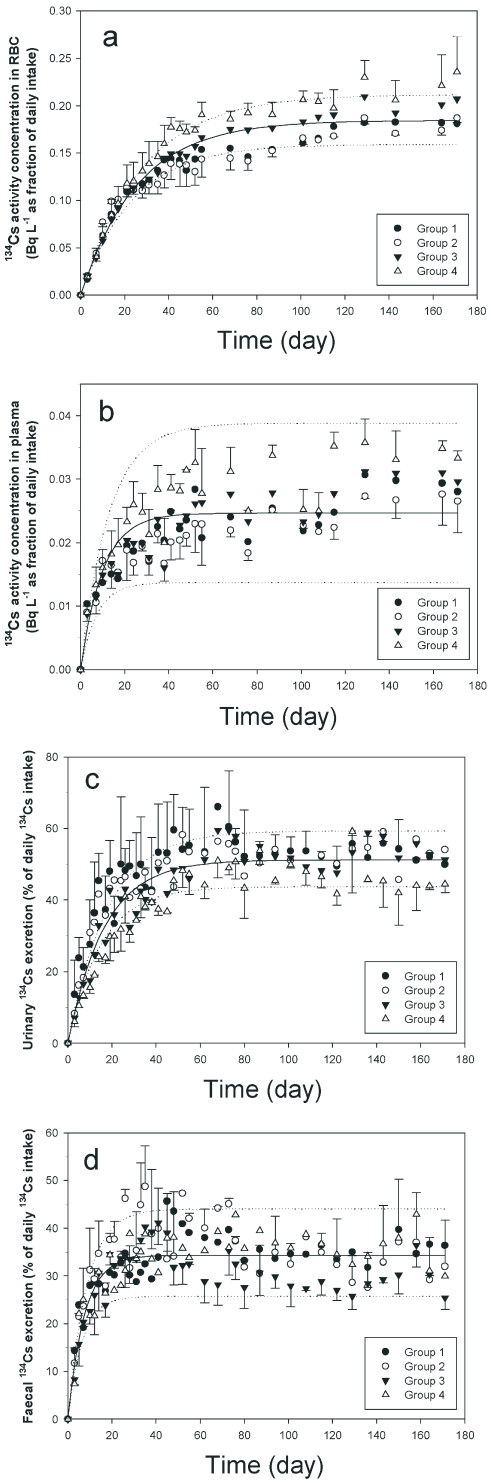

Fig. 2 Activity concentrations of ^{134}Cs in **a** RBC, **b** plasma (Bq l^{-1}), **c** daily urinary and **d** fecal ^{134}Cs excretion (Bq day^{-1}) in the four groups of animals in experiment 2, normalised to the average ^{134}Cs administration rate (Bq day^{-1}). *Dots* represent observed values. One-sided standard deviations are shown for two groups of animals. *Curves* represent fits by the model to the average of all animals (with 95% confidence intervals)

Table 4 Estimated fractional absorption of radiocaesium in RBC and plasma or excretion with urine and faeces (*a*), and biological half-lives ($T_{1/2}$), in the model for ^{134}Cs absorption and excretion in reindeer on lichen diet (Eq. 3). Mean±SE, and model R^2

Compartment	a, 10^{-3}	$T_{1/2}$ (days)	R^2
RBC	7.2±0.2	17.8±0.7	0.988
Plasma	2.4±0.3	7.0±0.9	0.72
Urine	31±1	11.4±0.5	0.97
Faeces	47±3	5.1±0.4	0.87

Experiment 2

The body masses of the animals (Table 2) increased on average by 2.5±2.4 kg. Urine production in groups 2, 3 and 4 was significantly higher than in the four control animals (P=0.001) and significantly different between the treatment groups (P=0.001), but was not related to the ^{134}Cs administration rates. There were no significant differences between the treatment groups in faecal production.

Analysis of faeces and urine from the control animals in experiment 2 during the first 9 weeks indicated ^{134}Cs concentrations ca. 0.5%–2% of those for treatment animals at equilibrium. No correction for background values was therefore found to be necessary. Figure 2 shows the observed ^{134}Cs activity concentrations in RBC and plasma, and the daily excretion of ^{134}Cs with urine and faeces, all normalised to the average daily ^{134}Cs intake. Two-way analyses of variance showed significant differences between the groups (P<0.01). Normalised activity concentrations in RBC and plasma in group 4 were higher than in the other groups, while the excretion of ^{134}Cs with urine (and urine production) was lowest in these animals. Group 3 animals had significantly higher ^{134}Cs concentrations in plasma, and lower urinary and faecal excretion, than groups 1 and 2 (despite higher urinary production in group 3 compared to group 1, and comparable urinary production to group 2). These differences did not appear to be related to the ^{134}Cs administration rates. Since the lichen diet provided a low mineral intake, it was assumed that individual differences in feed and water intake caused the differences in activity concentrations and excretion observed between the groups. Thus, Table 4 presents the results derived on the basis of average ^{134}Cs concentration and excretion data for all animals.

Non-linear regression analyses of RBC data resulted in estimates for a for individual animals ranging from 0.0060±0.0001 l^{-1} to 0.0087±0.0004 l^{-1}, while estimated $T_{1/2}$ values ranged from 13.2±0.8 to 24±1 days. The individual transfer coefficients calculated were independent

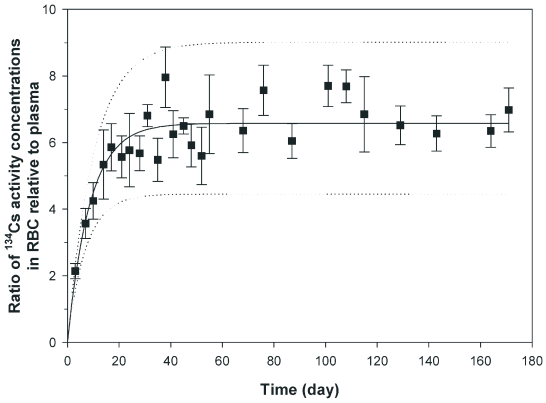

Fig. 3 Ratio between ^{134}Cs activity concentration in RBC and plasma. *Dots* show average and standard deviation for all 8 animals. The curve is the fitted non-linear regression model in Eq. 3, with upper and lower 95% confidence intervals

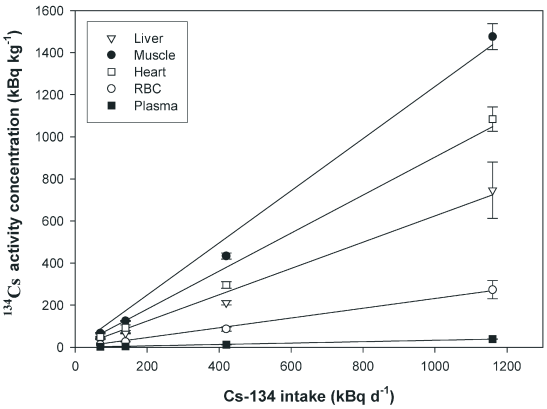

Fig. 4 Examples of activity concentrations in different tissues of reindeer calves slaughtered after being fed constant amounts of ^{134}Cs for 171 days. Average values ±SD and linear regression lines. The values for muscle are averages for the four muscle groups sampled

of estimated a values, but significantly correlated to the $T_{1/2}$ (R^2=0.52).

Figure 3 shows the ratio between observed radiocaesium activity concentrations in RBC and plasma, together with estimated curves using Eq. 3. A period of 40–50 days was necessary for equilibrium to be established between activity concentrations in RBC and plasma.

Estimated apparent absorption coefficients (A_a) for the individual animals ranged from 0.58 to 0.74 (average of observations from day 143 to 171 for each animal), with a total average of 0.67 for ^{134}CsCl sprayed on lichen, while the model results in Table 4 suggest an A_a of 0.65.

Figure 4 shows ^{134}Cs activity concentrations determined in some tissues of the calves at the end of the feeding experiment, and indicates linearity between the amount of ^{134}Cs fed and taken up in different tissues. Figure 5 presents the determined ^{134}Cs activity concentrations in all sampled tissues, normalised to daily ^{134}Cs intake. As mentioned in connection with RBC and plasma results, and as indicated for muscle, heart and liver in Fig. 4, samples of animals from group 4 had generally higher normalised ^{134}Cs values than the other groups.

Additionally, Fig. 5 shows transfer coefficient values for the different tissues, assuming that equilibrium ^{134}Cs activity concentrations were attained after 171 days of feeding. For the different muscle groups sampled, average F_f values ranged from 1.00±0.06 days kg^{-1} for triceps to 1.09±0.06 days kg^{-1} for neck. The average F_f for all muscle samples was 1.04±0.03 days kg^{-1}. Normalised activity concentration values in muscle were on average 5 times the concentration in RBC.

Discussion

Several authors have demonstrated the appropriateness of more than one exponential term in models for radiocaesium excretion (e.g., [19, 20, 27). Non-linear regression was also attempted using a model with two exponential terms, but was found applicable to only a few cases. Therefore the single exponential model in Eq. 3 was applied throughout to describe increases in radiocaesium activity concentrations and daily urinary and faecal radiocaesium excretion in the present experiments.

An inherent limitation of feeding experiments of this design, where constant daily administration rates to animals are used to study radionuclide kinetics, is that analyses using models such as Eq. 3 do not allow independent estimates of the fractional absorption (or excretion; a) and biological half-life ($T_{1/2}$) to be made. The parameter values estimated will be strongly correlated, and since they determine the transfer coefficient value (Eq. 4), their correlation may introduce variability in transfer coefficient estimates. Estimated fractional absorption factors a to RBC differed by a factor of 2.9 in the two experiments, while the difference between estimated transfer coefficients was a factor of 4. Furthermore, the individual transfer coefficients estimated for the 8 animals in experiment 2 were significantly correlated to the $T_{1/2}$ (R^2=0.52). Thus, acknowledging the influence of the biological half-life on the transfer coefficient values may help reduce the variability in transfer coefficient estimates.

Tissue distribution

The observed linearity between daily ^{134}Cs intake and activity concentrations in tissues in experiment 2 showed that there was no saturation of Cs absorption mechanisms in the alimentary tract or other tissues. The highest administration rate of 1,160 kBq day^{-1} corresponded to a Cs mass of about 0.024 μg, which is comparable to the stable

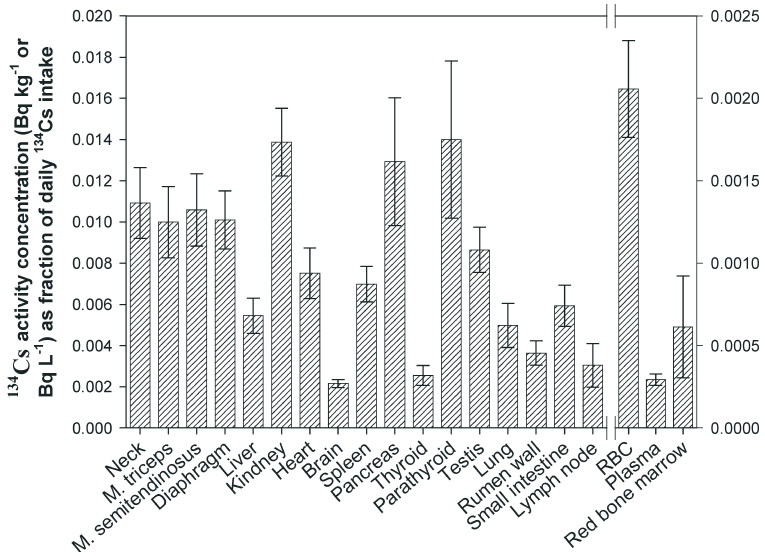

Fig. 5 Activity concentrations of ^{134}Cs in reindeer tissues at the end of experiment 2 normalised to the ^{134}Cs administration rate (mean ±SD of all 8 animals). RBC, plasma and red bone marrow concentrations are given on the right axis. Neck muscle samples were taken dorsal to the spine at the level of the vertebra 3–5

Cs content in 0.01–1 g vegetation [28, 29]. The maximum ^{134}Cs activity concentration in kidney tissue of 1,900 kBq kg^{-1} corresponded to 0.039 μg kg^{-1}. In comparison, stable Cs concentrations in ewes and lambs of 140–200 μg kg^{-1} have been reported [30]. There were some differences in absorption and excretion of ^{134}Cs between the groups of animals in experiment 2, but these were thought to be due to individual differences in feed and water intake since no systematic pattern related to administration rate was evident.

Caesium is taken up from the gastrointestinal tract and distributed to and between the various tissues by the blood. Absorption and excretion rates vary in different tissues in the body, and radiocaesium concentrations in muscle relative to other tissues will change relatively slowly when intake changes (e.g. [31]). Relative radiocaesium concentrations in tissues in a continuous feeding experiment may therefore not be comparable to quasi-equilibrium activity concentrations obtained during variable conditions of natural grazing. Nevertheless, the tissue distribution of radiocaesium in the present study was in general agreement with the results of Åhman [31] and Rissanen et al. [32]. Higher concentrations were reported in [32] for kidney, brain and thyroid tissue relative to muscle, while relatively low concentrations in liver, kidney and heart and high concentration in rumen wall were reported in [31] for grazing reindeer. Our study provided results on testis, parathyroid and the small intestine, e.g. showing that radiocaesium accumulation in testes, one of the most radiosensitive tissues, was similar to muscle.

The ratios between radiocaesium activity concentrations in RBC and muscle in the present experiments at the time of slaughter were similar (0.18–0.20), and comparable to those reported by Åhman [33] (0.17±0.04) and Eikelmann [34] (ratio RBC:muscle of 0.21 and 0.24 for calves and females, respectively). Furthermore, the ratio between concentrations in plasma and muscle in experiment 1 (0.028) was comparable to that in [33] (0.024± 0.004).

Absorption and excretion

The estimated values for the fractional absorption or excretion (a) and the biological half-life ($T_{1/2}$) presented in Tables 3 and 4 indicate that there were significant differences in absorption and excretion of radiocaesium between reindeer in experiments 1 and 2. Reindeer given a pure lichen diet with ^{134}CsCl had 40% longer $T_{1/2}$ and 2.9 times higher a values for RBC than reindeer feeding on fallout contaminated lichen and concentrates. The pure lichen diet also resulted in 60% longer $T_{1/2}$ and 4.8 times higher a values for excretion of radiocaesium with urine, and 40% longer $T_{1/2}$ for excretion of radiocaesium with faeces, but the diet in experiment 1 resulted in 3.5 times higher a values for faecal excretion. As expected, the difference in $T_{1/2}$ in RBC was less in the current experiment than in [26] (i.e., 72%) where the potassium supplementation simulated the high intake in spring. Although not directly comparable, the half-life estimates in the current experiment appear to be in agreement with those by Holleman and Luick [35, 36], but relatively short compared to those by Hove et al.[23] and Birke et al. [26]. The discrepancies may be due to methodological differences (e.g., model parameterisation, length of experimental periods, number of animals and data transformation).

Estimates of apparent absorption (A_a) of radiocaesium in the experiments gave values of 0.17 and 0.65–0.67 for experiments 1 and 2 respectively, indicating that

the availability for absorption of Chernobyl fallout ^{137}Cs was on average about 25–26% that of ^{134}CsCl sprayed on lichen. Another estimate of bioavailability can be derived from the estimated values of fractional absorption in RBC. The a value for fallout ^{137}Cs in lichen was 35% of that for ^{134}CsCl. The difference between 25–26% and 35% demonstrates the dependency of absorption and bioavailability estimates on the biological half-life (Eqs. 3 and 4). Correspondingly, the earlier estimated "absorption factors" of 20–30% for nuclear weapons test fallout ^{137}Cs in lichen [19, 36] may appear higher than the A_a in experiment 1 due to longer $T_{1/2}$ in those studies (due to lower potassium intake). The estimated bioavailability of Chernobyl radiocaesium in lichen collected in 1988 of 35% of that for ^{134}CsCl is in agreement with other studies of radiocaesium bioavailability in lichen different years after the Chernobyl fallout, indicating that the physical-chemical forms of the fallout intercepted in lichen does not change significantly with time [37]. This is in contrast to plants, in which the radiocaesium bioavailability increased the first years after deposition (e.g., [20]).

Conclusions

1. The model and parameterisation applied in this study demonstrate the importance of considering the influence of the biological half-life on transfer coefficient estimates, as this may aid the identification of sources of variability in experimental results.
2. Both diet composition and radiocaesium source contributed to the activity concentrations of the tissues of reindeer calves following prolonged feeding with feeds contaminated by radiocaesium. The half-life for radiocaesium accumulation in RBC was about 40% longer with the pure lichen diet, while corresponding differences in the half-lives for urinary and faecal excretion were about 60% and 40%, respectively. Transfer coefficients (F_f) to reindeer meat were estimated to be 0.25±0.01 and 1.04±0.03 days kg^{-1} for fallout radiocaesium and ^{134}CsCl, respectively.
3. Assessed on the basis of fractional absorption of radiocaesium in RBC it was concluded that the bioavailability of the Chernobyl radiocaesium fallout in lichen in 1988 was about 35% of that of ^{134}CsCl, and similar to estimates of radiocaesium availability in lichen both for the nuclear weapons test and Chernobyl fallout in the literature.

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ERRATUM

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Absorption, retention and tissue distribution of radiocaesium in reindeer: effects of diet and radiocaesium source

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The scale of the vertical axis in Fig. 5 is too low by a factor of 100. The correct scale is given here.

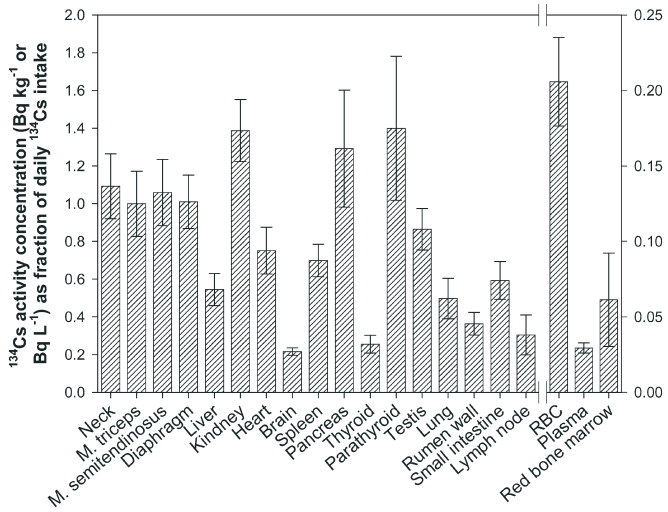

Fig. 5 Activity concentrations of ^{134}Cs in reindeer tissues at the end of experiment 2 normalised to the ^{134}Cs administration rate (mean ±SD of all 8 animals). RBC, plasma and red bone marrow concentrations are given on the right axis. Neck muscle samples were taken dorsal to the spine at the level of the vertebra 3–5

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Paper II

Review paper: Physiological parameters that affect the transfer of radiocaesium to ruminants

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REVIEW

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Physiological parameters that affect the transfer of radiocaesium to ruminants

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Abstract Recently there has been a renewed interest in biological scaling relationships between parameters, such as those between, for example, body mass, dry matter intake and biological half-times of radionuclides that are useful in predicting the transfer of radiocaesium to different animal species, particularly to wild animals. However, there is still a considerable unexplained variability in transfer coefficient estimates between individuals of the same species. This paper discusses the physiological parameters that affect the transfer of radiocaesium to ruminants, and it shows how a better understanding of these parameters may help to reduce the within-species variability in radiocaesium transfer coefficients. In light of the improved understanding during the past 10–15 years of the importance of source-dependent bioavailability on absorption of radiocaesium from the gastrointestinal tract, it is concluded that further studies are required on the effects of feed digestibility and physiological factors on absorption and endogenous faecal excretion of radiocaesium to better understand the variability in transfer coefficients.

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Introduction

Uptake of radiocaesium and other radionuclides by animals from feed varies according to species, size, age and growth rate, and according to the digestibility of the feed and, in the case of lactating species, the milk yield (see, e.g. [1]). Nevertheless, general relationships between radiocaesium turnover expressed by biological half-times, and animal mass have been found both between and within species (see, e.g. [2–4]). Similarly, a relationship between radiocaesium transfer coefficients and body mass has been reported [5]. However, animals of similar mass, even of the same species, may exhibit pronounced differences in metabolism due to factors like growth rate, lactation and season—e.g. in reindeer [6]. The influence of such factors on radiocaesium metabolism and transfer will not be adequately accounted for when only using body mass as the explanatory or predicting variable.

To account for physiological factors such as growth rate and lactation, dry matter intake was suggested as a more generally applicable parameter to predict radiocaesium transfer [7]. This is in agreement with Holleman et al. [4] who stated that "animals, feeding on forage of uniform potassium concentration, should exhibit caesium kinetics proportional to feed intake or metabolic rate", since potassium intake apparently had a direct influence on caesium retention by reindeer in their experiment. Presumably, feed intake, which varies with physiological needs, will reflect sources of variability in radiocaesium transfer not accounted for by body mass alone.

Recently, the use of biological scaling or "allometric" relationships to predict the transfer of radionuclides to different wild animal species has come to the fore (see review by Beresford et al. [8]). On the basis of such relationships, it is suggested that radiocaesium concentration ratios (i.e., the ratio between whole body and dietary radiocaesium concentrations) in different animal species are similar [9]. This paper focuses on the transfer

coefficient concept and on the within-species variability, discussing how different physiological parameters influence radiocaesium uptake as well as excretion in animals (particularly ruminants). It is pointed out how a better understanding of these parameters may reduce the unexplained within-species variability in observed radiocaesium transfer coefficients.

Mathematical description of radionuclide transfer

Transfer of radionuclides from feed to animal products is commonly described using transfer coefficients (F). Transfer coefficients are defined as the amount of the animal's daily radionuclide intake that is transferred to 1 kg of the animal product at equilibrium [10]. F takes the subscripts f or m when referring to meat (flesh) or milk, respectively. Assuming first order kinetics, and using the notation in [10], an expression for the transfer coefficient can be derived from the equilibrium condition where uptake equals excretion:

$$a \cdot I = k \cdot q$$

where a is the fractional absorption (also called assimilation fraction [11]; kg^{-1}), I is the intake rate for the animal (Bq d^{-1}) , k the effective biological rate constant (loss from tissue, compartment or whole body; d^{-1}), and q the activity concentration of radionuclide in tissue or compartment (Bq kg^{-1}).[1] The transfer coefficient is given by:

$$F = \frac{q}{I} = \frac{a}{k}$$

Expressing the biological rate constant by the biological half-time gives:

$$F = \frac{a \cdot T_{1/2}}{\ln 2}. \quad (1)$$

Equation 1 explicitly shows the influence of $T_{1/2}$ on F. The same expression may be derived for an animal tissue or compartment whose radionuclide retention can be described by first order kinetics:

$$q(t) = q_0 \cdot e^{-\ln 2/T_{1/2} \cdot t}$$

where q_0 is the initial activity concentration in a tissue. With $q_0 = 0$, a constant daily intake I of a radionuclide, of which a constant fraction a is absorbed from the gastrointestinal tract, the increasing activity concentration in the tissue can be described by:

$$q(t) = I \cdot a \cdot \int_0^t e^{-\ln 2/T_{1/2} \cdot t} dt$$

$$= \frac{I \cdot a \cdot T_{1/2}}{\ln 2} \cdot \left(1 - e^{-\ln 2/T_{1/2} \cdot t}\right)$$

As time t approaches infinity, the equilibrium activity concentration approaches:

$$q_{eq} = \frac{I \cdot a \cdot T_{1/2}}{\ln 2}$$

Normalised to the daily intake I, this gives the expression for the transfer coefficient in Eq. 1 above. Similarly, when second order kinetics are more appropriate, the transfer coefficient will be given by:

$$F = \frac{a}{\ln 2}(c_1 \cdot T_1 + c_2 \cdot T_2)$$

where c_1 is the fraction of the absorbed activity being excreted with a half-time T_1 and similarly for c_2 and T_2.

Fractional absorption (a) equals the gastrointestinal absorption fraction (f_1) when considering the absorption of radiocaesium from the gastrointestinal tract per se. When studying the transfer of radiocaesium to a specific compartment, the fractional absorption (a) is the product of the gastrointestinal absorption fraction (f_1) and the fraction transported to the specific compartment (f_2) [10]:

$$a = f_1 \cdot f_2 \quad (2)$$

Absorption from the gastrointestinal tract

Absorption from the gastrointestinal (GI) tract has often been expressed as the difference between dietary radionuclide intake and faecal excretion relative to dietary intake ("apparent absorption coefficient"; A_a). However, circulation of absorbed caesium between the GI tract and body fluids [12] results in the endogenous excretion of absorbed radiocaesium from blood to faeces, which is not accounted for by apparent absorption. Therefore, methods for determining "true absorption coefficients" (A_t) have been developed (see, e.g. [13]), which additionally gives a better measure of radiocaesium bioavailability than apparent absorption [14]. True absorption coefficients have been important tools in improving our understanding of the influence of source-dependent bioavailability on transfer of ingested radionuclides to ruminant-derived food products during the last 10–15 years [14]. According to Howard and Beresford [15], the endogenous secretion of radiocaesium into the gut of ruminants is typically 20–25% of the total absorbed radiocaesium, but this can be affected by a number of dietary and physiological factors [16]. Mayes et al. [13] found that A_t for CsCl was similar for sheep and cattle and therefore concluded that the tenfold

[1] Ward and Johnson [10] discussed the caesium transfer coefficient from feed to milk (F_m). They denoted q the activity in milk, and divided it by the mass M of milk. For the discussion of transfer to meat (F_f) in the context of this mathematical derivation, the variation in meat mass has been ignored.

difference in transfer coefficients for sheep and cattle was not due to differences in absorption. Howard and Beresford [15] concluded that A_t would not be expected to vary between different ruminant species, but, as they stated, the limited available data did not unequivocally support this hypothesis.

Studies with sheep indicated that although radiocaesium absorption (A_t) did not vary with age, lactation, size or breed, A_t showed some variability between individuals and might be reduced with low digestibility diets [14]. For instance, lower A_t was found for radiocaesium in heather than for radiocaesium in grass, while A_a values remained relatively constant [14], and a higher crude fibre content in the diet of cows (lower digestibility) resulted in a substantial decrease in radiocaesium transfer coefficients [17, 18]. Furthermore, significantly decreased apparent absorption (A_a) of radiocaesium in milk (and relatively constant A_t) was observed when the lambs' diet was changed from pure milk to mixed milk and herbage diet, which was ascribed to a substantial increase in endogenous faecal excretion of radiocaesium with the increased faecal dry matter output [19]. In the same study, A_a decreased with age, due to increased endogenous faecal excretion, while there was no change in A_t with age [19].

Furthermore, some higher apparent absorption of radiocaesium due to increased metabolic activity has been inferred from studies of increased milk production [10] and exercise [20]. Both studies concluded that metabolism and exercise had no effect on transfer coefficients, while $T_{1/2}$ was reduced (e.g. 13% due to exercise [20]).

Accumulation in different body compartments

In regard to the accumulation of radiocaesium in different body compartments (factor f_2 in Eq. 2), the study by Assimakopoulos et al. [21] is worth mentioning. In this study, radiocaesium transfer coefficients were compared in different tissues of cows, sheep and goats, normalised to the transfer coefficient to blood, on the basis of results of different feeding experiments. No significant differences in transfer coefficients to different tissues were detected between animals or due to physiological status (i.e. whether the animal was lactating or not), except the kidneys [21]. By normalising transfer coefficients for different compartments to that of blood, these results showed that f_2 is similar for most of the tissues studied in the selected animal species.

Transfer coefficients and biological half-times

Several studies on radiocaesium transfer and biological half-times have been carried out (see, e.g. [2–5]). In a study on variation in individual metabolism of radiocaesium in adult female sheep, a correlation coefficient of 0.75 between F_f and $T_{1/2}$ was found [22] (a correlation approaching 1.00 would indicate that the fractional absorption a was constant, cf. Eq. 1). A stepwise regression analysis of other possible factors influencing the observed F_f revealed that:

- Body mass change was the best regressor ($R^2 = 0.46$). The correlation coefficient was −0.68, i.e. ewes with larger body mass change had lower F_f.
- Feed intake was the second best regressor, and together with body mass change gave $R^2 = 0.72$. The correlation between F_f and feed intake was −0.65, i.e. ewes with larger intake had lower accumulation. The correlation between body mass change and dietary intake was not given, but would have been expected to be positive.

When $T_{1/2}$ was added to the regression model containing body mass change and feed intake, R^2 increased to 0.92. The half-times were similarly correlated to body mass change as F_f (correlation coefficient = −0.68). In a separate experiment with the same animals, F_f was positively correlated to individual values of A_t ($R = 0.57$; see Eq. 1), while there was no significant correlation between A_t and $T_{1/2}$ [22].

Differences in transfer to individual ewes could not be explained by differences in body mass in this study [22], while a significant correlation between transfer and body mass was found in a later study with lambs [7]. Potential reasons for the lack of correlation in the study with ewes [22] are

- Limited variability in body mass among the animals studied. A correlation between $T_{1/2}$ and body mass was reported for reindeer weighing 48–112 kg, but the results also show a pronounced variability in $T_{1/2}$ for animals of similar body mass [4]. Thus, a relatively wide range of body masses may be needed to separate the effect of body mass from individual variability in metabolic rates in animals of similar mass.
- Potential effects of individual body masses were masked by differences in metabolic rates related to differences in growth rates during the experimental period.

In experiments where reindeer calves were given radiocaesium in different diets, a fourfold difference was found in F_f and transfer coefficients to red blood cells [23]. The majority of the difference in transfer coefficients could be accounted for by a 2.9-fold difference in radiocaesium bioavailability (expressed by fractional absorption, a), while differences in $T_{1/2}$ values (1.4-fold difference) accounted for the remainder. These results demonstrate the usefulness of parameterising the transfer coefficient (Eq. 1) when studying reasons for its variability.

A study on lambs indicated a possible link between protein and radiocaesium turnover rates [7]. Protein turnover rates are influenced by thyroid hormones, and earlier studies have shown increased radiocaesium

excretion with increased levels of thyroid hormones in rats [24–26]. The effect of thyroid hormones on radiocaesium turnover may be due to increased concentration and activity of Na^+K^+ ATPase following a thyroid hormone–induced elevation of K^+ efflux (and Na^+ influx) from cells [27]. A study on the effect of thyroxin in suckling rats showed that increased metabolism reduced radiocaesium retention without being secondary to higher fibre intake [26]. Furthermore, since a fraction of intracellular radiocaesium appears to be bound to cell proteins (cf. [28]), increased protein turnover may potentially increase mobilisation and excretion of this bound fraction.

Summary and conclusion

The use of transfer coefficients to describe the transfer of radiocaesium to cows' milk was introduced as this parameter significantly reduced the variation between individual animals, as compared to expressing total milk ^{137}Cs activity as a percentage of intake [10]. Transfer coefficient values provide rough estimates of activity concentrations in animal products from estimates of dietary intake, particularly when estimates of absorption and biological half-time are unavailable. However, it has been questioned whether the transfer coefficient is the most robust parameter or not [9, 15]. For instance, the many physiological factors involved in determining the magnitude of the transfer coefficients, and the variability in each and every one of these factors, will lead to variability in estimated transfer coefficients.

In general the transfer coefficient can be described by:

$$F = \frac{(A_t - E_f) \cdot f_2 \cdot T_{1/2}}{\ln(2)}$$

A_t, the true absorption, is a function of radionuclide bioavailability (physical–chemical forms of the radionuclide) and feed digestibility. A_t increases with higher bioavailability and, for radiocaesium, decreases with lower feed digestibility. Average A_t values reported for vegetation typically range from 0.65–0.88, with a low value of 0.23 reported for pasture grass of low digestibility [14].

E_f, the endogenous faecal excretion (the difference between true and apparent absorption), increases with higher feed intake and lower digestibility (both contributing to high flux of undigested fibre in the GI tract), and is apparently reduced in physiological states involving elevated metabolic activity (e.g. milking and exercise). $(A_t - E_f) = A_a = f_1$ in Eq. 2. To our knowledge, no previous study has attempted to quantify the effects of digestibility and physiological factors on E_f. Differences in A_t and A_a estimates in sheep suggest that E_f values range from 18% of that absorbed from clover hay to 34% with a grass hay diet [13]. A twofold reduction in the transfer coefficient for radiocaesium to cows' milk due to higher fibre intake has been reported [17], but this reduction may be due to reduced A_t in addition to increased E_f.

f_2 is the fraction of absorbed radiocaesium transported to the body compartment of interest.

$T_{1/2}$, the biological half-time, is correlated to metabolic rates (e.g. body mass, lactation and exercise); in other words, $T_{1/2}$ is reduced by higher metabolic activity.

As a result of the relationships between these factors, a linear relationship between the transfer coefficient and biological half-time would not be expected (since both E_f and $T_{1/2}$ are influenced by metabolic activity). Furthermore, since many of the factors that have been studied will be expected to be correlated, the effects of partial correlation between the variables must be taken into account in an appropriate manner.

Dry matter intake has been suggested to be a parameter which could potentially be used as a 'generic' transfer coefficient determinant [7]. As such, feed intake would have to be corrected for digestibility, and the crude fibre content of the faeces would probably be more important than crude fibre intake, because of differences in digestibility [17]. Intake of fodder will vary with physiological needs, and any intake of feed will per se result in increased Na^+K^+ ATPase activity and metabolic rates (e.g. [6, 7]).

Our understanding of the importance of bioavailability on radiocaesium absorption in the GI tract has been considerably improved during the past 10–15 years. Further studies on the effects of feed digestibility and physiological factors on absorption and endogenous faecal excretion of radiocaesium are required to better understand the variability in transfer coefficient estimates within animal species. Indeed, feed digestibility was identified as an important factor back in the 1960s. In light of the improvements in the understanding of bioavailability and absorption, feed digestibility should be given more attention than has been the case over the past 40 years, due to its effect on both adsorption and endogenous excretion. However, the effects of other physiological factors will need to be duly considered in the same studies.

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Paper III

Transfer of ^{85}Sr and ^{134}Cs from diet to reindeer foetuses and milk

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Transfer of ^{85}Sr and ^{134}Cs from diet to reindeer foetuses and milk

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Abstract

Sr-85 and ^{134}Cs in aqueous solution of the chlorides were administered daily to four pregnant reindeer during the last part of gestation. Radionuclide concentrations were determined in calves sacrificed at birth, and secretion of the nuclides was measured in milk. Although gastrointestinal absorption of ^{85}Sr was low, an apparently higher transfer of the absorbed fraction of ^{85}Sr than ^{134}Cs from the mother to the foetus led to similar accumulation of ^{85}Sr and ^{134}Cs in foetuses. At birth 1.4 – 1.6 and 1.5 – 2.5 % of the total administered activities of ^{85}Sr and ^{134}Cs respectively were present in the calves' bodies. Transfer coefficients (F_m) for ^{85}Sr and ^{134}Cs from feed to milk were estimated at 0.0218 ± 0.0026 and 0.185 ± 0.025 day kg^{-1}, respectively, and the observed ratio ($OR_{milk\text{-}diet}$) for ^{85}Sr was 0.124 ± 0.037. Transfer of radiostrontium to reindeer milk was in agreement with previously reported relationships between Ca intake and radiostrontium transfer in ruminants. These relationships suggest that transfer of radiostrontium to foetuses and milk of free-ranging reindeer can be 3 – 4 times higher than observed in this experiment (due to lower Ca intake with natural forage), but the transfer to milk will not be as high as that of ionic ^{134}Cs. The concentrations of ^{85}Sr in milk suggested that the does mobilized skeletal stores of Ca and ^{85}Sr for milk production, although the diet appeared to satisfy the Ca requirements. In reindeer with radiostrontium intake during the whole year, radiostrontium concentrations in milk will therefore be higher than indicated by the F_m value observed in our study. No differences in half-times for ^{85}Sr and ^{134}Cs secretion in milk were detected. Both nuclides were secreted with short- and long-term half-times of 1 – 2 and 12 – 19 days, respectively. For ^{85}Sr, 80 – 90 % of the activity was excreted with the short half-time, whereas the corresponding figure for ^{134}Cs was 30 – 50 %.

Introduction

Reindeer and caribou (*Rangifer tarandus*) are known to accumulate both anthropogenic and natural radionuclides via various pathways in the environment [1-7]. The fallout in Scandinavia from the 1986 Chernobyl accident introduced the need for developing measures to reduce radiocaesium concentrations in reindeer before slaughter (e.g. [8]), and some 18 years after the 1986 Chernobyl accident, ^{137}Cs concentrations in reindeer in some districts of Norway continue to exceed the intervention limit of 3000 Bq kg^{-1} in Norway [6].

Skogland et al. [9,10] attributed a decrease in reproduction in the Rondane wild reindeer herd after the Chernobyl accident to in utero and postpartum exposure to the Chernobyl fallout. Although this finding was rebutted by others [11], such speculation demonstrates the need for experimental data on the transfer of radionuclides from the diet to foetuses and milk of reindeer. The placenta does not present a barrier to the transfer of Cs to the foetus, whereas Sr is discriminated in favour of Ca during transport across the placenta and other biological membranes [12].

Reindeer milk was a traditionally important Saami food product that was lost as part of the transition from a subsistence household-based economy to more extensive herding and cash economy [13]. Milking continued in some parts of Fennoscandia up to the early 1960s, and is presently practised only in the border areas between Russia, China and Mongolia [13]. Recently there has been a renewed interest in reindeer milk as a niche product in Fennoscandia [13]. Compared to milk from domestic ruminants, reindeer milk is high in fat and protein, with a moderate to high mineral element content (1 – 1.5 % [13]). Daily milk yields of 1 – 1.5 l day^{-1} together with the mineral element content appear to indicate that transfer of radiostrontium and radiocaesium per litre of reindeer milk will be higher than in milk of many other ruminants. Observed radiocaesium concentrations in reindeer milk in the 1960s did indicate high transfer (e.g., [14]). Holleman et al. [15] conducted a study on the transfer of radiocaesium from reindeer does to suckling calves, but quantification of the radiocaesium secretion in milk is not possible from their paper. It has been suggested that the higher radiocaesium concentrations in reindeer calves than adults in early autumn the first years after the Chernobyl accident (see for example [16]) was partly due to intake of highly bioavailable radiocaesium with milk [17]. No studies on secretion of Sr in reindeer milk have been published.

This paper presents a detailed study on the transfer of radiostrontium and radiocaesium from feed to reindeer foetuses and milk. Pregnant reindeer does were given ^{85}Sr and ^{134}Cs daily during the last part of their pregnancy. Their calves were sacrificed at birth, and the does subsequently milked. The results on ^{85}Sr secretion in milk were used to test the generic relationship between Ca intake and Sr transfer to milk of ruminants suggested by Howard et al. [18] and Beresford et al. [19].

Materials and methods

Sr-85 and ^{134}Cs were administered in daily doses of 289 kBq ^{85}Sr and 7.6 kBq ^{134}Cs to four semi-domesticated pregnant reindeer does (A – D) during the last 25 – 60 days of gestation (from 30 March 2001), and for 10 days postpartum. The does belonged to the experimental herd of the Norwegian University of Life Sciences (UMB), and were housed in metabolism cages for about 2 months prior to the experiment and until parturition, allowing individual

feeding and collection of urine and faeces. The animals were therefore accustomed to the experimental conditions. The does were offered a diet consisting of 500 g of lichens (fresh mass, FM) and pelleted concentrate (Rensfoder hög, Linkøping, Sweden, containing 14.8 % crude protein and 12.2 g K, 8.4 g Ca, 5.5 g P and 2.4 g Na kg^{-1}). All the lichen feed offered was consumed. Any remaining concentrates were collected every morning and weighed. All procedures relating to animal experimentation were according to Norwegian regulations for animal ethics.

Both radionuclides (obtained from standard solutions) were sprayed as chloride solutions (in 0.5 ml portions) on approximately 5 g of dry lichen. The contaminated lichen was administered to the animals in the morning after cleaning the feed troughs, and consumed immediately. Blood was taken from the jugular vein every 2 – 5 day using sodium heparin as an anticoagulant. Blood samples were centrifuged for 20 min at 3000 rpm, and plasma and red blood cells (RBC) were separated for determination of radionuclide activities (in approximately 5 ml of each). Urine and faeces were collected every 3 – 4 day. The urine and faeces produced during each sampling interval was weighed and mixed separately. However, urine and faeces were collected in the same container below the metabolism cage, and cross contamination between the two was therefore possible. Sub samples of 200 g FM faeces and 20 ml urine were taken for determination of ^{85}Sr and ^{134}Cs activities. Faeces were dried at 100 ºC for 24 hours and homogenized.

The does calved during the period 24 April – 28 May. Doe D gave birth to calf D after 25 days of radionuclide administration, calf C was born after 31 days, and calf B and A after 49 and 60 days, respectively. Calf B was female, the others were males. After parturition the does were held in a 0.4 ha outdoor pen and had free access to water, hay and pelleted concentrate. Milking was carried out as described by Gjøstein et al. [20] for 2 – 3 months, initially twice daily, later once daily. The contaminated lichen was administered during milking, and consumed immediately. Oxytocin was injected to elicit milk let-down, and was not expected to influence the chemical composition of the milk [21]. At intervals of 5 – 22 days saline water was used as placebo for oxytocin or the does received no injections. Concentrations of ^{85}Sr and ^{134}Cs were determined in all milk samples obtained during the first 16 – 22 days postpartum, and about every 5 days thereafter. Calcium concentrations was determined in milk sampled at days 3 and 10, which represented days of approximate maximum ^{85}Sr concentrations and the end of ^{85}Sr administration respectively. Moreover Ca concentrations were determined in milk sampled at day 22 or 24 for comparison with literature data [22].

The calves were sacrificed at birth, after sampling of blood. The tongue, lungs, heart, liver, kidneys, spleen, rumen, abomasum, small intestine, large intestine and gonads were removed for analysis, and samples of 3 muscles (M. quadriceps, M. longissimus dorsi and M. triceps), skin, metatarsus, metacarpus, mandible and teeth were also obtained. The remainder of the body was ground and five samples of each of the ground bodies were used to determine radionuclide activities in the remnants of the body. Together with the tissue samples this allowed estimates of total radionuclide transfer to the calves. All tissue samples from the sacrificed calves were dried to constant mass at 70 ºC and homogenized.

Concentrations of ^{85}Sr and ^{134}Cs were determined in UMB's Isotope Laboratory using a NaI(Tl) detector (faeces, urine and milk), at the Norwegian Radiation Protection Authority using HPGe detectors (plasma, RBC, milk and calf tissues), and by the Dept. of Radiation Physics at Lund University Hospital (Sweden) using HPGe detectors (calf tissues). A number of samples were analysed at different laboratories for inter-comparison.

All analytical results were decay corrected to the start of the experiment. Calcium concentrations in milk were determined using flame atomic absorption spectrometry at the UMB Dept. of Animal and Aquacultural Sciences.

Statistical analysis was carried out using SPSS for Windows (release 11). All data are presented as mean ± standard deviation (SD), unless otherwise specified. When *P*-values are not given, a significance level of 0.05 is understood. The estimated average daily urinary and faecal excretion was allocated to the middle day in the sample interval (day 2 when the interval was 3 days). The biological half-time of ^{134}Cs in RBC of pregnant does was estimated using a non-linear regression model:

$$A_{RBC}(t) = \frac{a \cdot T_{1/2}}{\ln 2} \cdot (1 - e^{-\ln 2 / T_{1/2} \cdot t}) \quad (1)$$

where $A_{RBC}(t)$ is the activity concentration in RBC, a is a measure of the fractional absorption of ^{134}Cs into RBC and $T_{1/2}$ is the biological half-time [23]. Biological half-times for ^{85}Sr and ^{134}Cs concentrations in milk after terminated radionuclide administration were estimated using an exponential model:

$$A_{milk}(t) = A_0 \cdot (f_1 \cdot e^{-\ln 2 / T_1 \cdot t} + f_2 \cdot e^{-\ln 2 / T_2 \cdot t}) \quad (2)$$

with one or two exponential terms. A_0 is the activity concentration when radionuclide administration was terminated and f_1 and f_2 express the fractions of this concentration decreasing with half-times T_1 and T_2 respectively. Log transformation of nuclide concentrations in RBC and milk and of regression models was used to harmonize variances in all regression analyses.

The transfer of the radionuclides from feed to milk was quantified using the transfer coefficient (F_m), which is defined as the amount of the animal's daily radionuclide intake that is transferred to one kilogram of the animal product at equilibrium [24]. The discrimination in transfer of Sr from the diet to milk compared with that of Ca was quantified using the observed ratio (*OR*) [25], given by:

$$OR_{milk-diet} = \frac{[Sr]_{milk} / [Ca]_{milk}}{[Sr]_{diet} / [Ca]_{diet}} \quad (3)$$

Results

Feed intake, urine and faeces production and radionuclide excretion

Does A and C had similar intakes of concentrates (Table 1), significantly lower than that of B and D (t-test, $P<0.001$). Correspondingly, doe D produced significantly more faeces than did A and C, and B more faeces than A. In addition, urine production by doe D was higher than by doe A (t-test, $P<0.05$).

Daily urinary and faecal excretion of the radionuclides was variable (Fig. 1), due to varying production of urine and faeces and variable radionuclide concentrations. For instance, the low faecal nuclide excretion by doe B on day 48 was due to low faeces production only, while the low urinary nuclide excretion by doe A on day 48 was due to a combination of low urine production and relatively low nuclide concentrations.

The observed faecal ^{85}Sr excretion shown in Fig. 1 indicated that no appreciable absorption of ^{85}Sr occurred in the gastrointestinal (GI) tract. The cumulative faecal excretions until the last day of faeces sampling were 96.0, 97.2, 97.8 and 98.9 % of the

administered ^{85}Sr activity for does A – D respectively. Due to the possibility of cross-contamination between urine and faeces, we cannot be conclusive on the extent of ^{85}Sr uptake in the GI tract on the basis of the observed ^{85}Sr content in urine.

^{85}Sr and ^{134}Cs in does' blood

There was no apparent systematic increase in ^{85}Sr concentrations in plasma following 6 – 7 days of administration (Fig. 2), but the concentrations were more variable than ^{134}Cs concentrations in RBC. Concentration of ^{134}Cs in RBC increased throughout the administration period, approaching an approximate equilibrium in does A and B at the end of their gestation.

By assuming that rates of radiocaesium absorption and retention did not change significantly during the final period of gestation, the data on ^{134}Cs in RBC (Fig. 2) were used to estimate biological half-times using Eq. 1. Short periods of ^{134}Cs administration (especially for doe C and D) gave significant uncertainties in the $T_{1/2}$ estimates (Table 2), and none of the $T_{1/2}$ estimates were significantly different from the others.

^{85}Sr and ^{134}Cs in foetuses

Concentrations of ^{85}Sr were highest in bone, and generally higher in tissues of calves A and B than the others (Table 3; paired t-tests, $P<0.05$), but the difference between calves was not linearly related to the total administered ^{85}Sr activity. Concentrations of ^{85}Sr in tissues relative to the total administered activity were lower in calf D than in the others (paired t-tests, $P<0.05$). Estimated total body contents of ^{85}Sr in the calves (Table 3) were 1.4 – 1.6 % of the total administered ^{85}Sr activity. Furthermore, the ratios between ^{85}Sr concentrations in plasma of calves and does A – D were 0.7, 0.5, 0.5, 0.3, respectively, i.e. lowest in the animals subjected to ^{85}Sr during the shortest period.

The ^{134}Cs concentrations in calf tissues increased from the first to the last-born calf (Table 3; paired t-tests, $P<0.05$), e.g. from an average muscle concentration of 6.0 Bq g^{-1} in calf D to 12.8 Bq g^{-1} in calf A. The increase in ^{134}Cs concentrations was lower than the increase in total administered activity to the does, as expected when concentrations reach equilibrium (cf. Fig. 2b). Estimated total body contents of ^{134}Cs in calves A – D were 1.6, 1.5, 1.6 and 2.5 % of the total administered ^{134}Cs activity, respectively. Concentrations of ^{134}Cs in RBC of calves A – D were 38, 7.3, 26 and 31 % higher than in does A – D, respectively (paired t-test, $P<0.05$).

^{85}Sr and ^{134}Cs in milk

For all does except A there was a rapid initial increase in ^{85}Sr concentrations to maximum values 1 – 2 days postpartum (Fig. 3). The concentrations in doe A's milk reached a maximum at the end of the period of radionuclide administration, at which time the milk production had nearly stabilized (Fig. 3). Following the increase, ^{85}Sr concentrations decreased in three phases: First there was a decrease during the period of continued ^{85}Sr administration in does B – D. Non-linear regression showed that half-times in this period were 39 ± 15 days in doe B, 8.54 ± 0.91 days in doe C and 14.5 ± 1.7 days in doe D. The two next phases followed the termination of radionuclide administration, when the activity

concentrations followed a double exponential decrease (Eq. 2) in all does: 72 – 87 % of the activity decreased with the short half-time of 0.86 – 2.8 days, while 13 – 28 % decreased with the longer half-time of 12.4 – 19.3 days (Table 4). The T_2 estimated for doe B and D were not significantly different from the half-time estimated during the period of continued ^{85}Sr administration (the variability in concentrations during the 8 – 9 days of continued administration made the estimated half-times during that period more uncertain).

The fraction of ^{85}Sr secreted in milk with the shorter half-time was used as an estimate of the fraction of ^{85}Sr transferred directly to the milk (and not via accretion in bone). The concentrations in milk at the end of nuclide administration (average of two last samples per doe) and the fractions of ^{85}Sr secreted with the short half-time (Table 4) suggested that 6290 ± 740 Bq kg^{-1} of the ^{85}Sr in milk was transferred directly from feed, corresponding to a transfer coefficient ($F_{m,Sr}$) of 0.0218 ± 0.0026 day kg^{-1}.

Individual feed intakes were not recorded while the does were held in the pen. However, from the total consumption of feed we estimated that the does had an average Ca intake corresponding to consumption of approximately 2 kg concentrate about one week postpartum, i.e., 16.8 ± 4.5 g Ca day^{-1} (the SD was derived from that of the individual intakes in Table 1). The average Ca concentration in milk was 2.95 ± 0.12 g kg^{-1} at day 10 (Fig. 3), which gave an ^{85}Sr/Ca-ratio of 2130 ± 270 Bq ^{85}Sr g^{-1} Ca. Calcium concentrations were on average 0.73 g kg^{-1} lower in the samples collected at day 3 (2.37 ± 0.21 g kg^{-1}) than at days 10 and 22 – 24 (paired t-test, $P<0.05$). The average milk production on day 10 postpartum was approximately 0.52 kg, i.e., approximately 1.5 g Ca day^{-1} was secreted in milk (equal to 9 % of the daily intake). From the estimated ^{85}Sr transferred directly from feed above, the observed Ca concentration in milk by day 10, daily administered ^{85}Sr activity and estimated Ca intake above, an $OR_{milk\text{-}diet}$ (Eq. 3) of 0.124 ± 0.037 was estimated.

For all does the ^{134}Cs concentration in milk increased towards a plateau, or equilibrium, concentration within 2 – 5 days postpartum (Fig. 3). From the average equilibrium ^{134}Cs concentration of 1410 ± 190 Bq kg^{-1} (average of all animals) a transfer coefficient ($F_{m,Cs}$) of 0.185 ± 0.025 day kg^{-1} was estimated. Following termination of radionuclide administration, the ^{134}Cs concentrations in milk decreased according to a double exponential model in all does except doe A. Non-linear regression indicated that 31 – 48 % of the ^{134}Cs activity in milk decreased with a short half-time of 0.8 – 1.5 days, while 52 – 69 % of the activity decreased with the longer half-time of 12.7 – 16.8 days (Table 4). The single half-time for doe A was in the range of the T_2 estimates for doe B – D.

Discussion

Transfer of ^{85}Sr and ^{134}Cs to foetuses

Transfer of Ca and Sr to the foetus increases in late pregnancy as the mineralization of the foetal skeleton accelerates towards parturition [12,26]. Calcium is transported actively across the placenta from the maternal to the foetal side, and Sr is probably transported by the same mechanisms. However, Sr is transferred across the placenta less efficiently than Ca, and in humans the placental discrimination is about 0.6 [12]. The observed ratios of 0.3 – 0.7 between ^{85}Sr concentrations in reindeer calves' plasma relative to the does' were in

agreement with such discrimination. The placenta does not present a barrier to the transfer of Cs to the foetus, and ratios between Cs concentrations in foetal and maternal tissues are fairly constant and independent of the time of Cs administration [12]. Correspondingly, similar Cs concentrations were observed in female caribou and their foetuses [15]. However, in the current experiment 7 – 38 % higher ^{134}Cs concentrations were observed in RBC of calves than of does.

The estimated total body contents of ^{85}Sr and ^{134}Cs in calves (Table 3) indicated that similar percentages of the total administered activity of the nuclides were transferred to the calves. From the apparently low ^{85}Sr absorption from the GI tract by the does (see below) it is therefore evident that a larger fraction of absorbed ^{85}Sr than of ^{134}Cs was transferred from the doe to the foetus. The higher transfer of ^{85}Sr was probably due to the higher skeletal mineralization in late pregnancy, whereas the total amount of ^{134}Cs in the calves increases nearly linearly with their body masses. Concentrations of ^{85}Sr in tissues of the calf born first (calf D), relative to the total administered activity, were lower than in the other calves, but this could possibly be due to doe D's larger feed and Ca intake. The transfer of Sr from diet to foetuses of free-ranging reindeer may be higher than observed in this experiment due to lower Ca intake (see discussion regarding $F_{m,Sr}$ in the next section).

The tissue distribution of ^{85}Sr (Table 3) was in agreement with that in one year old reindeer receiving constant daily amounts of ^{85}Sr for 8 weeks [27], except concentrations in RBC and plasma which appeared somewhat lower in our experiment. The distribution of ^{134}Cs was generally in agreement with previous reports (see examples and references in [23]), with relatively high concentrations in muscle and kidney, but differences between tissues were less pronounced than those in one year old reindeer after 171 days of constant ^{134}Cs administration [23].

Ca concentrations were not determined in calf tissues. However, from the low ^{85}Sr uptake (see below) there is no reason to assume that the Ca supplementation did not satisfy the does' and foetuses' requirements. No significant difference in the Ca content of long bones of foetuses or does would be expected [28], and the differences in ^{85}Sr concentrations in the present work were therefore not expected to be due to differences in nutritional Ca status in the animals.

Transfer of ^{85}Sr and ^{134}Cs to milk

The discrimination of Sr transport in favour of Ca in the GI tract and in transport across the mammary epithelial cells in the udder results in values of $OR_{milk\text{-}diet}$ for different ruminants ranging from 0.08 – 0.16, with most of them falling near 0.1 [25]. Howard et al. [18] and Beresford et al. [19] suggested 0.11 as a general value for ruminants. The value of 0.124 ± 0.037 estimated for reindeer is close to this general value, and lower than the estimate of 0.2 based on a bulked reindeer milk sample and an estimated summer diet in [29]. Estimating the *OR* from the total ^{85}Sr concentration (Fig. 3), not from the fraction secreted with the short half-time, gave a 20 % higher *OR* value. This could be explained by an increased mobilization of Ca and co-deposited ^{85}Sr from the readily exchangeable Ca pools in bone accumulated during the 25 – 60 days of administration before parturition (see next paragraph). Continuous accumulation of ^{90}Sr, year round in free-ranging reindeer, with subsequent mobilization and secretion in milk during lactation could thus potentially explain the higher *OR* estimated in [29].

The fraction of ^{85}Sr secreted in milk with the shorter half-time was used as an estimate of the fraction of ^{85}Sr transferred directly from feed to milk and as basis for estimating *OR* and $F_{m,Sr}$. This is a simplification, since ^{85}Sr taken up from the GI tract and circulating in the plasma and extracellular fluids will be involved in the continuous remodelling of the doe's skeleton. According to [30], mobilization of skeletal reserves of Ca during late pregnancy and lactation is achieved by an increase in the rate of bone resorption relative to that of bone accretion, which remains fairly constant. Therefore, such remodelling affects all estimates of $F_{m,Sr}$ for lactating animals and would therefore not introduce a significant bias into the estimated $F_{m,Sr}$ for reindeer milk in the present work. The initial decrease in ^{85}Sr concentrations in milk of does B – D, during the period of continued ^{85}Sr administration, was inferred as reflecting resorption of ^{85}Sr that the does accumulated before parturition. This was supported by a study showing that ewes are unable to absorb enough dietary Ca in late pregnancy and early lactation to meet the high demands, irrespective of their Ca intake, and that skeletal stores of Ca therefore are mobilized [30]. A similar mobilization of ^{90}Sr from skeletal stores to milk was indicated in a study with dairy cows [25]. Calcium concentrations in milk increased with time as opposed to the ^{85}Sr concentrations (Fig. 3), i.e., the initial decrease in ^{85}Sr concentrations in doe B – D was not related to changes in regulation of the Ca content in the milk. Furthermore, the lower ^{85}Sr concentrations in milk of doe A during the first days of lactation might reflect the lower demand for mobilizing skeletal stores during this period of low milk production.

The $F_{m,Sr}$ estimated in this work is similar to the value of 0.024 day kg^{-1} predicted by the general relationship between radiostrontium transfer coefficients to milk and the dietary Ca intake of dairy animals [18,19]. This prediction was based on a reported Ca content in milk of 3.6 g kg^{-1} by day 21 – 25 of lactation [22] and the estimated Ca intake from the current study. The Ca concentration in milk sampled 22 – 24 days postpartum (3.25 ± 0.34 g kg^{-1}) was comparable to the value in [22]. Staaland and Hove [31] estimated that the Ca intake in reindeer increased from approximately 5.2 g day^{-1} during winter to approximately 18 g day^{-1} during summer. Thus, at the time of calving in the spring, before fresh green vegetation is plentiful, considerably lower Ca intakes than in this experiment are likely. For example, with a Ca intake of 10 g day^{-1} the relationship in [18] suggests a doubling of the $F_{m,Sr}$ compared to that in this experiment, i.e., 0.04 day kg^{-1}.

The estimated $F_{m,Cs}$ of 0.185 ± 0.025 day kg^{-1} was higher than those reported in the literature for cows, sheep and goats (i.e., approximately 0.0079, 0.058 and 0.10 respectively [32]), as predicted on the basis of an expected secretion of roughly 10 % of the daily Cs intake in the milk [17].

This experiment did not reveal any significant differences in the half-times by which ^{85}Sr and ^{134}Cs was excreted in milk, only in the fractions of the activity being excreted with the different half-times. Probably the short half-time represents clearance of ^{85}Sr and ^{134}Cs from the plasma pool, and the slower decrease reflects secretion of mobilized ^{85}Sr and ^{134}Cs being released from intracellular pools, respectively. Doe D had the smallest fraction of ^{85}Sr being secreted with the longer half-time (13 %). Doe D received radionuclides the shortest period, and had least time for ^{85}Sr accretion in bone. On the other hand, doe A received radionuclides for the longest period, and had the largest fraction of ^{85}Sr secreted with the longer half-time (18 %). In addition doe A was able to retain more skeletal ^{85}Sr during the initial lactation period of low milk production which was thus available for resorption later.

Milk production by other does of the same herd (with calves at foot) peaked by week 3, producing about 1000 g day^{-1} [20]. Then production decreased linearly to about 200 g day^{-1} by week 25 postpartum [20]. Our does produced only about half this peak value (plateau production of 400 – 600 ml day^{-1}), and the production was maintained at this level for the duration of the sampling.

Absorption and excretion of ^{85}Sr and ^{134}Cs by pregnant reindeer

The review by Coughtrey and Thorne [33] showed that a maximum of approximately 10 % of Sr was absorbed by adult ruminants (i.e., cattle, sheep, goats), while a true absorption coefficient (A_t) of 0.12 was reported for $SrCl_2$ [34,35]. Variations in estimated A_t values for Sr are more related to differences in the Ca status of the animals than to the Sr source [35]. When Ca intakes are low or below requirement, A_t of Sr increases. Apparently the diet in the current experiment provided a sufficient Ca intake, making the estimated faecal excretion fluctuate by a factor of 2 (Fig. 1). Daily excretions of more than 100 % of the daily administered activity may be observed when small amounts are absorbed, partly due to endogenous excretion of radionuclides to faeces [36]. The variable ^{85}Sr concentrations in plasma (Fig. 2a) could be explained by small percentage changes in absorption of the large pool of ^{85}Sr in the GI tract.

In another experiment where reindeer calves were given 200 g of concentrates and 800 g (DM) of lichens daily, the animals excreted totally approximately 3 % of the administered ^{85}Sr with urine and approximately 86 % of the administered ^{85}Sr with faeces [27]. The apparently lower absorption in the current experiment may be due to a higher Ca content of the diet consisting predominantly of concentrates. The ^{85}Sr concentrations in plasma reported here were correspondingly low compared to those in the experiment in [27].

Faecal ^{134}Cs excretion was 10 – 15 % higher in this experiment than by calves in an earlier experiment with ^{134}Cs added to a 100 % lichen diet [23]. The higher excretion in the current experiment was probably due to the higher crude fibre content in the diet (see discussion in [37]).

In humans the biological half-time for Cs is reduced during pregnancy [12]. This effect was not evident for reindeer as the half-times estimated for ^{134}Cs in RBC (Table 2) were intermediate to those in reindeer calves fed lichens and concentrate and calves fed only lichens, respectively [23].

Conclusions

1. A significantly larger fraction of absorbed ^{85}Sr than of ^{134}Cs was transferred from the doe to the foetus. Distribution of ^{85}Sr and ^{134}Cs in tissues of newborn calves was generally similar to that reported in older individuals, suggesting that distribution of these nuclides after absorption from the GI tract does not change significantly with age of juvenile reindeer.
2. The low absorption of ^{85}Sr from the GI tract suggests that the diet satisfied the Ca requirements of the does. Nevertheless the change in concentrations of ^{85}Sr secreted in milk indicated that the does mobilized skeletal resources for their relatively low milk production.

3. The study supports a previously derived relationship between Ca intake and Sr transfer to milk of ruminants, and suggests that transfer of radiostrontium to the foetus and milk of free-ranging reindeer can be considerable higher than in this experiment with a Ca rich diet. However, transfer of radiostrontium in ionic form from diet to milk appears to be significantly lower than transfer of ionic radiocaesium.
4. This experiment did not indicate any significant differences in the half-times by which ^{85}Sr and ^{134}Cs were secreted in reindeer milk.

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Tables

Table 1 Feed intake and faeces and urine excretion during the experiment (mean ± SD). In addition to the concentrates each doe was given 500 g lichens FM daily.

	Doe A	Doe B	Doe C	Doe D
Concentrates (g day^{-1})	1190 ± 180	1450 ± 230	1220 ± 380	1730 ± 470
Faeces production (g DM day^{-1})	420 ± 60	529 ± 74	480 ± 120	630 ± 180
Urine production (ml day^{-1})	1180 ± 270	1240 ± 270	1490 ± 410	1570 ± 310

DM - dry mass

Table 2 Estimated biological half-times ($T_{1/2}$) and fractional absorption of ^{134}Cs in RBC (a) (Eq. 1). Mean ± SE, and model R^2. N is number of observations per doe.

Doe	N	$T_{1/2}$ (day)	a (Bq kg^{-1} day^{-1})	R^2
A	17	15.8 ± 1.6	34.6 ± 2.1	0.96
B	14	14.7 ± 1.2	33.5 ± 1.5	0.98
C	10	29.0 ± 7.9	22.8 ± 1.6	0.97
D	8	59 ± 37	18.3 ± 1.3	0.98

Table 3 Activity concentrations of ^{85}Sr and ^{134}Cs in sampled tissues and ground remnants of the calves (Bq g^{-1} DM), and estimated total body content (kBq). The body masses of calves A – D were 4.69, 4.25, 3.88 and 6.13 kg (FM) respectively. The SD given for ground remnants is SD for the 5 replicates; elsewhere SD reflects uncertainties due to counting statistics and geometry (small samples).

Tissue	Calf A		Calf B		Calf C		Calf D	
	^{85}Sr	^{134}Cs	^{85}Sr	^{134}Cs	^{85}Sr	^{134}Cs	^{85}Sr	^{134}Cs
M. quadriceps	nd	13.04 ± 6 %	nd	11.27 ± 6 %	1.10 ± 23 %	7.87 ± 5 %	nd	5.92 ± 6 %
M. triceps	nd	13.04 ± 5 %	1.20 ± 25 %	10.28 ± 5 %	nd	7.63 ± 6 %	nd	5.92 ± 7 %
M. longissimus	nd	12.42 ± 6 %	1.20 ± 16 %	9.161 ± 5 %	1.11 ± 30 %	7.26 ± 5 %	0.27 ± 35 %	6.04 ± 5 %
Heart	nd	9.41 ± 7 %	1.22 ± 10 %	6.822 ± 5 %	1.17 ± 28 %	5.44 ± 6 %	0.37 ± 30 %	4.24 ± 6 %
Tongue	nd	13.81 ± 6 %	3.59 ± 22 %	11.51 ± 6 %	1.12 ± 34 %	9.23 ± 6 %	nd	6.16 ± 6 %
Gonads	4.2 ± 27 %	9.91 ± 6 %	nd	6.592 ± 9 %	nd	3.45 ± 9 %	0.91 ± 22 %	4.23 ± 6 %
Liver	nd	8.03 ± 7 %	1.21 ± 38 %	6.568 ± 7 %	nd	6.16 ± 6 %	nd	4.61 ± 6 %
Kidney	nd	15.8 ± 7 %	1.21 ± 50 %	9.418 ± 6 %	nd	9.36 ± 7 %	nd	6.18 ± 5 %
Lungs	4.2 ± 26 %	8.54 ± 7 %	3.63 ± 10 %	8.178 ± 5 %	2.32 ± 13 %	6.54 ± 5 %	0.47 ± 52 %	5.70 ± 5 %
Spleen	nd	10.17 ± 6 %	2.42 ± 36 %	8.178 ± 6 %	nd	6.77 ± 7 %	0.95 ± 22 %	4.73 ± 5 %
Rumen	nd	10.67 ± 6 %	4.84 ± 14 %	8.178 ± 6 %	2.24 ± 28 %	7.14 ± 6 %	0.71 ± 25 %	4.27 ± 6 %
Abomasum	nd	12.18 ± 7 %	ns	ns	28.1 ± 17 %	8.65 ± 6 %	1.00 ± 24 %	5.85 ± 5 %
Small intestine	1.45 ± 42 %	11.83 ± 6 %	1.24 ± 14 %	7.200 ± 5 %	2.32 ± 19 %	7.41 ± 6 %	0.85 ± 14 %	6.07 ± 5 %
Large intestine	8.7 ± 14 %	10.07 ± 6 %	6.24 ± 17 %	11.31 ± 6 %	11.6 ± 9 %	7.16 ± 6 %	3.99 ± 12 %	5.97 ± 5 %
Skin	4.30 ± 18 %	4.40 ± 7 %	7.49 ± 8 %	1.615 ± 6 %	5.55 ± 8 %	1.48 ± 7 %	4.49 ± 6 %	1.70 ± 5 %
Metatarsus	821 ± 5 %	0.605 ± 13 %	764 ± 5 %	0.251 ± 17 %	485 ± 5 %	0.37 ± 12 %	253 ± 5 %	0.25 ± 15 %
Metacarpus	762 ± 5 %	0.416 ± 10 %	820 ± 5 %	0.251 ± 17 %	533 ± 5 %	0.37 ± 13 %	231 ± 5 %	0.12 ± 19 %
Mandible	791 ± 5 %	0.882 ± 9 %	639 ± 5 %	0.752 ± 17 %	509 ± 5 %	0.74 ± 8 %	185 ± 4 %	0.629 ± 3 %
Teeth	909 ± 5 %	0.479 ± 8 %	611 ± 5 %	0.376 ± 17 %	347 ± 5 %	0.37 ± 11 %	165 ± 5 %	0.12 ± 11 %
Metatarsus marrow	15.71 ± 5 %	6.207 ± 2 %	21.46 ± 4 %	4.629 ± 2 %	5.16 ± 6 %	5.07 ± 2 %	nd	2.29 ± 3 %
Metacarpus marrow	45.5 ± 4 %	5.03 ± 3 %	35.06 ± 4 %	4.015 ± 3 %	nd	13.61 ± 2 %	nd	2.31 ± 3 %
Plasma	0.472 ± 6 %	0.142 ± 4 %	0.444 ± 7 %	0.156 ± 5 %	0.382 ± 9 %	0.073 ± 9 %	0.127 ± 13 %	0.053 ± 8 %
RBC	nd	1.020 ± 2 %	0.082 ± 11%	0.649 ± 2 %	0.062 ± 13 %	0.664 ± 2 %	0.024 ± 20 %	0.460 ± 2 %
Ground remnants	244 ± 15 %	6.37 ± 7 %	257 ± 13 %	5.56 ± 8 %	169 ± 15 %	4.18 ± 10 %	80.9 ± 12 %	3.26 ± 5 %
Total body content	235 ± 14 %	7.34 ± 5 %	227 ± 12 %	5.53 ± 7 %	130 ± 14 %	3.82 ± 8 %	100 ± 11 %	4.81 ± 4 %

nd – not determined (nuclide content below detection limits)

ns – not sampled (calf B had been suckling. The abomasum contained milk and was therefore not sampled)

Table 4 Estimated biological half-times (T_1 and T_2; day) for the decline in ^{85}Sr and ^{134}Cs activity concentrations in milk of does A – D using Eq. 2. A_0 is the estimated activity concentration (Bq kg^{-1}) when nuclide administration was terminated, and f_1 and f_2 give the fractions of this activity concentration decreasing with the respective half-times. Mean ± SE, and model R^2.

Nuclide	Parameter	Doe A[1]	Doe B	Doe C	Doe D[2]
^{85}Sr	A_0	6330 ± 690	5590 ± 610	6270 ± 840	8750 ± 670
	f_1	0.82 ± 0.11	0.83 ± 0.10	0.84 ± 0.13	0.874 ± 0.076
	T_1	0.98 ± 0.12	2.76 ± 0.38	1.77 ± 0.30	1.72 ± 0.17
	f_2	0.177± 0.013	0.173 ± 0.030	0.160 ± 0.022	0.126 ± 0.013
	T_2	12.53 ± 0.66	19.3 ± 2.5	19.1 ± 1.5	12.6 ± 1.3
	R^2	0.986	0.979	0.984	0.993
^{134}Cs	A_0	994 ± 48	1670 ± 140	1240 ± 100	1540 ± 190
	f_1	1.00	0.367 ± 0.074	0.315 ± 0.078	0.48 ± 0.12
	T_1	14.67 ± 0.71	1.51 ± 0.53	0.79 ± 0.36	1.27 ± 0.50
	f_2	-	0.633 ± 0.034	0.685 ± 0.028	0.524 ± 0.037
	T_2	-	15.93 ± 0.61	12.69 ± 0.21	16.85 ± 0.59
	R^2	0.953	0.990	0.997	0.984

[1] Only one exponential term could be fitted to the ^{134}Cs data for doe A. Double exponential model gave insignificant parameters.
[2] The fit to the ^{85}Sr concentrations for doe D neglected the values at days 30, 42, 86 and 92 because the regression analysis gave significant positive residuals these days.

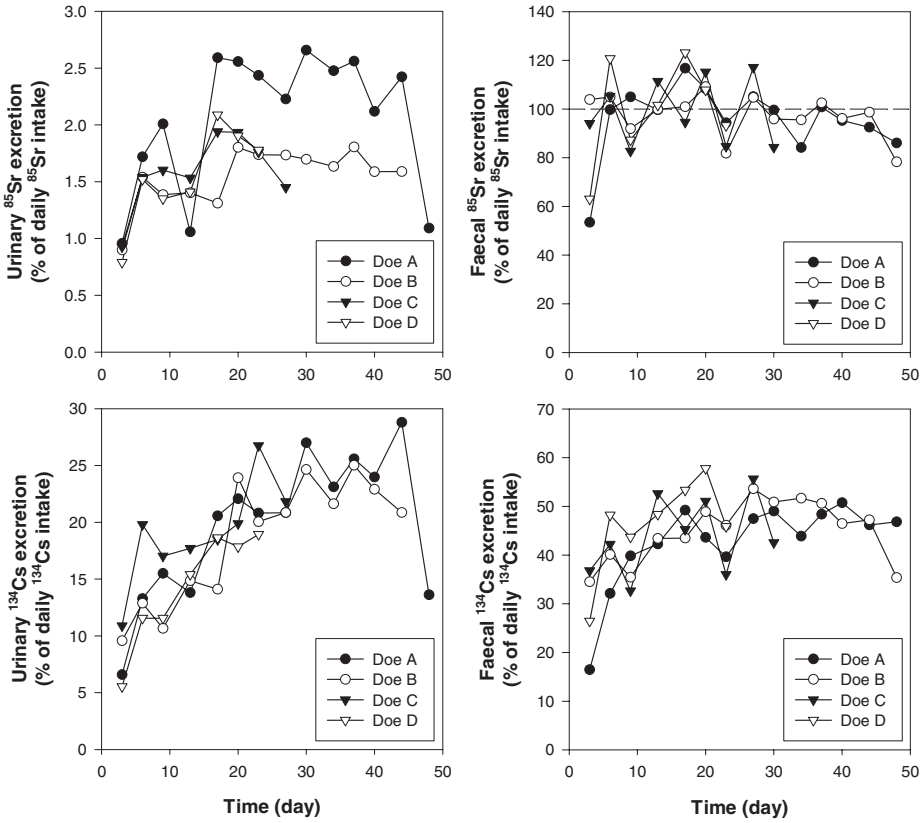

Fig. 1 Daily urinary and faecal excretion of ^{85}Sr and ^{134}Cs (Bq day^{-1}) by the does, normalised to the daily nuclide administration rates (Bq day^{-1}).

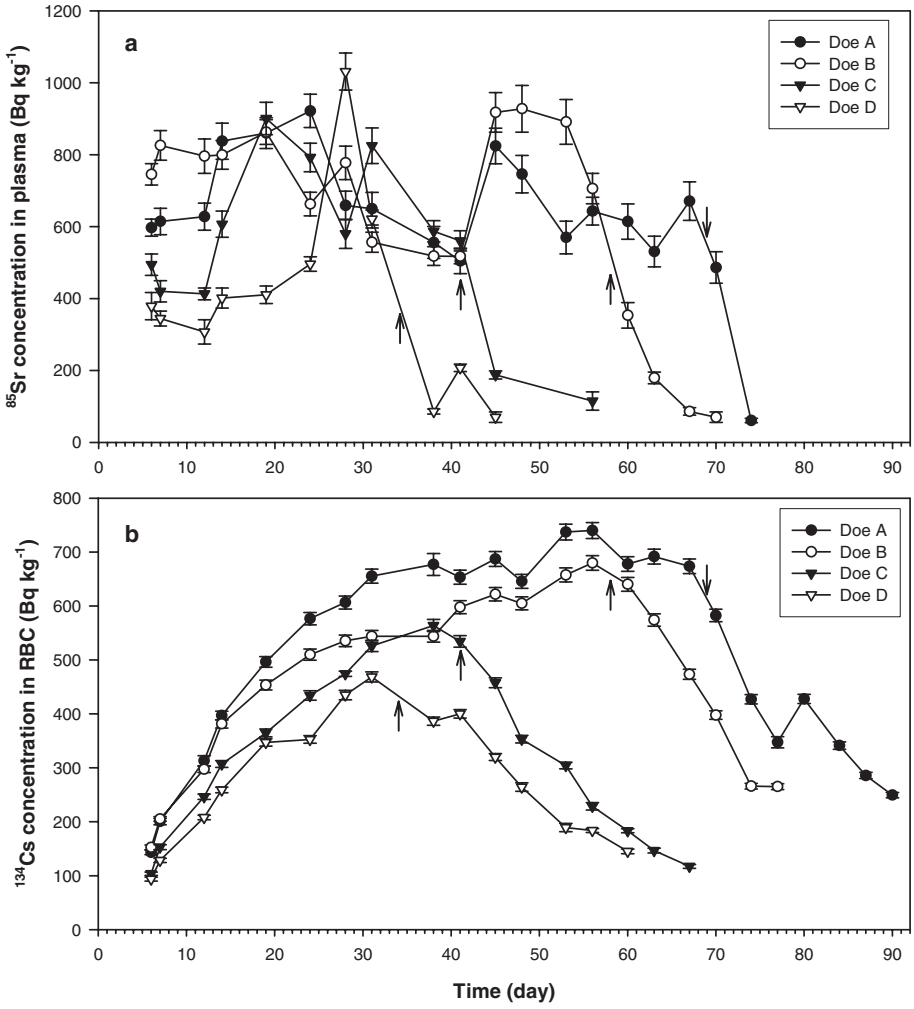

Fig. 2 Activity concentrations of (a) ^{85}Sr in plasma and (b) ^{134}Cs in RBC of the does (Bq kg^{-1}). The last day of nuclide administration is indicated with an arrow.

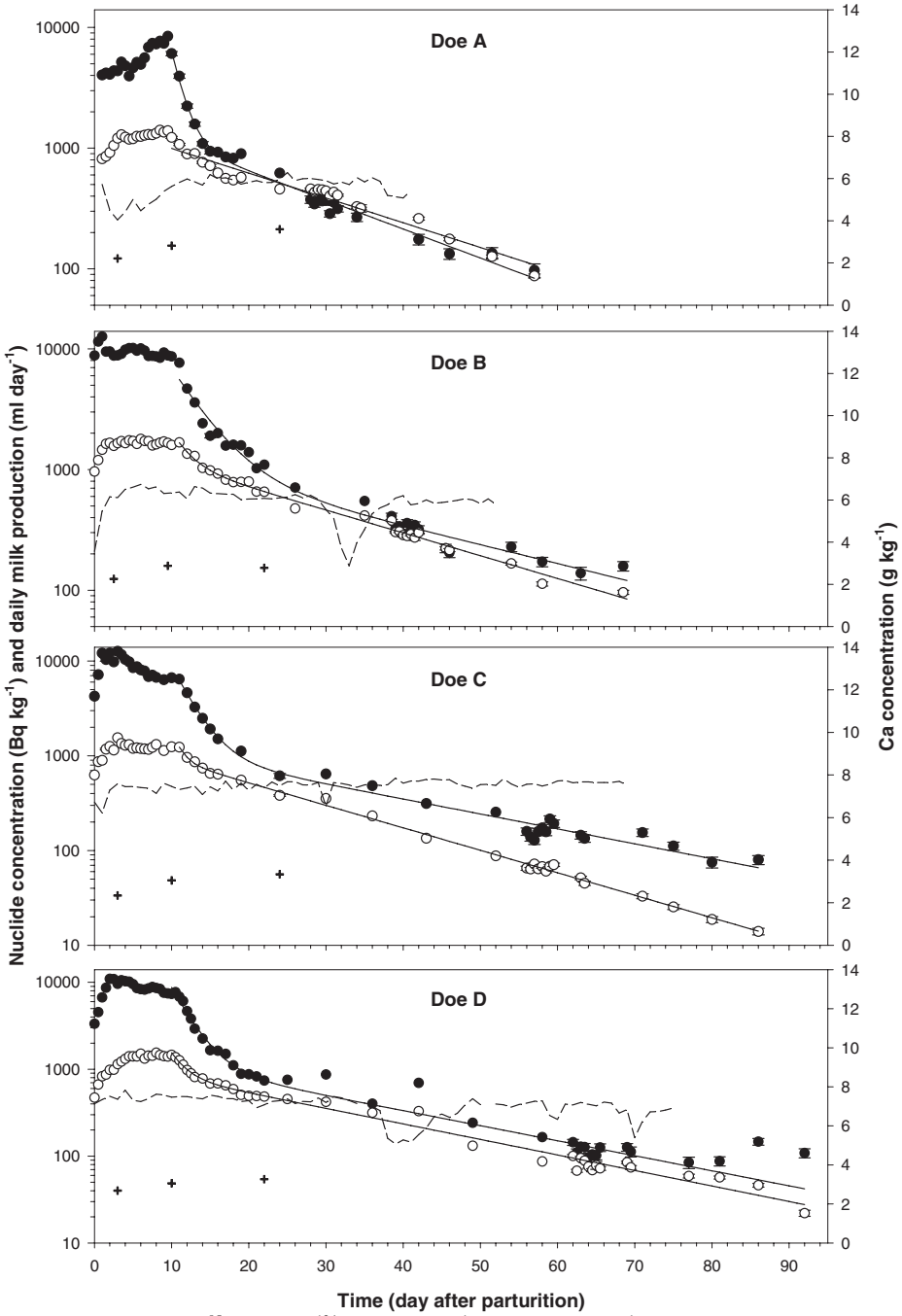

Fig. 3 Concentrations of ^{85}Sr (●) and ^{134}Cs (○) (Bq kg^{-1}) and Ca (+) (g kg^{-1}) in milk of does A – D. Curves represent fits by the model (Eq. 2; parameters in Table 4). Daily milk production (ml day^{-1}) is given by the broken line.

Paper IV

Chernobyl radioactivity persists in reindeer

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Chernobyl radioactivity persists in reindeer

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Abstract

Transfer of ^{137}Cs in the soil–plant/lichen–reindeer food chain was studied in central (Østre Namdal) and southern Norway (Vågå) during 2000–2003. Reindeer from these areas have been continuously subjected to countermeasure application since the 1986 Chernobyl accident. In both areas no decline in ^{137}Cs concentrations was detectable in reindeer slaughtered in autumn since 1995, or in reindeer slaughtered in winter since 1998–1999. Seasonal differences in ^{137}Cs concentrations in reindeer have been less pronounced in recent years, with ^{137}Cs concentrations occasionally higher in autumn than in winter. Soil-to-plant ^{137}Cs transfer was significantly higher in Østre Namdal than in Vågå. Climatic influences on lichen growth and abundance, and on soil properties that influence the availability of ^{137}Cs for plant uptake, are hypothesized to have a larger impact on long-term transfer of radiocaesium in the soil–plant/lichen–reindeer food chain than has been previously observed.

Keywords: Caesium-137; Effective half-life; Reindeer; Lichen; Plant; Food chain; Chernobyl

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1. Introduction

After radioactive fallout, reindeer contain elevated radiocaesium concentrations due to intake of contaminated lichens and other vegetation (Lidén, 1961; Hanson, 1966; Holleman et al., 1971; Gaare and Staaland, 1994; Staaland et al., 1995). Particularly, high concentrations can be observed in reindeer during the winter resulting from a combination of higher radiocaesium intake and lower radiocaesium excretion rate (e.g. Holleman et al., 1971). Contaminated lichens are generally considered to be the main source of radiocaesium intake during winter, and may be the main radiocaesium source during summer (Staaland et al., 1995). Additionally, fungi can be another important source of radiocaesium intake in the autumn (Hove et al., 1990).

Following fallout from the 1986 Chernobyl accident, individual radiocaesium activity concentrations in reindeer meat in the most affected areas in Scandinavia reached 150 kBq kg^{-1} (fresh weight, FW) (Strand et al., 1992) and declined with effective half-times of 3–5 years (Pedersen et al., 1993; Åhman and Åhman, 1994; Amundsen, 1995; Gaare et al., 2000; Åhman et al., 2001). In comparison, half-times of 6–9 years have been observed in northern Fennoscandia that received less Chernobyl fallout (Skuterud et al., 1999; Åhman et al., 2001; Rissanen et al., 2003). Observations of slower rates of decline in radiocaesium concentrations in several vascular plants compared to lichens (Gaare and Staaland, 1994; Gaare et al., 2000) indicated that the significant seasonal differences in radiocaesium concentrations in reindeer observed during the first years after the Chernobyl accident (e.g. Eikelmann et al., 1990; Pedersen et al., 1993; Åhman and Åhman, 1994) would be less pronounced with time (Gaare and Staaland, 1994). The point in time at which seasonal differences in radiocaesium concentrations become reduced might be dependent on the species composition of the reindeer diet, and may even vary between grazing areas.

About 58% of Norwegian territory is used as pasture by reindeer, a major part of which is used for reindeer herding (Norwegian Reindeer Husbandry Administration, 2004; Danielsen, J. Directorate for Nature Management, Trondheim, personal communication). The probability that any future ^{137}Cs contamination event in Norway will affect reindeer grazing areas is therefore great. A thorough understanding of reindeer radioecology is therefore essential in Norwegian nuclear emergency preparedness. However, most terrestrial radioecological studies in Norway were terminated during the 1990s, with the exception of routine monitoring of ^{137}Cs concentrations in semi-domestic reindeer in central and southern Norway where a continued requirement existed to ensure that meat complied with the national intervention limit of 3000 Bq kg^{-1}. Furthermore, by the end of the 1990s, studies indicated that rates of decline in ^{137}Cs concentrations in other components of the ecosystem were slower than previously reported (Jonsson et al., 1999; Smith et al., 2000). The present study was therefore initiated to obtain new data and more detailed knowledge about the long-term transfer of ^{137}Cs in the soil–plant/lichen–reindeer food chain that could help explain the persisting elevated ^{137}Cs concentrations in reindeer. The study was conducted during the period 2000–2003

with sampling of soil, vegetation and reindeer tissues in two of the areas in Norway that were most affected by fallout from the Chernobyl accident. Supplementary data on ^{137}Cs in reindeer from 1986 onwards were obtained from the Norwegian Reindeer Husbandry Administration.

2. Materials and methods

2.1. Study areas

The Vågå and Østre Namdal reindeer herding districts (Fig. 1) were selected because of the comparatively high levels of fallout received from the Chernobyl accident and the area's contrasting climates. Reindeer grazing in Vågå occurs mostly between 1000 and 1600 m above sea level (a.s.l.), whilst the Østre Namdal pasture is mainly situated between 450 and 1100 m a.s.l. Both areas lie in northern boreal and alpine vegetation zones, the low alpine zone starting at about 1200 m a.s.l. in Vågå and 750 m a.s.l. in Østre Namdal (Moen, 1999). However, the climate in Vågå is more continental with a total annual precipitation of 280–1200 mm in different parts of the range, compared to 1000–1500 mm of Østre Namdal, and slightly lower average temperature. Accordingly, Vågå is therefore botanically classified in the relatively continental 'indifferent' section whereas Østre Namdal is in the 'slightly oceanic' section (Moen, 1999). The growing season, defined as the period when the average temperature exceeds 5 °C, is on average 10–20 days shorter in Vågå than in Østre Namdal (Moen, 1999). There is a higher frequency of mires in Østre Namdal than in Vågå, and mires in areas with oceanic climates are generally nutrient-poor when compared to those in more continental climates. The Precambrian and Cambro-Silurian bedrock is of variable composition in both areas. Where the bedrock is not exposed, it is covered with till or a thin cover of superficial deposits (Moen, 1999).

The reindeer herds studied in Vågå and Østre Namdal both have about 2000 animals. While the Vågå herd graze mainly the southern and south-eastern parts of the area during summer, autumn and early winter, the Østre Namdal herd utilize the north-eastern corner of their area during these seasons. A combination of unfavourable climate and lichen availability causes the Østre Namdal herd to be transferred south-westwards in December–January (slaughtering is carried out in connection with this transfer). Similarly, in Vågå, winter grazing occurs in the northern parts of the area with less snow.

2.2. Fieldwork

Sampling of soil, vegetation and reindeer tissues was conducted during the period 2000–2003. Tissue samples (neck muscle, liver and kidney) were obtained during slaughter in August–September and December–February. Generally 10 calves and 10 adult females (about 10 years old) were sampled each time. Some additional samples were obtained in Vågå in August 2001 from animals hunted for surveillance

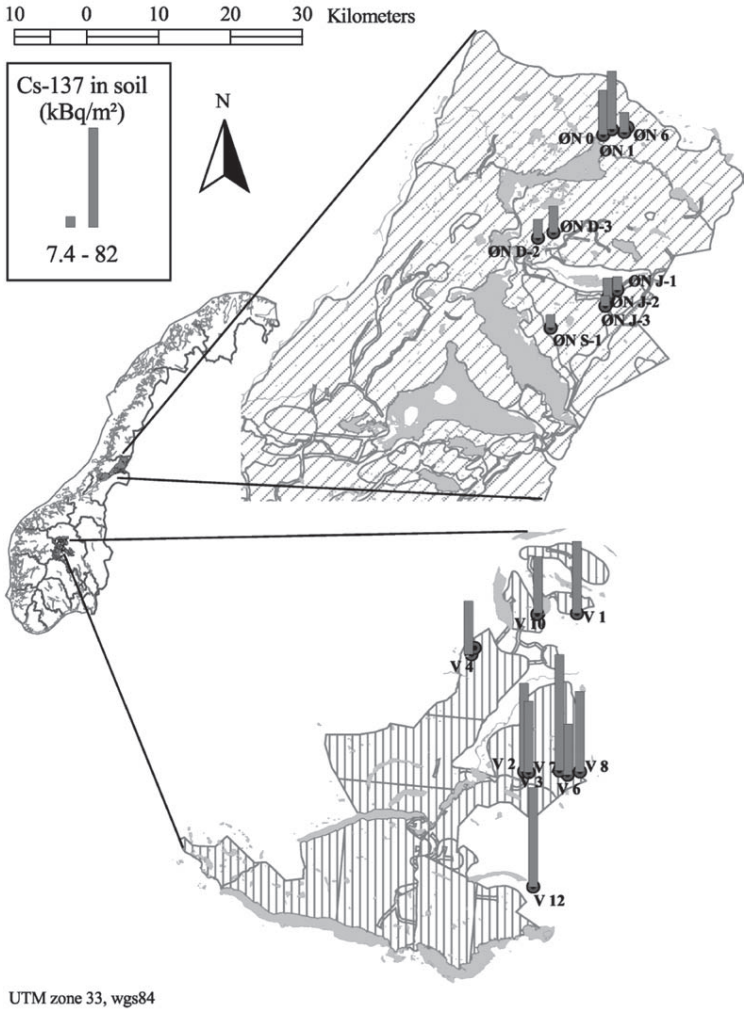

Fig. 1. The Vågå (V) and Østre Namdal (ØN) reindeer herding districts with soil sample locations and deposition density indicated. Bars represent geometric mean deposition density at each location (kBq m^{-2}; geometric SD were from 1.41 to 1.83). The total area of the Vågå pasture land is 1357 km^2 whereas the area of Østre Namdal is 6607 km^2 (only the north-eastern half of the Østre Namdal area is shown on a large scale).

of ^{137}Cs concentrations. Time series data on ^{137}Cs activity concentrations in reindeer meat samples or live monitored reindeer were obtained from the archives of the Norwegian Reindeer Husbandry Administration (the regional offices in Snåsa and Røros). These archives give the number of animals monitored at each slaughtering, with arithmetic mean, minimum and maximum values separately for calves and adults. Only ^{137}Cs concentrations in animals not subjected to countermeasures used

to reduce the body pool of ^{137}Cs are given here. The Vågå archive was supplemented by results of blood samples obtained in the field from free ranging animals of the same herd by one of the authors (K. Hove).

Soil and vegetation samples were taken from summer, autumn and early winter grazing areas in Østre Namdal, and in the autumn and winter grazing areas in Vågå (Fig. 1). Soil and/or vegetation samples were obtained from 12 and 11 locations within the Vågå and Østre Namdal pastures respectively, each location being roughly 50 m × 50 m and including different vegetation communities. Locations 1–11 in Vågå were sampled in August 2001 and location 12 in September 2002. Locations D-1 to D-3 in Østre Namdal were sampled in September 2001 and the remaining locations in July 2003. Most locations were at or above the limit of woodland. Soil cores (10 cm diameter, 5 cm deep or less if the humus layer was shallower) were collected from 5 arbitrarily selected sites within each location. Litter was removed in the field.

Samples of lichens and vascular plants were collected from across the area covered by the soil sampling (one sample per species per location). Fungal fruit bodies were only found in some Vågå locations. In total, samples of 29 different species of vegetation known to be eaten by reindeer (Staaland et al., 1995; Warenberg et al., 1997; Gaare et al., 2000) were collected, including lichens, fungi, grasses, herbs, heaths etc. Samples from trees and heaths consisted mainly of leaves, but as the sampled material was snatched (simulating grazing) some shoots were included. Samples of *Carex* and *Eriophorum* included parts of the rhizomes, which are also eaten by reindeer. Any soil adhering to these samples was carefully removed. The nomenclature of lichens and vascular plants follows Santesson (1993) and Lid and Lid (1994).

As a consequence of the climate, lichens are interspersed and partly covered with other food plants in the Østre Namdal locations. Accordingly, lichens were seldom found as dominating lichen carpets in Østre Namdal. *Cladonia stellaris* was not present at any of the sampling locations in Østre Namdal.

2.3. Sample preparation and analysis

All samples were dried to constant weight at 70 °C, with dead parts of lichen removed prior to drying. Samples of reindeer tissues and vegetation were homogenized, while soil samples were passed through a 2-mm sieve and mixed thoroughly. A 75-mm NaI(Tl) well detector and multichannel analyser were used to determine ^{137}Cs activities in reindeer tissues and vegetation samples. High purity Ge-detectors and multichannel analysers were used to determine ^{137}Cs in soil. No further characterization or analysis of the soils was carried out. Determined ^{137}Cs concentrations in soil and vegetation were decay corrected to 15 September 2002 to enable comparison of activities in samples collected at different times. All reported ^{137}Cs activity concentrations refer to dry mass (DM), except the reindeer time series data. Levels of ^{137}Cs in soil are referred to as deposition densities (kBq m^{-2}).

Data on ^{137}Cs in soil, vegetation and reindeer were log-transformed prior to statistical analyses (to harmonize variances), and the geometric mean deposition

density at each sampling location was applied when normalizing the ^{137}Cs concentrations in vegetation to the deposition densities. For all species but lichens this normalization was used to determine estimates of the aggregated transfer coefficient (T_{ag}, $m^2\,kg^{-1}$).

All statistical analyses (one-way analysis of variance (ANOVA), *t*-tests and non-linear regression using single and double exponential models) were carried out in SPSS for Windows release 11. The data on ^{137}Cs in reindeer from autumn 1986 were excluded from regression analyses since it was the year of the Chernobyl Accident and would have featured external contamination on plants. Concentrations of ^{137}Cs are presented as mean $\pm$ standard deviation (SD), whereas estimated effective half-lives of ^{137}Cs in reindeer are presented as mean $\pm$ standard error (SE).

3. Results

3.1. Soil

The deposition density in individual samples varied nearly 11-fold (18–190 kBq m^{-2}) in Vågå and 19-fold (4.2–79 kBq m^{-2}) in Østre Namdal, and there were significant differences in deposition densities between sampling locations within each area (Fig. 1; one-way ANOVA, $P < 0.001$). The average deposition density at the Vågå locations was more than three times that in Østre Namdal (factor 3.5 between the geometric mean values of 52 and 15 kBq m^{-2}; *t*-test, $P < 0.001$).

3.2. Vegetation

Activity concentrations of ^{137}Cs in the sampled vegetation are given in Tables 1 and 2 together with estimated aggregated transfer coefficients (T_{ag}; for simplicity, from here on "T_{ag}" is used also to describe the activity concentrations in lichen relative to the estimated deposition density although ^{137}Cs is not transferred from soil to lichens). The significantly higher deposition density in Vågå was not reflected in the concentrations in the vegetation. The only significant difference in ^{137}Cs activity concentrations was in *Vaccinium uliginosum*, which had higher concentration in Østre Namdal. Accordingly, the estimated T_{ag} values were significantly higher in Østre Namdal for 9 of the sampled species, reaching a 10-fold difference for *V. uliginosum*.

In Vågå there were differences both in activity concentrations and T_{ag} values between some plant species (Table 3). *Salix phylicifolia* generally had lowest activity concentration and T_{ag} values, and *Russula* spp. had highest concentrations. In Østre Namdal one-way ANOVA of log-transformed ^{137}Cs concentrations indicated that there were differences in mean concentrations in different plant and lichen species ($P = 0.045$), but multiple comparisons of species identified no significant differences. No differences in average T_{ag} values (ANOVA with log transformed values) were noted.

Table 1
Activity concentrations of ^{137}Cs (C, Bq kg^{-1} DM) and aggregated transfer coefficients (T_{ag}, 10^{-3} m^2 kg^{-1}) for vegetation species sampled in both grazing areas (arithmetic mean ± SD)

Group	Species		Vågå				Østre Namdal				P-value
			Mean ± SD	N	Minimum	Maximum	Mean ± SD	N	Minimum	Maximum	
Lichens	*Cladonia rangiferina*	C	2350 ± 1210	7	1460	4920	1530 ± 400	4	1120	1960	
		T_{ag}	46 ± 26	6	27	97	157 ± 32	4	130	200	<0.01
	Cetraria nivalis	C	1480 ± 820	9	687	3320	1350 ± 630	5	809	2240	
		T_{ag}	28 ± 18	7	14	65	139 ± 110	4	25	240	
	Cladonia arbuscula	C	1660 ± 910	10	450	3380	1930 ± 860	12	824	3580	
		T_{ag}	29 ± 16	9	11	66	131 ± 71	10	45	270	<0.001
Heaths	*Vaccinium uliginosum*	C	380 ± 280	4	153	795	1250 ± 670	6	560	2000	<0.05
		T_{ag}	8.4 ± 7.8	4	2.5	20	85 ± 62	6	38	210	<0.01
	Vaccinium myrtillus	C	1240 ± 940	6	323	2590	1580 ± 850	9	490	3010	
		T_{ag}	26 ± 20	6	6.4	62	108 ± 67	9	27	240	<0.01
	Empetrum nigrum	C	219	1			664	1			
		T_{ag}	3.5	1			72	1			
Trees	*Betula nana*	C	360 ± 200	7	165	743	510 ± 105	2	438	587	
		T_{ag}	7.6 ± 3.7	7	2.8	13	51 ± 12	2	42	59	<0.01
	Salix phylicifolia	C	180 ± 240	5	49	602	1290	1			
		T_{ag}	4.2 ± 6.1	5	0.96	15	130	1			
	Salix glauca +	C	710 ± 1050	9	70	2930	840 ± 580	6	179	1880	
	S. lapponum	T_{ag}	18 ± 28	9	1.4	78	68 ± 52	6	9.7	140	<0.05
	Salix herbacea	C	680 ± 630	5	45	1380	1141 ± 135	3	1040	1300	
		T_{ag}	14 ± 14	5	0.88	37	84 ± 12	2	75	92	
Herbs	*Solidago virgaurea*	C	1610 ± 1140	7	469	3380	2430 ± 790	5	1410	3460	
		T_{ag}	40 ± 34	7	7.6	94	160 ± 81	5	59	250	<0.01
	Rumex acetosa	C	3950 ± 5160	3	894	9910	2610 ± 2180	3	464	4820	
		T_{ag}	100 ± 150	3	14	270	108 ± 105	2	33	180	

(*continued on next page*)

Table 1 (*continued*)

Group	Species		Vågå				Østre Namdal				*P*-value
			Mean ± SD	*N*	Minimum	Maximum	Mean ± SD	*N*	Minimum	Maximum	
Graminoids	*Deschampsia flexuosa*	*C*	1090 ± 990	9	133	2970	1020 ± 660	10	159	2330	
		T_{ag}	20 ± 17	9	3.5	53	82 ± 59	9	28	180	=0.001
	Carex bigelowii	*C*	1020 ± 660	6	270	1760	1580 ± 1050	8	368	3550	
		T_{ag}	21 ± 14	6	4.3	37	106 ± 76	7	26	220	<0.01
	Eriophorum angustifolium	*C*	410 ± 490	2	67	761	1280 ± 1620	4	116	3670	
		T_{ag}	11 ± 14	2	1.1	20	70 ± 57	4	6.3	120	

Significant differences between mean values in the districts are indicated as *P*-value.

Table 2
Activity concentrations of ^{137}Cs (C, Bq kg^{-1} DM) and aggregated transfer coefficients (T_{ag}, 10^{-3} m^2 kg^{-1}) for vegetation species sampled in one grazing area only (arithmetic mean ± SD)

District	Group	Species		Mean ± SD	N	Minimum	Maximum
Vågå	Lichens	*Cladonia stellaris*	C	2040 ± 740	9	1200	3280
			T_{ag}	40 ± 14	7	24	62
	Fungi	*Leccinium* spp.	C	1730 ± 2060	6	440	5850
			T_{ag}	48 ± 61	5	12	160
		Russula spp.	C	5690 ± 4430	4	786	10500
			T_{ag}	78 ± 58	3	16	130
	Herbs	*Geranium sylvaticum*	C	350 ± 330	3	71	718
			T_{ag}	5.9 ± 5.0	3	1.4	11
		Alchemilla alpina	C	3370	1		
			T_{ag}	41	1		
		Menyanthes trifoliata	C	130 ± 100	2	59	206
			T_{ag}	2.2 ± 1.5	2	1.1	3.2
		Bistorta vivipara	C	1160	1		
			T_{ag}	19	1		
	Graminoids	*Carex rostrata*	C	837.0 ± 1.4	2	836	838
			T_{ag}	16.4 ± 6.4	2	12	21
Østre Namdal	Lichens	*Stereocaulon pascale*	C	1470 ± 130	2	1380	1560
			T_{ag}	131 ± 27	2	110	150
	Trees	*Betula pubescens*	C	552 ± 73	2	500	603
			T_{ag}	58.4 ± 5.8	2	54	63
	Herbs	*Oxyria digyna*[a]	C	3190	1		
			T_{ag}	230	1		
		Rubus chamaemorus	C	1730[b]	1		
			T_{ag}	180	1		
	Graminoids	*Eriophorum scheuchzeri*	C	401	1		
			T_{ag}	29	1		

[a] Sampled at same location as *Rumex acetosa* containing 2550 Bq kg^{-1} (DM) and with a T_{ag} value of 0.18 m^2 kg^{-1}.
[b] The concentration of ^{137}Cs in the whole plant (with berries). The concentration in berries was 1600 Bq kg^{-1} (DM) and in the rest of the above ground plant (i.e., leaf and stem) was 990 Bq kg^{-1} (DM).

Fig. 2 shows individual T_{ag} values for the most frequently sampled plant species, clearly indicating the higher transfer at the Østre Namdal locations. Estimated transfer coefficients were particularly high at Østre Namdal location J-1, a factor of 30–45 higher than those estimated at the location with lowest average values, Vågå 2 (Fig. 3).

3.3. Reindeer

The analyses of reindeer tissues (Table 4) showed that activity concentrations of ^{137}Cs were lowest in liver and highest in kidney (paired t-test with log transformed values; $P \leq 0.013$). Concentrations in muscle and kidney were highly correlated (with correlation coefficients ranging up to 0.99) at each sampling time, except Vågå December 2002, whilst concentrations in liver were only significantly correlated with those in the other tissues at roughly half the sampling times.

Table 3
Cross table with summary of tests (one-way ANOVA with post hoc multiple comparison) of significant differences in ^{137}Cs activity concentrations (indicated with asterisk) and T_{ag} values (indicated with dots) between all species of plants, lichens and fungi from Vågå

	Cladonia rangiferina ($n = 7/6$)	*Cetraria nivalis* ($n = 9/7$)	*Cladonia stellaris* ($n = 9/7$)	*Cladonia arbuscula* ($n = 10/9$)	*Leccinium* spp. ($n = 6/5$)	*Russula* spp. ($n = 4/3$)	*Vaccinium myrtillus* ($n = 6$)	*Solidago virgaurea* ($n = 7$)	*Rumex acetosa* ($n = 3$)
Vaccinium uliginosum ($n = 4$)						*			
Menyanthes trifoliate ($n = 2$)	*/•		*			**			
Betula nana ($n = 7$)	*		*			**			
Salix phylicifolia ($n = 5$)	***/•••	***/•	***/•••	***/••	**/•	***/•	*	**/••	**/•
Salix glauca + *S. lapponum* ($n = 9$)	**/•		**/•	*		***			
Salix herbacea ($n = 5$)						*			
Geranium sylvaticum ($n = 3$)						*			

All species of which more than one sample was collected were tested. No asterisk or dot: no significant difference; one asterisk or dot: $P \leq 0.05$; two asterisks or dots: $P \leq 0.01$; three asterisks or dots: $P \leq 0.001$.

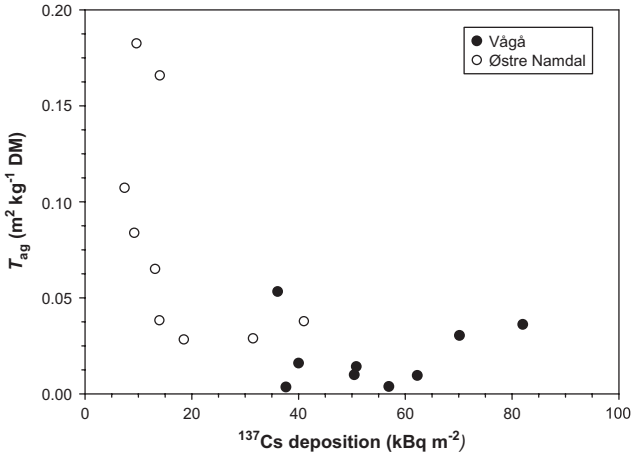

Fig. 2. Aggregated transfer coefficients for *Deschampsia flexuosa* at 18 locations in Vågå and Østre Namdal.

There were no significant differences between mean ^{137}Cs activity concentrations in muscle of calves and adults at any sampling time, but there were some seasonal and regional differences (one-way ANOVA). Highest concentrations were observed in Vågå in December.

The ^{137}Cs concentrations in reindeer after the 1986 Chernobyl accident have generally been higher in Vågå than in Østre Namdal (Figs. 4 and 5), e.g. in 14 of the

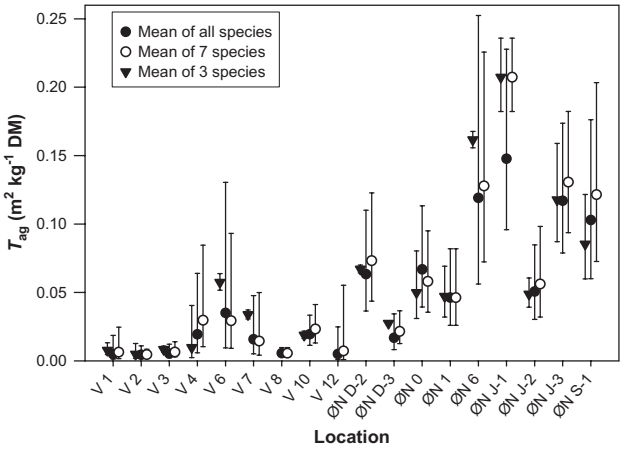

Fig. 3. Average aggregated transfer coefficients for plants at the locations in Vågå and Østre Namdal (geometric mean and SD). Dots represent means of all species sampled at each location, means of the 7 species most frequently sampled (*Descampsia flexuosa*, *Vaccinium uliginosum*, *V. myrtillus*, *Salix glauca* + *S. lapponum*, *S. herbacea*, *Solidago virgaurea*, *Carex bigelowii*), or means of the 3 species most frequently sampled (*Descampsia flexuosa*, *Vaccinium uliginosum*, *V. myrtillus*).

Table 4
Activity concentrations of ^{137}Cs (Bq kg^{-1} DM) in muscle, liver and kidney of calves and adult females (arithmetic mean ± SD)

District	Sample date	Calves			Adult females		
		Muscle	Liver	Kidney	Muscle	Liver	Kidney
Vågå	5 December 2000	10900 ± 1200	3920 ± 570	17900 ± 2400	11945 ± 1549	5300 ± 870	20300 ± 1900
	30 August 2001	7600 ± 2000 (4)	4300 ± 1800 (4)	19200 ± 5800 (4)	6820 (1)	5750 (1)	19200 (1)
	6 September 2001	7800 ± 2800	3800 ± 1700	12600 ± 3900	6200 ± 1200	2720 ± 740	13100 ± 2500 (9)
	1 December 2002	9800 ± 1100	4680 ± 490	17700 ± 2700	10500 ± 1500	4200 ± 1400	18300 ± 4400
Østre Namdal	2 February 2001	8400 ± 2200	4400 ± 1200	14200 ± 2700	8600 ± 1200	5300 ± 750	15600 ± 3500
	19 September 2001	5700 ± 1700	2190 ± 730	7600 ± 1400 (9)			
	12 December 2002	6946 ± 1176	3220 ± 830	10500 ± 2400	7129 ± 1590	3390 ± 570	10900 ± 1400

Number of samples is given in brackets ($n = 10$ when not specified).

16 years with comparable winter data. In Vågå, from 3 to 361 samples or live monitored animals were analysed at each sampling time (median $n = 21$), whilst the number of samples or live monitored animals in Østre Namdal ranged from 5 to 560 animals (median $n = 37$) per sampling time.

Until the second half of the 1990s there were relatively regular exponential decreases in ^{137}Cs concentrations in both herds, particularly during winter. Thereafter, the declines in concentrations in both herds were smaller (Figs. 4 and 5). We therefore attempted to fit double exponential models to the time series data from both herds, during both seasons, but these proved not to be applicable due to the variability and non-significant reductions in concentrations over the last few years. Thus, the results in the next paragraphs were derived after selecting years with local concentration minima as years when long-term trends changed.

In Vågå, autumn ^{137}Cs activity concentrations in reindeer declined with an effective half-time of 3.49 ± 0.44 years (mean $\pm$ SE; non-linear regression with log-transformed average values; $R^2 = 0.91$) until 1995, whilst there was no significant decline thereafter. The data from autumn 1988 were excluded from this analysis since this year, abnormally high abundance of fungi (Hove et al., 1990) was considered an

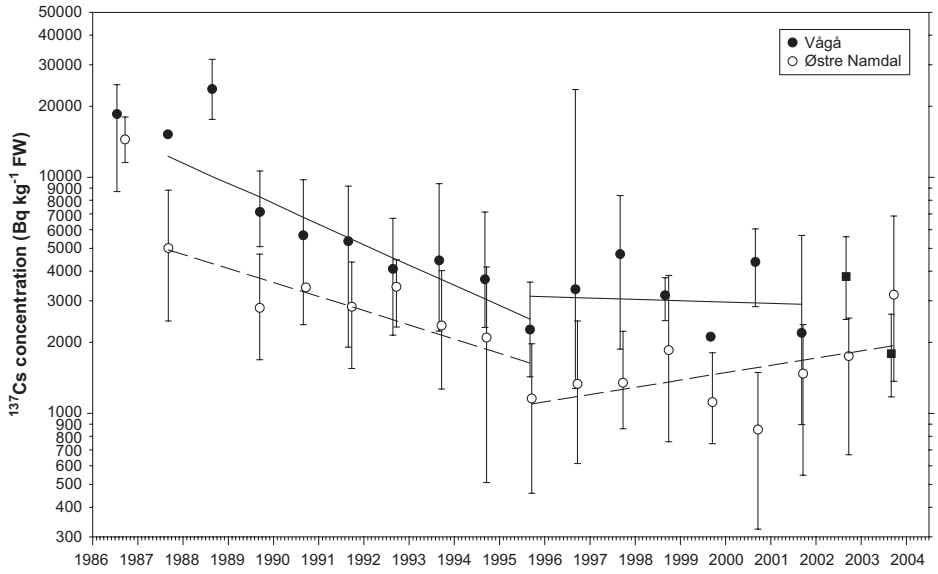

Fig. 4. Activity concentrations of ^{137}Cs in reindeer calves in the autumn in Vågå and Østre Namdal after the Chernobyl fallout: arithmetic mean values (Bq kg^{-1} FW) and ranges. Values in Vågå were from 11 August to 13 September and in Østre Namdal from 4 to 29 September (except in 1986 when sampling was done in July). No slaughtering or sampling occurred in Vågå in autumn 2002 and 2003 due to the abundance of fungi. The squares with bars in the figure indicate the expected ranges in average values in Vågå these years (estimated from observed average concentrations in the neighbouring district Lom, based on previously observed differences in ^{137}Cs concentrations in the two districts). The lines are fitted regression lines for the time periods 1987–1995 and 1995–2001 (Vågå) or 1995–2003 (Østre Namdal).

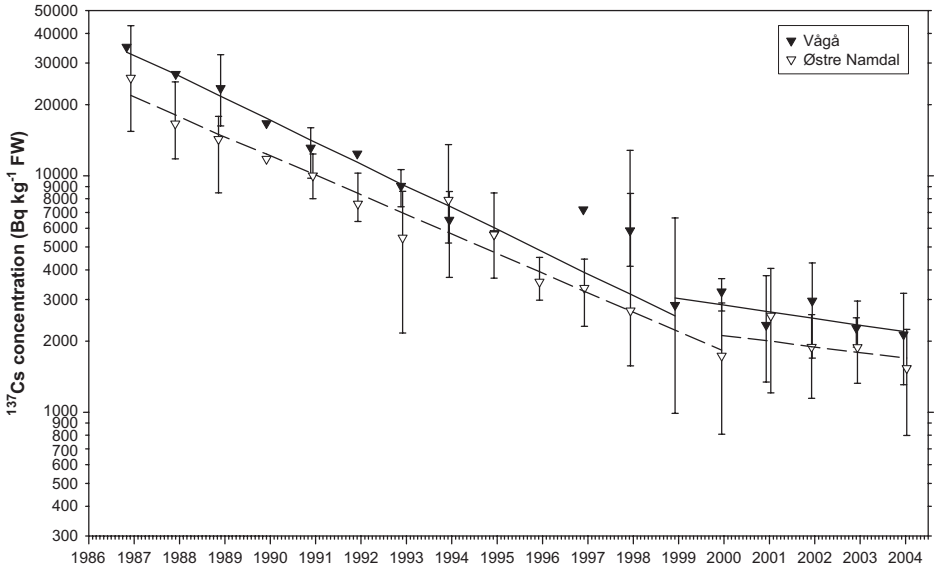

Fig. 5. Activity concentrations of ^{137}Cs in reindeer calves in early winter from Vågå and Østre Namdal after the Chernobyl fallout: arithmetic mean values (Bq kg^{-1} FW) and ranges. Values in Vågå were from 18 November to 16 December (except in 1986 when slaughtering took place on 1 November) and in Østre Namdal from 22 November to 13 December (except in 1988, 2001 and 2004 when slaughtering took place on 10 November and 11 and 12 January, respectively). The lines are fitted regression lines for the time periods 1986–1998 and 1998–2003 in Vågå, and the periods 1987–1999 and 1999–2003 in Østre Namdal.

'outlier'. Winter concentrations in Vågå continuously declined with a half-time of 3.24 ± 0.12 years until 1998 ($R^2 = 0.99$), with exception of the winter of 1996. Post 1998, there has been no detectable decline in winter concentrations.

Similar trends were observed in Østre Namdal, although activity concentrations in autumn were lower and declined continuously with a half-time of 5.0 ± 1.2 years ($R^2 = 0.74$) until 1995. Concentrations in winter declined until 1999 with a half-time of 3.63 ± 0.21 years ($R^2 = 0.97$), with no further decline observed thereafter.

Comparison of autumn and winter data in the periods where there has been no significant decline (i.e., in autumn during the period 1995–2001, and in winter during 1999–2003) showed that autumn ^{137}Cs activity concentrations in Vågå were on average about 2.6 times higher than in Østre Namdal, whilst the difference in winter was about a factor of 1.4 (paired t-test, $P < 0.05$).

3.4. Intake of ^{137}Cs by reindeer

The ^{137}Cs activity concentrations in sampled plants and lichen were used to estimate the daily ^{137}Cs intake by reindeer in autumn and winter (Table 5). The estimates indicate that, even when disregarding ^{137}Cs intake via ingested fungi, the autumn ^{137}Cs intakes would exceed those in early winter. However, the slower

excretion of caesium in reindeer during winter (Holleman and Luick, 1975; Skuterud et al., 2004) eliminates most of the seasonal difference in ^{137}Cs concentrations in reindeer. Using transfer coefficient (F_f) values for autumn and winter of 0.74 and 1.04 d kg^{-1}, respectively (adapted from Skuterud et al., 2004), ^{137}Cs concentrations in reindeer in autumn and winter would be 910 and 700 Bq kg^{-1} in Vågå, and 1100 and 730 Bq kg^{-1} in Østre Namdal. The estimates in Table 5 suggest that ingested lichens contributed to 65 and 53% of the winter ^{137}Cs concentrations in reindeer in Vågå and Østre Namdal, respectively, while the corresponding figures for autumn would be 14 and 9%.

4. Discussion

4.1. Long-term trends

In the analysis of long-term trends we chose to split the time period into two parts. The rationale behind this approach was that single exponential models could not satisfactorily describe the trends during the whole period (Figs. 4 and 5), and that non-linear regression analyses with double exponential models returned insignificant parameters. The selection of local concentration minima as years when long-term trends changed may have emphasized the insignificance in the decline in ^{137}Cs concentrations over the last years. Another approach could be to fit single exponential

Table 5
Estimated ^{137}Cs intake by reindeer in Vågå and Østre Namdal in the autumn and early winter based on consumption values adapted from Gaare and Staaland (1994) and average concentrations in different vegetation groups from Tables 1 and 2 (average of all mean values in each group)

District	Vegetation group	Activity concentration (Bq kg^{-1})	August–September		November–December	
			%[a]	Bq d^{-1}	%[a]	Bq d^{-1}
Vågå	Lichen	1900	21	170	71	460
	Graminoids	840	56	710	18	180
	Trees and heaths	480	13	94	11	63
	Herbs	1700	10	260	0	
	Total			1230		700
Østre Namdal	Lichen	1600	21	140	71	390
	Graminoids	1070	56	900	18	230
	Trees and heaths	870	13	170	11	120
	Herbs	2000	10	300	0	
	Total			1510		740

The resulting estimates of daily intake were corrected for the relatively low bioavailability of Chernobyl ^{137}Cs in lichen using an apparent absorption coefficient (A_a) of 17% for lichen (Skuterud et al., 2004). An A_a of 60% was assumed for vascular plants (Mayes et al., 1996). The daily intake values (Bq d^{-1}) in the table therefore refer to the net absorbed proportion of ingested activity.

[a] Percent of total consumption (2.5 kg DM d^{-1} in August–September and 2.0 kg DM d^{-1} in November–December).

models to different parts of the time period, e.g., as adopted by Åhman et al. (2001). As more data becomes available, double exponential models may prove to give appropriate descriptions of the long-term trends.

To our knowledge no previous studies have shown a similar change in rate of decline in winter ^{137}Cs concentrations in reindeer as suggested by the time series data from 1998–1999 (Fig. 5). However, Åhman et al., 2001 noted a slower rate of decline during 1995–2000 than during 1986–1991. A similar insignificant long-term decline in autumn ^{137}Cs concentrations, as that seen in Vågå and Østre Namdal from 1995 onwards (Fig. 4), was reported by Åhman et al. (2001) for an area of northern Sweden receiving less Chernobyl fallout. Furthermore, indications of significantly slower rates of decline from 1996 onwards have been noted from whole body measurements of the Saami population in central Norway in 1996, 1999 and 2002 (Mehli et al., 2000; Thørring et al., 2004).

The half-times estimated for the periods of continuous decline were within the range of 3–5 years, in agreement with previously reported values from Chernobyl fallout affected areas in Norway and Sweden (Pedersen et al., 1993; Åhman and Åhman, 1994; Amundsen, 1995; Hove and Staaland, 1997; Gaare et al., 2000; Åhman et al., 2001), whereas half-times of 6–9 years have been observed in the northern areas of Fennoscandia that had received less Chernobyl fallout (Westerlund et al., 1987; Skuterud et al., 1999; Åhman et al., 2001; Rissanen et al., 2003). As the long-term studies of radiocaesium in reindeer initiated in northern Fennoscandia in the 1960s were confined to areas botanically classified as 'slightly continental' (Moen, 1999), we hypothesize that differences in long-term trends among these northern areas and the Chernobyl affected areas is due to differences in climate. Lichens grow more rapidly when ample moisture is available (Kärenlampi, 1971), and the growing season is longer in the more southerly Chernobyl affected areas. This should encourage greater growth of lichen, which may result in shorter effective half-times for ^{137}Cs in these areas. However, in drier climates lichens constitute a larger proportion of the ground vegetation, and hence a larger proportion of the reindeer's diet. Lichens may therefore remain the dominant source of ingested ^{137}Cs for more years in the studied areas in northern Fennoscandia. Furthermore, the dominance of lichens as sources of dietary ^{137}Cs in more oceanic climates may be reduced by the generally higher ^{137}Cs concentrations in vascular plants in such areas due to higher abundance and lower nutritional quality of mires (Moen, 1999) (see below). If this hypothesis proves to be correct, it will have considerable influence on assessments of long-term consequences of radiocaesium fallout (such as in Howard et al., 2004) in large areas of Norway.

The similarity in half-times in autumn and winter slaughtered reindeer in Vågå until 1995 was probably due to significant lichen consumption during both seasons, as previously shown in this area (Staaland et al., 1995). The temporary deviations from the general decline in concentrations in early winter occurring in Østre Namdal in 1994 and Vågå in 1996 may be explained by grazing in areas with higher concentrations in lichen due to higher Chernobyl deposition density or in areas that had been grazed less intensively after the fallout occurred.

Due to generally shorter effective half-times for radiocaesium in lichens than vascular plants, it has been expected that the seasonal differences in ^{137}Cs

concentrations in reindeer would level out (e.g., Gaare and Staaland, 1994). The current results indicate that lichens were the primary source of ^{137}Cs to reindeer in winter in Vågå and Østre Namdal until about 1998–1999, whereas since then, vascular plants have become an important source of ^{137}Cs at this time of the year. This conclusion is supported by the similar ^{137}Cs concentrations in sampled lichens and vascular plants (Tables 1–3), and the ^{137}Cs intake estimates (Table 5).

Activity concentrations of ^{137}Cs in lichens decline due to washout, dilution by growth and by removal of contaminated parts by grazing (e.g., Martin and Koranda, 1971; Mattsson, 1975; Gaare et al., 2000). In *Cetraria nivalis*, *Cladonia mitis* and *C. stellaris* sampled in the Rondane and Dovrefjell mountains in southern Norway during the period 1987–1999, ^{137}Cs concentrations decreased with half-times of about 3.4–7.2, 5.2 and 3.6–6.4 years, respectively (Gaare et al., 2000). These half-time values are comparable to other estimates for lichens following fallout from the nuclear weapons tests as well as the Chernobyl fallout (see for instance summary by Synnott et al. (2000)).

According to Ehlken and Kirchner (1996) irreversible fixation of radiocaesium to soil particles would occur within approximately 3–7 years, whereas no aging process of caesium was apparent in peat. Similarly, the review by Rigol et al. (2002) concluded that two distinct patterns in time trends in radiocaesium transfer to plants were observed for organic soils. Most studies showed a decrease in transfer values with time, faster during the first 3 years and extremely slow afterwards (with soil-to-plant transfer remaining unchanged for crops grown 3 or more years after contamination of the soil). This pattern was very similar to that in mineral soils and agreed with that observed for the exchangeable fraction of 'organic-clay' soils (i.e., organic soils containing significant amount of clays (Rigol et al., 2002)). In contrast, other organic soils showed higher transfer values that remained almost constant over time (Rigol et al., 2002). Gastberger et al. (2000) summarized that remaining high transfer in semi-natural habitats was due to a combination of several factors, such as lack of dilution due to slow plant growth, climatic factors such as extended snow cover in combination with frozen soil profiles, runoff effects, water-logging and biological, physical and chemical soil characteristics. In addition, radiocaesium fixation in the biomass and cycling between living and dead parts of the vegetation are likely to be further reasons for higher availability of radiocaesium to plants (Gastberger et al., 2000).

Long-term studies of ^{137}Cs concentrations in individual plant species in mountainous areas of Scandinavia are limited. Concentrations of ^{137}Cs decreased by up to 80% from 1989–1990 to 1996–1999 in nearly all of 19 different vegetation species sampled in Vågå, although in three species (incl. *Betula pubescens*), concentrations increased by 10–35% (Liland et al., 2000). For species with deeper roots, an increase in plant concentrations can be expected as ^{137}Cs moves down the soil profile to the rooting depth (Belli et al., 1996). Concentrations in plants from the Rondane and Dovrefjell mountains showed pronounced variability between years, but concentrations declined significantly in *Betula nana* and *Deschampsia flexuosa* with effective half-times of 1.7–4.3 and 5.8 years, respectively (Gaare et al., 2000). Andersson et al. (2001) estimated the half-time in composite herbage samples

60–80 km south-east of Østre Namdal to be about 7 years during the period 1990–1997, while samples of *Carex*, *Betula* and *Salix* had relatively unaltered concentrations. In Irish blanket bog ecosystems radiocaesium concentrations decreased significantly during 1989–1997 in 26 out of 36 plant/site combinations (Synnott et al., 2000): estimated effective half-times for *Vaccinium myrtillus* were 3.6–6.2 years, *Eriophorum angustifolium* 2.9–22.0 years and *Eriophorum vaginatum* 11.2–17.7 years. Plants in the sedge and grass groups had slowest and least consistent rates of ^{137}Cs decline (Synnott et al., 2000).

Leccinium species sampled in Vågå showed relatively constant ^{137}Cs activity concentrations of 1.0–2.5 kBq kg^{-1} during the period 1990–1999 (Liland et al., 2000). Persistent ^{137}Cs concentrations in fungi and significant intake of fungi by reindeer when fruit bodies are available (Hove et al., 1990; Staaland et al., 1995) are likely reasons for higher ^{137}Cs concentrations observed in reindeer in autumn compared to those in winter in Vågå in 2000 and 2002, and in Østre Namdal in 2003 (Figs. 4 and 5).

The soil and vegetation samples in this study were collected during a period of 2 years, during which the ^{137}Cs activity decreased by 4.5% due to physical decay. The time period of 2 years was assumed to have negligible effects on the variability in observed ^{137}Cs concentration values, due to the above mentioned small changes in ^{137}Cs concentrations in several plants 15 years after fallout (with lichens being potential exceptions).

4.2. Differences in ^{137}Cs transfer

Significant differences in soil-to-plant transfer of ^{137}Cs in different areas have been shown in many studies (e.g., Ehlken and Kirchner, 1996). The most plausible explanation for the generally higher plant T_{ag} values observed in Østre Namdal is a likely combination of either lower clay mineral contents and higher organic matter content (see, for example, the reviews by Ehlken and Kirchner (2002), Kruyts and Delvaux (2002) and Rigol et al. (2002)) or more nutrient-poor soils (e.g. Varskog et al., 1994; Bunzl et al., 2000). In addition, differences in clay mineral species may be a significant factor, with illite and vermiculite being particularly effective Cs binders (Kruyts and Delvaux, 2002; Rigol et al., 2002). There is higher frequency of mires in Østre Namdal, and mires in this climate are generally more nutrient-poor than in more continental climates (Moen, 1999). The hypothesis about the importance of clay mineral and organic matter content is supported by an average 35% higher soil density in Vågå ($P < 0.05$) and field observations of generally thicker organic horizons in Østre Namdal.

The higher ^{137}Cs concentrations in lichens relative to the deposition density in Østre Namdal may be due to a higher proportion of the original Chernobyl fallout still remaining within the thalli. Mattsson (1975) suggested that grazing was the dominant factor for elimination of ^{137}Cs from lichen carpets in areas frequented by reindeer. However, lichens in Østre Namdal are generally interspersed and covered with other food plants, which probably reduces the grazing pressure on lichen thalli. Furthermore, leached ^{137}Cs from taller plants and trapped litterfall may have been absorbed by the lichen thalli, resulting in a slower decline in ^{137}Cs concentrations (as suggested by Synnott et al. (2000)).

The higher ^{137}Cs soil-to-plant transfer found in Østre Namdal for most species reduced the difference in ^{137}Cs reindeer concentrations between the two areas relative to deposition density. On the other hand, differences in concentrations in reindeer, especially in autumn, were larger than indicated by the mostly insignificant differences in concentrations in the sampled plants and lichens, and the resulting ^{137}Cs intake estimates. The estimated autumn concentrations in reindeer in Østre Namdal (Section 3.4) were in agreement with the lowest observed mean values (Fig. 4), which is reasonable since the intake estimates did not consider ingestion of fungi. However, the estimated ^{137}Cs concentrations in winter in Østre Namdal, and both seasons in Vågå, were 2–3 times lower than those observed. Choosing sample locations and sample vegetation representative of what was actually grazed by the reindeer in these vast pastures was virtually impossible, and the discrepancies between estimated and observed concentrations might be accounted for by grazing in areas of the pasture with relatively high deposition densities (average values from all locations within each district were applied in the intake estimates). In Vågå, samples from the northern winter grazing areas with relatively low deposition densities were included in the estimates, whereas no samples were collected in the more contaminated southern summer grazing area.

The calculations of ^{137}Cs intake by reindeer assumed similar diets in Vågå and Østre Namdal, and with non-significant differences in ^{137}Cs concentrations in food plants and lichens, the estimated intakes in the two areas were comparable (Table 5). The applied autumn diet in Vågå was in agreement with the preliminary results of rumen analyses in the current study (unpublished data), whereas the winter diet contained more lichen than the rumen samples indicated. No supplementary information is available on diets in Østre Namdal.

5. Conclusions

1. There has been no detectable decrease in ^{137}Cs concentrations in reindeer in Vågå and Østre Namdal during the last few years, and seasonal differences in ^{137}Cs concentrations in reindeer have been less pronounced than the first years after the Chernobyl fallout, probably because of persistently elevated ^{137}Cs concentrations in fodder plants. Furthermore, ^{137}Cs concentrations in the autumn have occasionally exceeded those in winter. The study suggests that future long-term decline in ^{137}Cs concentrations in reindeer will be governed principally by the physical decay.
2. The transfer of ^{137}Cs through the food chain from soil to reindeer appeared significantly higher in Østre Namdal than Vågå, especially in winter. Higher soil-to-plant transfer was demonstrated for 7 of 12 vascular plants sampled, and 2 of 3 lichen species in Østre Namdal contained relatively higher ^{137}Cs concentrations than in Vågå.
3. Although transfer of ^{137}Cs in the food chain leading to reindeer has been thoroughly studied and considered relatively well known, the slower rates of decline and regional differences in transfer demonstrated in this study, highlight

the need for continuing monitoring and research in this field of radioecology. It is hypothesized that climate can cause significant regional differences in long-term transfer of radiocaesium in the soil–plant/lichen–reindeer food chain. Without relatively elevated intervention limits for ^{137}Cs concentrations in reindeer meat, reindeer herding in the contaminated areas of Norway is likely to suffer practical consequences of the Chernobyl fallout (i.e., countermeasure application) for many years to come. Additionally, this situation may require special consideration in future emergency preparedness plans.

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Paper V

^{90}Sr, ^{210}Po and ^{210}Pb in lichen and reindeer in Norway

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^{90}Sr, ^{210}Po and ^{210}Pb in lichen and reindeer in Norway

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Abstract

Concentrations of ^{90}Sr, ^{210}Po and ^{210}Pb in lichen and reindeer were studied in central (Østre Namdal) and southern Norway (Vågå) during 2000–2003. The study focussed on potential differences in concentrations of these nuclides in reindeer of different ages. Concentrations of ^{90}Sr in bones of ~10 year old adult females were about 40% higher than those in calves' antlers (^{90}Sr concentrations in antlers and bones of calves are similar), while the available data from Vågå suggest that ^{90}Sr concentrations in reindeer calves decreased with an effective ecological half-time of 9.03 ± 0.06 years during 1988–2002. Furthermore, ^{90}Sr concentrations were 50–80% higher in bone of reindeer of a similar age from Vågå compared to those from Østre Namdal. Concentrations of ^{210}Po and ^{210}Pb in muscle and liver tissues were comparable to those reported for reindeer in other Nordic areas, with no significant

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difference in ^{210}Po and ^{210}Pb concentrations between adults and calves or between reindeer from the two different study areas.

Keywords: Strontium-90; Lead-210; Polonium-210; Reindeer; Lichen; Natural radioactivity

1. Introduction

Since the early 1960s, numerous studies have reported higher concentrations of nuclear weapons fallout ^{90}Sr, ^{137}Cs and the naturally occurring nuclides ^{210}Pb and ^{210}Po in reindeer and caribou compared to other animal species (Paakkola and Miettinen, 1963; Holtzman, 1966; Hill, 1967; Kauranen and Miettinen, 1967; Nevstrueva et al., 1967; Persson, 1971). Additional ^{90}Sr and ^{137}Cs contamination of reindeer in Norway occurred through the 1986 Chernobyl accident (Staaland et al., 1991; Skuterud et al., 2005). Although total ^{90}Sr releases from the Chernobyl accident were 11–12% of that of ^{137}Cs (CEC, 1998), ^{90}Sr deposition in Norway was only 1–3% of that of ^{137}Cs (Bjørnstad and Salbu, 1992) due to faster deposition of ^{90}Sr from the atmosphere compared to ^{137}Cs (CEC, 1998). Nevertheless, following the Chernobyl accident, the most contaminated areas in Norway received ^{90}Sr deposition densities comparable to the integrated nuclear weapons tests fallout (Bjørnstad et al., 1990).

Sr-90 is not bound as strongly as radiocaesium in lichens (Nevstrueva et al., 1967) and shorter effective half-times for ^{90}Sr in lichens have been found (Persson, 1971; Heinrich et al., 1999). In soil, a relatively large proportion of ^{90}Sr is available for plant uptake (e.g., Coughtrey and Thorne, 1983), and root uptake of ^{90}Sr in alpine pasture plants has been found to be comparable to or higher than that of ^{137}Cs (Bjørnstad et al., 1990; Gastberger et al., 2000; Schimmack et al., 2003). In animals, Sr accumulates in bone in a manner similar to calcium (as phosphates), and antlers of cervids have been used previously as monitors of environmental ^{90}Sr levels (e.g., Strandberg and Strandgaard, 1995; Schönhofer et al., 1994; Tiller and Poston, 2000). Sr-90 in antlers reflects the dietary intake during their period of growth, and concentrations are similar in both bone and antlers of reindeer calves (Hognestad and Lie, 1998). Differences in ^{90}Sr concentrations between selected larger bones in the reindeer body are insignificant (Staaland et al., 1991; Hognestad and Lie, 1998). Sr-90 concentrations in reindeer do not exhibit any seasonal trends (Staaland et al., 1991) as do concentrations of ^{137}Cs (e.g. Skuterud et al., 2005), ^{210}Pb and ^{210}Po (Kauranen and Miettinen, 1967, 1969).

Pb-210 and ^{210}Po are products of the ^{238}U decay series and are released into the atmosphere via the decay of ^{222}Rn. Pb-210 has a mean atmospheric residence time of about 29 days (Jaworowski, 1969; Salmon et al., 1998b) and is continuously deposited from the atmosphere in association with aerosols at a rate of about 55 Bq m^{-2} year^{-1} over Scandinavia (El-Daoushy, 1988). Generally, atmospheric ^{210}Pb concentrations are related to the size of the underlying landmasses, and oceanic areas including islands, ice and snow covered lands have reduced

atmospheric ^{210}Pb concentrations (El-Daoushy, 1988). However, deposition increases with increasing precipitation (Hill, 1960).

Vegetation is contaminated by ^{210}Pb and ^{210}Po predominantly by direct deposition (Ewers et al., 2003). Lichens are slow growing perennials that have high interception potentials for aerosols in precipitation, and therefore contain significantly higher ^{210}Pb concentrations than vascular plants (Holtzman, 1966; Kauranen and Miettinen, 1967, 1969; Jaworowski, 1969) and fungi (Skwarzec and Jakusik, 2003). The ^{210}Po/^{210}Pb activity ratio in lichen is typically ~1 as ^{210}Po approaches secular equilibrium with ^{210}Pb (Kauranen and Miettinen, 1967, 1969; Mattsson and Persson, 1972; Thomas et al., 1994). In animals, lead accumulates preferentially in certain soft tissues and bone (ICRP, 1994), particularly on bone surfaces (Salmon et al., 1998a), but skeletal accumulation of Pb occurs at a slower rate than for Ca and Sr due to the relatively higher accumulation of Pb in soft tissues (ICRP, 1994). Po-210 in reindeer is therefore due to both intake of ^{210}Po per se, and ^{210}Po originating from in situ ^{210}Pb decay (Kauranen and Miettinen, 1969; Thomas et al., 1994; Macdonald et al., 1996).

This paper presents results for ^{90}Sr, ^{210}Pb and ^{210}Po in lichen and reindeer (Rangifer tarandus tarandus) from two reindeer herding districts in Norway. The districts studied were selected due to differences in climate that could potentially affect radionuclide concentrations in reindeer, with one district exposed to a more oceanic climate than any areas previously studied in the Nordic countries. In order to test the hypothesis that age is an important parameter in determining concentrations of these nuclides in reindeer, a more systematic approach was adopted than previously employed. Furthermore, the study of ^{90}Sr extends one of the few available time series on ^{90}Sr contamination in Norway (Staaland et al., 1991; Hognestad and Lie, 1998). To our knowledge, no studies of ^{210}Pb and ^{210}Po in lichens and reindeer in Norway have been reported before. More extensive results on ^{137}Cs in soil, vegetation and reindeer from the same areas have been reported separately (Skuterud et al., 2005).

2. Materials and methods

2.1. Study areas and field work

Sampling of soil, vegetation and tissues of semi-domestic reindeer was conducted in the Vågå and Østre Namdal reindeer herding districts (Fig. 1) during the period 2000–2003. Both these areas received significant Chernobyl fallout, with current ^{137}Cs deposition densities of about 80 and 40 kBq m^{-2} in the Vågå and Østre Namdal districts, respectively (Skuterud et al., 2005). Direct measurement of ^{90}Sr deposition densities in the Vågå area is limited to one soil sample showing a total ^{90}Sr deposition density of about 5 kBq m^{-2} in 1987 (Bjørnstad et al., 1990). The climate in Østre Namdal is more oceanic, with higher precipitation (total annual precipitation about 1000–1500 mm compared to 280–1200 mm in different parts of the Vågå district) and slightly higher average temperatures. As such, Østre Namdal is botanically classified

Fig. 1. The Østre Namdal (ØN) and Vågå (V) reindeer herding districts with sample locations indicated. The total area of the Vågå district is 1357 km^2 whereas the total area of the Østre Namdal district is 6607 km^2 (only the north-eastern half of the Østre Namdal district is shown on a large scale).

in the "slightly oceanic section" whereas Vågå is categorised in the more continental "indifferent section" (Moen, 1999). Further details on the study areas are given in Skuterud et al. (2005).

Sampling of soil and vegetation was carried out in the summer, autumn and early winter grazing area of Østre Namdal, and in the autumn and winter grazing areas of Vågå. Samples were obtained from a total of 12 and 11 locations within the Vågå and Østre Namdal pastures, respectively, each location being roughly $50 \times 50\ m^2$ and encompassing different plant communities. Samples of lichens were collected from across each location (one sample per species per location). Most locations were at or above the limit of woodland. The sampling was carried out during July–September of 2001–2003.

Lichen samples from four randomly selected locations within each pasture were chosen for ^{90}Sr determination, whereas samples from five locations were submitted for ^{210}Po determination. Since differences between radionuclide concentrations can be significant in different lichen species growing at the same location (e.g. Thomas et al., 1994), only samples of *Cladonia arbuscula* (mostly the subspecies *C. arbuscula* ssp. *mitis* (Santesson, 1993)) were analysed to enable comparison of concentrations in the two districts. Dead parts of lichen were removed prior to drying to constant mass at 70 °C.

Samples of antlers, metacarpal bone, neck muscle, and liver tissues of reindeer calves (< 1 year) and older females were obtained during slaughter in autumn and winter. Age was determined from ear tags. The sex of individual calves was not recorded.

2.2. ^{90}Sr and Ca

Concentrations of ^{90}Sr were determined in antlers from calves and in antlers and long bones from adult females slaughtered during the winter of 2000–2001. Bones of calves were not analysed since ^{90}Sr concentrations in long bones were expected to be similar to those in antlers (Hognestad and Lie, 1998). In addition, antlers from calves slaughtered during the winter of 2002–2003 were analysed. Samples from four calves and four females from each sampling were analysed, except for females from Vågå in December 2000 ($n = 3$). Five to seven centimetre of the middle part of the right metacarpal bone (see illustration in Staaland et al. (1991)), and 10 cm of one antler tip were used for analysis. Soft tissues and marrow were dissected from the bone.

Determination of ^{90}Sr was carried out at Institute of Energy Technology (IFE, Norway). The samples were dried to constant weight at 105 °C, and ashed at 450 °C after addition of ^{85}Sr tracer. Bones and antlers were analysed using a modified HASL 300 procedure (USDOE, 1997; Varskog et al., 1997). Following chemical separation of Sr using decomposition in fuming HNO_3 followed by precipitation of hydroxide and chromate, the samples were left for in-growth of ^{90}Y. Yttrium was precipitated as oxalate, collected by filtration, and the ^{90}Y activity measured by low-level anticoincidence beta counters. Chemical yield was determined by ^{85}Sr tracer and by titration of Y with EDTA, respectively. Sr in lichen samples was separated and purified using an Sr-resin following a slight modification of Horwitz et al. (1992) and Eichrom (1999). Yttrium separation and counting was performed as

described for bones and antlers. Concentrations of ^{90}Sr were corrected for physical decay to the sampling date. Uncertainties in determined ^{90}Sr activities were 5–7.5%.

Calcium concentrations in antlers and bone were determined by flame atomic absorption spectrometry at the Department of Chemistry, Norwegian University of Science and Technology. Ashed antlers and bones (from IFE) were decomposed in fuming HNO_3, and lanthanum was added to reduce chemical interferences.

Concentrations of ^{90}Sr in bone and antlers are given as Bq ^{90}Sr per g Ca (Bq g^{-1} Ca).

2.3. ^{210}Pb and ^{210}Po

Concentrations of ^{210}Po were determined in muscle tissue of four calves and three adult females from Vågå in December 2000, and in muscle and liver tissue of calves and females from Vågå and Østre Namdal in December 2002 ($n = 7$ for females from Østre Namdal, otherwise $n = 10$). Pb-210 was determined only in liver tissue.

The samples from December 2000 were analysed at the Norwegian Radiation Protection Authority according to Chen et al. (2001). Other samples were analysed in the Department of Radiation Physics at Lund University Hospital (Sweden) according to Flynn (1968). Uncertainties due to counting statistics and uncertainty in tracer calibration in determined ^{210}Po activities in lichen amounted to 5.4–6.1%. The ^{210}Po activity in one calf muscle sample from Østre Namdal was below the minimum detectable activity (corresponding to ca. 4 Bq kg^{-1}) and was excluded from statistical analyses. Uncertainties in the determined ^{210}Po activities typically ranged from 6.8 to 12%, with two muscle samples with uncertainties of 17 and 19%. Uncertainties in determined ^{210}Pb activities in liver tissue ranged from 4.9 to 6.9%.

The contribution by "supported" ^{210}Po (from decay of ^{210}Pb via ^{210}Bi) to the ^{210}Po in muscle tissue was assumed negligible according to reported ^{210}Po/^{210}Pb ratios of 20–100 in muscle tissue during winter (Holtzman, 1968; Kauranen et al., 1971; Mattsson and Persson, 1972). Concentrations of ^{210}Po in muscle tissue were therefore obtained by decay correction to the sampling date. However, the concentration of ^{210}Po in liver tissue at the time of analysis (A_{Po}) was due to a combination of remaining "unsupported" ^{210}Po present at sampling date (A^0_{Po}) in addition to "supported" ^{210}Po (A'_{Po}) produced by the in situ decay of ^{210}Pb between the time of sampling and analysis:

$$A_{Po} = A^0_{Po} \times e^{-\lambda_{Po} t} + A'_{Po}$$

Here λ_{Po} is the decay constant for ^{210}Po (equivalently for ^{210}Pb and ^{210}Bi below). The supported ^{210}Po activity (A'_{Po}) was estimated from the ^{210}Pb activity at the time of sampling (A^0_{Pb}) using Batesman's equation (Ivanovich and Harmon, 1992):

$$A_{Po} = A^0_{Po} \times e^{-\lambda_{Po} t} + A^0_{Pb}\left(\frac{\lambda_{Bi}}{\lambda_{Bi}-\lambda_{Pb}}\ \frac{\lambda_{Po}}{\lambda_{Po}-\lambda_{Pb}} e^{-\lambda_{Pb} t} + \frac{\lambda_{Bi}}{\lambda_{Pb}-\lambda_{Bi}}\ \frac{\lambda_{Po}}{\lambda_{Po}-\lambda_{Bi}} e^{-\lambda_{Bi} t} + \frac{\lambda_{Bi}}{\lambda_{Bi}-\lambda_{Po}}\ \frac{\lambda_{Po}}{\lambda_{Pb}-\lambda_{Po}} e^{-\lambda_{Po} t}\right)$$

Solving the equation for A^0_{Po} gives the "unsupported" ^{210}Po activity at the time of sampling. The ^{210}Pb activity at the time of sampling (A^0_{Pb}) was determined by decay correcting ^{210}Pb activities from the time of analysis. Physical half-lives of the nuclides are 22.3 years, 5.01 days and 138 days for ^{210}Pb, ^{210}Bi and ^{210}Po, respectively.

Determined ^{210}Po concentrations in lichen were decay corrected to the sampling date assuming that ^{210}Po was in transient equilibrium with ^{210}Pb.

Statistical analyses (analysis of variance (ANOVA), Student *t*-tests, non-linear regression, Pearson's correlation) were carried out using SPSS for Windows release 11. Nuclide concentrations are presented as mean ± standard deviation (SD). Ratios of ^{210}Po to ^{210}Pb in liver were log-transformed prior to statistical analyses (to harmonize variances). All concentrations are given on a dry matter (DM) basis.

3. Results

3.1. Lichen

Table 1 shows determined ^{90}Sr and ^{210}Po activity concentrations in lichen samples together with the ^{137}Cs activity concentrations in the samples (from Skuterud et al. (2005)). There were no significant differences in concentrations of any of the nuclides between the two areas (*t*-test, $P > 0.05$), although the average ^{90}Sr concentration in Vågå (13.2 ± 5.6 Bq kg^{-1}) was nearly twice that in Østre Namdal (7.3 ± 1.9 Bq kg^{-1}; $P = 0.12$). Neither was there any correlation between individual nuclide concentrations in the lichen samples (Pearson's correlation coefficient, $P > 0.05$). The average ^{90}Sr/^{137}Cs ratios were not significantly different in the two areas (the ratio ranged from 0.25 to 1.1%; *t*-test, $P > 0.05$).

Table 1
Activity concentrations of ^{90}Sr, ^{137}Cs and ^{210}Po in the living part of *Cladonia arbuscula* at different locations in Vågå and Østre Namdal (Bq kg^{-1} DM)

District	Location	Sampling date	^{90}Sr	^{137}Cs	^{210}Po
Vågå	1	August 2001	9.1 ± 1.1	3470 ± 280	121 ± 4
	2	August 2001	12.1 ± 1.5	1460 ± 120	–
	3	August 2001	–	1115 ± 96	104 ± 3
	7	August 2001	22.6 ± 2.8	1910 ± 160	192 ± 6
	11	August 2001	12.0 ± 1.5	3060 ± 250	70 ± 2
	12	September 2002	–	1890 ± 160	212 ± 5
Østre namdal	D-2	September 2001	6.0 ± 0.7	1930 ± 160	119 ± 3
	D-4	September 2001	9.7 ± 1.2	1380 ± 120	171 ± 4
	0	July 2003	7.9 ± 1.0	3180 ± 250	111 ± 3
	1	July 2003	–	3510 ± 270	154 ± 3
	S-1	July 2003	5.5 ± 0.7	1430 ± 110	–
	J-3	July 2003	–	1570 ± 120	151 ± 4

3.2. Reindeer

Table 2 shows the ^{90}Sr concentrations in reindeer tissues. All females were 10 years old, with the exception of one of the three females sampled in Vågå December 2000 which was 7 years. As there was no apparent age-related difference in ^{90}Sr concentrations in the bone of these three females, the average of all three animals was used for statistical analyses.

Concentrations of ^{90}Sr were consistently higher in samples from Vågå than from Østre Namdal (two-way ANOVA, $P < 0.05$), whereas there was neither any difference between antlers from calves and females, nor was there any change in concentrations in calf antlers from winter 2000–2001 to 2002–2003. ^{90}Sr concentrations were on average 23% higher in female's bone than in their antlers (paired t-test, $P = 0.025$). There were no differences in Ca contents in either bone or antlers between the two districts or between age groups (two-way ANOVA, $P > 0.05$). Average Ca concentrations in antlers and bone were 166 ± 12 g kg^{-1} and 221.0 ± 5.3 g kg^{-1}, respectively.

Fig. 2 shows some of the values from Table 2 together with published data on ^{90}Sr in bones and antlers of reindeer from Vågå (Staaland et al., 1991; Hognestad and Lie, 1998). Hognestad and Lie (1998) reported ^{90}Sr concentrations in two animals born in 1989 and three animals born in 1991, which were slaughtered in 1996. The average ^{90}Sr concentration in bone of these animals aged 5 and 7 years (i.e., 3.1 ± 1.3 Bq g^{-1} Ca, Fig. 2) was not significantly different from that in the three females slaughtered in 2000 in this study. Furthermore, the mean ^{90}Sr concentration (i.e., 4.2 ± 1.3 Bq g^{-1} Ca) in antlers of three calves born in Vågå in 1996 (Hognestad and Lie, 1998) was not significantly different from that observed in this study for calves born in 2000 or 2002. However, by using ^{90}Sr bone concentrations reported for calves in 1988 (Staaland et al., 1991) as an estimate of ^{90}Sr antler concentrations (since concentrations in bone and antlers of calves are similar (Hognestad and Lie, 1998)), a significant decline in ^{90}Sr in antlers with an effective ecological half-time of 9.03 ± 0.06 years (mean $\pm$ SE; non-linear regression with log-transformed average concentrations; $R^2 = 0.99$) can be estimated (dotted line in Fig. 2).

Fig. 3 shows the ^{210}Po and ^{210}Pb activity concentrations in tissues versus age in Vågå and Østre Namdal in December 2002. There was a tendency towards higher

Table 2
Concentrations of ^{90}Sr in bone and antlers of reindeer in Vågå and Østre Namdal slaughtered in winter 2000–2001 and 2002 (Bq g^{-1} Ca; $n = 4$ except for females from Vågå where $n = 3$)

District	Sampling date	Females		Calves
		Bone	Antler	Antler
Vågå	December 2000	4.09 ± 0.97	3.6 ± 1.3	2.92 ± 0.44
	December 2002			2.77 ± 0.51
Østre Namdal	February 2001	2.69 ± 0.31	1.98 ± 0.29	1.89 ± 0.23
	December 2002			1.87 ± 0.16

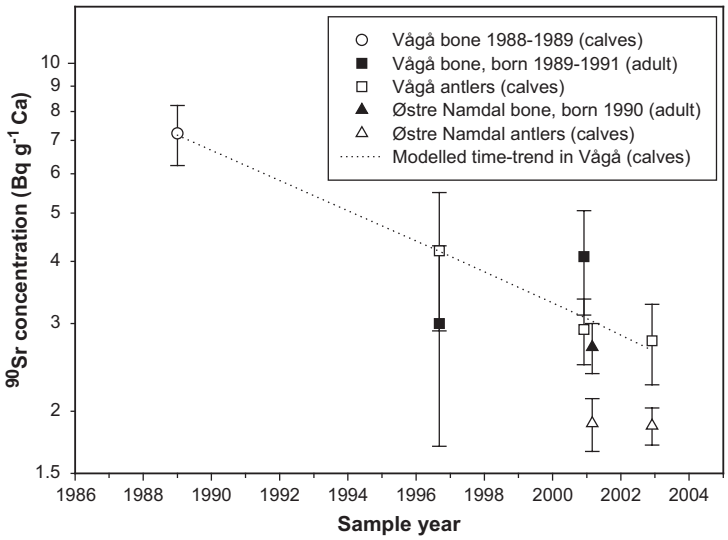

Fig. 2. Concentrations of ^{90}Sr in reindeer in Vågå and Østre Namdal after the Chernobyl accident, and estimated time-trend in concentrations in reindeer calves from Vågå. Concentrations in 1988–1989 were taken from Staaland et al. (1991), in 1996 from Hognestad and Lie (1998), and in 2000–2001 and 2002 from Table 2 (this study).

^{210}Po and ^{210}Pb concentrations in calves from Vågå than Østre Namdal, but the difference was statistically significant only for ^{210}Pb in liver tissue (t-test, $P < 0.01$). There were no significant differences in average concentrations of these nuclides between age groups within the same districts. However, there were statistically significant positive correlations (Pearson's correlation coefficient, $P < 0.05$) between concentration and age for ^{210}Po in muscle ($R = 0.46$) and in liver tissue ($R = 0.53$) in Østre Namdal, and ^{210}Pb in liver tissue in Vågå ($R = 0.45$). Geometric mean ^{210}Po/^{210}Pb ratios in liver tissue ranged from 3.6 in females from Vågå to 6.2 in females from Østre Namdal, with an overall geometric mean of 5.1 (geometric SD 1.9). There were no significant differences in ^{210}Po/^{210}Pb ratios between age groups.

Mean ^{210}Po activity concentrations in muscle tissue, December 2000, in Vågå were 23.7 ± 3.7 and 35.5 ± 9.2 Bq kg^{-1} DM in calves and females (>7 years) respectively, not significantly different from those in 2002.

The transfer coefficient F_f (i.e., the amount of the animal's daily radionuclide intake that is transferred to 1 kg of the animal product at equilibrium) (Ward and Johnson, 1986) for ^{210}Po to reindeer meat could be estimated from the determined ^{210}Po concentrations in lichen and reindeer muscle and an estimated lichen intake during winter of about 1.2 kg DM day^{-1} (Gaare and Staaland, 1994). This gave F_f values in the range 0.04–0.06 d kg^{-1} (FW).

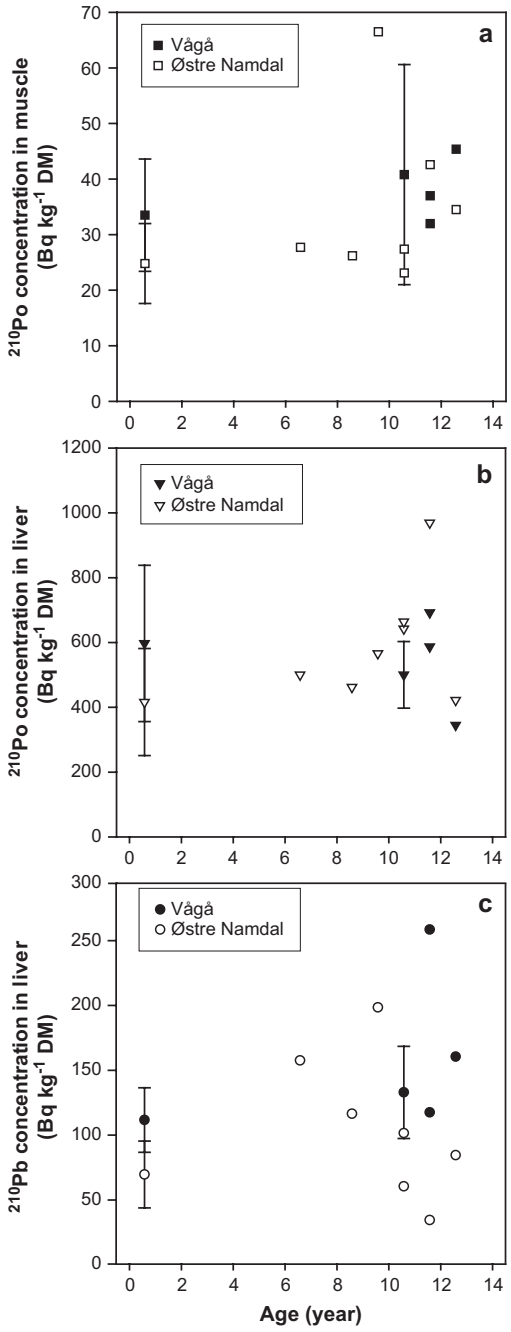

Fig. 3. Activity concentrations of ^{210}Po in (a) muscle and (b) liver tissue and ^{210}Pb in (c) liver tissue versus the age of the animal (Bq kg^{-1} DM, arithmetic mean $\pm$ SD). Calves: $n = 9$–10. Female average from Vågå: $n = 7$.

4. Discussion

4.1. ^{90}Sr

Strontium accumulated in trabecular bone is resorbed at a higher rate than that documented for cortical bone (ICRP, 1994), due to higher renewal rates (Ganong, 2001). In humans, trabecular and cortical bones are renewed at rates of about 20 and 4% per year, respectively (Ganong, 2001). Correspondingly, Farris et al. (1967) estimated that the turnover rates of stable Sr in metacarpal and femur bone of mule deer (*Odocoileus hemionus hemionus*) decreased from about 19% in juvenile to 6% in adult deer (corresponding to biological half-times of about 3.3 and 11 years, respectively).

In the relatively low number of samples from the respective years, no significant difference was detected in ^{90}Sr concentrations in bone of reindeer born around 1990 and slaughtered in 1996 compared to those slaughtered in 2000, or in ^{90}Sr concentrations in antlers of calves slaughtered in 1996, 2000 and 2002 (Fig. 2). Furthermore, Hognestad and Lie (1998) observed no decline in ^{90}Sr concentrations in bone from reindeer in Vågå born during the period 1987–1996 and slaughtered in 1996 (samples of 1–3 animals born each year). However, by including the data from Staaland et al. (1991) a significant decline in ^{90}Sr concentrations in antlers was estimated. Since some of the ^{90}Sr intake by calves (i.e., via milk) would have been originally accumulated by their mothers, the estimated effective ecological half-time of 9.03 ± 0.06 years does not reflect a decrease in ^{90}Sr concentrations in vegetation alone. The modelled time-trend from Vågå (Fig. 2) suggests that average concentrations in bone of reindeer born in 1990 decreased from about 6.1 Bq g^{-1} Ca (in 1990) to 3.0–4.1 Bq g^{-1} Ca by 5–10 years of age. This decrease in ^{90}Sr concentrations in adult female reindeer probably reflects a combination of declining ^{90}Sr concentrations in pasture vegetation and the effect of bone renewal (including mobilization of skeletal stores of mineral elements during pregnancy and lactation (Braithwaite, 1983)). However, any decrease will be off-set by the continuous accumulation of ^{90}Sr throughout the reindeer's lifespan (Farris et al., 1967).

In four lichen samples collected in central Norway (Nord-Trøndelag county) in the end of June 1986, ^{90}Sr activity concentrations ranged from 63–230 Bq kg^{-1}, 0.9–1.9% of the ^{137}Cs activity concentration (NRPA, unpublished). A decrease in ^{90}Sr activity concentrations and $^{90}Sr/^{137}Cs$ ratios in the lichen from Østre Namdal until 2001–2003 (Table 1) was expected due to the relatively short effective half-time of about 1.0 year for ^{90}Sr in lichen (Nevstrueva et al., 1967; Persson, 1971; Heinrich et al., 1999) and due to removal of ^{90}Sr from the lichen thalli by grazing. The ^{90}Sr activity concentrations in the lichen samples from Vågå appeared lower than the 55 Bq kg^{-1} determined in a composite lichen sample from Vågå in 1996 (Hognestad and Lie, 1998), but the difference might also be attributed to differences in ^{90}Sr concentrations between lichen species, or local differences in deposition densities. Activity concentrations of ^{90}Sr in vascular plants from the same area ranged from 30–67 Bq kg^{-1} DM in 1996 (Hognestad and Lie, 1998). Lichens may therefore be of lesser importance for ^{90}Sr intake by reindeer compared to intake of ^{137}Cs, ^{210}Pb and

^{210}Po. Nevertheless, the differences in average ^{90}Sr concentrations in reindeer from Vågå and Østre Namdal were nearly as large as the difference in average concentrations in lichen.

4.2. *^{210}Pb and ^{210}Po*

This study with selective sampling of reindeer calves and adult females could not identify any differences in ^{210}Po/^{210}Pb ratios in liver tissue related to age, and results on differences in ^{210}Pb and ^{210}Po concentrations in muscle and liver tissue due to age were equivocal. Sampling of tissues from reindeer of different ages was conducted to study the potential effect of higher ^{210}Pb contents in older animals on subsequent soft tissue ^{210}Po concentrations, as Thomas et al. (1994) estimated that redistributed ^{210}Po from skeletal ^{210}Pb decay accounted for the excess ^{210}Po found in soft tissues (i.e., when ^{210}Po/^{210}Pb ratios exceed 1). However, Salmon et al. (1998a) stated that no independent movement of ^{210}Po from the site of ^{210}Pb decay in bone has been observed. Furthermore, Macdonald et al. (1996) found no significant correlation between age and ^{210}Pb in caribou bone, and studies on stable lead in liver of Norwegian reindeer have given equivocal results with regard to age correlation (Kålås and Myklebust, 1994; Strand et al., 1995). In addition, any age effect could partly be masked by the higher absorption of lead at younger age (ICRP, 1994).

Since no significant role of age on ^{210}Po concentrations in muscle and liver tissue of reindeer was identified in the current study, we can provide no further evidence for the role of skeletal ^{210}Pb decay on subsequent soft tissue ^{210}Po concentrations. However, a 4-fold difference in ^{210}Po concentrations in muscle tissue between different seasons (Kauranen and Miettinen, 1969) indicates that dietary intake of ^{210}Po governs the soft tissue ^{210}Po concentrations. If ^{210}Po was continuously redistributed to soft tissues from the skeletal pool, higher ^{210}Po/^{210}Pb ratios would be expected during seasons with reduced lichen intake. However, no indication of such an increase in seasonal ^{210}Po/^{210}Pb ratios has been reported (Kauranen and Miettinen, 1969).

Although there was a tendency towards higher ^{210}Po and ^{210}Pb activity concentrations in calves from Vågå, no significant differences between the two districts in the concentrations of these nuclides were apparent. These results appear to reflect the comparable average ^{210}Po concentrations in lichen samples. It might have been expected that the more oceanic climate in Østre Namdal would generate lower atmospheric ^{210}Pb concentrations (El-Daoushy, 1988), but higher deposition of ^{210}Pb (Hill, 1960). Furthermore, a lower grazing pressure on lichens in Østre Namdal than in Vågå (Skuterud et al., 2005) would result in relatively high effective ^{210}Pb and ^{210}Po concentrations in lichens since the tops of the lichen thalli generally contain the highest ^{210}Pb concentrations (Persson et al., 1974). However, the hypothesised faster growth rates of lichen in the Østre Namdal climate (Skuterud et al., 2005) could potentially dilute ^{210}Pb concentrations in lichen, as observed in areas in Finland (Kauranen and Miettinen, 1967, 1969). Additionally, consideration should be given to the possible influence of differences in the composition of lichen species in the diet between the two regions, as Thomas et al. (1994) consistently found 1.4 to 2.8-fold higher concentrations of ^{210}Pb and ^{210}Po in *Cetraria nivalis*

than in *C. arbuscula* ssp. *mitis*. The difference was attributed to the morphological differences between the flattened thallus in *C. nivalis* relative to the more cylindrical branched structure of *C. arbuscula* ssp. *mitis*.

The ^{210}Pb and ^{210}Po activity concentrations determined in muscle and liver tissue from Vågå and Østre Namdal were similar to values reported from other Nordic areas (Kauranen and Miettinen, 1967, 1969; Persson, 1974). The concentrations of ^{210}Pb and ^{210}Po in lichen and reindeer in the Nordic countries are generally lower than those in lichen and caribou in Canada. The higher Canadian concentrations are due to the comparatively larger landmasses and additional local sources giving higher atmospheric ^{210}Pb concentrations (Tracy, 1993; Thomas et al., 1994; Macdonald et al., 1996).

The estimated transfer coefficient of about 0.05 d kg^{-1} for ^{210}Po to reindeer muscle is similar to the upper estimates for mule deer muscle in a feeding experiment with polonium chloride (Sejkora, 1982). Furthermore, the estimate herein is similar to the limited literature values available for sheep, but is about 10-fold higher than those for cattle (see review by Ewers et al. (2003)). The uncertainties in the estimated F_f are related to the proportion of lichen in the diet, lichen species composition in the diet and assumption of equilibrium between ^{210}Po intake and excretion. These uncertainties are difficult to quantify without more detailed knowledge of diets and ^{210}Po excretion in reindeer. Sejkora (1982) estimated that nearly 70% of the Po administered to mule deer was excreted with a half-time of about 1 day. Watters et al. (1971) observed a similar short-term half-time in goats, and a corresponding half-time of 3–4 days in cows. The assumption of an approximate equilibrium in reindeer slaughtered in December and February therefore appears reasonable.

5. Conclusions

Age was an important parameter in determining concentrations of ^{90}Sr in reindeer. Concentrations were higher in bones of adult females than in calves, but due to constant bone renewal and continuous ^{90}Sr intake, the current difference with age apparently did not reflect the differences in intake during their periods of growth ($<$2 years). Combined with previous data from Vågå this study suggests that ^{90}Sr concentrations in reindeer calves declined with an effective ecological half-time of 9.03 $\pm$ 0.06 years during 1988–2002. However, the decrease is not detectable in the relatively low number of antlers sampled during the same period.

The analyses regarding the effect of animal age on soft tissue ^{210}Po and ^{210}Pb concentrations in reindeer gave equivocal results. There were no significant differences in mean concentrations in reindeer calves and adult females, but correlation coefficients suggested that age accounted for 20–28% of the variability in concentrations in half the tissue – district combinations. We conclude that age did not have a major effect on concentrations of these nuclides in this material.

Differences in climate did not appear to have a significant effect on soft tissue activity concentrations of ^{210}Po and ^{210}Pb in reindeer from Vågå and Østre Namdal.

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Paper VI

Concentrations of ^{137}Cs in lynx (*Lynx lynx*) in relation to prey choice

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Concentrations of ^{137}Cs in lynx (*Lynx lynx*) in relation to prey choice

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Abstract

Concentrations of ^{137}Cs were determined in 747 lynxes killed in Norway during the period 1986–2001. Highly variable ^{137}Cs concentrations and aggregated transfer coefficient values were observed, probably caused by variable ^{137}Cs concentrations in prey and the lynx's extensive home ranges and roaming distances. Adult lynxes had higher ^{137}Cs concentrations than sub-adults, and lynxes killed in regions with extensive reindeer grazing areas were more contaminated than others. A model with ^{137}Cs deposition density, the year lynxes were killed, age, and extent of reindeer grazing area accounted for 50% of the variability in observed ^{137}Cs concentrations. The analyses were equivocal regarding the influence of stomach content on ^{137}Cs concentrations in lynx muscle, i.e., on the lynx's specialization in prey species. Gender was not significant. Information on caesium retention in lynx and better estimates of deposition densities in lynxes' home ranges are important for further elucidation of factors influencing ^{137}Cs contamination in lynxes.

Keywords: Lynx; Carnivore; Caesium-137; Reindeer; Roe deer; Food chain; Chernobyl

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1. Introduction

The Eurasian lynx (*Lynx lynx*) is the most numerous larger carnivore mammal in Norway with a population of 300–350 individuals in 2002 (Andersen et al., 2003), and is found in most parts of the country. The lynx is a solitary predator, preferring rugged forested terrain, with an average range in Norway of 250–1500 km^2 depending on prey density (Kvam and Jonsson, 1998). The population density of lynxes is only 3–10 animals per 1000 km^2, as there is little territorial overlap between individuals of the same sex (Solberg et al., 2003). Home ranges of male individuals are generally larger than of females; the largest recorded male home range in Norway reaching 3100 km^2 (Kvam and Jonsson, 1998).

Both male and female lynxes are fully grown by the age of 2 years, with average weights of 15–16 and 18–20 kg for females and males, respectively (Sunde and Kvam, 1997). The mating season lasts from the end of February through early April (Kvam, 1990, 1991) and female lynxes give birth for the first time at the age of 2 years (Kvam, 1990), with average litter sizes of 2–3 cubs. The cubs are born in May–June, and follow their mother for nearly a year (till February–May) before establishing separate home ranges. Six out of 18 tracked lynxes in Hedmark county established home ranges more than 150 km from their mother's (Andersen et al., 2000). Another sub-adult migrated about 400 km from Sarek (northern Sweden) to Steinkjer (central Norway). Average daily roaming distances, occurring mostly during the night, range from about 2 km for adult females to 5.3 km for adult males, with recorded extremes of 23.5 and 45 km. The latter was a male searching a female in the mating season (Andersen et al., 2000).

The size of lynx prey ranges from small rodents to larger cervids such as reindeer and moose. Stomach content analyses of 441 Norwegian lynxes showed that 67% had eaten cervids (mainly roe deer and reindeer), 25% small game (such as hare, capercaillie and grouse) and 8% other species (such as fox and rodents) (Sunde and Kvam, 1997). Roe deer is the favourite species. Lynxes take more cervids during winter than summer (Kvam and Jonsson, 1998; Andersen et al., 2000), and males can take slightly more cervids than females (Sunde and Kvam, 1997). For females, Sunde and Kvam (1997) found a tendency towards a body weight effect on food choice, indicating that the smallest individuals may have problems handling the largest prey species. Another study found no difference in prey choice due to gender and age groups (Andersen et al., 2000). Lynxes kill on average one animal every 5 days when cervids are the only prey (Kvam and Jonsson, 1998).

This study presents data on ^{137}Cs concentrations in lynx muscle in Norway from the 1986 Chernobyl accident up to the year 2001. Through knowledge of reindeer grazing areas and the results of stomach analyses of killed animals, we attempt to quantify the effect of prey selection and in particular reindeer predation, on ^{137}Cs concentrations in lynxes. Reindeer generally contain higher concentrations of radioactive caesium than other animals, especially during winter (e.g., Åhman and Åhman, 1994). Mohn and Teige (1968) found that most of the ^{137}Cs contamination in lynxes in Norway was traceable to reindeer, and Åhman et al. (2002) found higher

^{137}Cs concentrations in lynxes from reindeer grazing areas than in lynxes from areas without reindeer. Due to the expected higher ^{137}Cs concentrations in lynxes predating upon reindeer, systematic differences also in ^{137}Cs concentrations in lynx can be expected if individuals specialize in prey species (e.g., some predate only roe deer, others only reindeer).

2. Materials and methods

To assess potential ranges in ^{137}Cs concentrations in muscle of lynx predating upon animals with different ^{137}Cs concentration levels, a two-compartment model simulating ^{137}Cs uptake and retention in lynxes was developed in ModelMaker v.4. The model was similar to that used for wolf by Holleman et al. (1990). Parameter estimates were obtained by fit to a build-up curve for ^{137}Cs concentrations in a 20 kg animal with a biological half-time of 35 days (Mohn and Teige, 1968). Alimentary uptake of radiocaesium in ingested meat was assumed to be 100%.

The majority of bodies of lynxes killed, accidentally killed or found dead in Norway are routinely examined at the Norwegian Institute for Nature Research (NINA). Fig. 1 shows the number of lynxes from the different years included in this study, i.e. all lynxes with known location submitted to NINA by November 2001. In total 747 animals were sampled, of which two 1986 samples were animals killed prior to the Chernobyl accident. About 70% of the lynxes were killed during the period 1995–2001, and about 40% in the counties Nord-Trøndelag and Nordland. Hunting for lynx is only allowed between 1 February and 31 March, with a limit, that varies between regions, on the number of animals that can be killed each year. In this study,

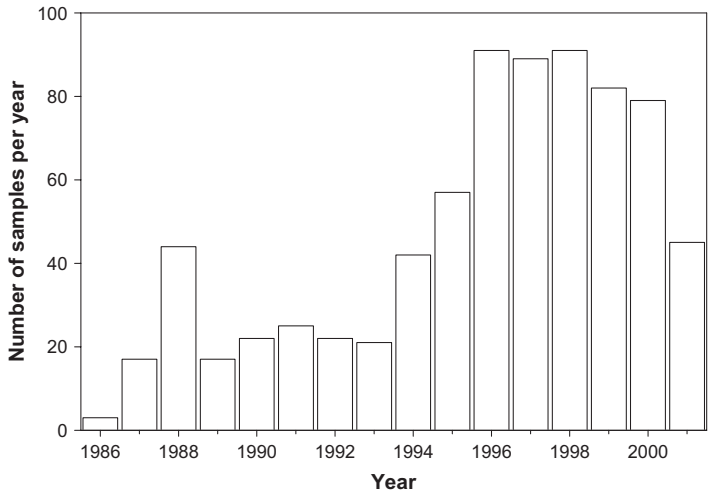

Fig. 1. Annual distribution of lynx muscle samples analyzed during the period 1986–2001.

approximately 93% of the available sample material was from animals killed during this period.

Muscle samples from the thigh (neck or shoulder if thigh was not available) were dried to constant weight at 70 °C and ground. A 3 inch NaI(Tl) well detector and a multichannel analyser (CompuGamma 1282) was used to determine ^{137}Cs. The detection limit was about 0.23–0.45 Bq depending on sample size. Obtained ^{137}Cs concentrations values were decay corrected to the date of killing, and the results are given in Bq kg^{-1} dry matter (DM).

Age (determined from teeth examination; Kvam, 1984) was used to categorize the lynx as sub-adult (0–24 months) or adult (>2 years). Information on killing date was used to classify the samples according to season: (1) winter from 1 December–31 May (in total only 3 animals from May), and (2) summer (1 June–30 November). Stomach contents were identified in 259 of the animals killed during 1986–1997 (Sunde and Kvam, 1997). The sampled animals were categorized according to their dominant stomach content: (1) reindeer, (2) roe deer, (3) hare, (4) unidentified cervid (including two cases identified as moose), (5) miscellaneous (e.g. birds (grouse, black grouse, capercaillie), mice, squirrel, dog, pig, sheep).

For some lynxes, killing date, gender or age was not determined, and different statistical analyses therefore involved different numbers of cases. In 45 cases no exact date of death was reported, and the date was set to 1 March in the respective years if there were no indications of killing outside the hunting season.

The Chernobyl fallout in Norway was highly heterogeneous, with average municipality ^{137}Cs deposition densities ranging from 0.06 to 100 kBq m^{-2} (Backe et al., 1986), with central Norway and the mountainous parts of southern Norway most contaminated. In an attempt to reduce variability in the dataset, ^{137}Cs concentrations in lynx were normalized by the total ^{137}Cs (i.e., ^{137}Cs from both nuclear weapons tests and Chernobyl fallout) deposition density in the municipality in which the lynx had been killed. The municipality (areas ranging from 74 to 4646 km^2) in which the lynx had been killed was taken as the lynx's home habitat. The total ^{137}Cs 1986 deposition density from Backe et al. (1986) was decay corrected to give separate deposition densities per year. The normalized values thus obtained are the aggregated transfer coefficients (*Tag*; unit m^2 kg^{-1}) (Hove and Strand, 1990; Howard et al., 1991).

Using manual overlay and visual inspection of 1:2 500 000 scale maps of municipality borders and grazing areas of wild and semi-domesticated reindeer, all municipalities were roughly classified as containing: (0) no grazing areas, (1) minor grazing areas, (2) larger grazing areas (up to 50% of the area), (3) grazing areas in most of the area, and (4) grazing areas all over the area.

Statistical analyses (analysis of variance, linear and non-linear regression) were carried out in SPSS release 11. Due to skewed data and in order to harmonize variances, the ^{137}Cs concentration, deposition and *Tag* values were log transformed prior to analyses. The influence of time after fallout and deposition level on concentrations in lynxes was reduced using analyses of the residuals obtained after regression of concentration values vs. time and deposition.

3. Results

3.1. Simulated ^{137}Cs concentrations in lynx

Fig. 2 shows model simulated ^{137}Cs concentrations in lynx muscle due to different diets: Concentrations in lynxes predating only reindeer, lynxes predating only other species, and lynxes eating reindeer for 5 and 10 days in December and January, respectively. The lynx's daily meat intake was set to 2 kg (fresh weight, FW) in accordance with the estimate of Mohn and Teige (1968) and corresponding to killing a cervid like an average sized roe deer every 5 days (Kvam and Jonsson, 1998). The calculations applied a concentration of 7000 Bq kg^{-1} (FW) in reindeer meat, representative of average concentrations in reindeer of Nord-Trøndelag county about 1989–1990 (Mehli et al., 2000). For other prey a concentration of 300 Bq kg^{-1} (FW) was applied, corresponding to data on moose from Nord-Trøndelag county (Ahlin, J.P. Norwegian Food Safety Authority, Namdal office, personal communication) and to estimated concentrations in roe deer in the same area using aggregated transfer coefficients from Johanson and Bergström (1994) and Kiefer et al. (1996). The difference in concentrations in prey determines the range in concentrations in lynx (in this simulation a factor of 23 difference). The simulation showed that a considerable variability in ^{137}Cs concentrations in lynx would be expected due to variable diets and variable ^{137}Cs concentrations in prey.

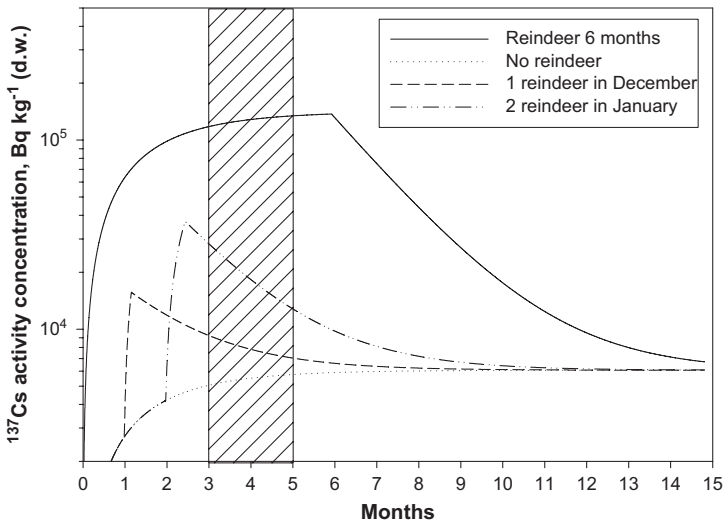

Fig. 2. Estimated ^{137}Cs concentrations in lynx muscle due to different diets. The simulation starts from 1 November (month 0). Solid line simulates concentrations in lynx killing only reindeer (containing 7000 Bq kg^{-1}) for 6 months; dot line simulates lynx not killing reindeer (prey containing 300 Bq kg^{-1}); dash line simulates lynx killing one reindeer in December, and dash-dot line simulates lynx killing two reindeer in January. The hunting season in February–March is indicated by the hatched area. See Section 3.1 for more details.

3.2. Observed ^{137}Cs concentrations and Tag values

Fig. 3a presents the ^{137}Cs concentrations in sampled lynx muscle. The simulated concentrations in Fig. 2 corresponded to those in the upper half of the range of observed values. The concentrations in the two animals killed prior to the Chernobyl accident were 670 and 800 Bq kg^{-1} (DM), while a maximum value of 125 000 Bq kg^{-1}

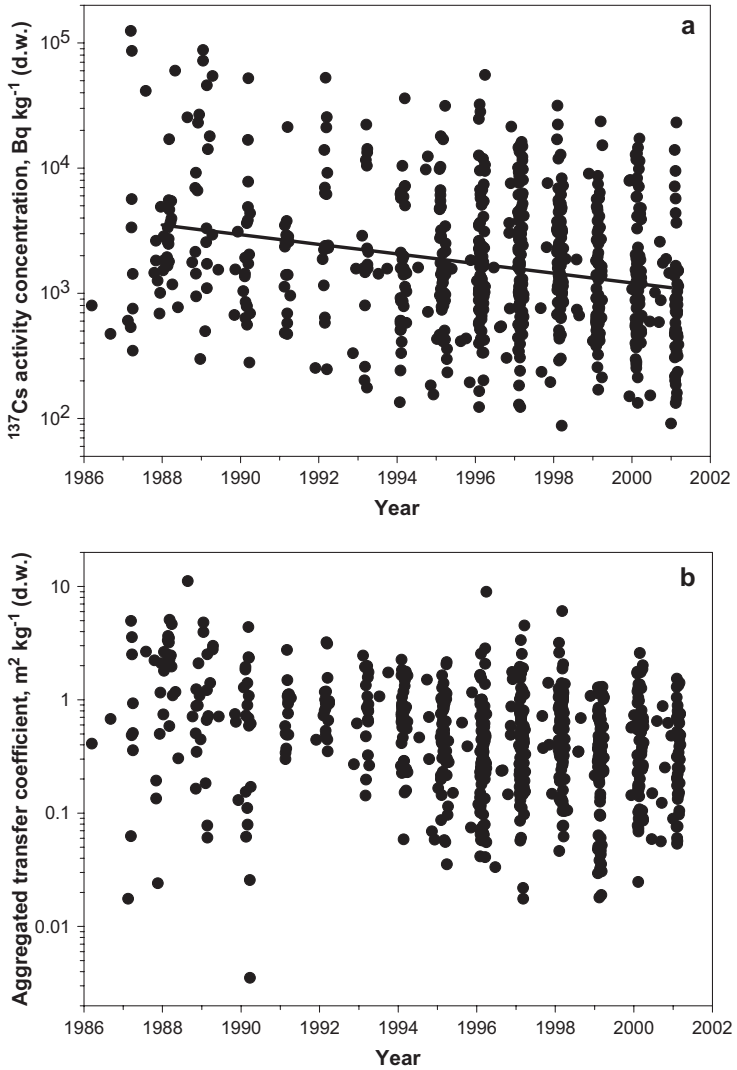

Fig. 3. (a) Concentrations of ^{137}Cs in lynx muscle and (b) aggregated transfer coefficients for lynx muscle in Norway during 1986–2001 (only samples with known date of killing). The line in (a) is the fitted linear regression model, Eq. (1).

(DM) was found in a 2 year old male lynx killed in Alvdal (Hedmark county) in March 1987. The highest average concentrations were found in 1988–1989. Annual maximum and minimum concentrations generally differed by more than a factor of 100. The difference was maintained throughout the whole 1986–2001 period, and was the same irrespective of season.

Normalizing the ^{137}Cs concentrations to the deposition values reduced the coefficient of variation (CV) (i.e., standard deviation in percent of sample mean) in the whole dataset from about 230% for the concentration values to 140% for the *Tag* values (Fig. 3b). *Tag* values ranged more than three orders of magnitude from below 0.01 to above 10 m^2 kg^{-1} (DM).

3.3. Factors influencing ^{137}Cs concentrations in lynx

Two-way analysis of variance was carried out to study the influence of ^{137}Cs deposition density, time after fallout (years), gender, age, season, extent of reindeer grazing areas, and stomach content on both concentration and *Tag* values. The analyses revealed that deposition density, age and extent of reindeer grazing area explained significant proportions of the variability ($P<0.01$, adjusted $R^2=0.82$). Furthermore, time after fallout was a significant factor ($P<0.01$). However, since deposition densities were obtained by decay correction of 1986 values, no degrees of freedom were left for assessment of the effect of time after fallout in two-way analyses of variance with deposition density included. Deposition density alone explained about 77% of the variability in the concentration values (adjusted R^2). Gender, stomach content and season ($P=0.12$) were found to be non-significant factors. Nevertheless, the 54 samples of lynx killed during summer were excluded from further analyses.

The residuals (e) from a linear regression of log transformed concentration values vs. deposition and time (after 1987) were assumed to be independent of the two factors, and were studied in an attempt to quantify the effect of the other factors. The obtained regression model was:

$$\ln A = 3.21(\pm 0.26) + 0.600(\pm 0.029)\ \ln D - 0.089(\pm 0.011)t + e \quad (1)$$

where A is the ^{137}Cs concentration in lynx (Bq kg^{-1} DM), D is deposition density (Bq m^{-2}) and t is time (years; $t=0$ in 1988) ($N=636$, adjusted $R^2=0.43$). The estimated model is illustrated by the lines in Figs. 3a and 4. Regression of *Tag* vs. deposition and time gave identical residuals to those for concentration values (since *Tag* values were obtained by decay corrected 1986 deposition densities, and $\ln \mathrm{Tag} = \ln A - \ln D$ where $\ln A$ is given in Eq. (1)).

Two-way analysis of variances in the residuals gave results similar to those above. Stomach content and gender were again found insignificant, whereas age ($P=0.029$) and extent of reindeer grazing area ($P<0.001$) accounted for 14% of the variability in the residuals (adjusted R^2). Concentrations of ^{137}Cs were on average 23% higher in adults than in sub-adults (mean residuals of 0.111 ± 0.053 and 0.093 ± 0.051 for sub-adults ($N=285$) and adults ($N=343$) respectively).

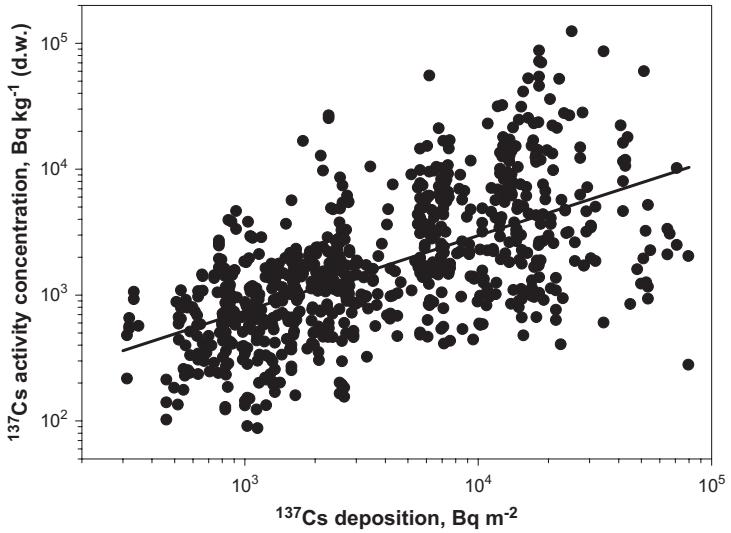

Fig. 4. Concentrations of ^{137}Cs in lynx muscle vs. deposition density. The line is the fitted linear regression model, Eq. (1).

The relationship between ^{137}Cs concentrations in lynxes and the extent of reindeer grazing area is illustrated in Fig. 5. Concentrations in lynxes killed in municipalities of category 4 were a factor of 2.6 higher than those of category 1 and a factor 1.6–2.0 higher than those of categories 0, 2 and 3 ($P<0.001$) (the factor is given by the difference in log transformed values (x) as e^x). Also, concentrations in lynxes from category 0 were higher than in those from category 1 by a factor 1.6 ($P=0.011$).

Summarizing the results into one model for ^{137}Cs concentrations in lynx muscle gives:

$$\ln A = 3.21 + 0.600 \ln D - 0.089t + c_{\text{age}} + c_{\text{area}} \quad (2)$$

where A is the concentration in lynx (Bq kg^{-1} DM), D is the deposition density (Bq m^{-2}; decay corrected 1986 values), t is time (years since 1988), c_{age} is a constant depending on lynx age category (equal to -0.111 for sub-adults and 0.093 for adults), and c_{area} is a constant depending on category of extent of reindeer grazing area (equal to -0.142 in municipalities with no reindeer grazing areas, -0.625, -0.383 and -0.38 for those with intermediate extents of grazing areas respectively, and 0.313 in municipalities with reindeer grazing areas all over the area). All c_{area} values are not significantly different (cf. Fig. 5) but serve as best estimates. The model R^2 was 0.52. As expected the R^2 was lower (i.e., 0.49) when the model (Eq. (2)) was used to estimate concentrations in all samples from 1986 to 2001. The model underestimates the highest concentrations and overestimates the lowest ones.

The concept of aggregated transfer coefficients assumes that *Tag* values are independent of deposition densities, but Fig. 4 and Eq. (1) show that ^{137}Cs concentrations in lynx muscle did not increase linearly with deposition. One-way

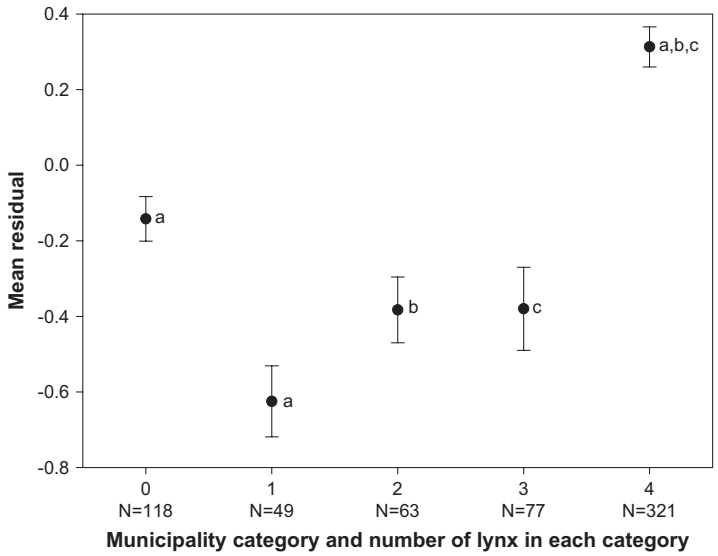

Fig. 5. Mean values (±SE) of residuals by extent of reindeer grazing areas. Category (0) no grazing areas, (1) minor grazing areas, (2) grazing area in up to 50% of the area, (3) grazing areas in most of the area, and (4) grazing areas all over the area. Mean values with similar letter were significantly different.

analysis of variance showed that stomach content had a significant effect on *Tag* values as well as residuals ($P=0.001$), and the mean *Tag* for lynxes with roe deer in their stomach was 45–48% of the *Tag* for lynxes with reindeer or unidentified cervid ($P=0.009$ and $P=0.012$, respectively, Fig. 6).

3.4. Effective ecological half-times

Regression analyses identified no systematic differences or pattern in effective ecological half-times for ^{137}Cs in lynx muscle with respect to extent of reindeer grazing areas or between counties, neither were there any differences in half-times in lynxes with different stomach contents. For the different grazing area categories and counties the analyses yielded results ranging from half-times of 3–4 years to no detectable decrease with time. Similarly, Åhman et al. (2002) estimated half-times in lynx of 2.2–4.2 years in some areas in Sweden in the period 1996–2001, while there was no significant decrease in other areas. The half-time estimated using all data (given by the time-dependent term in Eq. (1)) was 7.9 years (standard error range 6.9–8.9 years).

4. Discussion

The analyses of the dependency of aggregated transfer coefficients and ^{137}Cs concentrations in lynx muscle on stomach content gave equivocal results. One-way

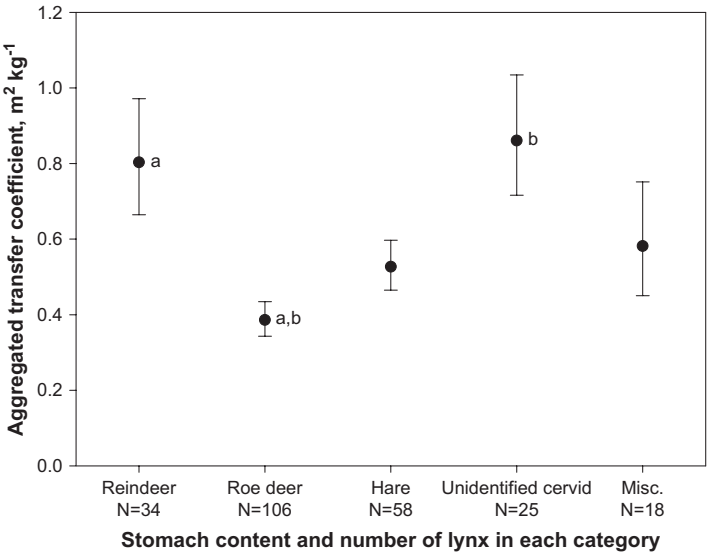

Fig. 6. Geometric mean (and SE) of *Tag* values for winter killed lynxes with different stomach contents. Mean values with similar letter were significantly different.

analyses of variance showed that stomach content was a significant factor, while two-way analyses of variance did not identify stomach content as significant. This indicated that the variable was correlated to other important factors. A slight but statistically significant correlation between stomach content and deposition ($R = -0.13$, $P = 0.044$) was the only possible explanation found (see further discussions regarding *Tag* below). Furthermore, if stomach content is insignificant, this may indicate that the lynx have a variable diet and do not specialize in prey species or that the stomach content at the time of killing did not give a representative indication of its diet. It may also reflect the variability in ^{137}Cs concentrations in lynxes resulting from spatial and temporal variable ^{137}Cs concentrations in prey species, as well as the extensive home ranges and mobility of particularly sub-adult lynxes and males during the mating season (which coincide with the hunting season). The variability in ^{137}Cs concentrations in reindeer following the Chernobyl fallout are illustrated in Åhman and Åhman (1994). Maximum concentrations in individual reindeer in central Norway reached 150 000 Bq kg^{-1} (FW) (Strand et al., 1992).

A reduction in the variability in ^{137}Cs concentrations in lynx due to different deposition levels in their home range was attempted by calculating *Tag* values based on average deposition densities, but the CV for *Tag* was also considerable. Since 21% of the municipalities where lynxes were killed have areas of less than 400 km^2, and 70% less than 1200 km^2, many lynxes will have home ranges extending beyond one municipality, and can even cross whole municipalities during one day's roaming. Furthermore, average deposition densities between neighbouring municipalities are up to a factor of 15–20 different (Backe et al., 1986) and a considerable variability in

Tag values would therefore be expected. Even lynxes living within the borders of one municipality may experience considerable heterogeneity in ^{137}Cs deposition.

Simulated ^{137}Cs concentrations in lynx muscle corresponded to those in the upper half of the range of observed values, which was reasonably since concentrations from some of the most contaminated areas in Norway were applied in the calculations. We know of no experiment on radiocaesium excretion rate or biological half-times in lynx or other felids that could be helpful in studying how variations in diet can cause variations in ^{137}Cs concentrations in lynx muscle. The biological half-time estimate of 35 days by Mohn and Teige (1968) was based on data from other species in Ekman (1967). In comparison, Holleman and Luick (1976) found half-times of 22–26 days in wolf and coyote. A shorter half-time than 35 days would give more rapid changes in ^{137}Cs concentrations in lynxes and lower maximum values than that indicated in Fig. 2, while the relative range in concentrations values due to differences in diet would be the same.

The *Tag* values estimated in this study were correlated to the deposition densities as indicated in Fig. 4 and Eq. (1). The correlation coefficient between log transformed *Tag* and deposition densities was $R = -0.44$ ($P < 0.001$). A substantial decrease from 1986 to 1995 in surface soil ^{137}Cs was reported by Bjerk et al. (1999) in most Norwegian counties. Thus the decay corrected values applied herein may have been overestimates, and could be one reason for the negative correlation with *Tag*. Another reason may be the importance of reindeer in the diet of lynxes in northern Norway receiving the lowest Chernobyl ^{137}Cs deposition, while other species may be more important prey in the most contaminated areas in central and southern Norway.

No definite explanation for the relatively high ^{137}Cs concentrations in lynx killed in regions with no grazing areas was found, but it appeared related to the fact that most of these municipalities are situated in southern Norway in areas of relatively low Chernobyl fallout and higher transfer to lynx (cf. the correlation between deposition density and *Tag*). Also, long ecological half-times for ^{137}Cs in the lynx's prey in these areas would contribute to relatively high concentration levels. Concentrations of ^{137}Cs in roe deer may increase significantly in autumn due to ingestion of fungi (e.g., Avila et al. (1999), and the ecological half-time for roe deer has been found to approach the physical half-life of 30 years (Johanson and Bergström, 1994).

Åhman et al. (2002) reported average *Tag* values for lynxes in Sweden during 1996–2001 corresponding to 0.16–0.28 $m^2\ kg^{-1}$ (DM) in areas with no or few reindeer, to 0.48–1.1 $m^2\ kg^{-1}$ (DM) in areas with reindeer present. These values are not directly comparable to those herein due to different methods of *Tag* estimation and categorization of presence of reindeer, but they appear lower than those estimated for Norwegian lynxes in the same period.

Prior to regression and calculation of residuals for concentrations vs. deposition density and year, a control was made of possible differences in time trends of concentrations in different reindeer grazing area categories, or in different counties. Shorter ecological half-times for ^{137}Cs in reindeer in areas with higher deposition density were reported by Åhman et al. (2001), thus a similar trend might be expected

if lynxes depended heavily upon reindeer predation. Since no systematic differences in half-times between counties or grazing area categories could be detected all concentration values were applied in the regression analysis. The resulting estimate of 7.9 years appears long compared to effective ecological half-times of 3–5 years estimated for reindeer in both Norway and Sweden (Amundsen, 1995; Gaare et al., 2000; Hove and Staaland, 1997; Åhman et al., 2001; Åhman and Åhman, 1994).

5. Conclusions

1. Adult lynxes attained the highest ^{137}Cs concentrations in Norway following the Chernobyl accident, and concentrations in lynxes from municipalities with reindeer grazing areas all over the area were higher than in other regions.
2. This study did not give clear indications of specialization in prey species, but unambiguous conclusions were possibly prevented by the variability introduced by the heterogeneous Chernobyl deposition (causing variable ^{137}Cs concentrations in prey) and the lynxes' considerable roaming distances, and the uncertainty in estimated ^{137}Cs deposition densities.
3. Deposition density, time after fallout, animal age and extent of reindeer grazing area could account for 50% of the observed variability in ^{137}Cs concentrations in lynx muscle.
4. Average effective ecological half-times for ^{137}Cs in lynxes in Norway were estimated to be in the range 6.9–8.9 years.
5. Experimental data on caesium retention in lynx, and independent estimates of deposition density in the lynx's home ranges, seem important for further studies of transfer of ^{137}Cs through the food chain to lynxes in Norway.

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