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99:19

An Integrated Approach to Radionuclide Flow in Semi-natural Ecosystems Underlying Exposure Pathways to Man

Final Report of the LANDSCAPE Project

Statens strålskyddsinstitut
Swedish Radiation Protection Institute

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DIVISION/AVDELNING: Department of Waste Management and Environmental Assessment/ Avdelningen för avfall och miljö.

TITLE: An Integrated Approach to Radionuclide Flow in Semi-natural Ecosystems Underlying Exposure Pathways to Man. Final Report of the LANDSCAPE project.

SUMMARY: The general objective of the LANDSCAPE project has been to obtain a basis for reliable assessments of the radiation exposure to man under different time scales from radionuclides in plant and animal products of representative forest ecosystems in Europe. The work has been focussed on radiocaesium, ^{134}Cs, ^{137}Cs. In particular, the project has included (i) to quantify some major processes which influence the radiocaesium contamination of vegetation and fungi, (ii) to quantify radiocaesium intake of key herbivores, particularly free ranging moose, relative to food availability and degree of contamination, (iii) to quantify the influence of forest management on radiocaesium dynamics, and (iv) to incorporate these processes in dynamic models.

The LANDSCAPE project has been the combined effort of eight research groups from five European countries, and this report describes the results obtained during 30 months of common work.

SAMMANFATTNING: Det övergripande syftet med projektet LANDSCAPE har varit att skapa en kunskapsbas för att göra tillförlitliga uppskattningar av stråldoser till människan från radioaktivt kontaminerade skogsekosystem i Europa i olika tidsperspektiv. Arbetet har främst behandlat radioaktivt cesium, ^{134}Cs, ^{137}Cs. Projektet har särskilt syftat till (i) att kvantifiera några viktiga processer som påverkar koncentrationen av radioaktivt cesium i växter och svamp, (ii) att kvantifiera intaget av radioaktivt cesium hos älg i förhållande till tillgång på föda och grad av kontaminering, (iii) att kvantifiera effekten av skogsbruk, i första hand gödsling, på radioaktivt cesiums uppträdande i ekosystemet, och (iv) att använda dessa kunskaper om skogsekosystemet i dynamiska beräkningsmodeller.

Projektet LANDSCAPE har varit ett samarbete mellan åtta forskargrupper från fem europeiska länder, och denna rapport redovisar de resultat som erhållits under 30 månaders gemensamt arbete.

SSI rapport: 99:19
Oktober 1999
ISSN 0282-4434

Statens strålskyddsinstitut
Swedish Radiation Protection Institute

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Final Report
Research Contract no F14P-CT96-0039
European Commission
Nuclear Fission Safety Programme

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INTRODUCTION

Semi-natural environments contribute to the exposure of man to radiation and are thus important from a radiation protection point of view. This realization existed among radioecologists as a result of the investigations performed after the atmospheric nuclear weapons tests. But it was not until after the Chernobyl accident in 1986 that the radiological effects of a contaminated semi-natural environment were clearly demonstrated to a broader community of scientists and especially that the effects became known to the general public. The LANDSCAPE project has studied the forest environment which is characterized by complexity and large variability in the radionuclide behaviour. A good understanding of the mechanisms behind the environmental transfer of radionuclides is a prerequisite for a realistic estimation of exposures and their uncertainties, and for ultimate quantification of the doses and risks involved. The radiation doses from products from the semi-natural environment can be high, and the dose estimates show large variations. The study of the consequences of contamination of these environments is also motivated by the need for nuclear emergency preparedness, where different uses of forest and forest products, forestry and forest industry are considered.

Objectives

The aim of the LANDSCAPE project has been *to obtain a basis for forming reliable assessments of the radiation exposure to man under different time scales from radionuclides in plant and animal products of some representative semi-natural ecosystems in Europe.*

The project employed both modelling and experiments according to the following summary of activities:

- to review and evaluate selected data, and define data sets for use in the development of dynamic ecosystem and landscape models which describe the behaviour of radionuclides at different levels of ecological resolution;

- to test model behaviours with data on radiocaesium from selected forest ecosystems in order to better predict the long term consequences of contaminated forest products, and the resulting radiation exposures, including estimates of the uncertainties involved;

- to measure relevant biological processes and links in selected semi-natural ecosystems in order to quantify the redistribution and reduction in time of radionuclides in plants and animals that constitute important exposure pathways to man;

- to quantify the influence of forest management, primarily fertilization, on the radiocaesium distribution, and to assess forest management as a possible countermeasure, or restoration technique;

- to quantify the impact of feed intake on temporal and spatial variation in animal populations and reduction in time of radionuclides in plants and free-ranging animals.

In this final report of the LANDSCAPE project the results are presented in two chapters summarizing the experiments, "vegetation dynamics" and "herbivore dynamics", and one chapter on "modelling of the forest ecosystem". The main achievements and conclusions as well as the implications for radiological protection and future research are discussed in a concluding chapter.

VEGETATION DYNAMICS

Introduction

More than a decade after the Chernobyl accident, most of the radiocaesium in the forest ecosystems still resides in the top soil and vegetation system. On a small scale, the uptake of cesium in plants is dependent on the mobility and bioavailability of cesium in the soil, which is affected by chemical and biological processes in the different compartments of the soil. The uptake also depends on the productivity and nutrient status of the soil as well as its water content. On a larger scale, a certain redistribution of cesium takes place by transport in surface and ground waters, and by movements of litter by means of wind erosion and surface runoff.

Man is using the forest for many purposes including recreation, as a food resource and for industrial purposes. Various forms of forest management are traditionally used in the forest. After a contamination of radionuclides, forest management may also be a way for restoration and for improving the radiological situation.

The dynamics of radioactive caesium in vegetation has been studied in LANDSCAPE and the results are presented and discussed in the following sections on ground vegetation, mosses and lichens, fungi, trees, litterfall and decomposion. One section is describing the results of large scale forest management. The chapter also includes sections on the vertical and horizontal redistribution of radioactive cesium in soil.

The field experiments[1] of LANDSCAPE have been carried out on sites in Finland, Sweden, and Belgium and these are described in Annex 1, Table 1.

Vertical redistribution of ^{137}Cs in soil

The vertical distribution of radiocaesium in soil directly influences the uptake in plants. It also influences the exposure of man from external radiation. From studies after the Chernobyl accident, it is known that most of the deposited caesium during the first years was found in the raw humus layers (see e.g. Fawaris and Johanson, 1994). Some later studies have shown a downward migration which is more rapid in deep organic soils (Rosén et. al, 1999).

The vertical distributions of ^{137}Cs in podzol soils in the regions of this study show a trend of redistribution from the vegetation and raw humus layers down to the mineral soil (Figure 1). In 1995 to 1997, between 20 and 40% of the deposition had reached the mineral soil. Although there is a general trend of downward migration, the depth distribution varies between the sites. The depth distribution is affected by differences in clay content in the mineral soil which in areas with high contents may trap caesium in the mineral fraction. The soil particle distribution affects the hydraulic conductivity and capillarity. Dense soil with low hydraulic conductivity and high capillarity will probably be more effective in retaining caesium in shallow layers due to slow percolation, high ion exchange capacity and an upward water flow during warm periods. Biotic factors may lead to variations in soil microflora which influence the caesium distribution. The indications of high uptake of radiocaesium in fungi suggests that mycelium may act as a sink for caesium. Variations in the vertical position of mycelium may accumulate caesium at different depths.

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[1] Unless otherwise described, standard methods are used for sampling and measurements.

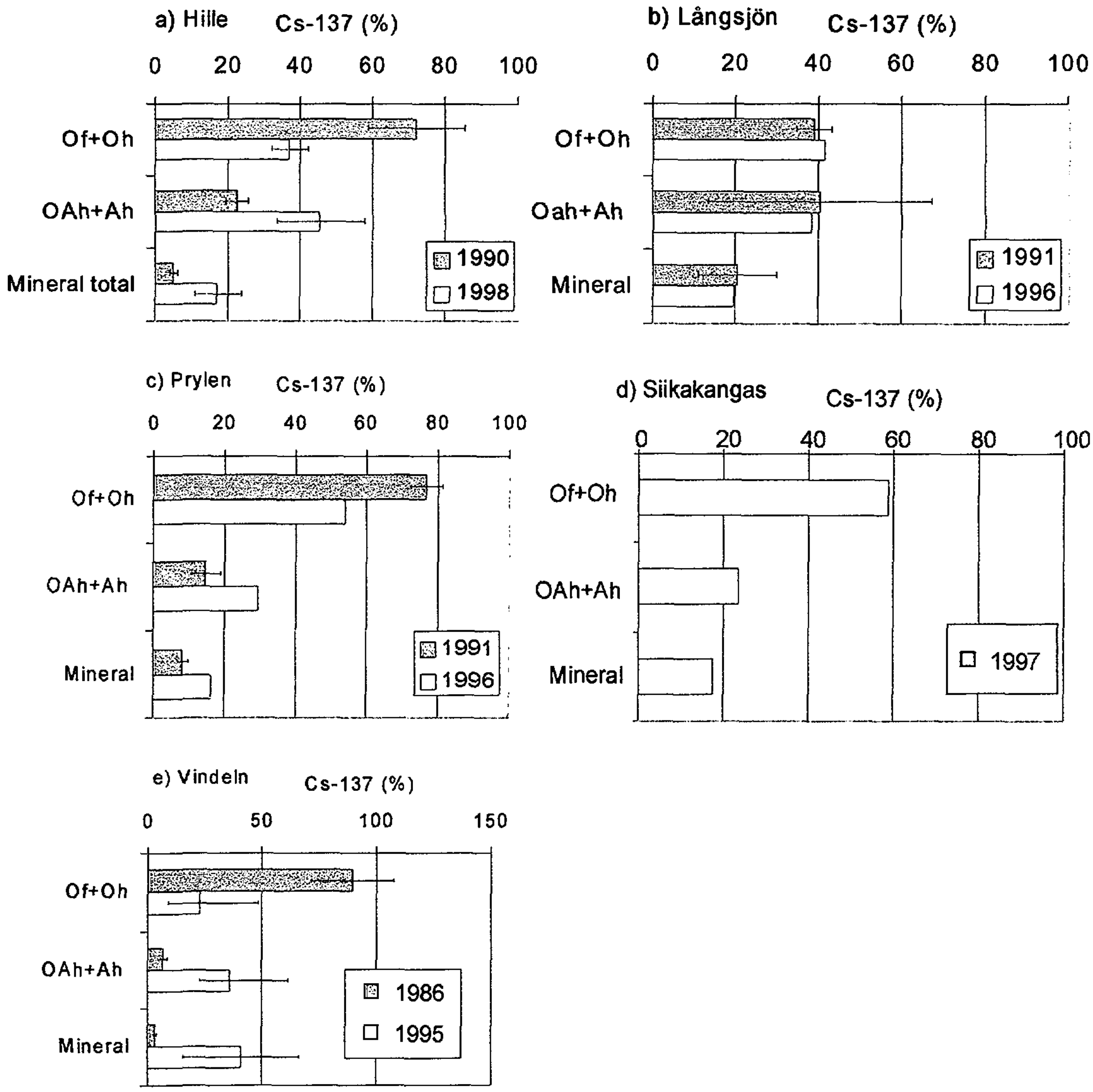

Figure 1. The vertical distribution (%) of ^{137}Cs in forest soils in Fennoscandia: a) Hille, b) Långsjön, c) Prylen, d) Siikakangas and e) Vindeln

In organic soils in bogs, the redistribution of ^{137}Cs is not so clear. In the studied sites, Vindeln and Liesineva, about 90% of the activity is still residing in the first 10 cm.

The successive vertical redistribution of caesium in podzol soils is also reflected as a decreasing rate in photon fluence. Changes in external exposition from the 662 keV photon peak were followed in different areas in northern Sweden from 1986 to the late 1990s (Table 1).

Table 1. Description of the areas where in situ gamma spectrometry measurements were performed.

Area	Stand	F/H (cm)	Soil type	Sub soil	Eff. Half-time *
A	Pine moor	2	Podzol	Sediment	13
B	Pine forest	6	Podzol	Moraine	12
C	Spruce forest	35	Humus podzol	Sand	9

*Effective half-time of fluence at 662 keV

The photon fluency at 662 keV decreased at least a factor of two faster than what can be expected from the physical half life of ^{137}Cs. The fastest decrease was found at the area with deeper organic soil (area C) which is illustrated in Figure 2. Except for the exponential decrease temporal fluctuations up to a factor of two can be noted. These are due to the variations in soil moisture and generate lower external doses during winter, spring and autumn compared to the summer.

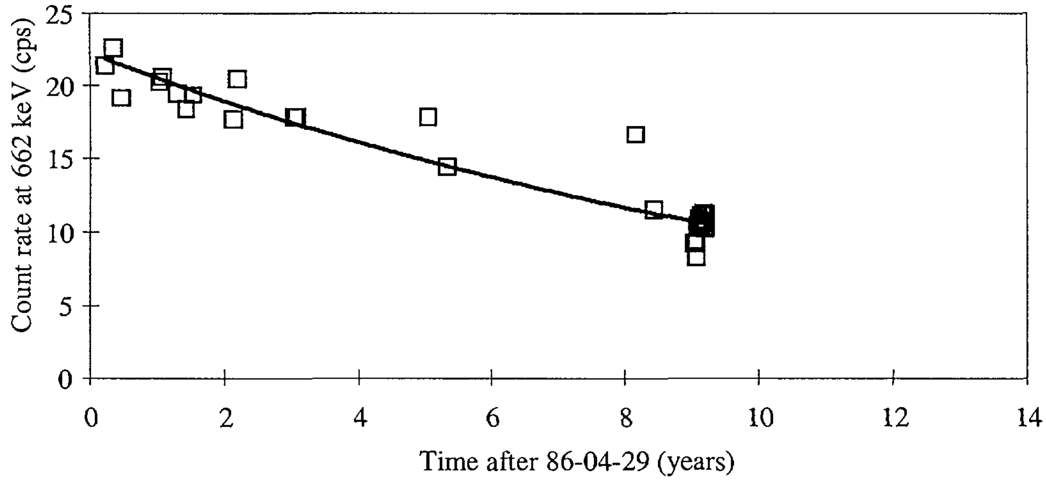

Figure 2. The photon fluence at 662 keV at 1 m height over a humus podzol soil (site C in Table 1).

Conclusions

There is a successive redistribution of ^{137}Cs in podzol soils slowly leading to an increased amount of cesium in the mineral soil horizon. This conclusion is based on measurements of soil profiles and is also confirmed by exposure measurements *in situ*. The situation on peat soils is less clear and there is probably larger variations between sites.

The change in the vertical distribution of ^{137}Cs in soil have implications for plant root uptake, which depending on species can lead to either an increase or decrease in uptake of caesium.

Horizontal redistribution of ^{137}Cs - Runoff

The concentration of radioactive caesium in water has earlier been studied in groundwater, in mires and in runoff from a catchment area in Vindeln (Nylén and Grip 1997) and the amount of radioactive caesium leaving the terrestrial compartment by water has been estimated. When the main deposition of Chernobyl caesium occurred on April 29th 1986, snow melting and runoff reached their yearly maximum intensity. The amount of ^{137}Cs discharged from the studied 0.5 km² catchment during this period was about 600 MBq, corresponding to 5% of the total deposition in the area. In the following years only relatively low levels of radioactive caesium were detected in the stream water, and in total about 10% had been lost during the period 1986-1994 (Figure 3).

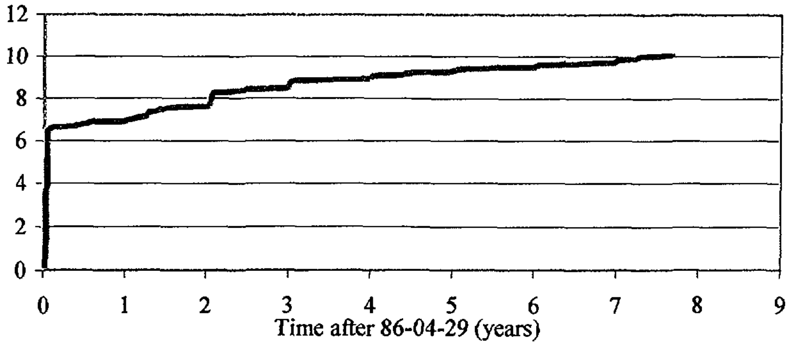

Figure 3. The cumulative amount (%) of ^{137}Cs *in stream water discharged from a catchment.*

More detailed calculations show that the initial loss during snowmelt in 1986 essentially occurred from the peat areas and amounted to about 40% of the total deposition in these areas. No significant loss from the pine and spruce forests occurred during the same period. From the autumn 1986 and onwards the annual loss amounts to 30% from the "wetter" part of the peat bog – i.e. the part often contributing to the surface runoff – while it is about 2% from the "drier" part of the peat area. The loss from areas of unsaturated mineral soil type is below 0.03% (Nylén and Grip, 1997). The corresponding ecological half-life (T_{eco}) for the boreal type ecosystem was estimated from the measurements on ground water (Nylén and Grip, 1997) and exceeded 4000 years (T_{eff} = 30 years) while the T_{eco} on the drier fractions of mires was about 34 (T_{eff} = 16 years). On the wet fractions of mires, that often contributes to saturated surface discharge, T_{eco} was about 2 to 3 years.

New measurements on surface and ground water show that a small fraction of the ^{137}Cs deposition has reached the ground water at about 1 m depth. Assuming that the total annual discharge of 330 mm passes this depth with a constant activity of 0.0018 Bq l^{-1} (Figure 4) the corresponding discharge via ground water would be 6 x 10^{-5} % per year which can be neglected compared to the discharge from the mire which on average has been 1-2% per year during 10 years.

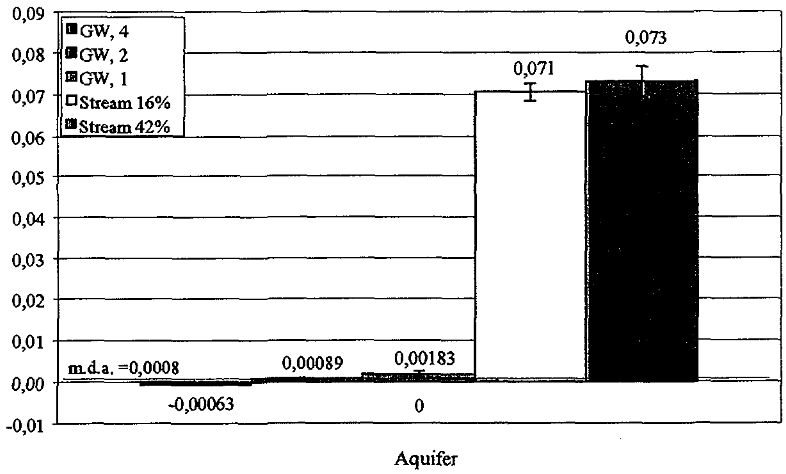

Figure 4. The activity concentration (Bq/l) of ^{137}*Cs in stream and ground water (GW) in January 1999 at the Vindeln site (the error bars represent 1 σ).*

Conclusions

More than a decade after the Chernobyl deposition, the amount of ^{137}Cs leaving the forest ecosystem by runoff is negligable except from the wetter fraction of mires. The caesium has now reached the ground water at 1 m height but the discharge from the system can be neglected.

Ground Vegetation

During the vegetation periods in 1997 and 1998, vegetation was sampled from seven sites: one in Belgium (Saint-Léger) three in central Sweden (Långsjön, Hille and Prylen), one in northern Sweden (Vindeln) and two in central Finland (Liesineva and Siikakangas) (Annex 1, Table 1). The results allow comparisons both within sites and between sites. This section mainly presents results on ^{137}Cs in understorey plants, but also gives some results on ^{137}Cs in current shoots of young trees as they are important forage for moose.

Temporal variations at a boreal site

In 1991, plants were sampled systematically from the forest cover at Hille, functioning as a benchmark station in Sweden (Guillitte *et al.*, 1994). In 1998, new samples were collected from the forest cover to monitor the evolution of the contamination over time, including factors involved in soil-plant transfer in herbaceous plants at the station. Samples were also taken from shrubs likely to be eaten by game, particularly by moose (Table 2).

Plants at Hille lost, on average, 70% of their activity between 1991 and 1998. In relative terms, however, major differences were observed in connection with this loss, which surprisingly tallied with the taxonomic families. For instance, most of the plants in the *Asteraceae*, *Fabaceae* and *Liliaceae* families were characterised by a loss far exceeding 70%. In contrast, the loss in most of the grasses and *Rubus* species was considerably less than 70%. The loss in *Ericales* and ferns was in the middle range. They maintained their position relative to one another over time.

Table 2. Evolution of the ^{137}Cs specific activity (kBq kg^{-1} dry wt.), aggregated transfer factors (T_{ag}) (10^{-3} m^2 kg^{-1}), and ranking of vascular plants according to their activity at the Hille site.

Species	**1991 activity**	**1998 activity**	**Ratio activity 1991/1998**	**T_{ag} 1991**	**T_{ag} 1998**	**1991 Ranking**	**1998 Ranking**
Dryopteris carthusiana	39	13.2	3.0	195	72	1	2
Melampyrum pratense	25	16.5	1.5	125	91	2	1
Pteridium aquilinum	19	7.2	2.6	95	40	3	3
Mainthenum bifolium	14	3.8	3.7	70	21	4	8
Hieracium laevigatum	14	2.6	5.4	70	14.3	5	15
Lycopodium annotinum	13	5.1	2.5	65	28	6	5
Luzula pilosa	11	5.2	2.1	55	29	7	4
Calluna vulgaris	11	4.8	2.3	55	26	8	6
Linnea borealis	9.8	3.0	3.3	49	16.5	9	11
Oxalis acetosella	9.0	1.7	5.5	45	9.3	10	19
Vaccinium vitis-idaea	8.7	2.7	3.2	43.5	14.8	11	13
Solidago virgaurea	8.2	1.7	4.8	41	9.3	12	18
Monotropa hypopitys	7.9	0.8	10.5	39.5	4.4	13	26
Deschampsia flexuosa	7.8	3.4	2.3	39	18.7	14	9
Trientalis europaea	6.9	2.7	2.6	34.5	14.8	15	14
Vaccinium myrtillus	6.7	2.5	2.7	33.5	13.7	16	16
Trifolium medium	4.5	0.5	9.0	22.5	2.7	17	27
Convallaria majalis	3.6	3.2	1.1	18	17.6	18	10
Hieracium umbellatum	3.5			17.5		19	
Epilobium angustifolium	3.3	4.0	0.8	16.5	22	20	7
Rubus idaeus	2.9	2.3	1.3	14.5	12.6	21	17
Vicia sp.	2.5			12.5		22	
Lathyrus linifolius	2.1	0.8	2.6	10.5	4.4	23	25

Table 2, continued......							
Species	**1991 activity**	**1998 activity**	**Ratio activity 1991/1998**	**T_{ag} 1991**	**T_{ag} 1998**	**1991 Ranking**	**1998 Ranking**
Viola riviniana	1.6	1.6	1.0	8	8.8	24	20
Potentilla erecta	1.6	0.3	5.0	8	1.6	25	29
Currania dryopteris	1.3	0.5	2.6	6.5	2.7	26	28
Rubus saxatilis	0.8	1.1	0.7	4	6.0	27	24
Calamagrostis	0.6	2.9	0.2	3	15.9	28	11
Sorbus aucuparia (twigs)		1.5			8.2		21
Betula alba (twigs)		1.4			7.7		22
Populus tremula (twigs)		1.1			6.0		23
Salix caprea (twigs)		0.1			0.5		30
Mean and sd	**8.5±8.4**	**4.1±5.7**	**3.4±2.7**	**42.7± 41.7**	**18.0± 19.7**		
Pinus sylvestris				16	39*		

* in 1997

The relationship between rooting depth and radiocaesium migration may partly explain the differences in the contamination trends of plants. This is undoubtedly the most likely explanation in the case of the shallow-rooting *Oxalis acetosella* and pine which has a deep tap root. Furthermore, the contribution of successive cohorts of ever deeper-rooting mycorrhizal fungi in this species cannot be excluded. The amount of radiocaesium stored in the *Liliaceace* runners was probably depleted and is now beginning to be found in decontaminated surface horizons. On the other hand, the fibrous root-system of grasses enables them to keep a good anchorage in deeper humiferous horizons that are still relatively heavily contaminated. *Ericales* and ferns often have running roots. However, they also produce numerous, more or less fibrous, adventitious roots that explore deep soil layers. One should also bear in mind that these plants are mycotrophic, and therefore the buffer role of mycorrhizal fungi in the decontamination of these species cannot be precluded.

Variations at a temperate site

A major investigation was carried out in the autumn 1997 on vegetation from the forest stations in Belgium and Luxembourg, which had already been investigated in a previous project[2]. The objective of this investigation was to monitor the evolution of contamination over time, including factors involved in soil-plant transfer, in some widely distributed plant bioindicators at a station in a temperate area but closely related ecologically to stations in the boreal forest (Saint-Léger station, pine stand on podzol soils).

On average (Table 3), forest plants lost 70% of their activity in the decade 1987–1997. This coincided with the average loss recorded at Hille in Sweden over the same period of time.

[2] EC-Contract: FI3PCT9200

Table 3. Evolution of ^{137}Cs specific activity (Bq kg^{-1} dry wt.) at Saint-Léger, Belgium. Aggregated transfer factors (T_{ag})) (10^{-3} m² kg^{-1}) are given in brackets.

Species	1987	1989	1990	1992	1993	1997
Beech (twigs of young trees)	260 (68)	105 (29)	260 (72)	100 (29)		76 (24)
Oak (twigs of young trees)	530 (139)		820 (227)	610 (175)		470 (149)
Birch (twigs of young trees)						105 (27)
Pine (one-year-old needles of young trees)	55 (15)		300 (83)	390 (112)		525 (166)
Dryopteris carthusiana (leaves of the fern)				1313 (376)	2670 (783)	1370 (434)
Calluna vulgaris (twigs)		1910 (520)			1170 (343)	760 (241)
Deschampsia flexuosa (leaves)		1230 (335)				377 (119)
Vaccinium myrtillus (twigs)	1000 (263)	910 (248)			730 (214)	560 (177)
Rubus idaeus (twigs)	500 (136)			120 (33)	70 (21)	43 (14)

Variations between sites

To fascilitate a comparison between the data, aggregated transfer factors, T_{ag} (Bq kg^{-1} per kBq m^{-2}), have been used. The T_{ag} values in vegetation ranged from 0.2 to 241 (10^{-3} m^2 kg^{-1}). The lowest values were found in young plants of deciduous trees and in herbs while the highest values were found in dwarf shrubs (Table 4). On average, the general trend for ^{137}Cs in the investigated vascular plants from the seven sites was: *Salix spp< Betula spp< Epilobium angustifolium< Deschampsia flexuosa < Pinus sylvestris< Vaccinium spp<Calluna vulgaris*, with same variations between the sites. The relation between *Vaccinium myrtillus* and *Calluna vulgaris* seems to be general (Guillitte et al,1994, Fawaris and Johansson, 1994, Nelin and Nylén 1994).

Table 4. Comparison of the ^{137}Cs aggregated transfer factors (T_{ag}) (10^{-3} m² kg^{-1}) between seven European sites in 1997-98.

Species	**Saint-Lèger**	**Långsjön**	**Hille**	**Prylen**	**Vindeln**	**Liesi-neva**	**Siika-kangas**	**Mean All areas**
Calluna vulgaris	241	51.5	24.8		81.4			100
Vaccinium myrtillus	177	16.7	12.6	14.8	25.9	148.4	57.8	65
Vaccinum vitis-idea			13.8		33.3	111	69.9	57
Pinus sylvestris	166	5.4	3.9	9.7	38.8			45
Deschampsia flexuosa		17.9	17.6	3.2	15.6		30.4	17
Epilobium angustifolium			20.6	4.6	6.8			11
Betula sp.	27	3	3.8	4	10.7			9.7
Sorbus aucuparia		4.4	7	4.4				5.3
Salix caprea		7.6	0.7	0.2	1			2.3
Salix fragilis		1.7		0.4				1.0
Mean for each area	**150**	**12**	**14**	**5.2**	**27**	**130**	**53**	
S.D.	**90**	**16**	**8.3**	**4.9**	**25**	**26**	**20**	

The comparison between the different sites further shows that Saint-Léger generated the highest mean T_{ag} values followed by Liesineva (peat land), Siikakangas and Vindeln while the sites Långsjön, Hille and Prylen had the lowest mean values for T_{ag}. Vindeln in northern Sweden, under boreal conditions, had a transfer factor up to 10 times greater than those in central Sweden. Explanations for this may be different soil properties (Annex 1, Table 2) (Rosén et al, 1999) and differences in rainfall (Annex 1, Table 1). It can be expected that the higher the rainfall the closer the fungi-plant relationship, which also underlines the importance of these organisms. However, the differences in annual rainfall are not very large.

The long term annual changes in aggregated transfer factors could be compared for three species and sites. The activity in heather (*Calluna vulgaris*) changed over time in a very similar way for the studied sites. Assuming an exponential decrease in activity, the half time for ^{137}Cs, 4 and 12 years after the Chernobyl accident, was very similar for the three sites and about 7 years in average (Figure 5).

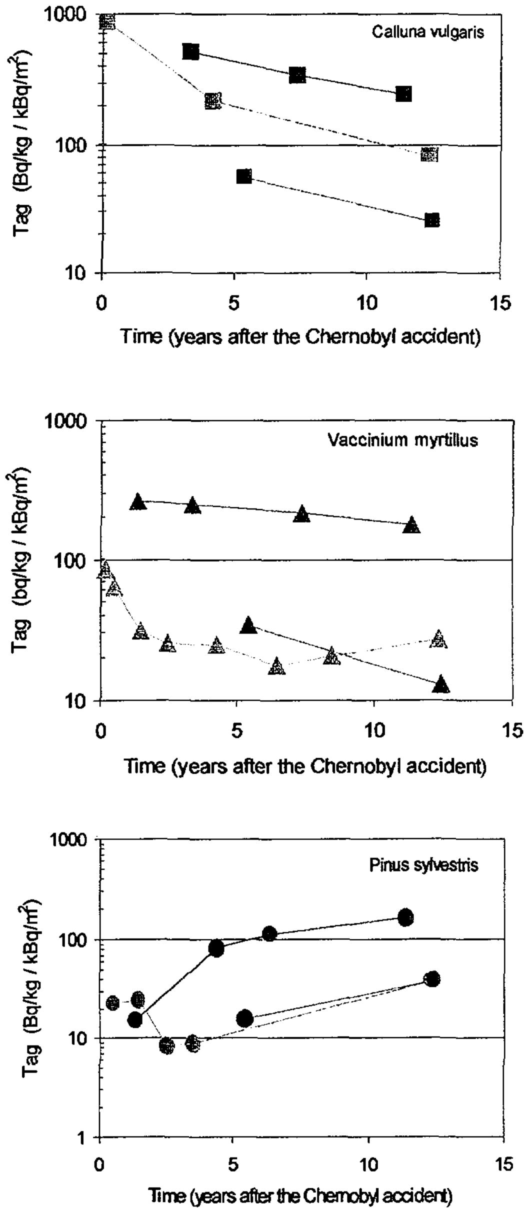

Figure 5. Temporal changes of ^{137}Cs (T_{ag}) in green parts of heather and bilberry twigs, and current shoots in young Scots pine for three regions: Saint-Léger, Belgium (red); Hille, Sweden (blue) and Vindeln, Sweden (green).

In bilberry twigs (*Vaccinium myrtillus*) ^{137}Cs behaved differently between the three sites. A monotonous decrease in T_{ag} was observed between 1989 and 1997 for Saint-Léger and Hille while in Vindeln the T_{ag} values seem to stabilise after 1989 (the relative S.D in Vindeln is >50% for n=20). For Scots pine (*Pinus sylvestris*) the T_{ag} values have increased dramatically since a few years after the deposition. Although the amplitudes are different the rate of increasing T_{ag} is equal between the sites indicating (as in the case of *Calluna vulgaris*) that a general process is acting. This can be a protracting redistribution of Cs in the forest floor from the mosses and litter layers down to the humus and mineral soil where pine roots are found.

The results indicate that the T_{ag} values of identical species from two biogeographical areas have either increased or decreased over time. The average ratio between plants from the temperate Saint-Léger station and the boreal Hille station increased only marginally, from 5.7 to 5.8, but the standard deviation of this average rose substantially, from 1.9 to 3.3. On the other hand, the gap between the individual T_{ag} ratios increased over time (Table 5). Although this development challenges the ratios determined by Guillitte *et al.* in 1994, it does not fundamentally alter the ranking of plants based on contamination level suggested by the same authors.

These differences in behaviour may again be considered in relation to rainfall. The impact of rainfall is particularly strong in the growing season on migration, which occurs at a faster rate in the temperate area, but also on mycorrhization, which is promoted in the temperate area by limited rainfall during the vegetative stage.

Table 5. Evolution of the ^{137}Cs aggregate transfer factor (T_{ag})) (10^{-3} m^2 kg^{-1}) at two pine stands on a podzol soil, at Saint-Léger, Belgium and Hille, Sweden, 1991–1997

Species	Year	Saint-Léger	Hille	Saint-Léger/ Hille
Dryopteris carthusiania	1991	775*	195	4.0
	1997	435	89.6*	4.9
Calluna vulgaris	1991	425	55	7.7
	1997	241	30.5*	7.9
Vaccinium myrtillus	1991	281*	39	7.2
	1997	119	21.6*	5.5
Rubus idaeus	1991	45*	14.5	3.0
	1997	14	12.9*	1.1
Pinus sylvestris	1991	95*	16	5.9
	1997	166	39	4.3
Mean value - standard deviation	**1991** **1997**			**5.7 ±1.9** **5.8 ± 3.3**

* linear extrapolation from the value measured in 1990 at Saint-Léger or in 1998 at Hille

Conclusions

Both at the temperate and boreal sites in this study, the data show an average decrease of 70% in plants during the last decade. The most likely explanation for this decrease is a downward migration of radiocasesium in soil.

Even though the transfer from soil to plant differs in absolute values, in some cases by more than an order of magnitude, some species show very similar temporal trends for the investigated sites.

The monotonously increasing T_{ag} values in young pines will be subject for further investigations since this may be of significance for external doses, internal doses via meat from moose and Gallinaceous birds, for forestry in general (biofuels) and the steady state levels for the different compounds in the forest ecosystem.

Mosses and lichens

Mosses and lichens can play a significant role in the contamination of food chains. In this respect, the lichen-reindeer-human chain is the most well known. The involvement of mosses and lichens in the food chain outside northern areas is negligible. They can nevertheless prove highly valuable as bioindicators of a radioactive deposition. The study of their ecological half-lives is therefore very useful. The results of measurements on mosses and lichens are summarized in Tables 6-8. Table 6 shows the evolution of the ^{137}Cs contamination of lichen and mosses at the Hille site. Table 7 compares the activity in some of these organisms and the deposited amounts retained in their mats at Vindeln. Table 8 compares their T_{ag} at the same locations.

Table 6. Evolution of the ^{137}Cs concentrations (kBq kg^{-1} dry wt) and deposition in the mat of various cryptogams at Hille

Species	**Microniches**	**1991**	**1998**	**1991/1998**	**Deposition in mat kBq m^{-2}**
Hypogymnia physodes	Trunk	36	8.4	4.3	
	Branches	65	14.4	4.5	
Cladonia sp.	Soil		5.5		4.9
Parmelia saxatilis	Rock		23.0		14.9
Dicranum sp.	Rock		17.9		29.3
	Soil	18	4.6	3.9	3.0
Pleurozium schreberi	Rock		10.7		20.7
	Soil	14	5.7	2.5	3.8
Ptilium cristatum	Soil	11	4.2	2.6	1.5
Polytrichum formosum	Soil	16	23.8	0.7	11.5
Sphagnum sp.	Soil		4.0		18.6

Table 7. ^{137}Cs concentrations (kBq kg^{-1} dry wt) and deposition in the mat of various cryptogams in 1998 at Vindeln

Species	**Microniches**	**Specific activity**	**Deposition in mat**
Cladonia sp.	Rock	5.6	3.8
	Soil	2.1	1.8
Dicranum sp	Rock	3.7	10.4
	Soil	1.8	2.0
Hypogymnia physodes	Trunk	4.5	
	Branches	4.6	

Table 8. Comparison of ^{137}Cs aggregated transfer factors, T_{ag} (10^{-3} m^2 kg^{-1}), of various cryptogams in 1998 at Vindeln and Hille

Microniches	**Species**	**Vindeln**	**Hille**	**Vindeln/Hille**
Rock	Lichens	324	124	2.6
	Mosses	213	97	2.2
Branches	Lichens	266	78	3.4
Trunk	Lichens	260	45	5.7
Soil	Lichens	121	30	4.0
	Mosses	104	25	4.2
Mean value and sd		**215 ±87**	**67 ± 40**	**3.7 ±1.3**

With regard to the specific activity, it appears that differences within sites continue to be more important than differences between sites. However, these latter differences are much smaller than those found in vascular plants. The value of deposited radiocaesium (in kBq/m^2) was similar to the specific activity. At the Hille station, only *sphagnum* plants retained large amounts of deposited radiocaesium in their mats — up to 9% of the initial amount — 12 years after the Chernobyl accident. This is probably attributable to their large biomass compared to other species. At the same site, other mosses and lichens retained 0.5–5% of the initial amount found in their mats, with average values varying from 3% under pine and 1.2% under spruce.

Mats collected from flat rocks were 3–5 times more heavily contaminated than those collected from the soil surface. Mats make up enclosed niches in which radiocaesium is strongly attached to organic debris accumulating at their foot, whereas on the soil surface the leached radiocaesium moves into the ground. In the 1991–1998 time period, lichens growing vertically on tree trunks and horizontally at the Hille station had the same decontamination level as lichens growing at temperate stations. Their ecological half-life of 4.7 years was of the same order of magnitude as that determined earlier for lichens growing in the temperate area. Over the same period, mosses lost 66% of their specific activity, which is close to the 70% recorded in vascular plants. This loss is equivalent to an ecological half-life of 6.4 years and similar to the half-life determined at the site Saint-Léger, in the temperate area ($T_{1/2}$=5.7 years). The small difference in behaviour between these two biogeographical areas was unexpected. It confirms an earlier hypothesis (Guillitte *et al.*, 1990) that a distinction has to be made between a phase of rapid decontamination — characterised by a highly variable half-life depending on their geographical location or location within the station — and the following phase of slower decontamination, with less variable half-lives.

The relative contamination of mosses and lichens on different substrates followed a similar pattern at Hille and Vindeln. The similarities were especially striking when comparing the specific activity and surface activity of fungal mats growing horizontally on rocks. This observation justifies the interest shown by Giovani *et al.* (1994) in this sampling method as a way of assessing the differences in depositions between areas.

Conclusions

Lichens and mosses are suitable bioindicators for radioactive contamination. A striking result found in this investigation is the similarity in ecological half-lives in the boreal and temperate areas for the long term. The halflife is approximately 5 years.

In the studied sites, the concentrations in mosses have decreased by approximately 65% during the last decade.

Fungi

Mushrooms are the most heavily contaminated organisms in forest ecosystems and many of them play a direct and significant role in the contamination of food chains by radioactivity. Successive sample collections made at the Hille site confirm the main trends described in a previous project[3]. Unfortunately, data from recent samplings in Belgian Lorraine and Luxembourg are still too fragmentary and heterogeneous to draw final conclusions.

The role of fungi is not restricted to their direct impact on food. Two complementary research approaches were followed: (i) identification of a contamination process of plants from the forest cover growing in the vicinity of the fruitbodies of contaminated species (experiment conducted *in situ* in

[3] EC-Contract: FI3PCT9200

three boreal forest stands); and (ii) identification of a contamination process of forest trees through mycorrhization (experiment performed in a controlled chamber).

The contamination and decontamination curves of mushrooms reflect the evolution of the activities in the various soil horizons (Moberg et al, 1998). This characteristic has to be considered in the light of the specific use of soil horizons by the mycelium of these organisms. The profile of litter-loving species shows an exponential decrease, whereas other species have a bell-shaped curve with the contamination peak moving further away from the time of deposition according to the depth of the mycelium. Whereas all the species from the temperate forest appear to have reached their contamination peak 10 years after the Chernobyl fallout (the partial data currently available do not contradict this statement), in the boreal forest at least one species, *Cortinarius albo-violaceus*, reached its maximum activity 12 years after the initial fallout. The mycelium of this mushroom usually develops in the lower part of the semi-organic horizons (Ah) and, in the case of moder humus, in the OAh horizon. This is a very shallow horizon which consists of highly decomposed organic matter and has a greasy texture. All analyses show that there is a significant decontamination of the holorganic horizon (O) at the expense of semi-organic horizons (Ah) and even mineral horizons (E or B). The migration of soil contamination to the bottom of the Ah (and/or OAh) horizons, combined with the particular organic texture of the lower part of these horizons, explains the high availability of radiocaesium for mushrooms growing mainly or even partially in these horizons.

Apart from *Cortinarius albo-violaceus*, the data show an overall reduction in contamination, in average approximately 70%, during 1991 to 1998 (Table 9). When comparing the average activity of obligate mycorrhizal species (usually characterised by a deep mycelium) with that of facultative mycorrhizal species (which have a more superficial mycelium) during this period, one notices a reduction of 60% and 80%, respectively.

Table 9. Evolution of ^{137}Cs specific activity ($kBq\ kg^{-1}$ dry wt.) in mushrooms from 1991 to 1998 at the site Hille. Between brackets: minimum and maximum values. Ecophysiological types: OM = obligate mycorrhizae, FM = facultative mycorrhizae, HS = humo-terricolous saprophytic mycorrhizae, LS = lignicolous saprophytic mycorrhizae.

Species	Types	1991	1993	1997	1998
Calvatia excipuliformis	SH		0.0		
Lactarius pubescens	MO			0.3	0.3
Telephora palmata	MO				0.4
Ramaria ochraceovirens	SH				0.6
Lactarius deterrimus	MO		0.8?		
Tricholoma saponaceum	MO		1.3		
Russula nigricans	MO				3.1
Laccaria bicolor	MF				4 0
Gyromytra esculenta	SH				7,7
Lycoperdon perlatum	SL	5			
Suillus bovinus	MO		5, 4 [2.2-8.5]		
Chroogomphus rutilus	MO		9.3		
Suillus granulatus	MO		14		
Collybia butyracea	SH			7	
Coltricia perennis	SL	9			
Leccinum scabrum	MO		0.8	11	
Ramaria invalii	SL	14			
Amanita muscaria	MO				12
Lactarius helvus	MO	17	93	28	
Russula obscura	MO		21		
Russula claroflava	MO		25 [23-28]		
Russula aeruginosa	MO	22	16		
Hygrophorus pustulatus	MO				20
Hygrophorus agathosmus	MO		27		
Melanoleuca sp.	SL	29			
Hydnellum aurantiacum	MO		15 [11-18]		31
Russula vesca	MO	42			
Boletus edulis	MO	48	21 [14-31]	23 [13- 44]	12
Amanita fulva	MF	52	9.5	29	
Amanita rubescens	MO	55			

Table 9, continued...					
Species	**Types**	**1991**	**1993**	**1997**	**1998**
Russula veternosa	MO	55			
Russula integra	MO	55			
Russula aurata	MO	55			
Russula xerampelina	MO		0.15?	41	
Cantharellus cibarius	MO	65	3.6	3.9	
Porphyrellus porphyrosporus	MO				50
Cortinarius camphoratus	MO			64	
Gomphidius glutinosus	MO	72	1.3?		
Chalciporus piperatus	MO		[7.1-73]		
Agaricus augustus	SH		76		
Lactarius rufus	MF	76	87 [28-149]	21	42
Russula atropurpurea	MO	83			
Russula emetica	MO		97		
Chroogomphus rutilus	MO	93			
Hygrophorus olivaceoalbus	MO	110			25
Russula decolorans	MO	115	47[40-55]	25 [20-40]	
Lactarius deliciosus	MO				97
Stropharia hornemanii	SL		140		
Tylopilus felleus	MF	139	118 [66-201]		
Amanita porphyria	MF	140			32
Suillus luteus	MO	140			
Cantharellula umbonata	SF				133, 97
Paxillus involutus	MF	172	30 [9- 57]		
Suillus variegatus	MO	177	368	85 [33-119]	
Tricholomopsis rutilans	SL	180			
Gymnopilus penetrans	SL	180			
Paxillus atratomentosus	SL	210			
Hygrophoropsis aurantiaca	SH	239	26		
Clitocybe clavipes	SH	260	438	20	
Cantharellus lutescens	MF	277	52.4 [30-106]		
Russula paludosa	MO	280	82	74 [71-77]	
Tricholoma album	MO		288		177
Cystoderma cfr amianthinum	SH				275
Hydnum repandum	MF	317	22 [9-35]	113 [49-154]	
Hydnum rufescens	MF		59 [20-39]		
Cortinarius elatior	MO			312	
Lactarius theiogalus	MF	340		85	
Dermocybe semi-sanguinea	MF		283 [33-488]	254	218
Dermocybe sanguineus	MF	340		249	219
Lactarius camphoratus	MF	373	247	92 [75-109]	101
Xerocomus badius	MF	390	222		
Cortinarius brunneus	MO	435	76		57
Cortinarius anomalus	MO		655 [465-844]	371 [337-405]	220
Cortinarius alboviolaceus	MO		360	558 [441-675]	676 [511-995]
Cortinarius camphoratus	MO			640 [631-650]	
Dermocybe cinnamomea	MF	950	384 [229-646]		
Rozites caperata	MO		438 [230-974]	217 [172-262]	152 [87-212]
Cortinarius violaceus	MO			1030	

The comparison of T_{ag}-values of nine bioindicator species at Hille and Vindeln shows a high degree of similarity in the behaviour of these species (Table 10).

Table 10. Comparison of the mean ^{137}Cs T_{ag} values (m^2 kg^{-1}) of 9 fungal species in 1997–1998, Hille and Vindeln.

Species	**Vindeln**	**Hille**	**Vindeln/Hille**
Cortinarius albo-violaceus	3.93	3.18	1.24
Cortinarius anomalus	2.49	1.53	1.63
Dermocybe sanguineus	1.97	1.21	1.63
Suillus variegatus	0.58	0.44	1.32
Russula decolorans	0.46	0.13	3.54
Rozites caperata	0.40	0.95	0.42
Russula paludosa	0.35	0.38	0.92
Lactarius rufus	0.23	0.16	1.44
Boletus edulis	0.03	0.09	0.33
Mean	**1.16**	**0.94**	**1.23**

This similarity is even more pronounced (r=0,96) than in other groups of bioindicators (Table 11). On the other hand, the ratio of the average T_{ag} values is lower in the group of mushrooms than in the other two groups. The T_{ag} of *Dermocybe sanguineus* (2.86) is the only data that can be used to compare the boreal stations with the Saint-Léger station. It indicates that, like the soil/plant transfer in other ecological groups, the soil/mushroom transfer is more important in temperate areas.

Table 11. Comparison of the mean ^{137}Cs T_{ag} values ($m^2\ kg^{-1}$) of three ecological groups in 1997–1998, Hille and Vindeln.

Ecological groups	Number of bioindicators	Vindeln	Hille	Vindeln/Hille	Correlation
Fungi	9	1.160	0.940	1.23	0.96
Mosses and lichens	5	0.215	0.066	3.26	0.85
Vascular plants	8	0.027	0.013	2.07	0.51

The analyses of the soil columns collected under the carpophores of several mushroom species with relatively bulky and contaminated carpophores and of soil columns collected away from fruitbodies at Hille and Vindeln in 1998 did not show any significant difference neither in specific activity nor in deposition in the various soil horizons (O, Ah, E or B, moss and lichen mats). Therefore, it would seem that no contamination spots occur in areas where the carpophores of these species usually grow, contrary to conclusions based on the measurements recorded at Vindeln in 1993. The only trend that was significant and consistent with the previous experiment was the systematically higher contamination in bilberry growing near carpophores (Table 12), it was 4 and 1.25 times higher at Hille and Vindeln, respectively.

Table 12. Comparison of the ^{137}Cs T_{ag} values ($10^{-3}\ m^2\ kg^{-1}$) of 3 Ericales species according to the location of their fruiting bodies, Hille and Vindeln, autumn 1998.

	Around fruiting bodies	Away from fruiting bodies
Hille: *Vaccinium myrtillus/Rozites caperata*	27.4	7.2
Vindeln: *Vaccinium myrtillus/ Cortinarius alboviolaceus*	33.3	24.2
Vindeln: *Vaccinium myrtillus/ Suillus variegatus*	41.2	30.5
Hille: *Vaccinium vitis-idea/Rozites caperata*	17.3	4.4
Hille: *Calluna vulgaris/Rozites caperata*	26.3	34.5

At Hille, cowberry (*Vaccinium vitis-idea*) and bilberry had exactly the same profile. On the other hand, *Calluna vulgaris* did not seem to be affected by the proximity of carpophores. This difference in behaviour probably has more to do with the competition between fungal species and surrounding vascular plants for the available caesium in the soil than with the accumulation of caesium over the years resulting from the successive appearance and decay of fruitbodies in identical areas, as was initially thought.

This experiment also provided a rough assessment of the proportion of caesium retained in fruitbodies in their growing area. The assessment was based on the average number of fruitbodies per m^2, i.e. from 4 to 8 fruitbodies per m^2, depending on their size. Thus, the average value in *Cortinarius alboviolaceusis* was about 5% of the deposition, whereas it varied from 1.3 to 1.7% in other mushrooms. (For mushrooms that have lost 70% of their contamination over the past 7 years, one can estimate that at least 5% of the total deposition had been retained at the time of maximum fruitbody contamination). These values are to be compared with a retention rate of 1–5% of the total deposition in moss mats at Hille, 2% in pine trees at the same station, 10% in moss and lichen mats at Vindeln, and 0.9% in bilberry mats at the same station.

Experiments carried out in a controlled chamber did not reveal any significant differences in contamination in the needles and wood of young spruces (collected at Hille) that are naturally associated with various mycorrhizal species. Although the selection of these plants was based on the fruitbodies growing in their vicinity, it was noticed prior to the analysis that most of the plants were associated with several types of mycorrhizae — and not exclusively with those present initially — thereby creating a more homogeneous rhizosphere around plants. This could have been the situation at the outset. However, the presence of these "new" mycorrhizae was probably due to the germination of the spores contained in the undisturbed soils. Because of the excessively small amount of mycorrhizal root apices, it is not possible to determine the specific activity in the various types of mycorrhizae involved. These apices have an average contamination of 12 kBq/kg, which is roughly 10 times lower than the average contamination of the fruitbodies of the collected species. This observation coincides with results based on the samples collected *in situ* in 1993 at Hille, where root apices associated with *Dermocybe* were also 10 times less contaminated than the carpophores of this mycorrhiza. By contrast, the non-mycorrhizal root tips (2.2 kBq/kg) of the plants under investigation were significantly less contaminated than the mycorrhizal root apices.

Because of insufficient biomass production, the mycorrhizal spruce seedlings inoculated artificially with *Lactarius rufus* — a strain from Hille — could not be separated into their various parts. After six months, mycorrhizal plants have a lower specific activity (-25%) than non-mycorrhizal plants; however, the weight increase of mycorrhizal plants (+56%) is such that the outcome is a global soil/plant radiocaesium transfer exceeding 18%. All seedlings present a lower contamination than shoots of older plants in relation probably with less deep root system. It is naturally impossible to extrapolate results on the basis of a single successful experiment using a single inoculum on a single host plant species at a very early growth stage. But the findings nevertheless support the need for continuing the investigations on the role of mycorrhizae in the active radiocaesium transfer from the soil to host plants.

Conclusions

The data available indicate an overall reduction of ^{137}Cs concentrations in mushrooms between 1991 and 1998. When comparing the average activity of obligate mycorrhizal species (usually characterised by a deep mycelium) with that of facultative mycorrhizal species (which have a more superficial mycelium) during this period, there is a reduction of 60% and 80%, respectively.

A comparison of T_{ag} -values of nine bioindicator species at Hille and Vindeln shows a high degree of similarity in the behaviour of these species at the two sites. This similarity is even more pronounced than for bioindicators as mosses, lichens and vascular plants.

A rough assessment of the proportion of caesium retained in fruitbodies in their growing area shows that the average value in *Cortinarius alboviolaceusis* was about 5% of the deposition, whereas it varied from 1.3 to 1.7% in other mushrooms. In mushrooms that have lost 70% of their contamination over the past 7 years, one can estimate that at least 5% of the total deposition had been retained at the time of maximum fruitbody contamination.

There is a need for continuing the investigations on the role of mycorrhizae in the active radiocaesium transfer from the soil to host plants.

Trees

Introduction

Trees are often the most significant component in a forest ecosystem from an economical point of view. Concentrations of radiocaesium in different parts of trees vary, and are changing with time. The use of different parts of trees in forest industry, and the question of access of people to heavily contaminated forests motivate analysing the distribution of man-made radionuclides in trees in the short and long term. The redistribution of ^{137}Cs in coniferous trees has been studied in boreal forests in four research areas.

The main process removing deposited radioactive material from the crown and trunk is weathering, including by definition also throughfall and stemflow, which can remove both metabolised radionuclides, and those retained on plant surfaces. Removal rates through weathering vary significantly with time after the deposition event. Litterfall includes removal of needles or leaves as well as pieces of branches and bark. Roots release litter in the soil. Early phase litterfall may remove a considerable fraction of the radioactive material in the crown (Bergman *et al.* 1991), but in a later phase the removed activity fraction is very small. Before the litterfall, trees return a considerable part of nutrients, e.g. potassium from old needles or leaves to other parts of the tree for later use. An analogous behaviour is probable for metabolised caesium.

Root uptake adds to the radionuclide content of the tree roughly in proportion to growth intensity, and thus increases the radiocaesium content of the tree which originates from metabolised, initially deposited radiocaesium. This caesium will be found in both above-ground parts and in roots, and is varying with season of year.

Growth dynamics causes fluctuation in radiocaesium contents in conifer needles of different ages during the growth period. Different growth conditions add to the variation between sites. One advantage of sampling the needles for the study on seasonal variation during 1996-1997 was the absence of needle age classes contaminated directly by the Chernobyl fallout.

In LANDSCAPE, long term or seasonal variations in ^{137}Cs concentrations were analysed in stands dominated by Scots pine or Norway spruce (Annex 1, Table 1). Compiling previous results and new data from the sites studied allow a retrospective look at radiocaesium in wood, bark, branches, twigs and needles. Using contents related with local deposition of radiocaesium facilitated comparison of the sites. The changes in ^{137}Cs contents of different parts of trees were used to compare radiocaesium and potassium dynamics in trees grown at different sites. The vertical distribution of ^{137}Cs in the trees as well as the dynamic redistribution between 1986 and 1997 were estimated.

Long term variations

The distribution of ^{137}Cs in trees has been studied in wood, bark, branches, shoots and needles of Scots pine. The aim has been to estimate the temporal changes in inventories of radiocaesium in the trees and to describe the vertical distribution and redistribution within the trees. Four trees were felled in 1991 and in 1997 at the Hille site. The trees were divided into bark and stem wood at different heights above the stump. The stem wood was also divided into wood created before 1986 and after 1986. Shoots/twigs and needles of available age classes, normally 4 or 5, were collected at the top, upper, middle and lower part of the canopy.

The results show that the total ^{137}Cs activity in bark is similar for the 1991 and 1997 trees, 7.0±2.8 and 7.2±2.5 kBq as an average of four trees with a total biomass of 10.8±3.5 and 10.3±1.8 kg for the two years. The concentration of ^{137}Cs in bark as a function of height above stump is shown in Figure 6.

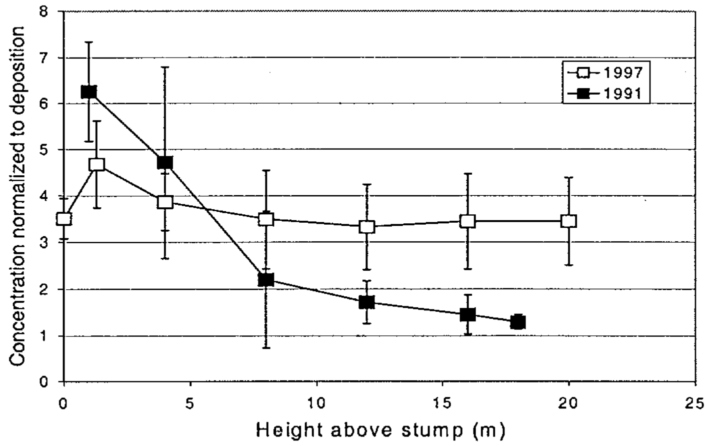

Figure 6. The normalised concentration of ^{137}Cs (Bq kg^{-1} dw per kBq m^{-2}) in bark at the Hille site as a function of height above the stump. Data are averages of four trees with the standard deviation of the mean.

The distribution seems to be more even in 1997. In particular, the concentration has increased at the top of the tree. This may be explained as a combination of the fact that the phloem, containing ^{137}Cs, in this case is part of the bark, and the effect of weathering/washoff. In 1991, there was still superficial ^{137}Cs on the bark, particularly at the lower parts of tree, while this has been largely washed off during the years after. At the same time the flow of ^{137}Cs through root uptake (phloem) has increased giving a more even distribution in the tree.

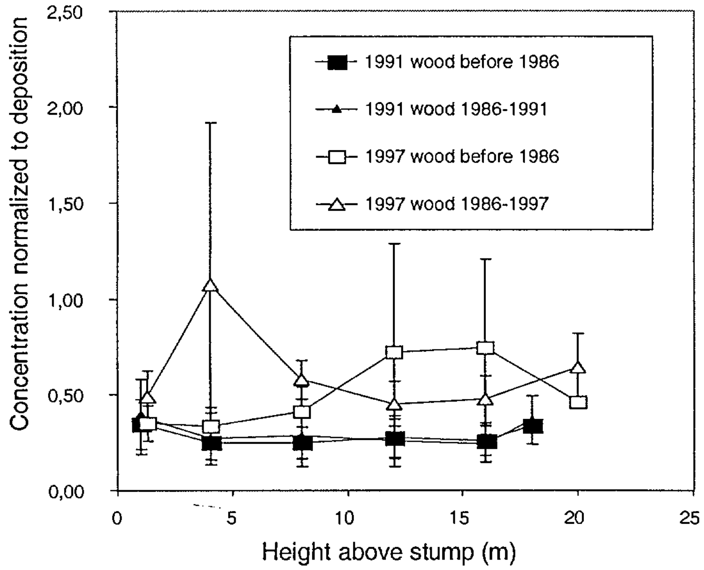

Figure 7. The normalised concentration of ^{137}Cs in wood (Bq kg^{-1}per kBq m^{-2}) at the Hille site as a function of tree height above the stub. Data are averages of four trees with the standard deviation of the mean.

The height distribution of ^{137}Cs in stem wood is shown in Figure 7. If the variability between different trees are taken into account, the concentration is rather evenly distributed with height. There does not seem to be any significant difference in caesium concentration between wood created before and after the Chernobyl accident in trees of the same age. However, there is a general increase of the ^{137}Cs concentration in stem wood between 1991 and 1997.

The concentration of ^{137}Cs in the new growth of twigs and needles of these trees has also increased during the period between the two samplings due to root uptake. The caesium concentrations are highest in the youngest twigs which is consistent with other measurements. (The temporal changes of ^{137}Cs in needles are also discussed in the sections on seasonal variations in needles and on ground vegetation.)

Based on the experimental results, the average activities of whole trees above ground have been estimated to 25.3 kBq and 39.6 kBq, in 1991 and 1997 respectively. With a tree density of 940 stems/ha, the total activity is then 2.4 kBq/m^2 and 3.7 kBq/m^2. The activity, in percentage of the ground deposition, stored in the above ground parts of the trees has increased from 1.1% in 1991 to 2.0% in 1997.

The temporal changes of ^{137}Cs concentrations in pine needles and the differences between the three sites Hille, Siikakangas and Vindeln are shown in Figure 8. The stands are all pine-dominated forests on mineral soil. The stands differ by age of dominant trees, which vary between 40 and 79 years (Annex 1, Table 1). The data for Vindeln show a rapid decline in ^{137}Cs concentrations due to weathering (loss of contaminated needles) in 1986. About ten years later the mean concentration at the same site is comparable to two other sites. The Hille site shows a clear increase in contents of ^{137}Cs from 1991 to 1997. The somewhat different average contents for the three sites in 1996-97 may reflect the ages of trees, as well as varying growth conditions at the sites.

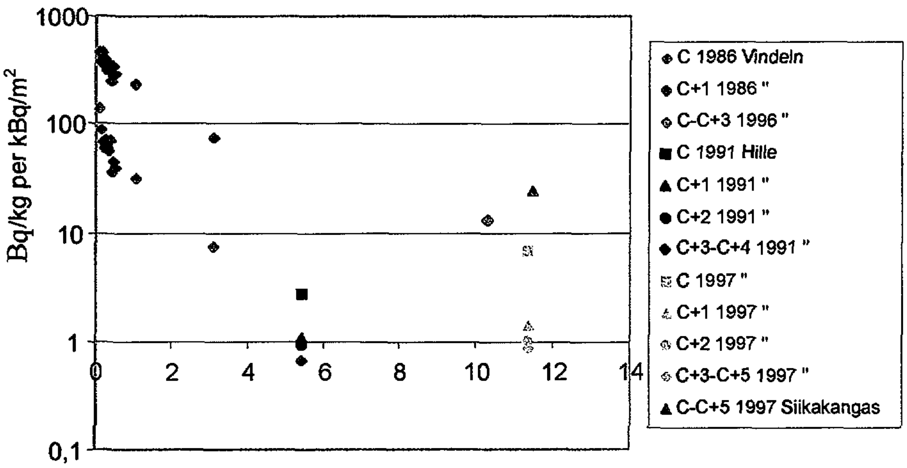

Figure 8. Long term variations in normalised activity concentrations of ^{137}Cs in pine needles at three sites after the Chernobyl accident in spring 1986.

Conclusions

Ten years after the Chernobyl accident the total inventory of ^{137}Cs in Scots pine (*Pinus sylvestris*) on mineral soils are increasing. At one site the inventory of ^{137}Cs has almost doubled from 1991 to 1997, which has implications on the industrial use of trees or tree parts.

Different contents of ^{137}Cs in pine nedles between sites in 1996-97 may reflect the ages of trees, as well as varying growth conditions at the sites.

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Seasonal variations in needles

Seasonal variations in concentrations and contents of ^{137}Cs and potassium were studied in Norway spruce and Scots pine needles in Finland and Sweden.

Needle samples were collected at one stand of Norway spruce (*Picea abies* (L.) Karst.) and one of Scots pine (*Pinus sylvestris* L.), in the main fallout area of Central Finland (Arvela et al. 1990)[4]. Samples were collected from 1 April 1996 to 28 February 1997 at two-week intervals during the growing season (May-October) and once a month during rest of the year. The tree-specific samples of current (C), one year old (C+1) and two years old (C+2) needles were collected from 25 randomly selected dominant or co-dominant trees per sample plot. The description of chemical analyses has been given in Raitio & Merilä (1998) and the measurements of ^{137}Cs in Klemola & Leppänen (1997).

In Sweden, needle samples were collected at a young Scots pine stand at Prylen (Annex 1, Table 1). Samples of needles were collected from 16 October 1995 to 4 November 1997 once a month during growing season and during dormancy once per year. The tree-specific samples of current and C+1 needles were collected from six trees, the same trees being followed throughout the study.

The foliar ^{137}Cs concentration (Bq/kg dry weight) decreased along ageing of needles for both Scots pine and Norway spruce. The ^{137}Cs concentration was highest in current needles at the beginning of growing season. For the comparison of sites and age classes of needles, the ^{137}Cs concentrations were related to the concentration in current needles on 14 October 1996 for Finnish sites (Ruovesi and Suupajoki) and on 21 October 1996 for Swedish sites (Prylen) (Figures 9, 10).

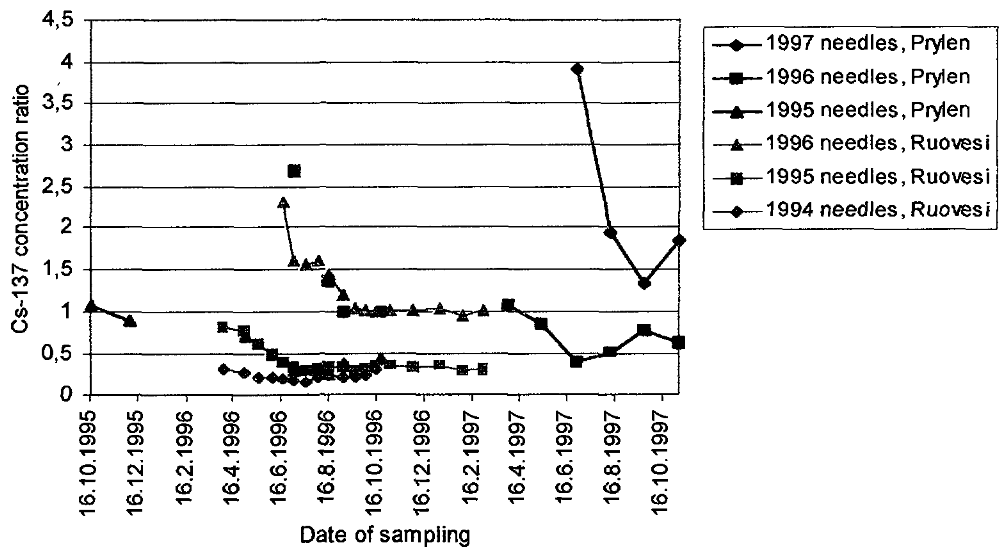

Figure 9. Seasonal variations of ^{137}Cs concentrations in Scots pine needles in a young stand at Prylen, Sweden and an advanced stand at Ruovesi, Finland. The ^{137}Cs concentrations (Bq kg^{-1} dw) have been normalised by dividing with the concentrations in 1996 needles on 21 October 1996 for the Swedish data and on 14 October 1996 for the Finnish data.

In C needles, radiocaesium concentration decreased during the growth period and levelled out during dormancy. In older needles, radiocaesium concentration decreased from the beginning of growing season until active growth ceased in late summer, and then stabilised. The ^{137}Cs content, mBq/needle,

[4] The stands belong to the Pan-European Intensive Monitoring Programme of Forest Ecosystems (ICP Forests Level II; UN-ECE 1995, 1997). Sampling and chemical analysis were made in connection of the pilot project (No 96.60.SF.003.0) financed by EC (Raitio & Merilä 1998).

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increased in growing current needles along the increase in dry weight. After the cessation of growth, radiocaesium content decreased in current needles and the decreasing trend of the content continued in older needles.

At the Ruovesi and Suupajoki sites, concentrations and contents of radiocaesium fluctuated more in growing needles than in older needles, whereas potassium varied seasonally in old needles as well. In all age classes, the seasonal variations were stronger in potassium and carbon compared to caesium. The correlation between the potassium concentration and dry mass of 10 needles was stronger than that between radiocaesium and dry mass. Among age classes, potassium showed closer dynamics to carbon or dry mass than did radiocaesium.

In general, seasonal variations in ^{137}Cs concentrations showed similar trends to those of potassium concentrations. However, trends in the potassium concentrations in C+1 and C+2 pine needles differed from those of radiocaesium in autumn by showing stronger increase before levelling out in winter.

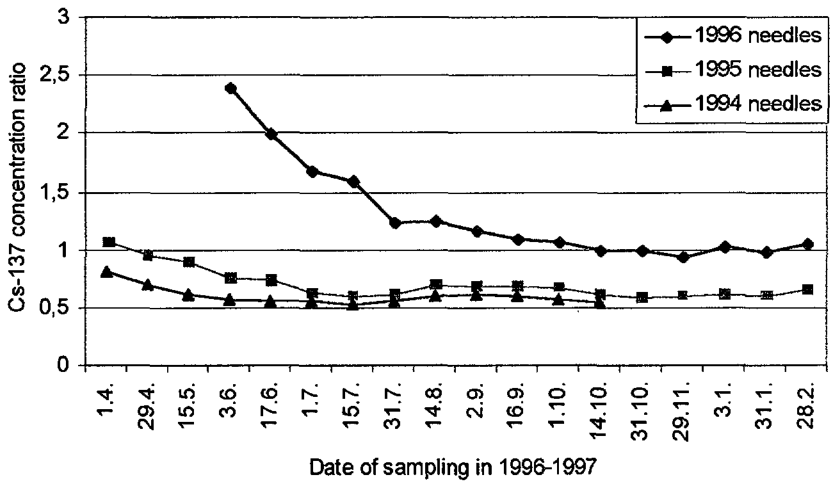

Figure 10. Seasonal variations of ^{137}Cs concentrations in Norway spruce needles in an advanced stand at Suupajoki, Finland. The ^{137}Cs concentrations (Bq kg^{-1} dw) have been normalised by dividing with the concentration in the 1996 needles on 14 October 1996.

Studies of the Chernobyl fallout in forests have shown that needles grown after initial deposition contain less radiocaesium than those directly exposed to airborne radioactivity (Nylén & Ericsson 1996, Nygren et al. 1994, Ertel & Ziegler 1991, Block & Pimpl 1990, Bunzl et al. 1989). Of the needle age classes grown after deposition, the current needles have a higher radiocaesium concentration than older needles as observed for Chernobyl derived ^{137}Cs (Nylen & Ericsson 1996, Raitio & Rantavaara 1994, Sombre et al. 1994, Nygren et al. 1994, Cousen 1991, Ronneau et al. 1991, Cousen 1989, Tobler et al. 1988), global fallout ^{137}Cs (Ertel & Ziegler 1991) and for stable caesium (Wyttenbach & Tobler 1988). A trend towards a similar, lower level of activity has been reported for older needle age classes (Nylén & Ericsson 1996, Sombre et al. 1994, Ertel & Ziegler 1991, Ronneau et al. 1991, Block & Pimpl 1990, Tobler et al. 1988).

Compared to annual variations, less research has been directed to seasonal variations of radiocaesium in needles (Nylén & Ericsson 1996, Sombre et al. 1994, Nygren et al. 1994, Ertel & Ziegler 1991, Tobler et al. 1988, Momoshima & Takashima 1981). These studies have revealed that in current needles the radiocaesium concentration is highest in the beginning of growth, decreases towards autumn when there can be a slight fluctuation in activity before levelling out in winter. The concentration in older needles changes also along the physiological changes in tree during the year.

Conclusions
In current needles the radiocaesium concentration is highest in the beginning of growth, decreases towards autumn when there can be a slight fluctuation in activity before levelling out in winter. The concentration in older needles also changes along the physiological changes in tree during the year.

There was no significant temporal variation neither in radiocaesium concentration nor in content after growing season for any of the needle age classes. Sampling of tree compartments in radioecological studies should therefore be carried out during dormancy, when the concentration of the determined substance is not fluctuating.

Litterfall and decomposition

Litterfall studies were performed at the sites at Hille and Långsjön from autumn 1996 to autumn 1998[5]. The results, the total litterfall, are expressed as kg/ha and as Bq/ m^2 (Table 13). The contribution from the needles alone are also given in the table. The relative contributions from the different parts of the litter varies during a year.

Table 13. The amount of litter generated by litterfall at the Hille and Långsjön sites 1996-1998

Site/Year	Total Litterfall			Needles only		
	kg/ha*y	Bq/ m^2	% of deposition	kg/ha*y	Bq/ m^2	% of deposition
Hille						
1996-97	3733	196	0.11	1438	31	0.02
1997-98	3329	204	0.11	2011	26	0.01
Långsjön						
1996-97	2550	140	0.14	1392	71	0.07
1997-98	2988	100	0.10	2340	62	0.06

Five years after the deposition, throughfall is of minor importance compared to the situation the first years, while litterfall still seems to generate a significant deposition (Nylén 1996). This is primarily explained by the loss of directly contimated needles and shoots during the 4-5 years after deposition. More than a decade after the deposition the contribution to the total deposition, corrected for radioactive decay, amounts to appr. 0.1% per year during 1996 to 1998, about the same at both Hille and Långsjön. The relative contribution from needles seem to be greater at the Långsjön site. In the northern boreal forest (Vindeln) the contribution from litterfall, the flux of ^{137}Cs, was found to be higher during years 5 to 8 after the accident compared to Hille and Långsjön. The mean value for these years was 0.6% (Nylén 1996) of the deposition. The flux of ^{137}Cs from the tree canopy to the forest floor at Hille and Långsjön agree well with the results (0.14%) obtained in 1992/93 in Ukraine (Belli and Tikhomirov 1996)

The decomposition of litter is a key process to mobilise elements including ^{137}Cs from the litter and make them available for plant uptake. In order to study the decomposition (primarily the rate of remobilising ^{137}Cs) a field experiment was performed at the Långsjön site. The litter was composed of

[5] Ten litter traps, with a total surface area of 2.87 m^2, were randomly placed on experimental areas of 30*30 m^2 and 25*25 m^2, respectively. The traps were emptied 3-4 times a year. During the second year at Hille only six traps remained and the data for the second year may therefore be somewhat more uncertain..

needles[6] from the site collected on 4 September 1996 and brought back in the form of litterbags (in total 288 bags) on 21 October. Bags containing litter needles from Långsjön have also been placed at a reference site (Lindholmen) with a very low ^{137}Cs deposition.

During two seasons studied, the weight of the litter needles has decreased to below 60% of its original value at Långsjön (Figure 11) while the activity concentration has increased to three times the initial value. The ratio between activity concentration and weight is also shown in the figure (right scale).

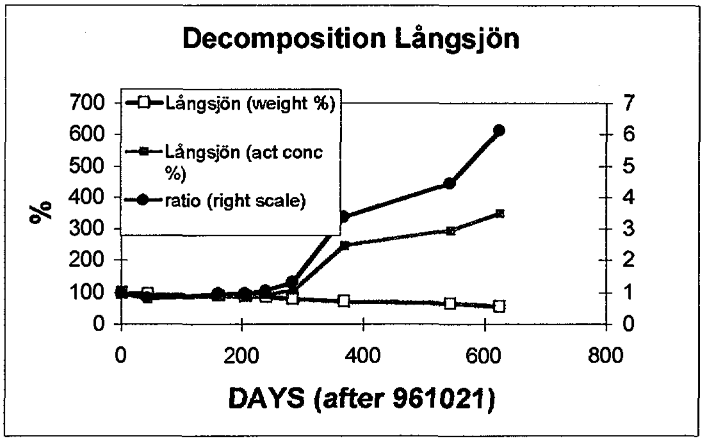

Figure 11. The decomposition of needle litter at the Långsjön site. Left scale shows ^{137}Cs concentrations and weight expressed as per cent of the initial values. Right scale shows the ratio between activity concentration and weight.

Figure 12 shows the corresponding results from Lindholmen, the reference site. The weight has decreased to less than 40%. At the same time also the activity concentration has decreased to about 50% of the initial values. A slow increase of activity concentration starts after 500 days but it is still below the starting value.

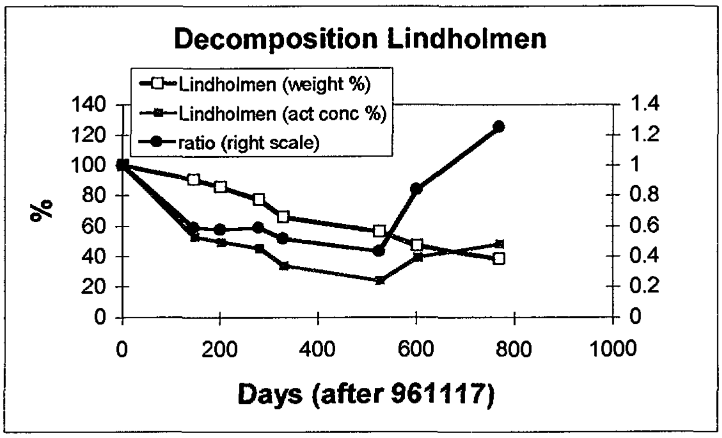

Figure 12. The decomposition of needle litter at the Lindholmen site. Left scale shows ^{137}Cs concentrations and weight expressed as per cent of the initial values. Right scale shows the ratio between activity concentration and weight.

[6] The needles have been dried, sorted, weighed and the intial content of radioactive caesium has been measured on representative samples. After packing in litterbags (11*13 cm^2, mesh 1mm), each bag containing about 1.5 grams, the bags have been replaced on the ground at the site (961021) in groups of 12 bags at 24 randomly chosen areas (1*1 m^2) within the test site (25*25 m^2). Before measuring the ^{137}Cs concentrations the collected bags have been grouped in samples from the same six areas at each occasion in order to obtain acceptable counting statistics. That means that each data point is the average of six measured samples.

The decomposition rate, expressed as the decrease of weight with time, is faster at Lindholmen than at Långsjön. This probably reflects the fact that Lindholmen is more diverse or nutrient rich than Långsjön, and it is also situtated 150 km south of Långsjön. The concentration of ^{137}Cs in litter at Långsjön decreases during the first winter but starts to increase during the following summer, while the ^{137}Cs at Lindholmen decreases until the second summer when a slow increase is observed.

The increase of ^{137}Cs in the litterbags in Långsjön is in accordance with earlier findings (Belli and Tikhomirov, 1996). This is not a concentration effect as the weight loss by decomposition does not exceed the loss of ^{137}Cs (Figure 9). The explanation for the increase is rather an importation of ^{137}Cs from outside the bag. The most probable vector for this importation is fungi. As a consequence only a fraction of the ^{137}Cs in the needles is initially released/mobilized. After two years, there is an increase of activity in the litter. These experiments can not reveal the relative contributions from caesium initially in the needles and the caesium imported. The different behaviour of the needles at Lindholmen may simply be a consequence of a much lower ground deposition, and accordingly less ^{137}Cs available. The needles initially had a content of ^{137}Cs corresponding to the situation at Långsjön.

Conclusions

The annual contribution from litterfall to the ground deposition a decade after the deposition is only a few tens of a percent of the total deposition.

In the high deposition area, there seems to be an initial release/mobilisation of ^{137}Cs, but then a substantial increase in the content of ^{137}Cs in the litter. This is probably caused by the interaction with fungi.

Forest management

Introduction

When assessing the influence of the different forest ecosystems on the exposure of the human population to radiation, it is necessary to know how radionuclides cycle also in forest ecosystems that have been subjected to various management measures. The most common management measures are fertilisation, liming and site preparation.

Potassium is probably the most interesting fertiliser affecting the distribution of radiocaesium in forest ecosystems. Generally, peat soils contain quite small amounts of potassium compared to the other main nutrients and compared to the amounts bound in the tree stands. In the root layer (0-20 cm) there may be less potassium than in a tree stand with 150 m^3 of stem wood. It has been estimated that there may be about 1.4 million hectares of drained peatland forests in the need of potassium fertilisation for their sustainable development in Finland where this study was performed. Preliminary results indicate that trees with good potassium nutrition concentrate ^{137}Cs and ^{134}Cs into wood, bark, branches and needles less than trees suffering from potassium deficiency. They also indicate that ^{137}Cs and ^{134}Cs leach downwards in the peat profile faster in potassium fertilised than unfertilised stands and thus out of the active element cycle in a peatland ecosystem. Potassium fertilisation seems to have two advantages on peatlands; (i) it increases tree growth and (ii) it reduces the root uptake of ^{137}Cs and ^{134}Cs by vegetation.

Mineral soils, on the other hand, normally contain enough potassium for the tree growth. Instead, nitrogen is quite often a growth-limiting factor on podzol soils. However, potassium fertilisation has not been observed to have harmful effects on tree growth especially if used together with nitrogen.

The aim of this study of forest management was to investigate the effects of potassium fertilisation on the radiocaesium distribution in Scots pine stands in a sub-dry site on mineral soil and on a transformed tall-sedge pine fen. The aim was also to assess whether fertilisation can be a possible countermeasure or restoration technique in a radioactive fallout situation.

Material and methods

The fertilisation experiments are located in western Finland, where the radioactive fallout resulting from the Chernobyl accident was relatively high (Arvela et al. 1990). The first of the experiments (Siikakangas) was established by the Finnish Forest Research Institute in 1980 to investigate the effect of nitrogen fertilisation on the growth of Scots pine stand growing on a sub-dry site. The experiment consists of sample plots differing from each other in terms of fertilisation treatments. Control and NPK-fertilisation (N 180, P 80, K 300 kg/ha) treatments with two replicates were chosen for this study. The experiment was refertilised in 1985 by applying the same amounts of nutrients as in 1980.

The second experiment (Liesineva) was located on a peatland experimental area drained with open ditches (ditch spacing 60-100 m) in 1934-36 and supplemented with covered ditches (20-60 m) in 1949-51. These were opened and the open ditches cleaned in 1987. The site type is transformed tall-sedge pine fen with a fully stocked Scots pine stand. The experimental area was PK-fertilised between 1958 and 1962. The effect of potassium was negligable in 1976, and a refertilisation experiment was established in the area in the same year. The two plots chosen for this study were fertilised with PK (P 55, K 96 kg/ha) in 1976 and with K (80 kg/ha) in 1989. The two control plots have not been treated since 1962. The general description of the sites is found in Annex 1, and a scheme showing the sample compartments is given in Figure 13.

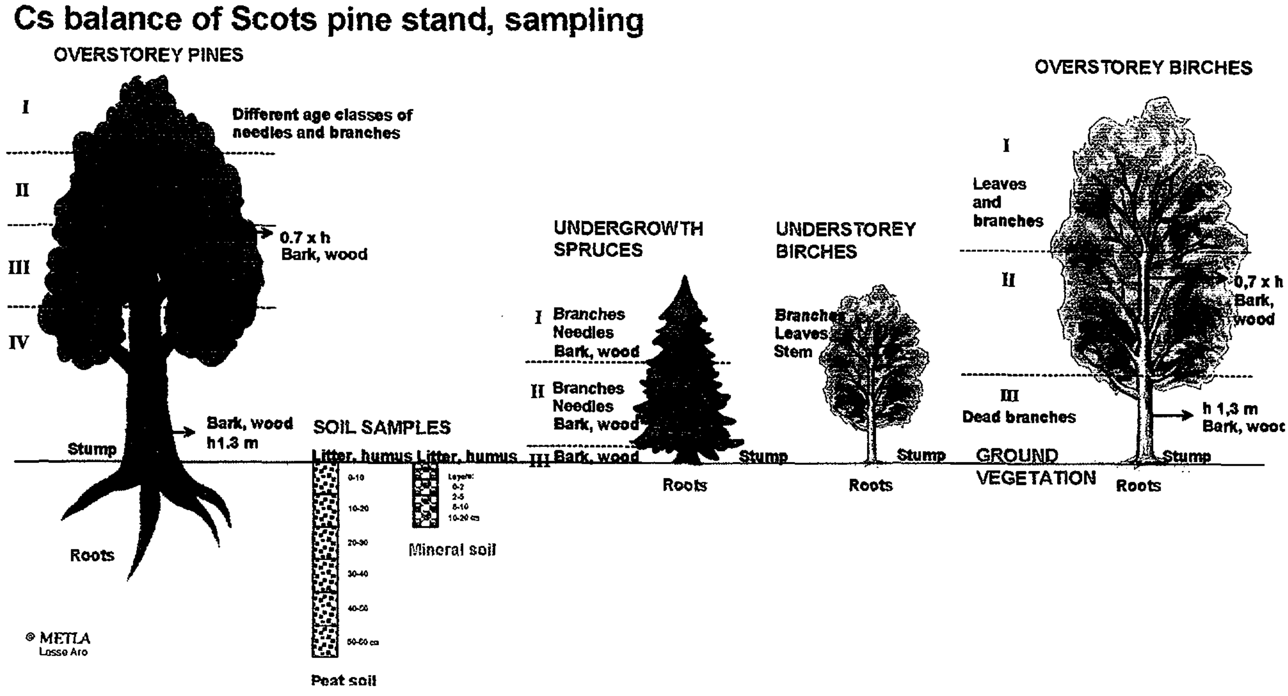

Figure 13. Sampling of vegetation and soil on mineral soil and peatland site

Sampling of soil and ground vegetation. Plant and soil samples were collected systematically from different parts of the ecosystem in both experiments (plot size varied from 800 to 1300 m^2). Volumetric (the area of the sampling device was 88.2 cm^2 in mineral site and 36 cm^2 in peat site) litter (litter+Ol), humus (Of+Oh), mineral soil and peat samples were taken from nine subpoints and combined by layers in each plot. The mineral soil profile was subdivided to 0-2 (mainly OAh+Ah), 2-5, 5-10 and 10-20 cm and the peat profile to 10 cm thick layers to the depth of 60 cm.

Ground vegetation was collected from nine squares in each plot (total area of 9 m^2) and combined by species groups: lichens, mosses, mushrooms, grasses, lingonberry, bilberry, other dwarf shrubs, tree seedlings (h < 1.3 m), dead vegetation and litter. The total area of vegetation collected for this study was 72 m^2. Species were separated and dried at 60°C one week before milling and analysing them. Vegetation species were quite typical of Vaccinium site type in control plots at the mineral soil site. The most common moss was *Pleurozium schreberi. Vaccinium vitis-idaea* and *Calluna vulgaris* were the most common shrubs in field layer. Fertilisation had changed the vegetation compared to control plots. More grasses were found, and especially *Deschampsia flexuosa* was a very common species. Instead of that lingonberry and other dwarf-shrubs were quite rare. There were also changes in bottom layer, that is the most common species of mosses were *Dicranum* and *Polytrichum. Polytrichum, Dicranum* and *Sphagnum* species were the most common mosses at peatland forest sites. *Carex* sp., *Dryopteris carthusiana, Eriophorum vaginatum, Melampyrum sylvaticum* and *Deschampsia flexuosa* were the most common grasses. Drainage had changed the composition of vegetation resembling more that of mineral forest soils. Species typical of more fertile sites dominated in fertilised plots.

Sampling and measurements of trees. Small birches (i.e.of no use in forest industry, $d_{2.5m}$ < 6 cm) and undergrowth spruces were also measured and sample trees taken from nine circles (r=1.5 m) in each plot. The crowns of the sample spruces were divided into two as long parts from which needles, branches, stemwood and stembark were separated for biomass, nutrient and radiocaesium determinations. Furthermore, bark, wood and dead branches were analysed from a stem under living crown. The small sample birches were processed as whole trees from which living and dead branches, leaves and stem including both wood and bark were separated for analyses.

Height, crown limit and stem diameter at breast height were measured from all of the overstorey trees in each of the 4 plots. Five sample trees were chosen from each plot. All trees in a plot were sorted in ascending diameter order into five groups (two groups in birches), so that the basal area of each group was equal (Kukkola 1994). One tree was then chosen randomly as sample tree from each group. The total number of sample trees was 40 for pine and 8 for downy birch. Height and length of living crown of each felled sample tree were measured. Diameters with and without bark, heartwood diameters, number of annual rings in sapwood and all annual rings were measured from discs in a laboratory. The discs were taken from stump height, breast height, lower limit of living crown, six meters height and relative heights 0.3xh and 0.7xh.

Additional stem samples were taken from breast height and 0.7xh for stemwood and bark biomass, nutrient and radiocaesium determinations. The living crown was divided into four as long parts. The fresh mass of living and dead branches in each part of crown was weighed in the field as well as the fresh mass of the whole stem. One branch with living needles (or leaves) and one dead branch from each quarter were sampled and weighed in the field. Dry mass of leaves and different age classes of needles and branches (C, C+1, C+2, C+3, rest of needles and branches) were weighed and analysed for their nutrient and radiocaesium concentrations in the laboratory.

Root and stump samples for pine and birch were taken from a tree belonging to the middle group of measured trees in each plot. Roots and stumps of spruces and small birches were collected from two different sized trees in each plot. Root samples were divided into diameter classes 0.1-1.0, 1.1-5.0 and more than 5-cm-thick roots. The finest roots (diameter less than 0.1 cm) were included into the soil samples. Bark was not separated from wood.

Nutrient and radiocaesium analyses. Particle-size distribution, clay content, pH (H_2O, $CaCl_2$) and organic matter content in soil and the concentrations of nutrients (N, P, K, Ca, Mg, S, B, Mn, Cu, Zn, Fe, Al) in soil and vegetation were measured by the methods routinely used at the Finnish Forest Research Institute (Halonen et al. 1983).

Nitrogen analyses were done using the *CHN-LECO* analyzer, boron was analysed spectrophotometrically with *azomethin-H* staining, while the rest of the elements were determined using ICP and nitric-acid-hydrogen-peroxide wet-combustion method (TJA Iris Advantage ICP-emission-spectrometer). However, total nutrients of soil (a dry-combustion method) and their proportions of extractable acidic ammonium acetate (pH 4.65) were analysed using ARL 3580 ICP-emission-spectrometer.

Radiocaesium in dried, homogenised samples or sometimes in ashed samples was determined with a low background gamma spectrometer (Klemola & Leppänen 1997).

Balance calculations. Stem volume of Scots pine stands was calculated with KPL-programme (Heinonen 1994) and divided into proportions of stemwood both in and under living crown by using spline functions. Dry weight of different needle and branch compartments in sample branches were multiplied with a coefficient which was calculated for each crown quarter and sample tree separately by dividing the fresh mass of crown quarter by the fresh mass of the sample branch. Biomass functions for needles and branches of pine were calculated for each plot and crown quarter separately with regression method in which breast diameter, height, lower limit of living crown and the crown ratio with different transformations were used as independent variables. Dry masses of different compartments in dominant birches were also determined with regression method. The total biomass of tree roots and stumps was estimated by regression functions available in forest literature (Hakkila & Mäkelä 1973, Mälkönen 1974, 1977, Paavilainen 1980, Issakainen 1988, Finér 1989). The dry mass of each compartment was calculated for one hectare (kg ha^{-1}). Radiocaesium and nutrient balances were calculated by multiplying biomass with different nutrient or ^{137}Cs concentrations.

Statistics. All analyses have been made separately for the mineral soil site and for the peatland site. Differences in ^{137}Cs and nutrient concentrations and dry matter content of different tree compartments, understorey vegetation and soil between fertilised plots and control plots were studied using *t*-test for independent samples. Also the components of the radiocaesium and nutrient budgets were tested. For the unequal variances situation, Cochran and Cox (1950) approximation of the probability level of the approximate t statistic was used. Correlation of ^{137}Cs with nutrients in vegetation and soil was studied separately for fertilised and control plots by Pearson correlation coefficient.

Results and discussion

Effect of fertilisation on dry mass accumulation

The total amount of dry mass varied from 243300 kg ha^{-1} in the control treatment to 270500 kg ha^{-1} in the fertilisation treatment on mineral soil site and from 393500 to 417800 kg ha^{-1} on the peatland site, respectively (peat layers from 20 to 60 cm excluded). The above ground parts of the stands represented about 40% of the total dry mass on the mineral soil site and about 17% on the peatland site (Figure 14).

Fertilisation had no effect on the amount of organic matter in the surface layers (0-20 cm) of mineral (about 86000 kg ha^{-1}) or peat soils (about 254000 kg ha^{-1}). Fertilisation did not increase the dry matter content in humus and litter layers significantly although clear differences were observed (from 41900 kg ha^{-1} to 47900 kg ha^{-1} on the mineral soil site and from 52600 to 63900 kg ha^{-1} on the peatland site).

NPK+NPK-fertilisation decreased significantly the total biomass of understorey vegetation on the mineral soil forest site (2600 kg ha^{-1} in control and 380 kg ha^{-1} in fertilisation treatment, p=0.0018). The result was similar to earlier investigations concerning sub-dry sites in Finland (Mäkipää 1994).

The largest reduction was observed in the dry mass of mosses: in fertilisation treatment the dry mass was only 7-19% from that of control treatment (p=0.0066). Fertilisation decreased clearly the dry mass of lingonberry and other dwarf shrubs whilst according to Mälkönen et al. (1982) N-application will increase the covering of dwarf shrubs for longer time than that of herbs.

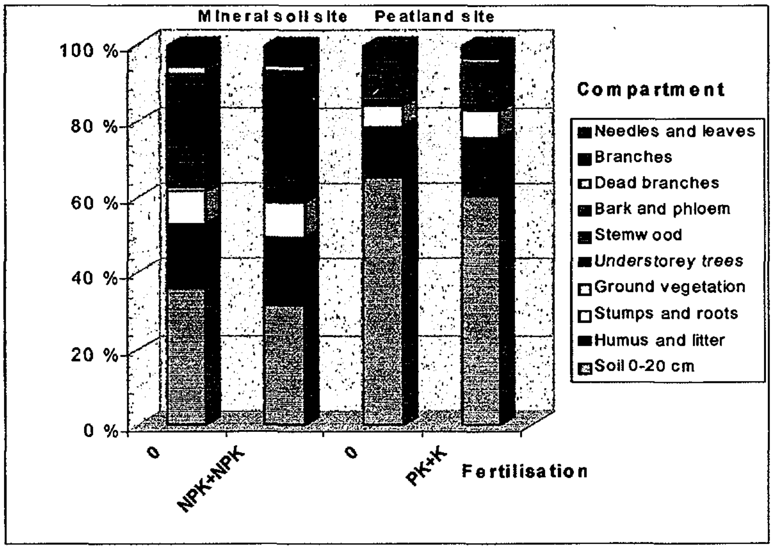

Figure 14. Relative distribution of dry matter in forests at the two experimental sites.

On the peatland site, PK+K-fertilisation increased significantly the dry mass of understorey vegetation (1670 kg ha $^{-1}$ in control and 2310 kg ha^{-1} in the fertilisation treatment, p=0.0231). This is typical of sites poor in mineral nutrients (e.g. Paavilainen 1980).

The total dry mass of undergrowth spruces varied from 500 to 11300 kg ha^{-1}. The dry masses of needles or branches varying from 200 to 3300 kg ha^{-1} were quite close to each other if compared in the same plot. The dry mass of dead branches was only some tens of kilos per hectare and that of stem wood with a bark 130-3200 kg ha^{-1}.

The dry mass of above ground parts of small birches ($d_{2.5}$ < 6 cm) was 4800 kg ha^{-1} in control and 7800 kg ha $^{-1}$ in fertilisation treatment on the peatland site. The below ground dry mass of stumps and roots varied from 300 to 2000 kg ha^{-1}. Fertilisation increased both the dry mass of stems, branches, leaves, stumps and roots but the increments in dry masses were not significant.

The effect of fertilisation on stem volume of Scots pine stands was clear in both experiments. However, the difference between the control and the fertilised plots was not significant. The total dry mass of Scots pine stand was 111000 in the control and 130000 kg ha^{-1} in the fertilisation treatment on the mineral soil forest site. About 80% of the dry mass was accumulated in the above ground part of stands. The fertilisation increased the dry mass of Scots pine bark and wood both in and under the living crown. NPK-refertilisation increased also the dry mass of needles, branches, roots and stumps. On the same site the dry mass of needles and branches varied from 16200 to 25600 kg ha $^{-1}$. The dry masses of underground compartments of the Scots pine stand varied between 19100 and 26000 kgha^{-1}.

On the peatland site, the total dry mass of the Scots pine stand was 68000 in control and 88000 kg ha^{-1} in the fertilisation treatment. Fertilisation increased the dry mass of needles significantly (p=0.0129). The effect of fertilisation on the other compartments of the pine stand was also clear but not significant.

Effect of the fertilisation on ^{137}Cs concentrations in soil

The radiocaesium in soil varied from 24 to 43 kBq/m^2. On peatland, more than 50% of the caesium was found in the raw humus layer, which often was quite thick (>5 cm). On the mineral soil, over 50% of the caesium was present in the litter and humus layers (Figures 15, 16) in all plots. On the peatland site about 10% or less of the ^{137}Cs activity had penetrated below the uppermost 10 cm of peat, whereas on the mineral soil site less than 10% had penetrated below 5 cm of mineral soil.

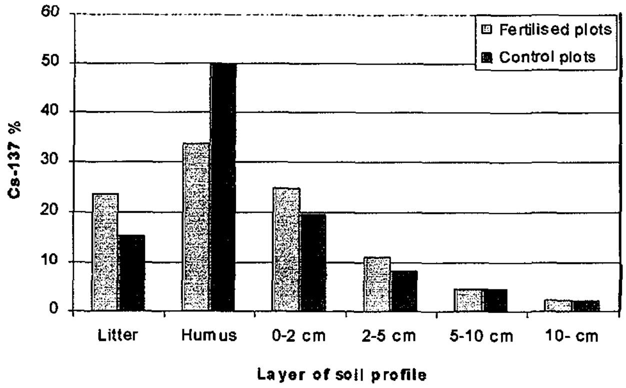

Figure 15. Percentage depth distribution of ^{137}Cs in Siikakangas podzolic soil at two fertilised plots and two control plots.

On the mineral soil site, the ^{137}Cs concentration did not differ significantly ($p>0.05$) on the fertilised plots compared to the control plots in litter, humus and in most layers of the mineral soil (0-2 cm, 5-10 cm and 10-20 cm). For one layer of mineral soil (2-5 cm), the ^{137}Cs concentration was significantly higher ($p=0.0064$) at the fertilised plots compared to the control plots.

On peatland, the potassium fertilisation decreased significantly the ^{137}Cs concentration in the raw humus layer and increased it in the peat layers. However, the effect was significant only in surface peat layers (0-10 cm and 10-20 cm) and in the middle of the profile (30-40 cm). In the deepest layers (40-60 cm) the radiocaesium concentration was not affected by the fertilisation. The total amount of ^{137}Cs in the 60-cm profile was independent of fertilisation. The small amounts of Chernobyl-related radiocaesium in the deeper layers of the soil show that the effect of fertilisation on downward migration is small, although it can be detected with detailed measurements and careful data analysis. Therefore, the removal of the Chernobyl-derived radiocaesium out of the system through vertical transport is also very slow. The effect of fertilisation on the external radiation from the soil still has to be established, but these results do indicate a small effect. Generally potassium concentrations correlated weakly with the radiocaesium activity in the studied soil layers.

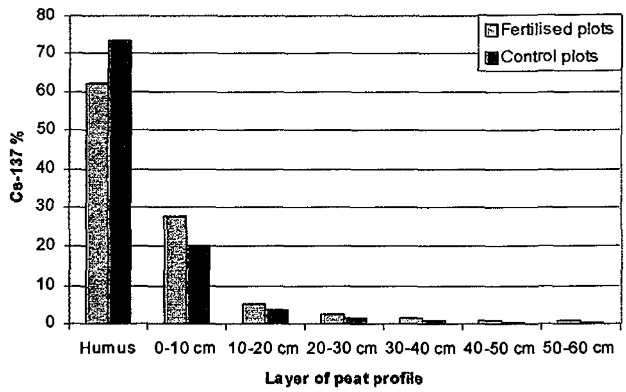

Figure 16. Percentage depth distribution of ^{137}Cs in Liesineva peatland soil at two fertilised plots and two control plots.

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Effect of the fertilisation on ^{137}Cs concentrations in vegetation
For Scots pine, the ^{137}Cs concentrations in bark, wood and current, C+1, C+2, C+3 needles as well as many samples of branches of different ages were lower in the fertilised plots compared to the control plots both on the mineral soil site and on the peatland site. For Scots pine, dead branches, dead parts of living branches, some samples of branches of different ages and for more than four years old needles (C+4), the ^{137}Cs concentrations did not differ between treatments.

On the mineral soil site, the ^{137}Cs concentration was lower in *Deschampsia flexuosa* and *Vaccinium myrtillus* on the fertilised plots compared to the control plots. For grasses, herbs, lichens, mosses and other dwarf shrubs (*Empetrum nigrum* and *Calluna vulgaris*) no statistically significant difference in the ^{137}Cs concentration between the treatments was observed. However, at the fertilised plots the concentrations were only 15-50 % of those at the control plots in these species.

On the peatland site, *Vaccinium myrtillus*, tree seedlings, lichens, mosses and other dwarf shrubs (*Empetrum nigrum* and *Calluna vulgaris*) had lower concentrations of ^{137}Cs at fertilised plots. In herbs, grasses, dead vegetation, *Dryopteris carthusiana* and *Vaccinium vitis-idaea* the concentrations at fertilised plots were 38–68 % of those at control plots, although the differences were not statistically significant.

For overstorey birches on the peatland site, the ^{137}Cs concentration was lower at the fertilised plots in living branches, bark and wood. Also in wood, leaves and branches of understorey birches on the peatland site, the ^{137}Cs concentration was lower at fertilised plots compared to control plots.

Correlations of ^{137}Cs concentrations with nutrients varied along the treatment and site type as well as for the different vegetational components.

Effect of fertilisation on radiocaesium distribution
The ^{137}Cs distributions on fertilised and control plots on the mineral soil site and on the peatland site (Figures 17, 18) show significant changes due to the treatments. The activity fraction in vegetation

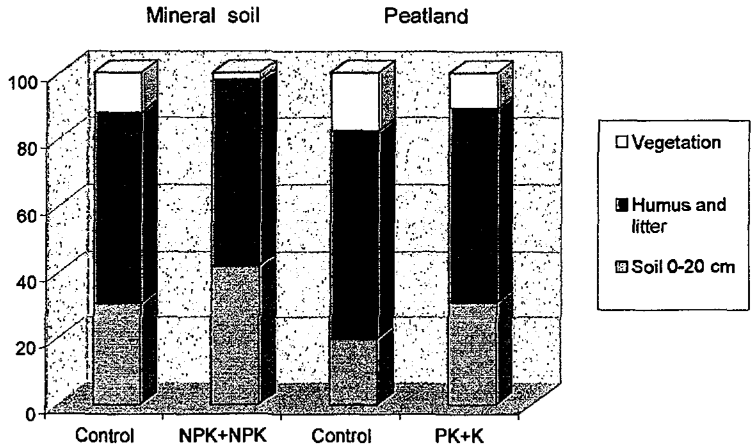

Figure 17. The relative distribution of ^{137}Cs in pine dominated forest on mineral soil and on peatland

decreased and the activity in the soil increased on fertilised plots. On the mineral soil, fertilised vegetation contained less than 20% of the ^{137}Cs activity found in vegetation on control plots. On peatland the corresponding figure was 60%. In above ground parts of vegetation the reduction in ^{137}Cs activity in vegetation was slightly more than when roots and stumps were included in vegetation. A slight increase in activity fraction due to fertilisation occurred in understorey trees on the mineral soil (16%), where the simultaneous biomass increase due to the treatment was about tenfold. On peatland a distinct decrease of one third of the activity content in understorey trees was found. The corresponding

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change in biomass was less than 30%. The contribution of overstorey trees to the radiocaesium budget changed as that of the whole vegetation compartment on mineral soil, and was somewhat lower on peatland.

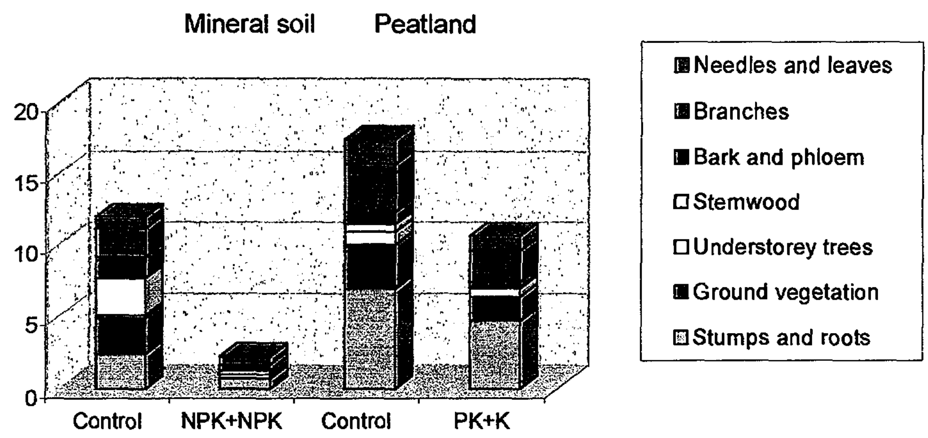

Figure 18. Distribution of ^{137}Cs in vegetation (in per cent of the ^{137}Cs budget, Cf. Figure 17) of a pine dominated forest on mineral soil and on peatland.

Conclusions

Radiocaesium uptake in vegetation can be reduced by site specific nutrient fertilisation on nutrient poor sites. The more efficient reduction of ^{137}Cs content of vegetation on fertilised mineral soil compared to peatland may reflect the excess of nutrients. On peatland the dose of K-fertiliser was less abundant in relation to the prevailing nutrient status of the stand.

Dry mass accumulation can be increased especially in stemwood by fertilisation. The results on mineral soil site show clearly, that the reduction in Cs uptake cannot be explained with growth response alone, but that is also related to root uptake.

The growth response of trees to the fertilisation is normally observed in the growing season following next to that of the treatment. As to the timing of treatments in relation to Chernobyl fallout, the observed reduction in uptake of ^{137}Cs on the mineral soil site may represent near to the maximum reduction that can be achieved with the doses used. However, the effect of fertilisation on the dry mass accumulation of vegetation will last only some years on sub-dry mineral soil sites and 15-20 years on peatland sites poor in mineral nutrients.

The results indicate the benefits of fertilisation for restoration of contaminated forests in a severe fallout situation. The availability of timber to forest industry can be essentially increased with long term treatments of forests. Through multiple use of forests, pickers of wild berries and mushrooms, and hunters receive less radiocaesium through foodstuffs from fertilised forests than otherwise.

The results of the study motivate further research on the applicability of forest management methods as countermeasures in a fallout situation. Such methods offer possibilities to follow the principles of sustainable forestry, as the treatments do not change ecosystem in a radical way.

HERBIVORE DYNAMICS

Introduction

The semi-natural ecosystems of northern Europe have abundant wild large herbivores, which are a significant food source for man. After the Chernobyl accident, large variations in radiocaesium levels have been found in the tissues of free-ranging herbivores culled from the same areas. Although in herbivores there is relatively little scope for variation in the degree of absorption and metabolism of pollutant radionuclides, our ability to predict the intake of radionuclides by herbivores living in heterogeneous environments is poor. It is expected that the radionuclide intake by herbivores would depend critically on the composition of the diet in terms of plant parts and plant species and the overall levels of environmental contamination as it varies over a range of spatial scales. The need to quantify diet composition is especially acute where herbivores ingest fungal fruiting bodies, which can contain very high concentrations of radiocaesium and other pollutants. These are most abundant and may be ingested in late summer and autumn, prior to the culling seasons of large ruminants, whose radionuclide contamination levels may hence be strongly affected. This part of the LANDSCAPE project developed and applied methods to (i) measure the intake and diet composition of moose in the Boreal forest, and (ii) assess the use of space by moose, and considers these in relation to likely patterns of deposition and distribution of radionuclides in vegetation. This facilitates examination of the likely contributions of these factors to variability in radionuclide intake and contamination level of moose, against the wider background of variability due to pattern of deposition.

Diet and intake of wild ruminant herbivores in forest ecosystems

Intake and diet composition – technique development and testing

A method has been previously developed to quantify diet composition and intake by domestic ruminants grazing temperate grassland (Mayes et al 1986, Dove & Mayes, 1991). The non-invasive technique relies on comparing faecal concentrations of natural plant hydrocarbons with the similar compounds, usually long-chain n-alkanes, which are orally dosed at known rates. The method has hitherto not been applied to wild ruminants, because of the requirement to administer a daily oral dose, nor has it been applied to ruminants foraging on diets of woody browse or mixtures of woody and non-woody plants, typical of wild Boreal herbivores. The availability of intra-ruminal, controlled release devices (CRDs), which release n-alkane marker over a period of several weeks, provides the opportunity to apply the method to wild herbivores. Three experiments were conducted to test and validate the method using CRDs under conditions relevant to foraging by Boreal forest ruminants. First, the appropriateness of the method of using CRDs for estimating intake and diet composition, was tested using captive red deer (*Cervus elaphus*) eating a mixed diet of woody browse and non-browse components. Second, the use of CRDs to deliver alkane markers for the measurement of intake and diet composition was also tested in captive moose (*Alces alces*) eating the two main forest species, *Betula pubescens* and *Pinus sylvestris.* This second experiment tested the method in the exact same animal and woody plant species in which it was applied in the field situation (see below). The third experiment, using housed goats, evaluated the potential of the common fungal sterol, ergosterol, as a faecal marker for dietary fungi.

Experiment 1: A validation of the technique of measuring intake of birch in summer by red deer, using natural plant markers and intra-ruminal controlled release devices.

Material and methods

The overall approach taken was to offer birch trees (*Betula pendula*) to each of ten adult female red deer (hinds), and measure the intake of birch using CRD-delivered alkane markers. The intake measured by this method was then tested against the mass eaten from trees, and hence the total daily intake of birch, which was estimated by the frequency distribution of bite diameters, the mass associated with each of which was calibrated with a sample of dissected trees.

This experiment was conducted in July and August 1998. Each hind was dosed with a CRD delivering 50mg/day of each of C_{32} and C_{36} n-alkanes. After insertion of the CRD, a one-week period elapsed to allow equilibration of the passage of alkanes through the gut. During the first 3 days of this period, the hinds had free access to heather (*Calluna vulgaris*) plots and some supplementary feed, given as an unmeasured but approximately constant daily amount of grass pellets. For the final 4 days of this equilibration week, hinds were penned individually in a 3 x 3 m pen. On each morning for these four days (and subsequently), each hind was permitted a three hour period of grazing heather in one of six available 0.25ha enclosures. They then received a supplement of approximately 0.8 x maintenance requirements of dried grass pellets in the afternoon. During this four-day preliminary feeding period their feeding regimen was identical to that during the experiments, with the exception of the unavailability of birch trees. For the final 2 days of the preliminary feeding period, and the following morning (ie the day of the first experiment), faecal samples were collected from the floor of the pen, or as the hinds were transferred to their grazing areas. These samples formed a set of zero birch faecal samples and provided a measure of the alkane concentration of the faeces on the baseline diet and feeding regimen, in the absence of any birch.

On two consecutive days, each hind was allowed three hours of grazing heather plus birch, before being returned to their individual pen when they received the same level of supplementary grass pellets as previously. For the three hour grazing period, a single hind was given access for one hour each to three 0.25ha enclosures each containing an array of 25 birch trees. The trees had been grown in pots to a height of 1.2-2.2m. From dissection of a calibration sample of 24 birch trees, relationships were formed between the twig diameter bitten and the mass and n-alkane concentration of leaf and twig associated with a single bite of this diameter. For each of the 75 trees offered to hinds on both days, the number of bites of each bite diameter was summed and the mass eaten was estimated. During the two experimental days, faecal samples were collected whenever possible and their time of collection and of production, if known, was recorded. For a period of five further days following the two experimental days, the individual penning, feeding and faecal collection continued as before, except birch trees were not available during the grazing period, and in this period the hind was in the company of others whilst grazing.

All samples were stored at -20°C and either oven dried at not more than 60 °C (vegetation samples), or freeze dried (faecal samples), prior to heptane extraction and analysis by gas chromatography.

Calculation of the intake of birch by red deer was carried out by using a modification of the method of Duncan et al (1999). The n-alkane C_{25} was chosen as the birch dietary marker because of its high concentrations in birch relative to its levels in other dietary components. The faecal concentrations of C_{25} alkane were plotted against time for a period of 5 days from the start of the birch feeding period of the trial. This period was sufficient for the concentrations of birch alkanes in the faeces to return to baseline levels. The area under this curve was then calculated and compared with the area beneath the line described by the plot of concentration against time, for the dosed alkane, over the same time period. C_{36} alkane was chosen as the dosed marker, since its concentrations in the diet were negligible.

Results

Both the mass associated with different segments and the concentrations of alkanes varied strongly with the diameter of the segment (Table 14).

Table 14. Variation in leaf mass, twig mass, and leaf and twig concentration of C_{25} n-alkane, associated with segments of different twig diameters of silver birch, Betula pendula.

	Twig segment					
	0-1mm	1-2mm	2-3mm	3-4mm	4-5mm	>5mm
Leaf mass (gDM)	0.027	0.180	0.788	1.668	2.884	4.519
Twig mass (gDM)	0.011	0.108	0.648	1.677	3.292	5.935
Total mass (gDM)	0.038	0.288	1.436	2.345	6.176	10.454
Leaf C_{25} conc. (mg/kgDM)	617.8	472.1	436.2	436.7	334.3	464.6
Twig C_{25} conc. (mg/kgDM)	1038.7	495.2	335.6	165.7	131.0	74.0

This was particularly true of the twig fraction. The alkane concentration of the twig fraction, decreased strongly and consistently with increasing twig diameter. The alkanes on the twigs are probably distributed on the exterior and hence this effect results from the geometry of the twigs and the decreasing surface area in relation to volume or mass at the larger diameters. Although the mass of leaves associated with different twig segments increased with the diameter of the segment, there was no clear relationship between the diameter of the segment and the leaf concentration of C_{25} alkane. The net effect was that bites taken by red deer at different twig diameters resulted in an overall negative relationship for alkane concentration in all tissue, with the diameter at which the twig was bitten (Figure 19). We would hence expect that our estimation of diet composition and intake using the method would be sensitive to variation in the diameter of twigs bitten by the animal. In further calculations, the modal class of bite diameters for each individual hind was applied as a representative average. For all hinds the modal class of bite diameters was 2mm which represented between 52.6% and 64.6% of total bites for all 10 hinds.

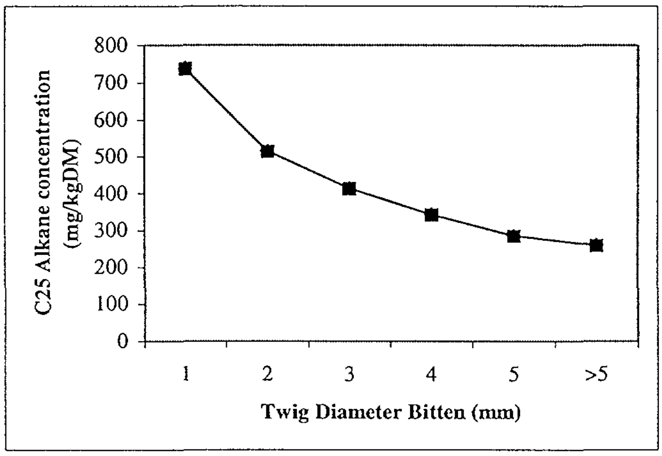

Figure 19. The concentration of C_{25} alkane in overall leaf and twig tissue, in relation to the diameter of twig bitten by red deer. Calculated by summation, from analyses of leaf and twig components associated with different twig diameter segments 0-1mm, 1-2mm etc.

The daily intake of birch calculated using the alkane method corresponded well with the intake calculated from the measurement of clipped bites, for all but one of the ten experimental hinds (Figure 20).

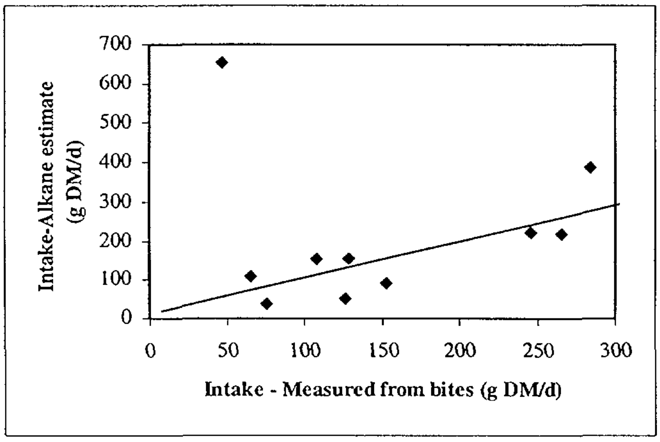

Figure 20. The daily intake of birch estimated from the alkane technique, in relation to the intake measured from the bites taken by each of 10 red deer hinds. The line shown has a gradient of 1.0, and it that which would be obtained if the measurements from the two methods corresponded exactly.

The deer represented by the outlying point had either regurgitated its CRD, or the CRD had failed to function. No dosed alkane was detected in the faeces of this hind and the CRD was never found.

For the nine hinds for which the CRD functioned well, the regression line fitted through the origin was:

Intake (alkane technique) = 0.9995 x Intake (bite measurements).

The slope of this line was very close to 1, indicating good agreement between the methods (Figure 19). The fitted line differed significantly from zero ($F = 17.49$, df 1,8, $p < 0.01$), and explained 69% of the variation ($r^2 = 0.686$).

Experiment 2. Validation of marker techniques using intraruminal controlled-release devices, for determining diet composition and intake in captive moose given winter browse diets.

Materials and Methods

At Grimsö research station in central-southern Sweden an experiment was conducted in which a single CRD containing C_{32} n-alkane was inserted into the rumen of two captive moose cows. It was intended to study an additional moose, but this bull was behaviourally unsuited to the experimentation.

The experiment was conducted during February 1998 during snow cover. Prior to the experiment the moose occupied a large enclosure of about 15 ha, and were accustomed to foraging on both birch and Scots pine. Large volumes of fresh browse, in the form of whole trees of Scots pine and downy birch (*Betula pubescens*) were brought to the enclosure a few times each week. The CRD was introduced into each moose on day 0 of the study and a sample of faeces was collected from each moose on days 0,2,4,6,8 and 10. Faecal samples were collected by searching the enclosure and removing all faeces visible on the snow surface. On day 10 the moose were placed in a smaller divided pen of about 0.2 ha and for a period of 7 days, during which all browse offered and refused was weighed. During this period, dummy feed samples were weighed to permit adjustment for non-browsing weight loss, and all faeces produced were collected, weighed and sub-sampled for oven drying and marker analysis. The amounts offered to the two individuals were adjusted so that different browse species predominated in the diets of the two moose. The amounts eaten were thus measured gravimetrically. The actual diet composition of the moose, namely the percentages of birch and Scots pine in the diet, measured by the weighing method were compared with the values calculated by the marker technique. The calculated diet composition values were used to estimate total diet marker concentrations, enabling intake estimates to be calculated on the basis of dietary markers, using the method of Mayes et al (1986).

The active delivery period of the CRD was checked against that given by the manufacturers (which was for use of the device in cattle), by continuing to collected faecal samples after the intensive feeding period, at approximately two-day intervals until day 36 after dosing.

Results

An initial analysis of the plant material offered in this experiment showed that Scots pine contained very low concentrations of n-alkanes, which hence, as a group, were unlikely to be suitable markers for calculation of composition and intake of diets which included this species. The winter diets of moose were especially likely to include these species. The leaf material of birch species was particularly rich in fairly short-chain length n-alkanes, around 25 carbon atoms long. The unsuitability of using alkanes as sole markers in the experiment was confirmed by the calculation of diet composition on the basis of alkanes, of the two Grimsö moose, which was showed poor agreement with the measured proportions of the two components (Table 15). However, further analysis of the Scots pine and other plants available as winter forage for moose and other large herbivores, demonstrated that long-chain fatty alcohols were present in, and varied between, the available plant species. In birch (*Betula pubescens*), these alcohols were mainly primary alcohols, with carbon chain lengths, in the range C_{22}-C_{28}. In the case of Scots pine there was a high concentration of C_{29} alcohol, hydroxylated at the 10-carbon position; the concentration of 1869 mg/kgDM contrasted starkly with its concentration in *Betula pubescens*, which contained only 7 mg/kgDM. This secondary alcohol, is also present in juniper (*Juniperis communis*), which can also be a component of the winter diet of moose. However, juniper also contains high levels of C_{33} and C_{35} n-alkanes allowing its presence in the diet to be easily established. An assessment of the potential of long-chain fatty alcohols as markers in diets containing conifers was made. The agreement between the diet composition measured using the alcohols, and that measured gravimetrically was very good (Table 15). The suitability of the long-chain fatty alcohols as dietary markers in situations where conifers are included in the diet, means that estimation of total intake of such diets can be made. This can be achieved by using the faecal concentrations of these alcohols and those of dosed alkanes as the respective reference dietary and dosed markers, when the recoveries of both are known. However, the estimates of intake using the marker techniques were not consistently in agreement with the actual measured values (Table 6). This discrepancy is thought to have been introduced into the calculation of intake, by errors in the approximation of the faecal recovery of the alcohols, the values for which were those measured in goats (Ashton 1998). This error does not exist in the calculation of diet composition, which only requires an estimate of recovery of the alcohols relative to each other.

Table 15. The dietary proportions of birch and Scots pine, and daily intake, both measured and calculated using the marker method based on n-alkanes and alcohols, for two captive moose in February 1998.

Moose	**Dietary composition (% DM)**			**Intake (kgDM /d)**	
	Actual	Estimated (Alkanes)	Estimated (Alcohols)	Actual	Estimated (Alcohols)
Algros	*Birch* 63 *Pine* 37	*Birch* 64 *Pine* 36	*Birch* 64 *Pine* 36	3.7	2.74
Erika	*Birch* 17 *Pine* 83	*Birch* 3 *Pine* 97	*Birch* 15 *Pine* 85	4.5	4.16

The period of effective operation of the CRDs was from days 5-20 in one of the moose and days 8-26 after dosing, in the other. In the moose with the narrower window of effective operation of the CRD, the sudden cessation of the presence of the dosed alkane in the faeces, without subsequent detection, suggests that the device may have been regurgitated.

Experiment 3. The potential of using ergosterol as a faecal marker for determining the contribution that fungal fruiting bodies make to the diet of moose.

Material and Methods

Following chemical analysis of fruiting bodies from about 20 species of fungi found in forest and pastoral environments, the sterol, ergosterol, was considered as a promising candidate as a marker to indicate the presence of fungi in the diet. Although such a marker would not allow discrimination of different fungal species in the diet, some of the fungi investigated contained additional sterols, which suggested that a degree of discrimination may be possible.

For a compound such as ergosterol to be used as a faecal marker, its recovery in faeces must be determined. Goats were used in this experiment, as they are the most readily available domestic ruminant with characteristics of browsing animals; they are hence a suitable model study system for wild browsing ruminant such as moose. An experiment was initiated in 4 goats, housed in metabolism cages, to determine the faecal recovery of ergosterol using commercial mushrooms, which contained 5300mg/kg dry-matter ergosterol, as the dietary source. Unfortunately, this experiment had to be abandoned since the goats consumed none of the offered mushrooms.

In order to determine the recovery of ergosterol in the faeces, four goats were housed in metabolism cages and dosed twice daily for 12 days, with gelatin capsules containing 50 mg ergosterol (absorbed into compressed paper tissues). The goats received mixed diets consisting of perennial ryegrass (*Lolium perenne*), heather (*Calluna vulgaris*), bilberry (*Vaccinium myrtillus*), goat willow (*Salix caprea*), woodrush (*Luzula sylvatica*) and Scots pine (*Pinus sylvestris*). A further 8 goats in metabolism cages received similar diets but no ergosterol.

Results

Ergosterol was present in all the investigated fungal species at relatively high concentrations (1000-7000 mg/kg dry-matter) and virtually absent in browse and herbage species. No ergosterol was detected in the faeces of the goats which had received the oral doses of ergosterol, which suggested that it is not a suitable marker for establishing evidence of ingestion of fungal fruiting bodies in ruminants. However, it is known that dietary sterols, such as β-sitoserol and cholesterol can be hydrogenated in the digestive tract. Further studies are required to ascertain if hydrogenation products of ergosterol are found in the faeces of animals receiving ergosterol or fungal fruiting bodies. If so, then ergosterol might still have potential as a marker representing fungal ingestion.

Conclusions

The application of CRDs to wild ruminants provides considerable opportunity, where the species composition of the diet is relatively simple, to measure intake of total dry matter of food and hence intake of any of its chemical constituents, including radionuclides. The method has the advantage over previously used marker methods, that the animals need not be captured more than once for the dosing of the markers.

Previous results have suggested that the behaviour of the dosed hydrocarbons in the digestive tract, is similar for all species of ruminant, and is similar to that of natural plant wax constituents (Mayes et al 1995). The expectation that the method would apply equally well to moose as to domestic ruminants, was borne out, but further work is needed on the quantification of exact faecal recoveries of all the markers used, in order to obtain accurate measurements of food intake.

The validation experiments conducted with red deer and captive moose suggest that the methodology is equally applicable to ruminants ingesting diets of woody browse plants, in both summer and winter seasons. There was good quantitative agreement between the results of the hydrocarbon marker method, and measurements of diet composition by weighing, in the case of moose. There was also good agreement between the hydrocarbon marker method of intake estimation with detailed calibration of mass intake via browsing damage, in the case of red deer.

N-alkanes are suitable dietary markers for birch in leaf, although the concentrations in twig material are low, they are sufficient to be used as markers in winter, when leaves are absent. Scots pine did not contain significant amounts of n-alkanes; hence the use of long-chain fatty alcohols is suggested when this is a likely dietary constituent.

Because of the negligible recovery of ergosterol itself in the faeces of goats, it is unlikely that it can be used as a suitable marker of dietary fungi in ruminants. The ergosterol is presumably being absorbed or transformed biochemically, in the rumen or elsewhere in the digestive tract. The possibility of using metabolites of ergosterol in faeces as an alternative qualitative marker for dietary fungi in ruminants, requires further investigation

Estimation of intake and diet composition in wild moose

The method of plant hydrocarbon markers was applied to wild moose in northern Sweden. The field facilities, moose tracking by radio telemetry and GPS and dosed the moose with CRDs are described in the section "Spatial and temporal scales of herbivores". The objective was to estimate the intake and diet composition of moose in two seasons, winter and summer. The numbers of animals dosed, fitted with GPS collars and from which sufficient faecal samples were collected to estimate diet composition, are given in Table 16. In effect, many faecal samples were collected in the field from moose other that those which had been dosed with the CRDs, as, for example, moose were usually aggregated into groups of about 4 or 5 individuals in winter. The whole group was then back-tracked and samples collected from all group members, the faeces of the dosed moose being indistinguishable at this stage. In summer an area occupied by the CRD-dosed moose, in the previous day, was identified, and samples of the fresh faeces from within a large 50 x 50m area were collected. All samples collected were given a preliminary screening extraction and GC analysis, for the dosed the alkane (C_{32}), and these samples were analysed further in more detail following identification on this basis. The dosed moose occupied different areas, hence in this study there was no confounding of samples from different individuals.

Table 16. Numbers of moose dosed with CRDs, fitted with GPS collars and faecal samples successfully obtained, within the period of functioning of the CRDs, in the Robertsfors area of northern Sweden.

Season/Year	No Dosed with CRDs	No GPS collars	Successful faecal collection
Winter 1997	3 cows	4 cows	2 cows
Winter 1998	4 bulls	5 bulls, 2 cows	3 bulls
Summer 1997	-	-	-
Summer 1998	1 bull	3 bulls	1 bull

Location of the alkane-dosed moose was carried out using the radio telemetry facility of the GPS collars in the winter of 1997 (Feb-April), whereas in winter and summer 1998, the GPS system was applied for the location of dosed moose for collection of faeces.

Samples of the vegetation available to the moose were collected in all three seasons in order to provide information on the range of hydrocarbons and long-chain fatty alcohols that may be ingested by moose. In both summer and winter the species sampled and anlaysed for potential markers and occurrence in the faeces of moose, were birch (Betula pubescens and B. pendula), Rowan (Sorbus aucuparia), Willow (*Salix caprea*), Aspen (*Populus tremula*), Scots pine (*Pinus sylvestris*), Norway spruce (*Picea abies*), Juniper, and the dwarf shrubs *Ledum groenlandicum, Vaccinium vitis idaea, Vaccinium myrtillus, Empetrum nigrum and Calluna vulgaris.* The availability of the dwarf shrubs was restricted by snow cover in winter. Horsetails *(Equisetum sylvaticum)*, and the herb *Chamaenerion angustifolium,* were also collected during the summer.

Results - Winter

In winter, estimates of diet composition and intake were obtained for two cows in 1997 and three bulls in 1998 (Table 17). High levels of C_{29} alcohol, combined with low levels of dietary n-alkanes in the faeces, suggested little birch or juniper in the diet of the dosed moose. The winter diet of all the dosed moose consisted predominantly of Scots pine, particularly for the two cows in 1997 (Table 17). Examination of the faecal content of dosed alkanes showed that in 1997, the CRD given to cow number 515 had ceased to function by 12 days after initial dosing, possibly due to it being regurgitated, and the CRD of cow number 810 ceased to function between 23 and 28 days after initial dosing. Hence in 1998, no faecal samples were collected later than 23 days after insertion of the CRD.

Table 17. Results of the diet composition and intake of alkane-dosed moose in winter 1997 and winter 1998.

Year/ Moose	Sample collection date	Number of samples	Diet composition (%DM)			Daily intake (kgDM)	
			Birch (mean)	Pine (mean)	SD	mean	SD
1997							
04 (515)	10: 3: 97	4	0	100	0.0	3.0	0.18
	13: 3: 97	8	0	100	0.0	3.2	0.09
95 (810)	18: 3: 97	1	0	100	-	3.4	-
	21: 3: 97	7	0	100	0.0	3.6	0.13
	25: 3: 97	10	0	100	0.0	4.2	0.74
	28: 3: 97	6	0	100	0.0	5.1	1.18
1998							
BD	24: 2: 98	13	13	87	2.8	3.5	0.60
	6: 3: 98	6	5	95	1.6	4.1	0.34
	11: 3: 98	10	7	93	1.9	4.2	0.30
5B	27: 2: 98	10	0	100	0.8	5.2	0.54
	28: 2: 98	1	2	98	-	4.8	-
	10: 3: 98	5	0	100	0.1	6.6	0.87
CO	24: 2: 98	9	20	80	2.8	2.3	0.33
	2: 3: 98	13	19	81	2.1	3.3	0.20
	3: 3: 98	2	17	83	4.5	3.4	0.53

Results – Summer

A total of 34 fresh faecal samples were collected from within the home range areas of known moose at Robertfors, Umeå, Sweden between June 22 and 5 October 1998. Only three of these samples were known to be from the alkane-dosed moose bull. The summary of the results presented, consists of the mean dietary proportions calculated from samples collected in each of the months of June, August and September. The possibility that the samples may be derived from the same animals precludes full statistical analysis as the data cannot be considered to be independent. The September samples (n=18) included four that were collected between 1-5 October 1998.

A preliminary inspection of the faecal samples showed that freshly produced faeces were rapidly colonised by invertebrates including dung beetles. Samples chosen for analysis excluded visible invertebrates. Faecal samples were analysed for both n-alkanes and long-chain fatty-alcohols, and qualitative analysis of these showed absence of Scots pine or Juniper from the summer diet. In very few of the samples, the presence of horsetails (*Equisitum sylvaticum*) was identified, as was an uncharacterised plant species, probably a grass. Both formed an extremely low proportion of the average diet, and along with *Ledum palustre* and *Empetrum nigrum* which were absent from the faecal samples, were excluded from the analysis. The quantitative analysis of diet composition was based on analysis on n-alkanes only, from which we could discern the main species in the diet. The only exceptions to this, were the poor discrimination between *Salix caprea* and aspen (*Populus tremula*), and between *Betula pubescens* and *Betula pendula.* In addition to these seven species mentioned, samples of Scots pine, Norway spruce, juniper, Due to the similarity in alkane profiles between them, which is expected from their taxonomic relatedness, both pairs were combined for presentation of the mean diet composition. This combination is justified by the ecological and radioecological similarity, between the two members of each pair.

Table 18. The mean diet composition of moose at Robertsfors in three months, determined by n-alkane analysis of fresh faeces samples. Results are expressed as % of dry matter in the diet. Standard deviation is given in parentheses and n is the number of samples analysed.

	Month, n		
Species	**June,** 9	**August,** 6	**September,** 18
Betula pendula + B. pubescens	36.0 (13.9)	6.7 (8.6)	0.2 (0.2)
Sorbus aucuparia	0.0 (-)	26.9 (17.2)	43.7 (13.9)
Salix spp + Populus tremula	30.9 (14.0)	39.1 (13.0)	33.8 (19.7)
Calluna vulgaris	0.0 (-)	0.6 (0.5)	1.9 (0.5)
Vaccinium vitis idaea	0.6 (1.1)	1.6 (1.6)	3.1 (3.8)
Vaccinium myrtillus	5.7 (6.2)	17.0 (10.2)	13.0 (11.7)
Chamaenerion angustifolium	26.6 (20.7)	8.1 (19.6)	4.3 (15.6)

The results (Table 18) show that the diet of moose in summer consists mainly of broad-leaved trees, the leaves of which are stripped from the twigs. In early summer birch, willow and aspen are the major components, with the birch being replaced by rowan in August and September. The other large contributors to the diet are fireweed or rose-bay willow herb (*Chamaenerion angustifolium, Syn. Epilobium angustifolium*) which decreases from June to September, and blaeberry (*Vaccinium myrtillus*). None of the plant species identified as the main components of the summer diet, is exceptional in its propensity to contain high levels of radionuclides; hence dietary switches between them are unlikely to result in significant variation in radionuclide intake. The uptake of radiocaesium by heather (*Calluna vulgaris*) is known to be higher than by most other plant species in the Boreal forest. Thus, ingestion of heather by moose in summer or early autumn, prior to or during the hunting season could result in significant contamination of moose meat. However, the proportion of the diet formed by *Calluna vulgaris* is consistently low in all months.

Conlusions

On the basis of the results presented for our intensively studied moose at Robertsfors, Northern Sweden, the winter diet of both moose bulls and cows is dominated by Scots pine.

In contrast the summer diet was dominated by broadleaved tree species, primarily birch (*Betula spp.*). Some samples showed evidence that heather (*Calluna vulgaris*) was present in the diet, but the levels were such that it was unlikely to have significant radioecological consequences at the population level.

Overall, it is unlikely that variation in diet composition of moose in late summer and autumn, prior to and during the hunting season, is a major determinant of variation in radionuclide uptake. The exception to this remains the ingestion of fungal fruiting bodies, which can lead to significant uptake of radionuclides, but the ingestion of which is unpredictable due to their ephemeral temporal and spatial distribution.

It seems likely that the greatest variability in radionuclide uptake by moose is brought about by spatial variation in deposition, in conjunction with normal movements of the animals (see next section), rather than being due to variations in diet selection or intake *per se*.

Spatial and temporal scales of herbivores

Materials and methods

The work was done in two areas in Sweden, Robertsfors parish, Västerbotten county, and in Abisko, Kiruna parish, Norrbotten county.

The Robertsfors area consisted of a mixed boreal forest with bogs and wetlands and with smaller areas of farmlands. The soil is mainly of podzolic type and the vegetation is dominated by mixed stands of *Picea abies* and *Pinus sylvestris* with understorey vegetation ranging from dry lichen heath to more moist *Vaccinium myrtillus* covered and moss dominated habitats. The population density of moose is about 7-8 individuals per 1000 hectare and hunting occurs annually from September to December. Ground deposition of ^{137}Cs ranged from 2- 30 kBq/m^2.

Abisko is in the Swedish mountains (up to 2000 m) with a subarctic vegetation dominated by mountain birch (*Betula bubescens vs. tortuousa*) intermixed with shrubs like *B. nana* and *Salix* spp. The forest is distributed up to 600 m above sea level and above the tree line a subarctic and arctic ground vegetation predominate with *Empetrum hermafroditum* and *Vaccinium myrtillus*. The density of the moose population is considerably higher than in the Robertfors area and varies seasonally. Exclusive hunting rights belong to the Sami people in the area and only few animals are shot each year. The ground deposition is below 2 kBq/m^2 and is mainly from the atmospheric nuclear weapons tests in the 1960's.

Sampling of vegetation and animals. For the measurement of local vegetation availability and usage by moose, three transects (20 m long and 2 m wide) were established for each moose. In each of these transects the number and species of trees and shrubs, their stem and crown diameter was recorded. The diameter of twigs clipped by moose on different tree species was measured in winter. In total for all moose there were 66 sampling locations. Additional data processing involved the establishment of a vegetation map of the moose study area at Robertsfors and at Abisko. In September 1997 and 1998, local hunters collected muscle samples in the Robertsfors area from sixty freshly killed moose each year.

Immobilisation of animals. The project started in January 1997 with an experimental test of various types of immobilisation drugs for moose to be used to tranquillise wild animals. This work was performed on captive moose at the Grimsö Wildlife Research Station under supervision of a veterinarian in January-February 1997. In this experiment two animals were fed a mixed diet of pine and birch. The commonly used Immobilon® caused tetanus of oesophagus that prevented insertion of controlled release devices (CRD) into the rumen. A new drug, Pentanyl proved to be satisfactory for our purpose and for safe administration of CRDs into reticulo-rumen via the oesophagus.

The moose were located from helicopter and shot with an anaesthetic dart with 2-4 ml Pentanyl (see below) dependent on the size of the animal. The animals were usually tranquillised within 5 minutes from the hit of the dart. The animals were weighed and body measurements taken as well as three blood samples from each animal. The handling required a time about 20-30 minutes. Vital signs were monitored during these procedures.[7]

CRDs and GPS collars. The immobilization, the CRD insertion and the attachment of GPS-collars on free-living moose were carried out in February 1997, 1998 and in June 1998. In total, CRDs were inserted into eight animals and 20 animals were equipped with GPS-collars during the project (model GPS-1000, Lotek Inc. Canada). All collars that were on moose for a period up to a year have been successfully retrieved. Several of the collars were in an "emergency" mode indicating battery failure and some needed technical service at the manufacturer in Canada in order to obtain the stored data. A few collars were damaged by water and data could not be retrieved. The actual operating time was about 50% of the expected battery lifetime.

Calculation of home ranges. A minimum of 30 fixes with a similar time between fixes were used to determine the home range using the utilisation distribution (UD) produced by the kernel method with least squares validation. The UD was mapped at three probability levels, 30%, 70% and 95%, representing levels of utilisation from a core area to an estimate of the total home range. Calculations were made using the Arcview (ERSI Inc., Redlands, CA) extension written by Hooge and Eichenlaub (1998).

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[7] The Ethics Committee at the University of Umeå approved all parts of the handling of animals used in this project (ref. 1997-02-28).

To eliminate poor fixes due to satellite geometry etc., a series of quality control checks was imposed on the data. This system was developed after considering the results of trials from the data taken from a stationary collar at a known point. This scheme filtered the data using DOP (Dilution of Precision – an estimate of the accuracy of the fix based on the geometry of the satellites used), the Fix Status (whether it was a 2D location - minimum of 3 satellites, a 3D location - minimum of 4 satellites) and whether it was possible to differentially correct the location.

Collection schedules. Data collection schedules were performed to accommodate the collection of faecal samples in 1997 and 1998:

In Robertsfors 1997 - Before mid-March a fix was taken every 6 hour, during collection period (mid-March to mid-April) fixes were taken every 20 minutes during 2-3 days a week, other times fixes every 6 hours. After the collection period, six days with fixes every 6 hours and one day (Wednesday) with one fix every hour.

In Robertsfors 1998 - Before collection period (late February to mid March) fixes every 6 hours, during the collection period every hour and after collection period as above but one day a week fixes at every 30 minutes.

In Abisko 1998 - From capture of moose (mid-February) to early March fixes every 6 hours. From early March six days at every 6 hours and one day a week at 30 minutes. From September until March 1999 fixes were taken every 6 hours.

Results

Moose positions were collected during 1997 to 1999. In spite of technical problems, 12996 positions (Table 19) have so far been successfully downloaded. Data from the collars that were retrieved in February 1999, but not yet processed, are expected to add approximately 6000 positions.

Table 19. Summary of GPS moose data 1997-1999.

Year	No. moose	M/F	GPS	Diff. GPS%	Days
1997	4	0/4	4854	0.60	304
1998*	11	5/2	8142	0.68	980
1999#	5	5/0	-	-	-
Total	20	10/6	12996		1284

*Data until September 1998.
Data, September 1998-March 1999, to be processed.

Robertsfors
Results from the Robertfors area calculated from three moose during the period 5 March to 28 June, 1997 show variations in utilization distributions (UD) on all resolutions (probability levels) i.e. 30%, 70% and 95% of data points, respectively (Table 20).

Table 20. Home range areas for different utilization distributions for three female adult moose at Robertsfors during March 5 to June 28, 1997. Areas in km^2.

Moose	n*	30%	70%	95%
04	264	0.7	6.1	19.6
95	266	6.3	19.5	61.4
BD	295	12.4	60.9	144.1
Mean		6.47	28.83	75.03
SD		5.85	28.57	63.36

*Number of records used to estimate UD.

In 1997 moose started to move longer distances in late May and early June. Two moose cows (BD, 95) started their move at the same time and moved the same direction, but along different routes (Figure 21).

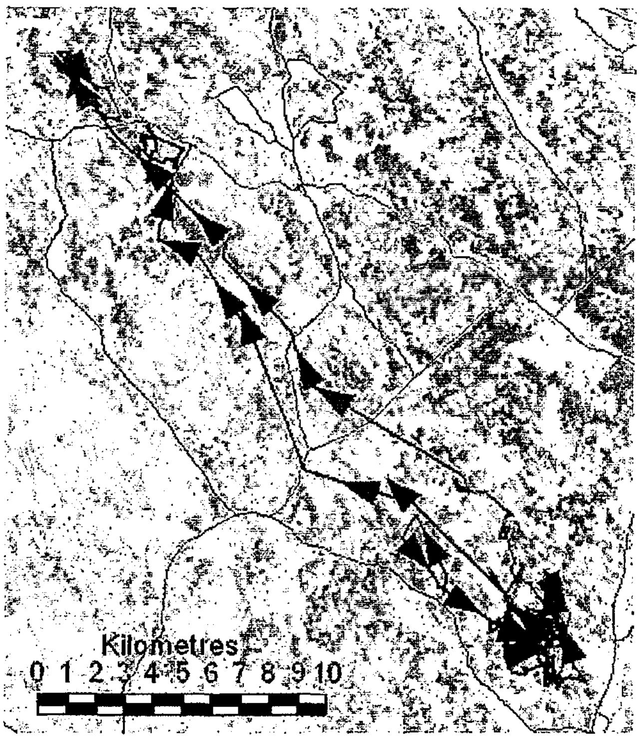

Figure 21. Walks taken by females BD (purple) and 95 (blue). There was a shift in both cases from the SE to the NW in mid March 1997.

The reasons for this variation in size of the home range area among individuals and the long distance movements are not fully understood. Probably, the animals are "programmed" into a seasonal migration that is contributing to the synchronised timing of longer moves. However, not all animals undertake extended movements. For instance moose 04 remained in its original area during the study period. Although the data are limited, the results indicate that there is large enough variation among individuals in movement patterns to be a driving variable in variation of caesium intake.

In 1998 the home ranges of two cows and five males were established for three different time periods. Weather information collected during the project period at the Umeå airport was used to delineate three datasets (Set 1-3) based on the snowfall and a 7-day moving average mean temperature. The period of time encompassed by Sets 2 and 3 was kept to approximately 65 days and was similar therefore to the period obtained for Set 1 which was set by the snowfall period:

Set 1 - Winter period with mean temperature below 0 °C (Capture to 20 April). Precipitation as snow.
Set 2 - Spring and mean temperature above 0 °C and generally between 5 and 10 °C (21 April to 24 June). All precipitation in the form of rain.
Set 3 - Peak summer period, mean temperature above 10 °C (25 June to early September)

Although the data are limited, seasonal variations in the size and morphology in the home range areas are indicated (Table 21, Figures 22-24). However, the difference between seasons was not significant at Kernel estimate at 95% confidence limit (ANOVA, $F = 2.94$, $p < 0.096$), but was so for Kernel estimate at 70% (ANOVA, F-ratio=5.049, $p<0.028$) with spring home range significantly larger than winter (Tukey, $p < 0.024$).

Also Kernel estimate at 30% gave significant effect by season on home range area (ANOVA, $F = 4.382$, $p < 0.04$). There were significant differences between 1997 and 1998 winter home ranges for Kernel 30% and 70% (Kruskal-Wallis, $p < 0.03$ and $p < 0.03$ respectively).

The 95% UD extrapolates the home range estimate than the lower value distributions (e.g. 70% and 30%). As such it indicates the potential total distribution.

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Table 21. Kernel estimate of home range at different seasons with 95%, 70% and 30% UD of points used. Data for 1998, Robertsfors.

Season	**N***	**30%**	**SD**	**70%**	**SD**	**95%**	**SD**
Winter	7	0.54	0.46	2.44	2.12	10.63	10.29
Spring	4	4.33	3.94	16.58	12.95	51.98	49.71
Autumn	3	0.9	0.7	5.4	4.25	19.83	12.58
Total	**14**	**1.7**	**2.6**	**7.11**	**9.14**	**24.41**	**31.37**

* Number of moose

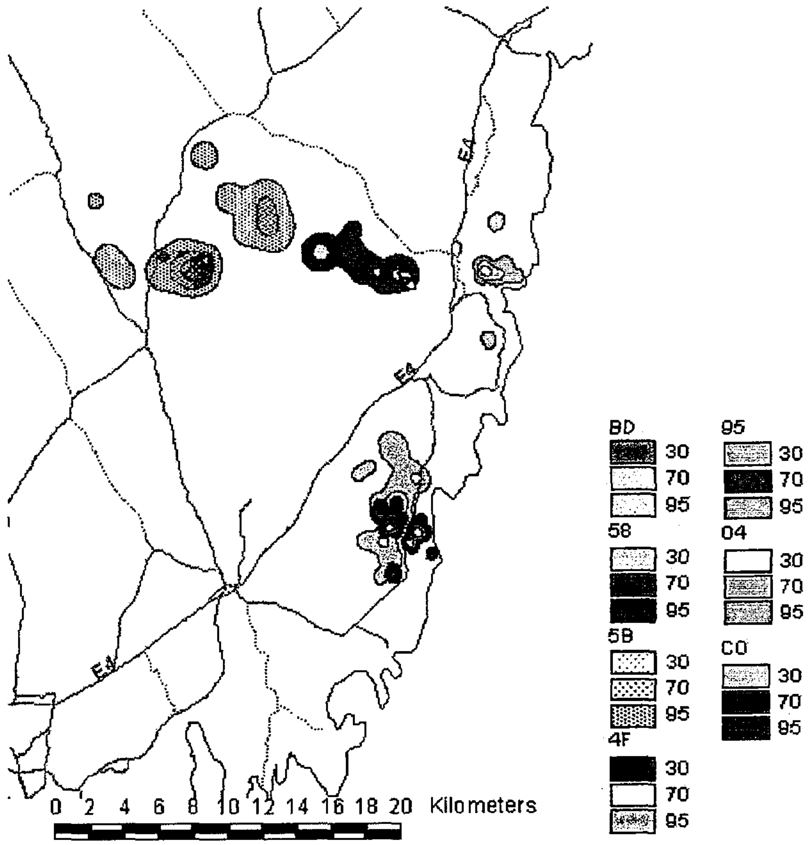

Figure 22. Home ranges for moose for the period early February to late April 1998. Robertsfors

In each case where it has been possible to create home range estimates for moose in the period February to March and also in April to June the size of the home range increased (70% UD) from winter to summer. This was exaggerated for female moose 95 owing to a shift in the home range (north to south, c.f. Figures 23 and 24). Unfortunately no further data was obtained for this moose. During the summer the two males (4F and 5B) and the female (04) had a smaller home ranges (70% UD).

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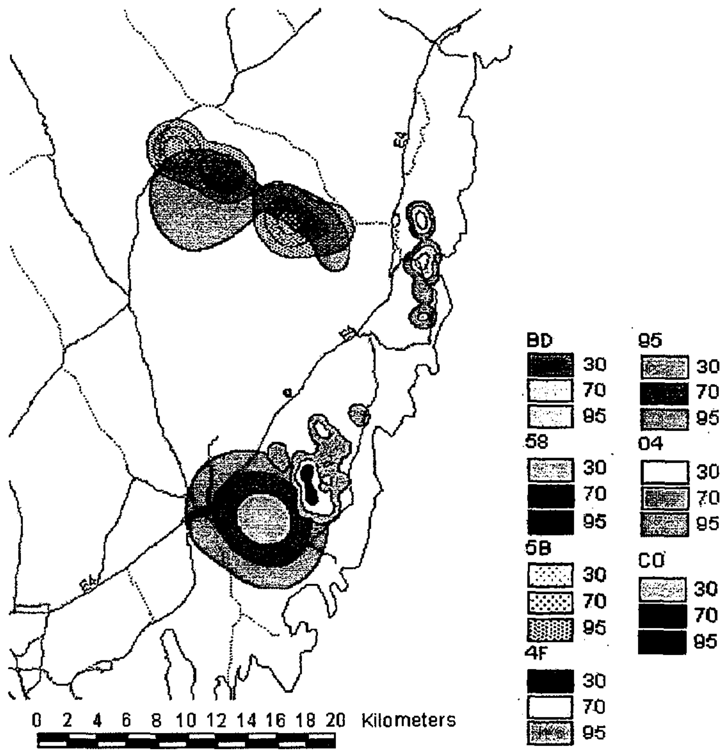

Figure 23. Home ranges for moose for the period late April to late June 1998. No data for C0 or BD. Female 95 shows a shift in her home range and female 04 shows significant range expansion. Robertsfors

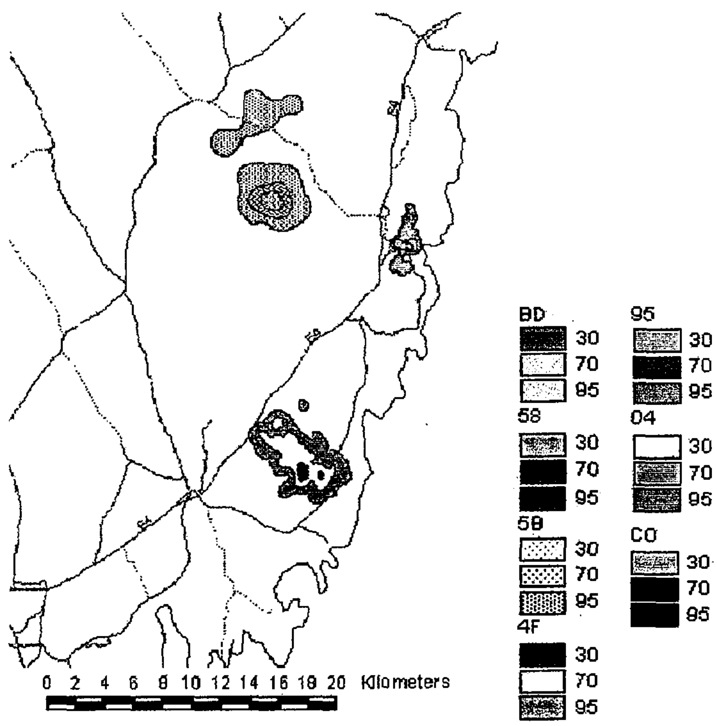

Figure 24. Home ranges for moose for the period early late June to early September 1998. Robertsfors

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Abisko

In Abisko the moose were not used in the alkane experiments so the GPS schedules were less complicated. The first download period in late March to early April 1998 was successful. Following the thaw the moose moved a considerable distance and contact was lost, contact was again established in February 1999 and collars retrieved.

As with the Robertsfors data sets with a minimum of 6 hours between fixes was used for the estimation of home range size in order to avoid problems with autocorrelation (Table 22, Figure 25). There was a considerable home range overlap for moose in the Abisko area.

Table 22. Mean and standard deviation of home range size for moose at Abisko.

Moose no.	N#	95%	70%	30%
53	166	30.2	8.0	2.0
55	77*	22.0	8.3	1.6
56	155	11.4	2.8	0.8
61	127	20.0	4.2	1.1
Mean		21.2	5.8	1.1
SD		9.4	2.7	0.8

*Collar was dropped on, or about, 23 February 1998. It remained operational and was retrieved 16 July 1998.
Number of records used to estimate UD.

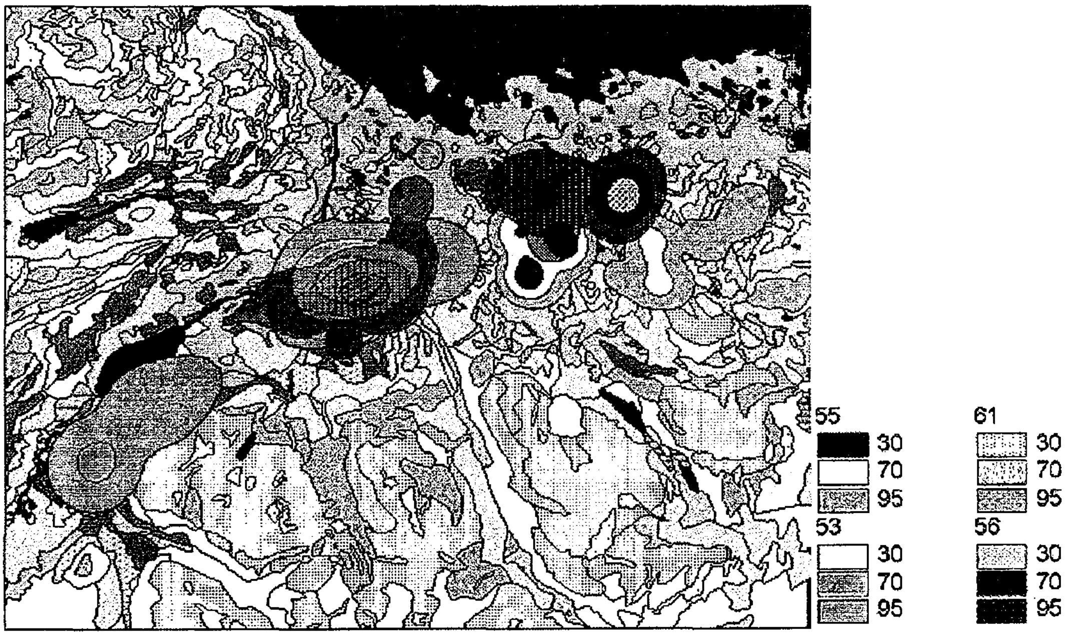

Figure 25. Home range morphology for moose captured in the Abisko Valley 1998.

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Utilization of Vegetation

Robertsfors

LANDSAT imagery provided by the projects "*Remote Sensing for the Environment*" and "*National Forest Inventory Data*" enable analysis of vegetation distributions within the Robertfors study area. The data set includes the total biomass and volumes per hectare of pine, spruce, birch and some other less common tree species. This data was used to analyse the vegetation used by moose for late winter/spring in 1997 and 1998 and for summer for 1998 (Table 23). A minimum convex polygon was created to encompass the moose locations and random set of data points was then generated to examine the expected utilisation if the area was used without bias (Figure 26).

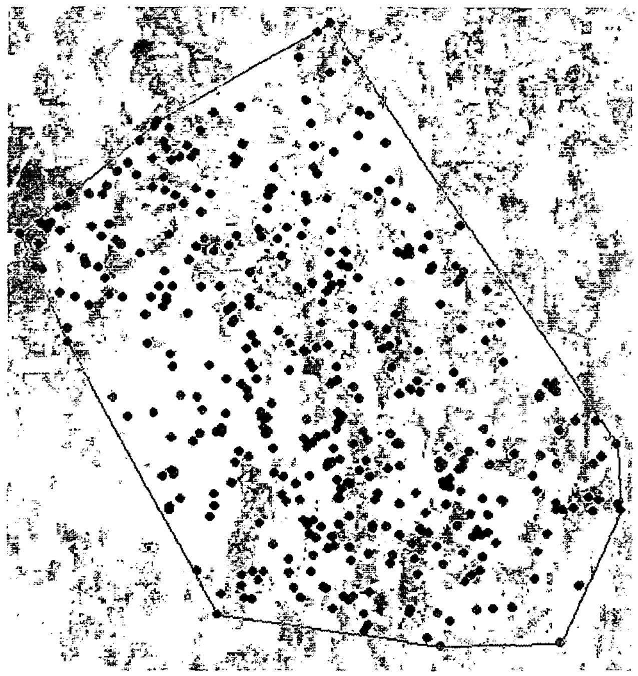


Figure 26. Example of the use of minimum convex polygon to generate random data points that provide an estimate on the unbiased selection of habitat. Original data (red) for moose 04 in summer and random points (blue).

The utilisation patterns based on the species and biomass present were similar between years for the winter period although the level of under- and over-utilisation varies among individuals, reflecting local variations in availability. Importantly, the presence of discrimination on species and available biomass has occurred within a polygon encompassing the location records for that single individual during that season.

In all situations examined the moose utilised areas with low volumes and levels of biomass as this probably represents the browse within reach. A large biomass would indicate a mature forest and much of the browse might be out of reach depending on the nature of the understorey. As the technique used to determine the vegetation type uses satellite imagery it is not possible to predict the type of understorey vegetation.

Vegetation types used in summer were in the same proportion as the winter on an individual basis as no significant differences were found when the winter distribution was used as the expected distribution for summer.

*Table 23. Utilisation based on 6-hour records from LANDSAT derived estimates of vegetation. Individual Chi^2 (**less than expected**, **<u>greater than expected</u>**). Data collection period: 24 June to end of August/September 1998 for moose 5B, 4F and 04.*

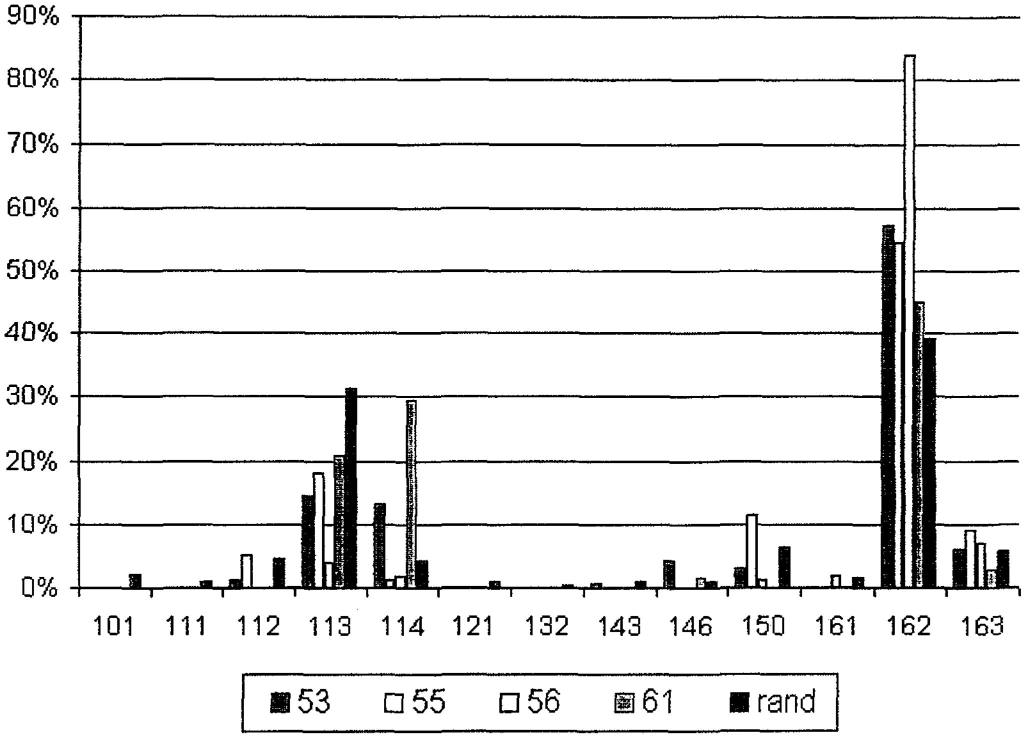
	5B					4F				
Biomass (t/ha) Volume (m^3/ha)	Biomass	Birch	Spruce	Decid	**Pine**	**Biomass**	**Birch**	**Spruce**	Decid	Pine
0-25	<u>*23%*</u>	81%	<u>*58%*</u>	100%	<u>*37%*</u>	<u>*36%*</u>	88%	55%	100%	<u>*44%*</u>
26-50	<u>*12%*</u>	13%	12%	0%	21%	8%	11%	**10%**	0%	17%
51-75	15%	4%	10%	0%	25%	10%	**1%**	**7%**	0%	10%
76-100	33%	2%	8%	0%	12%	18%	0%	<u>*9%*</u>	0%	18%
101-125	**10%**	0%	6%	0%	4%	**13%**	0%	**4%**	0%	**6%**
126-150	4%	0%	4%	0%	2%	9%	0%	<u>*6%*</u>	0%	3%
151-175	4%	0%	2%	0%	0%	3%	0%	<u>*4%*</u>	0%	1%
176-200	0%	0%	2%	0%	0%	2%	0%	**3%**	0%	0%
201-225	0%	0%	0%	0%	0%	1%	0%	**2%**	0%	0%
>225	0%	0%	0%	0%	0%	0%	0%	1%	0%	0%

	04				
Biomass (t/ha) Volume (m^3/ha)	Biomass	Birch	Spruce	Decid	Pine
0-25	22%	98%	56%	100%	25%
26-50	12%	2%	13%	0%	17%
51-75	23%	0%	9%	0%	28%
76-100	21%	0%	4%	0%	9%
101-125	12%	0%	6%	0%	17%
126-150	8%	0%	3%	0%	4%
151-175	1%	0%	7%	0%	1%
176-200	1%	0%	1%	0%	0%
201-225	0%	0%	1%	0%	0%
>225	0%	0%	0%	0%	0%

Abisko

The proportion of time spent in each vegetation type was determined for each location using the same 6 hourly data set as for the home range analysis (Figure 27). The proportion of time spent in each vegetation type was compared to the random distribution of 496 locations within a minimum convex polygon encompassing all locations for the four moose.

Fresh heath and the heath type birch forest with mosses vegetation were both used more than expected and dry heaths were avoided (Table 24). Variation between was noticeable even though moose were constrained within a relatively small geographic area, e.g. compare the utilisation observed of extremely dry heath and birch with mosses by moose 56.



Code	Description	Code	Description
101	Blocky areas and bedrock outcrops	143	Dry fen
111	Grass heath	146	Mosaic of bog and hummock vegetation and dry fen
112	Dry heath	150	Willow
113	Extremely dry heath	161	Birch forest - heath type with lichens
114	Fresh heath	162	Birch forest - heath type with mosses
121	Meadow with low herbs	163	Birch forest - heath type with tall herbs
122	Meadow with tall herbs		
132	Extreme snow bed		

Figure 27. Proportion of total number of 6-hour records that occurred in each vegetation type. Moose are represented by their collar number and the random distribution is shown as 'rand'.

*Table 24. Observed percentage utilisation of vegetation types for moose 53, 55, 56 and 61. Results of individual Chi^2 tests (**less than expected**, ***greater than expected***).*

Vegetation type	Moose 53	55	56	61
Blocky areas and bedrock outcrops	0%	0%	0%	0%
Grass heath	0%	0%	0%	0%
Dry heath	**1%**	2%	**0%**	0%
Extremely dry heath	**14%**	**8%**	**4%**	9%
Fresh heath	***13%***	1%	2%	***13%***
Meadow with low herbs	0%	0%	0%	0%
Extreme snow bed	0%	0%	0%	0%
Dry fen	1%	0%	0%	0%
Mosaic of bog and hummock vegetation and dry fen	**4%**	0%	0%	1%
Willow	3%	5%	**1%**	**0%**
Birch forest - heath type with lichens	0%	0%	2%	0%
Birch forest - heath type with mosses	***57%***	***25%***	***78%***	19%
Birch forest - heath type with tall herbs	6%	4%	7%	1%

Discussion

Moose dwell over large geographical distances and thus encounter gradients in the distribution of fallout and heterogeneous vegetation. Animal movement patterns reflect behavioural adaptations among which foraging strategies are one important factor. Differential survival and reproduction is assumed to be the mechanism by which foraging behaviour evolve. The extent to which foraging is limited is related to energy demand, forage quality and availability. Changes in digestibility, forage structure and depletion of food resources are linked to physiological constraints that could be reflected in movement patterns. One goal is to find the strategy that optimizes the energy intake over some period of time (Stephens and Krebs 1986, Belovsky and Schmitz 1991). The solution is not just to eat more but also to adjust foraging behaviour temporally and spatially to maintain a long-term positive energy and nutritional balance.

Seasonal shifts in feeding range, crossing of habitat boundaries, selection of food plants and tissues are important factors for intake of radionuclides from natural vegetation and further transfer in food webs. We anticipate that animal foraging activity along movement tracks and duration of feeding bouts in particular vegetation types are important mechanisms for intake of radionuclides. More specifically, one aim was to identify factors of importance for individual variability in intake of radionuclides. Several likely candidates for such variability can be identified. Diet composition, digestibility, duration of foraging bouts, habitat size and types and radiological gradients are some of them. It was previously shown on theoretical grounds that diet composition could be one important factor for radionuclide intake and variation among individuals (Palo et al. 1991, Palo and Wallin 1996, but see also Avila 1998). Here the focus has been on habitat selection, composition and spatial distribution as one additional factor for variability in radionuclide intake.

Based on a model developed by Lundberg and Palo (1993) it is possible to analyse intake of radionuclides with respect to forage density and diet quality. According to this model and to energetic principles it is expected that moose occupy more dense habitats during winter in order to minimize energetic cost for movement (Moen et al. 1997). In summer, according to model predictions, the moose should be more selective which might result in more extended movements and hence larger home range (Lundberg and Palo 1993).

The data in this project show an extension of home range in summer compared to winter at the Robertfors site, in line with model predictions, although the variation in home range size between individuals is large. Further, at Abisko, moose show considerable overlap in home range possibly due to aggregation in the most suitable and profitable habitats in this area, a result that is expected also from a theoretical point of view.

Although the moose at Abisko aggregated within the area dictated by the steep valley sides, there was a considerable individual variation in the use of the available vegetation types. Three of the four moose used the birch forest with mosses at a rate greater than expected. However, utilisation of this vegetation type ranged from 19% to 78%. Wide variation in resource selection at the individual level might be reflected in a high variance in cesium intake.

In Robertsfors, feeding by moose in winter is dominated by the availability of pine and the selection of food intake is similar for each moose. Vegetation types were used in winter in the same proportion as the summer on an individual basis. Since the variation in utilisation occurs within the MCP home range this implies that the moose are highly selective at a fine scale, but can be considered unselective at coarse scales.

It is reasonable to assume that food intake and radionuclide intake is directly proportional and that food intake is directly related to forage density and quality. What consequences does the variation in these parameters have on caesium intake over larger spatial scales? If a uniform deposition over the landscape is assumed, then only habitat characteristics such as availability of forage would be important for intake of radionuclides. If animals could compensate for reduced forage quality by searching a more abundant resource then how would foraging time be adjusted? Of interest from a radiological point of view are those behaviours constrained by ecological factors that determine intake of radionuclides. More specifically, how frequently do moose forage in habitats that are associated with low/high transfer rates and what are the factors associated with nested selections?

The goal of the forager is to maintain the intake rate even though quality is reduced, it can do so by reducing its walking time, i.e. staying longer in the better habitat. From these arguments it could be concluded that intake rate of radionuclides is less in poor habitats (mainly due to increased walking time) than in more superior habitats given the same deposition. The data presented here show that moose restrict their home range in winter at the same time as food quality and availability are reduced. Under these circumstances we expect decline in radionuclide intake. Nelin (1994) and Avila (1998) found lower activity concentrations in moose during winter than summer in line with predictions.

The habitat composition in relation to utilization of moose is not yet fully analysed. At a coarse scale, diet intake appears to be dominated by a few species, Scots pine in winter and birch in summer. This is seen both at habitat level and at diet selection level. The role of fungi notwithstanding, diet selection, in terms of the species utilised, does not explain the variation in cesium seen in moose muscle samples. The experiments have demonstrated that there is significant variation between moose at many levels of scale. This variation results from different migration patterns, interactions between moose for a home range, selection of within the home range for habitat patches and selection of the browse item itself.

Ritchie and Olff (1999) have recently suggested a common principle, a synthetic theory for animal and plant distributions that builds on spatial scaling. Here the encounter rate of food is dependent on the degree that resources fill space and the ruler length is the body size. Thus habitat size and resource concentration, i.e. degree of resource fragmentation, could be expected to be important for population density as well as for habitat encounter and food intake.

There is a tendency in radionuclide research to concentrate on the temporal aspects and to average out the spatial variation. Understanding the effect of the spatial heterogeneity imposed by individual choice of food resources may make an important contribution to the knowledge of the transfer of cesium to man via wildlife products.

Conclusions

This study shows that while it is possible to make useful predictions about radionuclide exposure for regional areas, e.g. Västerbotten county, great care should be taken before interpolating the results. The factors that operate at the population level are largely irrelevant when discussing the exposure at an individual level. To understand this one must also begin to explore individual-based models of intake. This is evident because the variation induced by tertiary (within home range) and quarterly (within plant) selection (Johnson 1980) will be reflected in the components of browse selected and hence the intake of radionuclides. A limited step has been taken in this direction, but it is believed that an integrated approach, combining basic ecological facts with geostatistical and individual-based models would give new insights to radioecology. The present approach, fully explored, could open a new avenue in understanding pattern and exposure of pollution in wildlife populations.

MODELLING OF THE FOREST ECOSYSTEM

Ecosystem models

A review of existing models

The modelling of radionuclide migration and transfer in forest ecosystems has been an area of scientific investigation since the birth of radioecology in the late 1950´s and early 1960´s. The earliest mathematical models of the migration and transfer of radionuclides in forest ecosystems were developed from data obtained in experiments with caesium inoculation (Olson, 1965) and from studies of the transfer of stable isotopes in the forest (Jordan *et al.* 1973). Other early models were based on the knowledge obtained during studies in forests contaminated by fallout from tests of nuclear weapons (Croom and Ragsdale, 1980), by the Kyshtym accident (Prokhorov and Ginzburg, 1973; Alexakhin *et al.* 1976) and by releases during the Manhattan project (Garten *et al.* 1978; Van Voris *et al.* 1990). The Chernobyl accident lead to radioactive contamination of large territories covered by forests. The knowledge obtained in studies carried out in these forests has been used in the development of several new models: the FOA model (Bergman *et al.* 1994), FORESTPATH (Schell *et al.* 1996), RIFE.I (Shaw *et al.*, 1996), FORESTLIFE (Shaw *et al.*, 1996) and ECORAD (Mamikhin *et al.* 1997). In LANDSCAPE a review of these models was carried out with emphasis on those models developed after the Chernobyl accident (Avila *et al.*, 1998, Riesen *et al.* 1999).

One aim of the model review was to identify which transfer processes have been included in the existing models and in which way. A list of the considered transfer processes and their relative importance in different phases after an aerial contamination is presented in Figure 28. This list was created applying a systematic approach involving the use of interaction matrices (Avila and Moberg, 1999), which was developed in the project. A second aim of the review was to evaluate to what extent the existing models can be used to study the factors responsible for horizontal patterns of the contamination, such as: the type and season of the deposition, the type of forest (deciduous and coniferous), biomass growth, age of the trees and the soil characteristics. The main conclusions of this review can be summarised as follows:

Static models have a limited applicability in forest ecosystems. This is primarily because the transfer of radionuclides in forests is a non-equilibrium process. Many years (decades) after a deposition, however, a quasi-equilibrium state can be reached concerning the uptake of radionuclides by forest plants. At this stage, transfer factors can be applied with a better confidence. A combination of transfer factors with dynamic models as implemented in some models (e.g. FORESTLIFE and RIFE.I) provides one way of using the available data on aggregated transfer factors. The existing dynamic models include the most relevant transfer processes, both for the acute and the long-term phases of the contamination. Most of the transfer processes are, however, described with rate constants. These are in many cases highly variable and include several transfer processes. They are, therefore, difficult to interpret, measure and estimate.

The existing models can hardly be applied to explain and predict differences in the behaviour and distribution of radionuclides in different types of forests. First, all models (except FORESTPATH) are site specific and the characteristics of the forests used for calibrations are usually not provided, which basically makes it difficult to apply them for assessments in other areas. The parameter values in FORESTPATH are given for generic forest ecosystems, but some of them vary within two or more orders of magnitude. Second, the models, with few exceptions, do not allow considering the factors responsible for differences in radionuclide behaviour and distribution in different forests, such as soil characteristics, type of forest (coniferous, deciduous), age of the forest, biomass density and productivity, etc. None of the existing models included forest game as a compartment, probably due to lack of knowledge of the game diet.

Atmosphere (Air)	1.2 Interception	1.3 Interception	1.4 Interception	1.4	1.5	1.6 Interception	1.7 Interception	1.8 Interception
2.1	2.2 Tree leaves	2.3 Translocation	2.4 **Leaves fall** Weathering	2.5	2.6	2.7 Weathering Interception	2.8 Weathering Interception	2.9 Ingestion
3.1	3.2 **Translocation**	3.3 Tree other	3.4 Weathering Interception	3.5 Fertilisation	3.6 Fertilisation	3.7	3.8 Weathering Interception	3.9 Ingestion
4.1 Resuspension	4.2	4.3 Rain splash	4.4 Litter	4.5 **Litter Decomposition,** Percolation	4.6 Percolation	4.7 Root uptake	4.8 Rain splash	4.9 Ingestion
5.1	5.2	5.3 **Root uptake**	5.4	5.5 Soil organic **Adsorption/ desorption**	5.6 Percolation, **Diffusion/ Advection**	5.7 **Root uptake**	5.8 **Root uptake**	5.9
6.1	6.2	6.3 Root uptake	6.4	6.5	6.6 Soil mineral **Adsorption/** desorption	6.7 Root uptake	6.8 Root uptake	6.9
7.1	7.2	7.3 **Root uptake (Mycorrhizae)**	7.4 Fertilisation	7.5 Fertilisation	7.6 Fertilisation	7.7 Fungi	7.8 **Root uptake (Mycorrhizae)**	7.9 **Ingestion**
8.1	8.2	8.3	8.4 Leave fall, Weathering Interception	8.5 Fertilisation	8.6 Fertilisation	8.7	8.8 Understorey	8.9 **Ingestion**
9.1	9.2	9.3	9.4 Fertilisation	9.5	9.6	9.7 Consumption	9.8 Consumption	9.9 Wild animals

Figure 28. Matrix description of the migration of ^{137}Cs in a forest ecosystem. Transfer processes relevant for the acute phase are marked green and for the long term phase yellow.

The general conceptual model presented in Figure 28 and the results of the review of existing models were used in the development of the three LANDSCAPE models: FOA by the Swedish Defence Research Establishment, LOGNAT by the University of Trieste and FORESTLAND (FRLD) by the Swedish Radiation Protection Institute[8]. These models have been tested within the project (Avila et al. 1999) as well as in the frame of the IAEA Programme BIOMASS. The three models developed in LANDSCAPE are based on the compartment model principles and first order kinetics for the turnover of caesium in the boreal forest. However, by the individual strategies chosen for describing and structuring the ecosystem and the different emphasis put on certain transfer and turnover processes, the

[8] The development of FORESTLAND was performed in close cooperation with Drs S. Fesenko, S Spiridinov and R Alexakhin, Russian Institute of Agricultural Radiology and Agroecology, Obninsk, Russia. (SSI project RYS 6.15)

resulting models exhibit in essence different characteristics. The main features of the three models are described below.

The FOA-model

In the FOA model (Bergman *et al.* 1993) the major regulators of energy flow, as well as of caesium turnover, are related to primary production and its constraints on the growth capacity. Certain fundamental physiological processes governing the metabolism of living matter in the biotope are also considered as well as (i) maximum attainable total biomass; (ii) dynamics of age dependent net productivity; and (iii) successional stage, stem density and age of the forest stand; which are all essential factors for the site specific growth dynamics.

The theoretical analysis is based on compartment theory and first order kinetics for the turnover of caesium in the boreal forest. The calculated time dependent change of the ^{137}Cs content in perennial vegetation has been compared to that actually observed at different local study sites with focus particularly on bilberry, pine and birch.

The estimated transfer factors (Bergman *et al* 1993, Bergman *et al.*1998) are based on the actual results for the time-dependent redistribution of ^{137}Cs from secondary sources in a Scots pine canopy by throughfall and needlefall (Nylén and Grip 1989, Nylén 1996), in addition to the release to the environment of ^{137}Cs deposited on the moss and lichen carpet.

After the Chernobyl accident loss from the system by runoff is less than that due to physical decay – from 1987 and onwards – and is therefore disregarded in the model.

The explanatory model structure of the FOA model includes principally components and interactions known or expected to be of major importance for the caesium circulation and the redistribution dynamics. The relatively minor amount of vegetation consumed by moose in boreal forest ecosystems does not motivate inclusion of a specific compartment to consider its herbivory in the explanatory model. However, moose (and part of its feed, i.e. bilberry) belong to boreal forest food-chains of importance for transfer of radioactive caesium to man.

By incorporation of compartments for moose (adjusted to the chosen moose population density) and representative components of its major feed during different seasons (twigs of bilberry, birch and pine), the FOA model has been adapted for descriptive purposes, simulating early and long-term dynamics of the caesium activity concentrations in the compartments according to the model structure illustrated in Figure 29. The main structure and corresponding compartments of the ecosystem are shortly summarized.

Tree

Compartments *Needles* and *Throughfall*

Interception of deposited radioactive caesium by the tree canopy is accomplished by input to the *Throughfall* compartment. The *Needle* compartment provides the loss to the forest floor by litterfall. The transfer of radioactive caesium by litterfall as a function of time after deposition is adapted to experimental data (Nylén 1996).

Compartment *Stem&Roots*

The rate of exchange of caesium over the fine roots in soil is assumed to be governed by growth of tree biomass. The time-dependent uptake from compartment *Soil "available"* thus depends on tree age and site specific primary productivity (the level regulated by a parameter). Transfer in the opposite direction, i.e. from tree to soil, is assumed to occur with the same dynamics as for the understorey vegetation.

Compartment *Branches*
The *Branches* compartment constitutes an intermediate link between needles and the stem. The transfer dynamics has been tuned for predicted transfer by litterflow and throughfall to agree with the results from measurements.

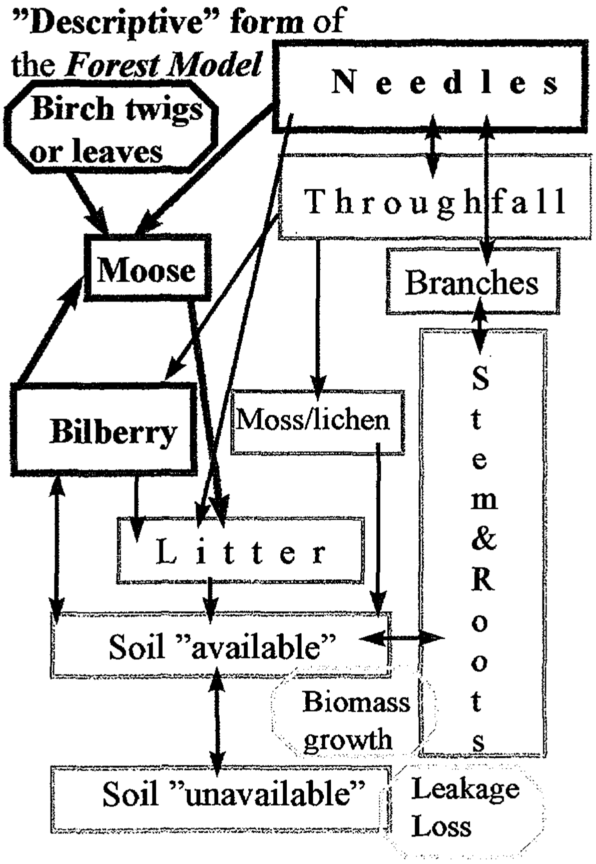


Figure 29. Structure of the FOA model for descriptive applications concerning total compartmental contents and concentrations of ^{137}Cs in e.g. moose meat, understorey vegetation, and different parts of pine trees. Compartments and flows directly related to the compartment "moose" are indicated by blue lines.

Forest floor
The "forest floor" in the model contains separate compartments for the understorey, litter and moss or lichen carpet respectively.

Compartment *Bilberry*
The bilberry (called understorey compartment in the explanatory model) compartment in the descriptive model is contaminated by direct interception during the period of deposition and by throughfalling rain.

Compartment *Litter*
Input to the litter compartment is due to litterfall from the tree canopy and understorey vegetation as illustrated in Figure 29. Decomposition of litter is assumed to accomplish transfer of radioactive caesium to the organic soil layers constituting the compartment defined as *Soil "available"*.

Compartment *Moss/Lichen*
Generally, although with notable species specific differences, moss and lichen retain deposited radioactivity relatively long – i.e. of the order of several years – before successive released to other ecosystem components. A moss or lichen carpet acts as a secondary source, giving rise to a protracted feeding of its share of the initial ^{137}Cs deposition into circulation. This particular feature of a secondary source, thus may influence the dynamics of redistribution depending on how much of the ground surface that is covered by the carpet.

Soil
Compartments *Soil "available* and *Soil "unavailable"*
Soil "available" obtains input at the fine root interfaces with understorey vegetation and the trees, as well as from litter decomposition. Furthermore, exchange of caesium is assumed to occur with components of soil not directly accessible for uptake by the root system, i.e. the compartment *Soil "unavailable"*.

Moose
The caesium content in the moose compartment relates to the food consumption, and excretion for an average moose (averaged over the whole population of adult and calves) times the population density. Compartments and associated interactions directly involved in transfer of caesium through the moose compartment are indicated by bold blue lines in Figure 29.

The LOGNAT model

The ^{137}Cs circulation model LOGNAT is derived from a former version developed under a previous project[9]. This model is addressed to assess the circulation of ^{137}Cs in the compartments of a generic forest ecosystem, expressed as percentage values of the initial deposition. The model is quite simple-structured and with limited input parameters, bearing in mind that it should be the numerical processing basis between GIS input and output codes, and that territorial data at a regional scale are often uncomplete, or not easily manageable, or even missing.

The LOGNAT model is available in software STELLA II and DOS-QBasic and is a simple compartment-type model assessing the transfer of ^{137}Cs from an initial event (deposition at t=0) as a function of time (years). The variable state is expressed as a fraction of the initial amount of ^{137}Cs (adimensional), intended as Bq/ m^2. The main assumptions of the model are:

- The model calculates the circulation of ^{137}Cs in a closed system (forest), assuming an initial deposition in the litter, holorganic and leaves compartments as input values.
- No losses (sink) from the system are accounted for.
- Corrections are made for radionuclide decay, with a ^{137}Cs half-life of 30 years (k = 0.023).
- Transfer between compartments are expressed as first order kinetics, in form of dimensionless parameters (fraction of total amount, 1/yr).
- Transfer parameters have been derived from experimental data (litter decomposition, soil and leaves sampling, etc.) and literature.
- The forest biomass evolution at long-term scale (i.e. 50 years) is neglected during simulations, because of the mature stage of the forest stand under analysis.
- Uptake rates are considered as functions of the standing biomass. The uptake rate is expressed as a fraction parameter per standing biomass unit (1/ (yr * kg standing biomass)).

Model structure, experimental scenario

The compartments and the flow scheme of LOGNAT is shown in Figure 30. The input profile for LOGNAT consists of (i) the fraction of ^{137}Cs deposition in each compartment at t=0, and (ii) the total

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[9] EU-FI3PCT920050, 1992-95

standing biomass for each vegetation component (deciduous, conifers, understorey). The parameters were calculated both from experimental data (soil and litter) and from literature data, especially concerning biomass distribution and standing biomass/productivity estimation. The calibration of LOGNAT was performed using the experimental scenario of Tarvisio, where experimental data were recorded in two forested plots, the first mixed, the second mostly dominated by conifers. The environmental characteristics of the test scenario area are described in Table 25.

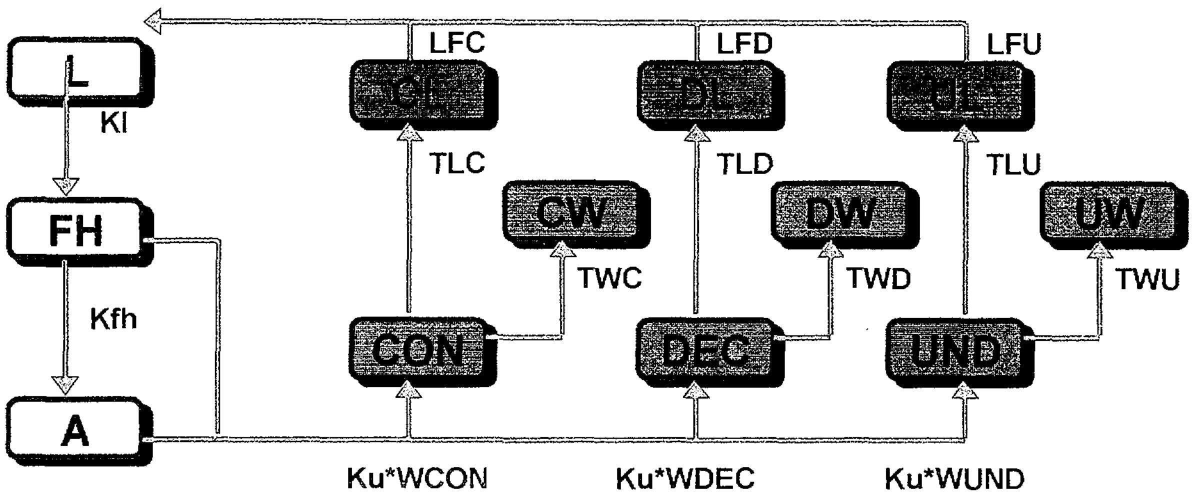

Figure 30. The functional structure of the LOGNAT model. Symbols are: L-litter, FH-holorganic layer, A-mineral, CON-Conifer, DEC-Deciduous, UND-Understorey, CW-Conifer wood, DW-Deciduous wood, UW-Understorey wood, CL-Conifer needles, DL-Deciduous leaves, UL-Understorey leaves. Red labels are the transfer parameters (WCON, WDEC, WUND: standing biomass of Conifers, Deciduous, Understorey)

Table 25 Characteristics of the Tarvisio forest area

LOCATION	Forest of Tarvisio, Rutte Piccolo, Friuli-Venezia Giulia, N-E Italy, near the border to Slovenia and Austria, 870 m a.s.l. Experimental area has 1 ha of extension.
SOIL TYPE	Brown earth on detritic glacial alluvium
ORGANIC LAYER DEPTH	3-5 cm
SOIL DEPTH	20-25 cm
FOREST TYPE	Mixed forest, mainly dominated by *Picea excelsa* (Red Fir, 77%) and *Fagus sylvatica* (Beech, 17%). Minor presence of *Larix decidua* and poor understorey, almost totally characterized by *Vaccinium myrtillus*, with no remarkable saplings of the main trees.
FOREST AGE	80 to 100 years, natural reserve area with some zones under forestry practices.
STANDING BIOMASS	15 to 20 kg/ m^2, net primary productivity about 0.5 kg/ m^2 yr, forest growth neglible during experiment.
CLIMATE	Temperature: year average 5°, winter minimum –20° to –25°, summer maximum 25° to 30° C Rainfall: year average 1500 mm, mainly in spring and autumn Snow cover: deep from November-December to April-May.

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The FORESTLAND model

FORESTLAND is a dynamic model for the prediction of temporal and spatial patterns of the radioactive contamination of forest ecosystems (Avila *et al.*, 1999). The model is focused on migration pathways leading to internal and external radiation doses to the population. FORESTLAND can be applied to both the acute and long-term phases of the contamination after an aerial radioactive deposition. The present version of the model consists of six individual models:

FORBIO: A model of the biomass dynamics of trees and the understorey vegetation,
FORGAME: A dynamic model of the long-term migration of radionuclides in forest food chains, including wild animals,
FORACUTE: A dynamic model of the migration of radionuclides in forest ecosystems during the acute phase of the contamination,
FORTREE: A model of the long-term migration of radionuclides in forest trees,
FOREXT: A dosimetric model for calculation of gamma dose rate in the forest,
FORDOSE: A model for calculation of the internal and external doses to the population (presently under development).

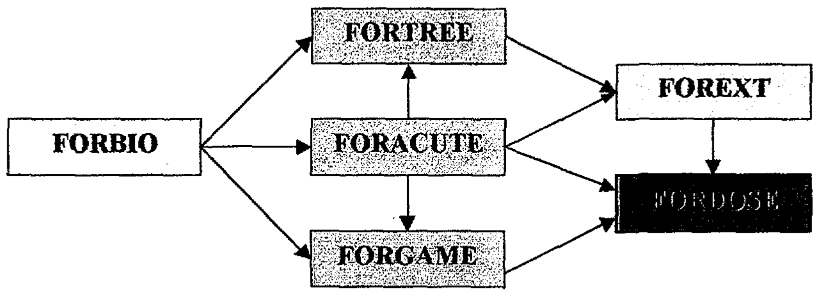

Figure 31 . Structure of FORESTLAND

These models can be used stand alone or as a sequence within FORESTLAND. The model FORBIO, Figure 31, provides the migration models (FORTREE, FORGAME and FORACUTE) with input parameters for calculation of transfer rates. The values of biomass density (kg m^{-2}) estimated with FORBIO are used to obtain the concentration of radionuclides in different forest components (Bq kg^{-1}) from the values of the radionuclide content in these components (Bq m^{-2}) calculated with the migration models. The initial distribution of radionuclides in different forest components needed in FORTREE and FORGAME (initial conditions) are calculated with FORACUTE. Alternatively the user can define directly the initial conditions. FORACUTE and FORGAME provide the values of activity levels in different forest components needed for calculation of the dose rates with FOREXT. A similar connection between FORTREE and FOREXT is being implemented. FORBIO provides FOREXT with some parameters needed for calculation of the attenuation and scattering of gamma rays. Part of the input values needed by FORDOSE are generated by the migration models and FOREXT.

A classification of forest ecosystems into four different categories has been adopted in FORESTLAND. Each category corresponds to a different type of tree (coniferous or deciduous) and landscape (automorphic or hydromorphic). A set of model parameters, consisting of a best estimate value and an interval of variation, is estimated for each forest category. A range of values is defined for each model parameter, which reduces the uncertainties of the parameter values selected for each specific application of the model.

In FORBIO a simple approach for describing seasonal and long-term biomass dynamics of trees and understorey vegetation has been applied. For the understorey vegetation and mushrooms, biomass growth is simulated with a logistic model, while an exponential decrease is assumed during senescence. Differentiation is made between summer and autumn mushrooms and between fruits of berries and the whole plant (animal feeds). The biomass growth of an individual tree is also described with a logistic model while an exponential equation is used for calculation of tree mortality. A linear differential equation, obtained by combining the equations for growth and mortality, is used for simulating the long-term changes of trees biomass density (kg m^{-2}). For tree leaves (needles) distinction is made between seasonal and long-term biomass dynamics. The yearly values of leaves (needles) biomass depend of the age of the tree. It is assumed that the contribution of leaves (needles) to the total tree biomass decreases from 10-15 % for a 15 years old tree to 1-2 % for a 100 years old tree. The seasonal variation of the leaves biomass is described with a logistic model during the periods of growth and senescence.

FORACUTE is a dynamic model of the migration of radionuclides during the acute phase of the contamination, lasting a few years after an aerial deposition. The model describes the primary interception of aerial deposited radionuclides by the above ground vegetation and their subsequent redistribution by transfer processes like weathering, secondary interception, translocation in the tree and the understorey vegetation, and root uptake from the upper soil-litter layer. The model also permits evaluating the dynamics of the radionuclide levels in forest products consumed by man, including forest game. To describe the primary and secondary retention of the radionuclides by the above-ground phytomass, the latter is viewed as a set of four successive filters: the tree leaves (needles), the tree bark, the understorey vegetation and the upper soil-litter layer. The interception by the understorey vegetation is calculated with an exponential function of the biomass density (Chamberlains equation). A method similar to the one commonly used for evaluating the passage of light though tree crowns is used to simulate the initial retention of radionuclides by trees. It is assumed that the initial retention by trees is proportional to the "projective cover" of the tree crowns which can be calculated from the crown closure (relative area of crowns) and the crown tracery coefficient, which depends on the tree species.

The model FORGAME (Avila, 1998) is a dynamic model to predict seasonal and long-term changes of ^{137}Cs activity concentrations in forest food chains (Figure 32). A set of 20 coupled differential equations describes the net accumulation of the radionuclide in the compartments over time. Since the model is focused on forest food chains, the migration in tree is described in a simpler way than in FORTREE. The transfer rates corresponding to the processes of root uptake and translocation in trees are, for instance, described with ordinary rate constants. The soil, on the contrary, is modelled in more detail (10 compartments) with the purpose of describing the influence of roots and mycelia location on root uptake by the understorey vegetation and mushrooms.

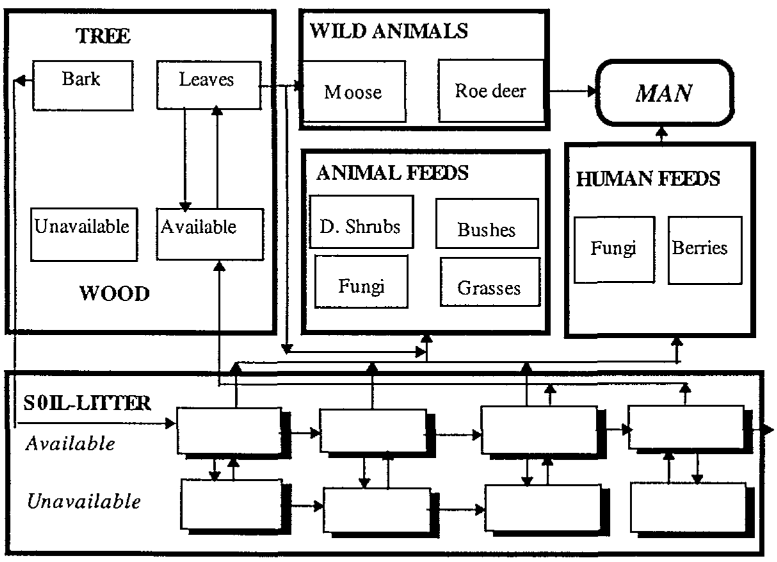

Figure 32 Conceptual structure of the FORESTGAME model

Root uptake by the understorey vegetation is described as a function of root distribution in soil, the available fraction of ^{137}Cs in soil, the soil-to-plant concentration ratio (CR) and the biomass growth rates. The CRs are related only to the available fraction of the radionuclide in soil and have, therefore, the same values for all soil-litter layers. The intake rate of ^{137}Cs by wild animals (roe deer and moose) is described with a function of the total feed intake, the share of different feeds in the daily animal diet and the activity concentrations in different feeds. It is assumed that the radionuclides incorporated by the animal via ingestion are instantly distributed in the animal body and that the elimination rate from muscles (edible meat) is proportional to the activity levels in this part of the animal body.

FORTREE is a dynamic compartment model of the migration of radionuclides in the tree (Figure 33). The main purpose of the model is interpretation and prognosis of the long-term kinetics of ^{137}Cs activity concentrations in wood. Another endpoint of the model is the seasonal change of activity concentrations in leaves (needles). Five compartments are used to describe the change of the quantity of available ^{137}Cs in soil. The first soil layer (0-10 cm) contains the most active parts of the root and is responsible for root uptake during the first several decades after the radioactive contamination. Each soil layer is divided into available, unavailable and fixed fractions with the purpose of considering sorption/desorption and fixation processes occurring in the soil. The tree is described with the following compartments: two compartments for the living part of the wood (liquid and solid phases of the wood xylem), one compartment for the dead (structural) wood, one compartment for leaves (deciduous trees) and four compartments for needles of different age class (coniferous trees). Root uptake is a function of the water flux (transpiration flow) through the xylem during the vegetative period. The main part of this flux is due to evaporation from leaves, which is proportional to the leaves (needles) biomass. The root uptake rates are, therefore, expressed as a function of the leaves (needles) biomass, which depends on tree biomass and age.

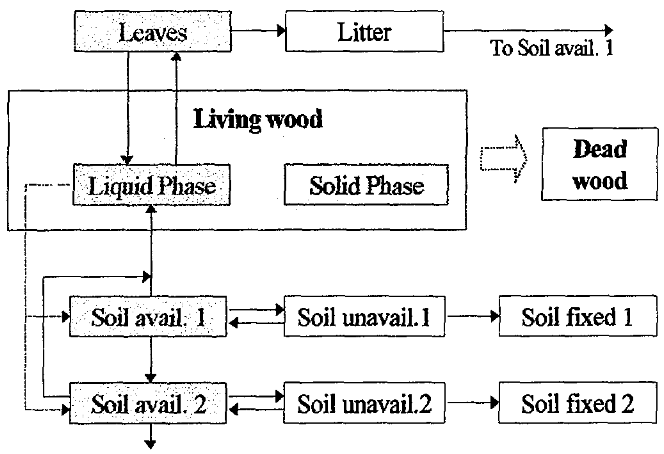

Figure 33 Basic structure of the FORTREE conceptual model for deciduous and coniferous trees.

FOREXT is a dosimetric model for calculation of the dose rates in forests contaminated with gamma emitting radionuclides. Although FOREXT can be used to calculate the exposure rates at any height and for any gamma emitter, in FORESTLAND the model is only adjusted for estimation of dose rate from ^{137}Cs at 1 m from the soil surface. The vertical column of the forest is divided into 7 successive layers with different average densities. The soil is described with four layers, one for each soil horizon (L, Of, Oh and A). The fifth layer covers the range from the soil surface to the average height of the understorey vegetation. The frontier between the sixth and seventh layer is set at the average height of the bottom of the tree canopy. Each layer is considered as a plane source of finite thickness. The activity of each source is calculated from the values of ^{137}Cs activity concentrations in different components of the layer provided either by FORGAME or FORACUTE. The variation of these values with time leads to time variation of the estimated dose rates. The attenuation and scattering of the photons in different layers is taken into account in the dose rate calculations.

Model predictions, tests and validation

The three LANDSCAPE models were developed by different teams and on the basis of experimental data obtained in different geographicla areas. This has resulted in differences in the model structures and in the values assigned to different model parameters. To what extent do these differences lead to disagreement in the predictions made with the models? What would be the degree of agreement between the model predictions and experimental observations made independently in other areas, i.e. not used for calibration of the models[10]. To answer these questions an intercomparison exercise was carried out in two steps (Avila *et al* 1999) with the following specific objectives:

- To determine the degree of agreement between the predictions of radiocaesium levels in different forest components after an aerial deposition (model-model comparison).
- To assess the agreement of the model predictions with experimental data obtained in field studies (model-data comparison).

The experimental data were from pine forests of the Bryansk region of Russia that were contaminated by the Chernobyl accident and these data were collected independently during 1986-1994. The experimental values were not shown to the modellers until they had returned the results of their calculations. The persons that developed the scenario and had access to the experimental data were not directly involved in the model calculations.

Scenario for the comparisons[11]

The models were tested on a contamination scenario consisting of an aerial deposition of ^{137}Cs over a 500 km^2 area covered by forest. The deposition occurred within one day the 26 of April 1986. The total precipitation during this day was below 0.1 mm and thus it can be assumed that ^{137}Cs reached the forest mainly by dry deposition. The deposition density in the area at the end of the day was 850 kBq/m^2. The main characteristics of the contaminated forest are given below.

Soil characteristics

The main type of soil of the contaminated forest is soddy–podzolic loamy sand formed from fluvial-glacial sand accumulation. The soil density is 1.2 g/m^3. The main soil mineral is quartz, with a clay content of about 0.5 %. The soil has a low natural fertility, a high water permeability and a low water-holding capacity. The litter is of mor-moder type and has a thickness of 3.5 cm. The organic matter content in litter varies from 22.6 (Oh layer) to 43 % (Ol layer).

Trees

The prevailing tree species is pine (*Pinus silvestris*) with an average age of 50 years and a height of 25 m at the time of the deposition. The average volume of wood per m^2 was 0.019 m^3. The tree biomass follows approximately a logistic growth rate of 0.0002 y^{-1} reaching a maximum biomass of 14 kg/m^2 by the age of 100 y.

Understorey vegetation and game

The total biomass of the understorey is about 1.0 kg/m^2. Shrubs include rowan-tree (*Sorbus aucuparia*), alder black (*Alnus nigra*), buckthorn alder (*Fragula alnus*). The prevailing species of dwarf-shrubs are red raspberry (*Rubus idaeus*) and blackberry (*Rubus trivialis*). The main species of mushrooms are *Boletus edulis, Leccinum scabrum, Cantharellus cibarius* and *Russula* species. Grasses are rather sparse. Mosses cover 90 % of the territory and true mosses (*Bryales*) are the prevailing species. The main existing game species are moose (0.08 animals per km^2), wild-boar (0.18 animals per km^2) and roe deer (0.06 animals per km^2).

Variables used in the comparisons

The modellers were asked to make predictions on a yearly basis for the first 50 years after the contamination event for as many forest components as possible. The results presented by the modellers were then transformed into variables suitable for the comparisons (see Table 26).

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[10] It should be noted that part of the data used for parameterisation of FORESTLAND was collected in experimental plots situtated in Bryansk area.

[11] The data for the scenario were kindly put at our disposal by Drs S Fesenko and N Sancharova, RIARAE, and their assistance in formulating the scenario is very much acknowledged.

Table 26. Variables used in the model-model and model-data comparisons.

Variable	**Definition**	**Units**	**Models**
Litter	Fraction of initial deposition that is in the mosses and litter layers (mosses + Ol + Of + Oh)	Bq/m^2 per Bq/m^2	FOA, FRLD, LGN
Litter/soil	Fraction of total content in the soil-litter layer that is in the mosses and litter layers (mosses + Ol + Of + Oh per total in soil)	Bq/m^2 per Bq/m^2	FOA, FRLD, LGN, Exp. data available
Needles	Activity concentration in needles.	Bq/kg dry weight per $1 kBq/m^2$	FOA, FRLD, LGN, Exp. data available
Wood	Activity concentration in wood.	Bq/kg dry weight per $1 kBq/m^2$	FRLD, LGN, Exp. data available
Tree	Activity concentration in the whole tree, including bark, needles, branches and wood.	Bq/kg dry weight per $1 kBq/m^2$	FOA, FRLD, Exp. data available
Berries	Activity concentration in fruits of berries	Bq/kg dry weight per $1 kBq/m^2$	FOA, FRLD, Exp. data available
Moose	Activity concentration in meat of moose (average value in September)	Bq/kg fresh weight per $1 kBq/m^2$	FOA, FRLD

Results and discussion

Model-model comparison

For all variables there was a relatively good agreement between the predictions made with the models (Table 27).

Table 27. Reliability Indices (RI) showing the degree of agreement between the model predictions (K_{m-m}) and between the models predictions and the experimental data (K_{m-d}). A RI of 2 corresponds to an agreement within a factor of 2. The smaller the RI the better the agreement.

Variables	K_{m-m}	K_{m-d}
All variables	-	2.1
Litter	1.9	-
Litter/Soil	1.9	1.15
Needles	2.2	1.8
Tree	1.9	3.0
Wood	2.9	3.2
Berries	1.6	1.5
Moose	1.4	-

The best agreement between the different predictions was obtained for the variables Moose and Berries and the worst for the variable Wood. For all studied variables, except Moose and Tree, the agreement between the predictions decreased with time. There were also more pronounced differences in predictions of the form of the time dependencies than in predictions of the absolute values. One reason for this is the existing conceptual difference between the models, especially in the description of radiocaesium transfer in the tree and the soil. Consequently, there may be larger differences between predictions made with these models for other contamination scenarios.

Model-data comparison

Plots of the model predictions against the experimental data are shown in Figures 34 and 35. The experimental and predicted values for different variables had approximately the same ranking:

Litter/Soil > Berries ≥ Needles > Tree > Wood. The reliability index calculated using all model predictions and experimental data was 2.1 (Table 27), which can be considered to be a very good agreement for this type of comparisons. The best agreement was observed for the variable Litter/Soil and the worst for the variables Wood and Tree. The reliability indices for different models varied between 1.5 and 3, but the differences between the models were not statistically significant.

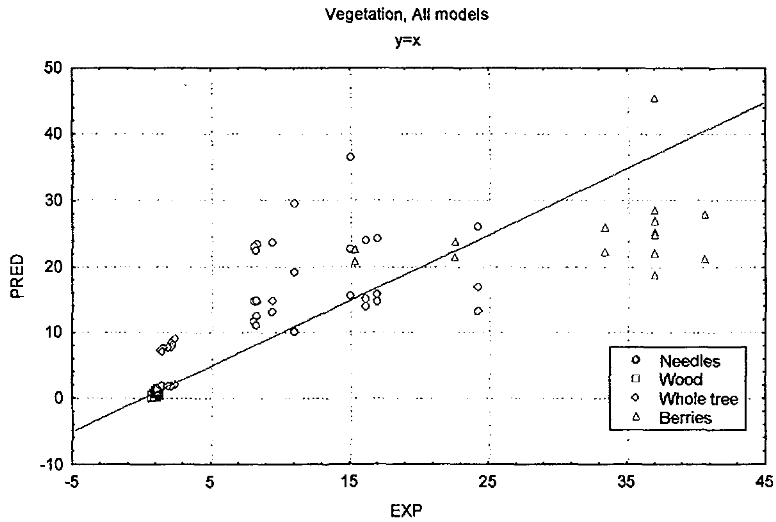

Figure 34. Experimental values versus model predictions for the variables Needles, Wood, Tree and Berries and for all three models.

The model predictions for all variables except for Berries were above the experimental values. The model overestimations of radiocaesium levels in the tree (Needles, Wood, Whole tree) may be explained by a slower migration downwards in the soil (higher values of the variable Litter/Soil) predicted by the models as compared to the experimental data. At the same time almost all predictions for Berries were below the experimental values. The roots of berries are probably located at the same level or above the tree roots and thus we also expected to obtain an overestimation by the models for Berries. The reason for this contradiction probably lies in the variability of the experimental data used in the comparison and for the calibration of the models.

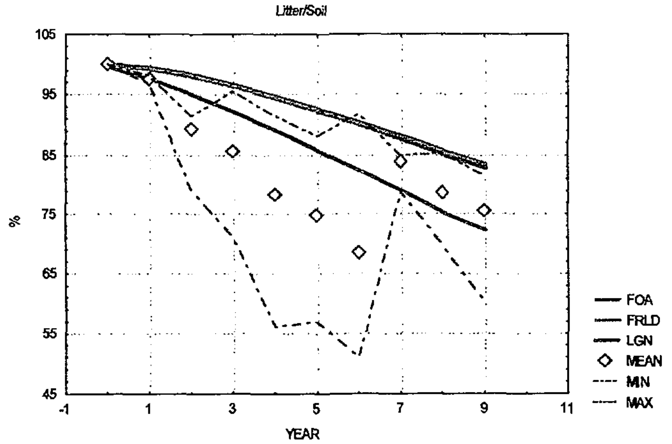

Figure 35. Model predictions and observed values of the fraction of the total radiocaesium content in the soil-litter layer that is in the mosses and litter layer (Litter/Soil). The dotted line corresponds to the interval of variation of the experimental values.

The agreement between the models as a rule decreased with time. This means that the agreement of the model predictions with "real" values obtained for times beyond the compared period (experimental data collected in the future) may be worse at least for some of the models. At the same time, as shown above, the agreement between the model predictions was within a factor of 2.9 during the studied period. Thus, if the predictions made with any of the models for future times (up to 50 years after the deposition) agree with the experimental data as well as for the first ten years (within a factor of 3) after the deposition, then it should be expected that the predictions with the two other models will agree with the experimental data at least within a factor of 10.

The comparison with the experimental data confirms that a main question that remains to be solved is whether the models predict correctly the kinetics of radiocaesium levels in different forest components. The experimental data used in the comparison did not allow to answer this question due to the high spatial variability as compared to the observed time variations (see Figure 35). Any similar data set collected with validation purposes will probably have the same problem. To obtain an answer to this question larger time series need to be used. Alternatively, field or laboratory experiments can be carried out to test the hypothesis behind the models and from this draw conclusions about the correctness of the time dynamics predicted.

Conclusions

The predictions made with the models for the studied scenario were in a relatively good agreement (within a factor of 1.4 to 2.9) for all studied variables. There was also good agreement between the model predictions and the experimental data (within a factor of 1.15 to 3.2). There were no significant differences between the studied models on how they reproduced the experimental data. This suggests that, at least for the studied scenario and for the first 10 years after the deposition, anyone of the models can be equally used if the final aim is to estimate absolute concentrations in different forest components. The agreement between the models, however, decreases with time and there were differences in the form of the time dependencies predicted by the models, especially for wood. This may lead to larger differences, although probably within a factor of 10, between the model predictions and the experimental data for times beyond the period for which data were available for comparison. From the experimental data (due to the high variability) it was not possible to discern which model predicts the kinetics best. Furthermore, examination of the models performance for predictions of the radiocaesium kinetic in different forest components and especially in trees is thus still needed.

Landscape models

Introduction

Geographic Information Systems (GIS) are software tools to represent and analyse ecological processes at landscape or territorial scale. Basic territorial data are expressed by means of maps per area unit, as geological characterictics, soil types, climatic data, vegetation types and productivity, social and demographic components, technological networks and so on. In this way, the landscape under study is read as a series of layers, each of them bringing a set of different information referred to the land unit area. Maps are input by digitizing, or scanning aerial photos or satellite images, and normally land units are classified by main components of each variable set. Numeric models are then applied to the basic classified data, each of them representing an input profile for the models; the results are one or more data sets for each system state variable taken in consideration, that are again processed by GIS software to generate new maps of calculated variables for territorial analysis and simulations in space and time.

The main components of GIS consist of:

- a series of basic (ancillary) data concerning the different environmental aspects of the area under analysis (physical, ecological, socio-economic), derived from cartography, or remote-sensing techniques, and organized as multi-layer spatial databases;
- one or more numerical models, combining and processing the basic data, thus producing index values or state variables in function of time per area unit;
- a series of maps from the previous, representing the evolution of the state variables or indexes in space/time.

This gives especially (i) estimation of particular variables at the present state, as function of different factors, (ii) evolution of the same variables with time, and (iii) simulation of alternative scenarios, whereas environmental factors undergo changes in spatial pattern and/or intensity.The GIS method is particularly advantageous for landscape analysis and management, when it is necessary to shift from system models to territorial models (i.e. to know about production or pollution within a defined area with time) and tools required for decision support.

Two applications of GIS methodology has been produced in the LANDSCAPE project. In the first one the dynamics and quantity of radiocesium in different compartments of the forest ecosystem has been predicted on a regional scale and at different time points. The second application has used information about radiocesium dynamics to predict the radiological consequences from the use of biofuels that come from a contaminated forest.

The two applications are fundamentally different but have a complimentary aspect. Both of them tend to answer questions that are related to the area management in case of accidents. The information obtained in the first study of dynamics can be used as a basis for policy makers and in decision making based on the radiological considerations. The second application has already resulted in a policy decision.

The GIS-application regarding ^{137}Cs dynamics in the forest ecosystem has been carried out in the NE Italian Alps. The second application regarding biofuels has been evaluated for a forested area in Sweden.

GIS application in N-E Italy

Problems and methods

The GIS application in Nort-Eastern Italy covers an area in the Tarvisio Forest in the Carnic Alps, region of Friuli-Venezia Giulia. The area is approximately 30x20 k m^2, and has the highest deposition from the Chernobyl accident among the Italian regions (20 to 40 kBq/ m^2).

The key steps of the GIS application were:

- to collect and arrange all the basic data from the area, both by existing maps and by remote-sensing techniques;
- to elaborate and classify the main vegetation types and green biomass classes;
- to apply the ^{137}Cs circulation model LOGNAT to these data;
- to produce mapped data of Cs% of the initial deposition in soil and vegetation on the whole test area, simulating the evolution for 50 years after time of deposition.

The basic GIS data included (a) a geological map, (b) DEM, Digital Elevation Model, (c) a slope map, (d) an aspect map, (e) a vegetation types map, and (f) a green biomass map.

The orography of the study area was first described by incorporating into the data base a DEM with 25 m resolution. From this, slope and aspect maps were derived. Information about vegetation types and net primary productivity (green biomass) were derived by analysis of remote-sensing data of Landsat TM images, no other ancillary data sources being available. Satellite data, with respect of traditional methods of vegetation mapping techniques, allow to cover large areas in short times and at reduced costs. The Landsat TM radiometer sensor detects the reflected radiation from earth surface in 7 wavelength bands, from the visible to infra-red, with 30 m of spatial resolution. Full images were acquired for the whole area for two days in July and October 1995, in order to have no clouded or shaded images at the highest vegetation productivity time of the season.

From the acquired images of the analysed area, a geo-reference in UTM was first cut out by software IDRISI, assuming control points as mountain peaks, crossroads, etc. This brought to the definitive cut of the area image, that was transferred to the ERDAS system format.

The vegetation map was obtained by classification of remote-sensing data. Twelve areas were identified along a N-S transect, from Mount Osternig (2052 m) to Cima Cacciatore (2071 m), and analysed in-field, representing to a major extent the environmental variability of the study area. The transect is characterised by a 1300 m altitude excursion, including mountain, subalpine and alpine vegetation layers, and different geological substrata from sedimentary and partially siliceous rocks to calcareous dolomite. Lithological and climatic variations along the transect determine different vegetation types, that human activity has modified mainly by grazing and forestry management. The vegetation analysis first produced 14 classes for the map classification, each of them characterized by a spectral signature, and these were further re-assigned into 8 definitive vegetation types, i.e.: (1) Beech woods, (2) Mixed woods, (3) Beech/Fir woods, (4) Fir woods, (5) Fir/Pine woods, (6) Grassland/meadows, (7) Urban/unvegetated areas, (8) No data.

Vegetation biomass (net primary productivity) was estimated for the vegetation types classified by means of the relationship between NDVI (Normalized Difference Vegetation Index, Rouse et al.,1974) and LAI (Leaf Area Index), that is the ratio of leaf area to soil area. LAI is proved to be linearly correlated to above-ground net primary production and stand volume growth of forests (Gong et al.1995).

Results

In LANDSCAPE, a general GIS methodology was used for estimating the ^{137}Cs circulation in a study area with time based on (i) generation of vegetation and productivity maps of the Tarvisio area; (ii) application of LOGNAT model to the forest types and biomass classes identified; and giving resulting maps of ^{137}Cs in the whole area per forest type and productivity class with time, simulated for different time periods (i.e. 1, 2, 10, 20 and 50 years after deposition).

The information given by the two maps were combined and synthesized. Vegetation classes were reduced by aggregating into three main forest types (i) deciduous forest, mainly beech type, (ii) coniferous forest, mainly spruce, fir and pine types, (iii) mixed forest, mainly beech-spruce-fir type. Other vegetation types such as grassland and crops were neglected, as were other classified areas (urban, water surface, bare rock). Results are indicated in Table 28 including the extension of each type and productivity class (hectares).

It is evident that mixed and coniferous forests cover most of the forested area, and pure beech stand are limited. Also, the productivity classes indicate that beech stands are generally more productive than fir and spruce, and mixed stands have intermediate values, as foreseeable. The stand productivity values ranges are 81-93 g/ m^2 (beech, average 88.9±1.9), 17-50 g/ m^2 (conifers, average 43.5±3.8), 27-72 g/ m^2 (mixed, average 63.5±5), indicating also great variability in stand density over the study area.

Table 28. Extension (hectares) of the productivity classes per main forest types in the Tarvisio area

PRODUCTIVITY CLASS (g/ m^2)	DECIDUOUS	CONIFEROUS	MIXED
0 - 20	0	25	0.63
20-40	0	1988	61
40-60	0	11274	2547
60-80	0	0	13592
80-100	1871	0	0

The input parameters of the LOGNAT model are (a) the deposition fraction in soil and vegetation compartments at deposition time, (b) the estimated standing biomass for each type (coniferous, deciduous, understorey), depending on the dominance of each type. No growth of vegetation was accounted for, as the whole forested area is in a mature stage (60 to 100 years). The results of modelling are expressed as dynamic curves for each forest type considered, describing the variation of the fractional content of ^{137}Cs of each compartment (litter, organic and mineral soil, total vegetation) with time. These are input parameters for GIS, whose final results (Figures 36 and 37) are:

- Map of the vegetation types of the Tarvisio forest area;
- Primary Productivity map of the Tarvisio forest area;
- ^{137}Cs in the total vegetation, expressed as % of initial deposition, 1, 2, 5, 10 and 50 years after deposition;
- ^{137}Cs in soil, expressed as % of initial deposition, 1, 2, 5, 10 and 50 years after deposition.

This methodological approach made it possible to estimate the ^{137}Cs content of the total area, expressed as average % of initial deposition on the land unit area, resulting from the average over all the pixels of the images. In this sense, current activity levels 13 years after the Chernobyl event could be reasonably estimated to about 10% for vegetation (3 kBq/ m^2) and 70% for soil (21 kBq/ m^2) in the whole area.

Conclusions

A general GIS methodology has been used to estimate the ^{137}Cs circulation in a study area of the Tarviso forest in Italy with time based on (i) generation of vegetation and productivity maps of the area. and (ii) application of the LOGNAT model to the forest types and biomass classes identified. The result is maps of ^{137}Cs in the whole area per forest type and productivity class with time, simulated for different time periods (i.e. 1, 2, 10, 20 and 50 years after deposition).

This made it possible to estimate the ^{137}Cs content of the total area, expressed as average % of the initial deposition on the land, and to estimate the activity levels 13 years after the Chernobyl accident to about 10% for vegetation (3 kBq/ m^2) and 70% for soil (21 kBq/ m^2) in the whole area.

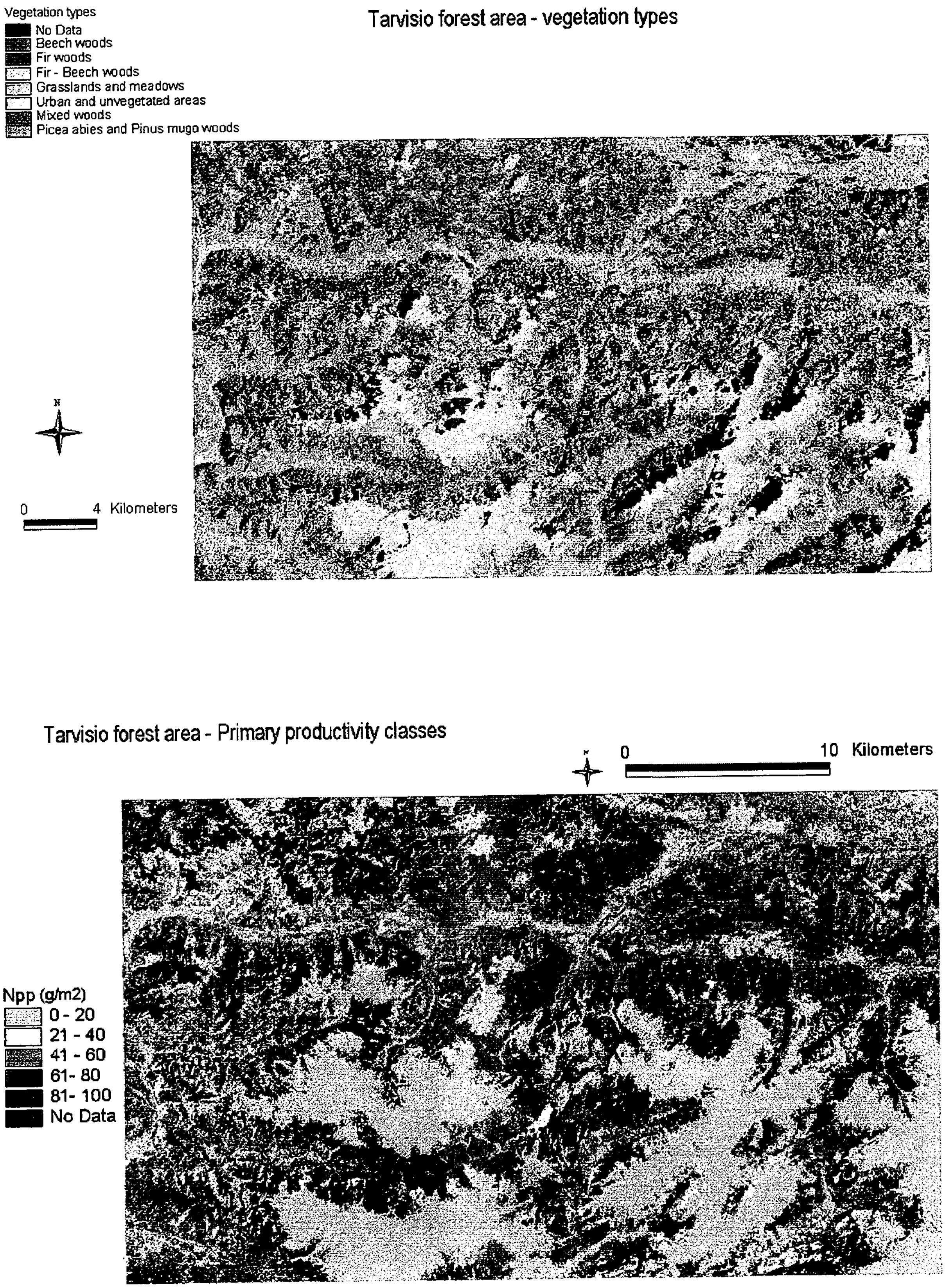

Figure 36. GIS vegetation types map (above) and net productivity map (below) of the Tarvisio forest area

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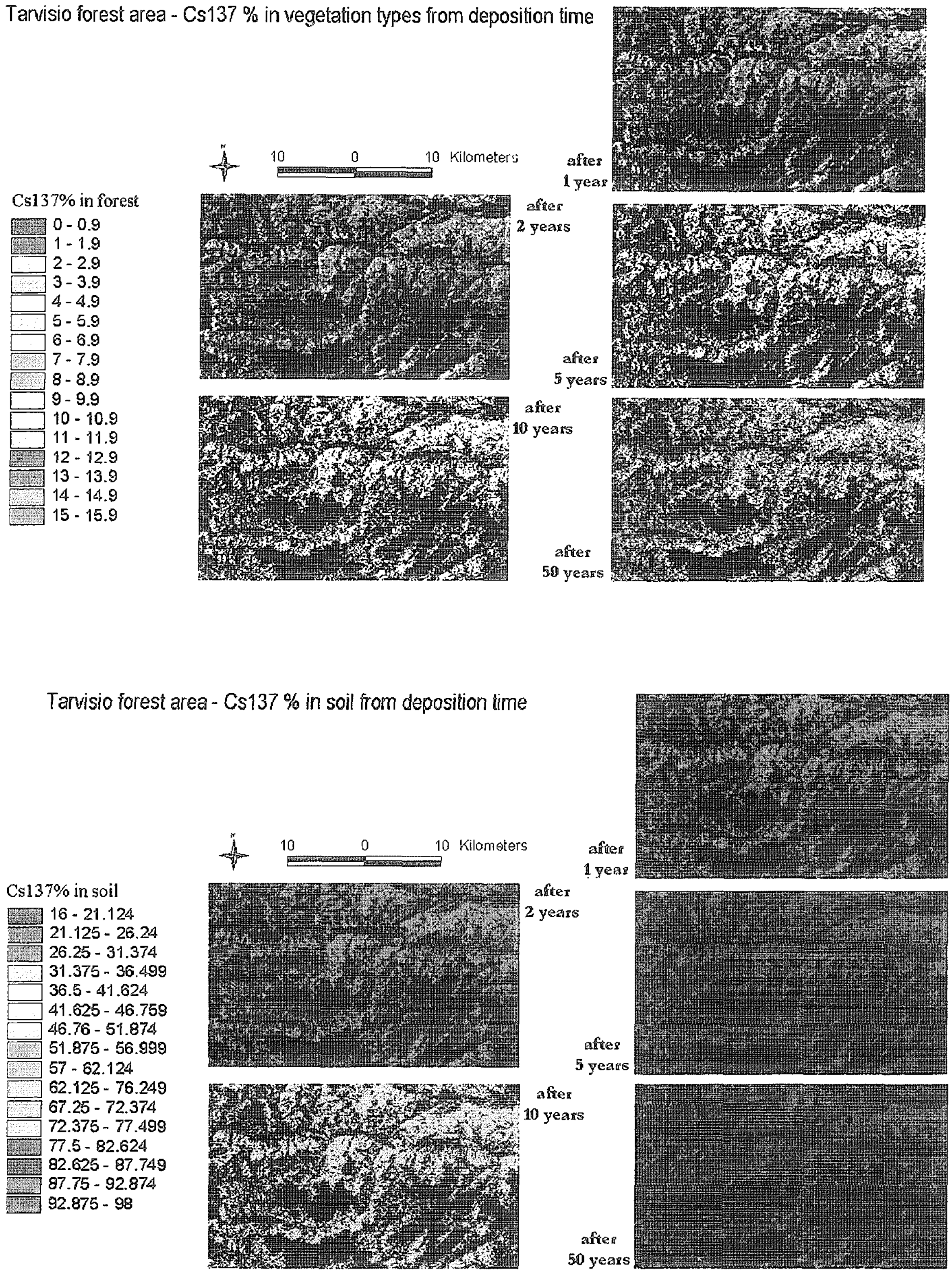

Figure 37. GIS maps of the estimated ^{137}Cs in % in vegetation (above) and soil (below) with time

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GIS application in Sweden - Occurrence of ^{137}Cs in forest biofuels

Introduction

The radiological consequences following energy production from the use of radioactively contaminated biofuels from forests has been investigated. The investigation has concentrated on the enrichment of ^{137}Cs in the waste products. The goal from the beginning of the investigation was to formulate a policy regarding radiation protection during the use of forest biofuels for energy production, and the subsequent handling of the waste products. The initial task was to fill the gaps in the existing knowledge. That need was countered by the general need for a policy concerning the radiological consequences from a growing industry in Sweden.

The use of biofuels for energy and hot water production has increased significantly in the past decade. Approximately twenty percent of the gross energy production in Sweden in 1996 came from biofuel powered plants, the majority (85%) of that amount came from forest biofuels. Two related factors have influenced the increase in the use of biofuels for energy production. One is the countrywide decision in 1980 by vote to eliminate nuclear power in Sweden. The other is that forest biofuels are renewable in Sweden, which contains 27×10^6 hectares of forest. That is enough surface area to successfully manage farming of the forests. Thus, forest biofuels in Sweden are renewable and their use comprises a natural recycling of the forest biomass, as it is taken from the forest, incinerated at the plant, and recycled back to the forest through the spreading of ashes for nutrition.

It is a well-known phenomenon that combustion can lead to waste products enriched in radionuclides. In those areas of Sweden that received the most fallout from Chernobyl, significant amounts of ^{137}Cs have transported to different parts of the forest ecosystem. The transport has occurred primarily through root uptake of ^{137}Cs that distributes to different parts of the tree. The highest concentration in the prevalent pine trees is in the bark, with decreasing concentrations respectively in the branches, needles and wood. The use of the needles, branches, and bark for biofuels is an increasingly common practice, with the wood left for lumber. Thus, the parts of the trees most concentrated in ^{137}Cs are often used for biofuel. It has been established that the concentration of ^{137}Cs in the trees in the Chernobyl-affected forests in Sweden is still increasing.

The radiological consequences to different members of the population depend on an estimation of both the internal and external radiation doses. In order to estimate those doses it is necessary to answer the following two questions with reasonable accuracy: (i) How much of the forested land area in Sweden will produce ashes from the biofuel combustion at different levels of ^{137}Cs contamination? (ii) How much land can be fertilised with the ashes produced at those different concentrations of ^{137}Cs?

GIS materials

These questions have been addressed using GIS maps and databases partially collected within the scope of the LANDSCAPE project. The ARC/INFO and ARCVIEW software have been used in this study. The maps containing the type of land use in the study area and the digital height map for the study area were obtained from the Swedish National Land Survey. Airborne gamma-ray spectrometric measurements of natural (K, U, Th) and other (such as ^{137}Cs) gamma-ray emitting isotopes have been obtained from the Swedish Radiation Protection Institute and the Geological Survey of Sweden. For the purpose of this study, these maps are used to give a quantitative estimation of the geographically distributed caesium deposition (see Figure 38).

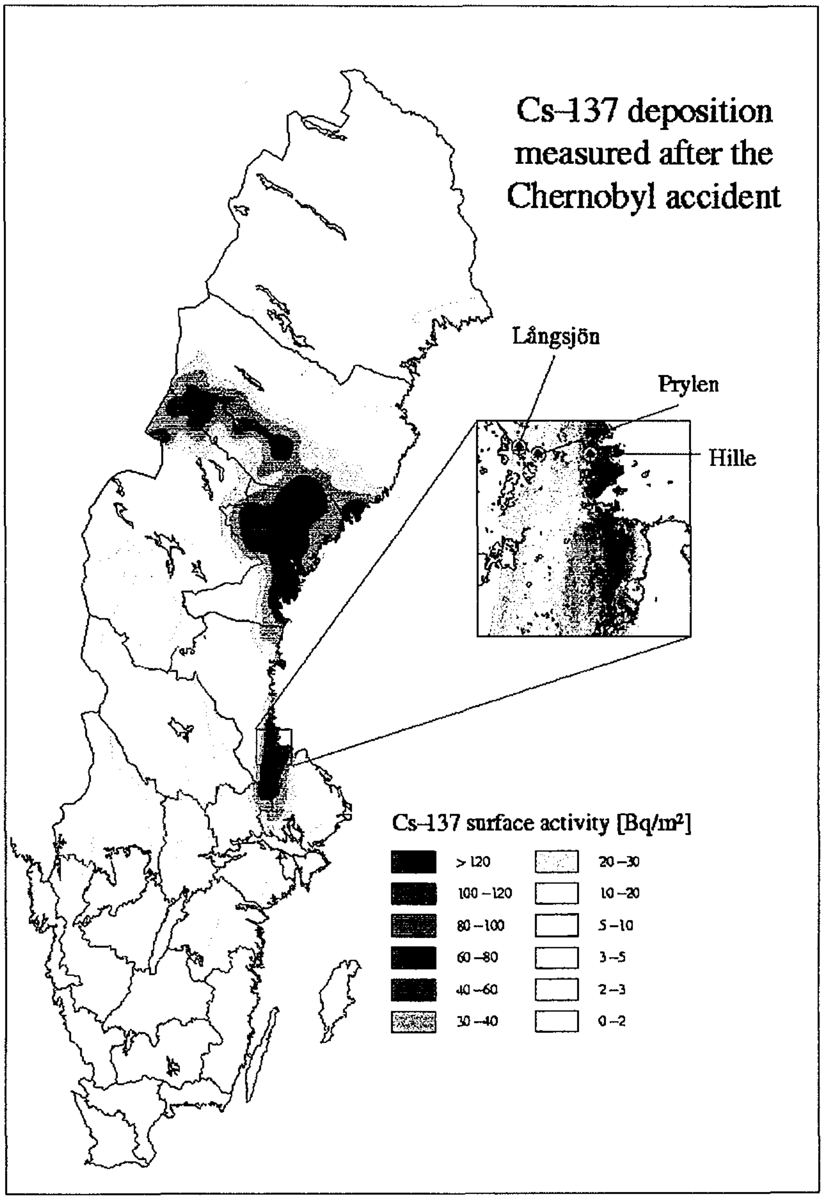

Figure 38. ^{137}Cs ground deposition in Sweden as measured by airborne measurements performed in May-October, 1986. The experimental sites of SSI are marked.

^{137}Cs content in the ashes of biofuels from forested areas in Sweden.

Wood, ash and soot samples from burning in single family dwellings were collected from different geographical co-ordinates in central Sweden and their ^{137}Cs content was measured and entered with their coordinates into a GIS database. This database was compared with a GIS map of the Chernobyl deposition over all of Sweden from airborne gamma-ray spectrometric measurements in 1986, which is shown in Figure 38. A surprisingly good linear correlation was found between the ^{137}Cs content in the ashes and the Chernobyl deposition on the forested land. No significant correlation was found however between the wood biofuel samples and the Chernobyl deposition.

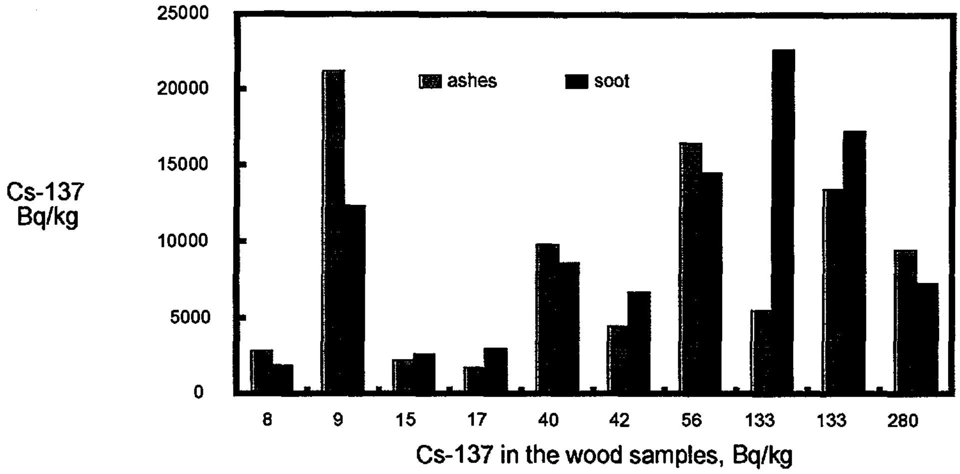

Figure 39. Burning of wood in homes: ^{137}Cs in wood vs ^{137}Cs in ashes and soot.

This can be partially explained with the use of Figure 39. The ^{137}Cs concentration in each of ten wood biofuel samples is shown on the x-axis. For each wood sample, the concentration of ^{137}Cs in the ash and soot sample from the same geographical location is shown on the y-axis. No convincing correlation can be seen between the ash and soot samples and the wood samples. It is known from the variety of measurements from studies on forest radioecology that wood samples have different ^{137}Cs concentrations that depend primarily on the age, type, and part of the tree measured as well as the soil conditions and the local ^{137}Cs deposition.

Samples of wood biofuels are thus random grab samples and it would require a statistically significant number of them from a given location before a correlation with deposition could be found. On the other hand, the ash samples, which contain a large number of the wood grab samples, have an integrating character that erases the dependencies on the other parameters. Figure 40 shows the linear fit of the ^{137}Cs concentration in the ashes as a function of the ^{137}Cs ground cover from the airborne gamma-ray spectrometric measurements from 1986. The calculated linear fit gives a relationship between ^{137}Cs in the ashes and the ^{137}Cs deposition from Chernobyl.

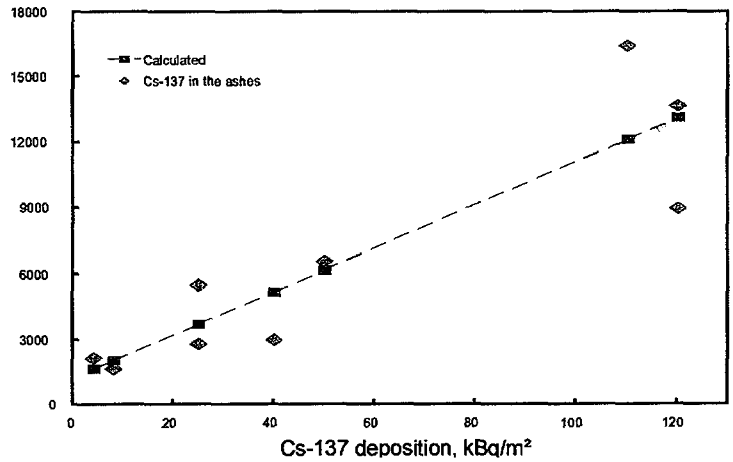

Figure 40. Linear fit of ^{137}Cs in the ashes as a function of ^{137}Cs ground cover from the airborne measurements performed in 1986.

This correlation has been used to predict, from a GIS map of all the forested land in Sweden and the map of the Chernobyl ^{137}Cs deposition over the whole of Sweden, which land areas in Sweden will produce ashes from biofuel burning which lie in different categories of ^{137}Cs-contamination. Figure 41 shows a local area of that map, Gävle city and county in Sweden. The Gävle area was one of the highest contaminated areas in Sweden from the Chernobyl accident.

An estimation of the radiation doses to various critical groups that would result from fertilising the forest with ashes of different ^{137}Cs content has been performed. These results have assisted SSI in a policy decision regarding the use of ashes from forest biofuels as fertiliser. Figure 42 shows a map of Sweden with the forested areas in red that are predicted to give ashes with ^{137}Cs concentrations over 5000 Bq/kg. The SSI policy states that over this limit the ashes should be deposited and not used as fertiliser.

The amount of ashes with more than 5000 Bq/kg that could occur with a fully developed biofuel activity are 15000 – 30000 ton ashes/a. This number takes into account that a maximum of approximately 10% of the forested land is forested each year. This allows for a constant regeneration of the forests. In 1996, 5 - 10 % of the maximum ash potential was produced, which gave 750 – 3000 ton ashes that could contain more than 5000 Bq/kg.

Conclusions

The radiological consequences following energy production from the use of radioactively contaminated biofuels from forests have been investigated. Based on GIS maps of the ground deposition of ^{137}Cs and the land use in combination with an empirical correlation between ^{137}Cs in ashes and the ground deposition, it has been possible to estimate the expected levels of ^{137}Cs in the ashes of biofuels from forested areas. In particular, a map showing the areas in Sweden giving rise to a ^{137}Cs concentration exceeding 5000 Bq/kg has been produced. The results of this investigation have also been used in the formulation of a radiation protection policy regarding the use of ashes from forest biofuels as fertilser.

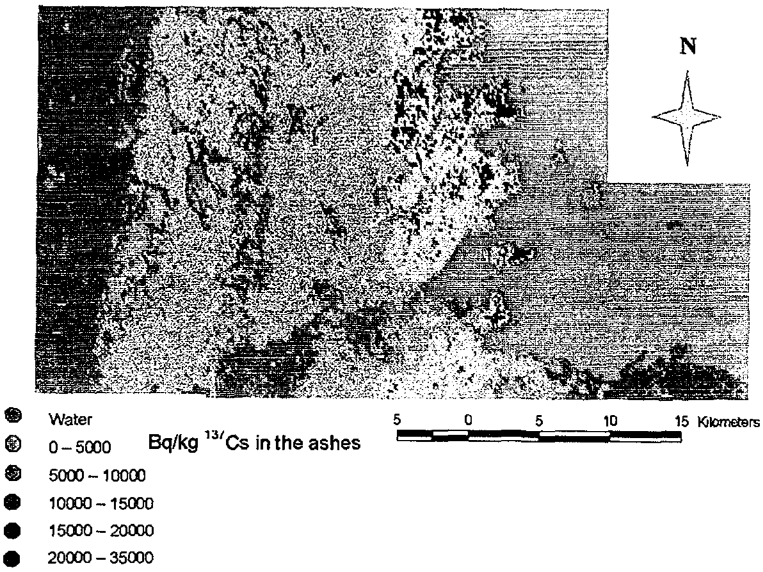

Figure 41. Gävle city and county, Sweden. Expected levels of ^{137}Cs (Bq/kg) in the ashes of biofuels from forested areas, determined from the ^{137}Cs deposition (Bq/m^2)

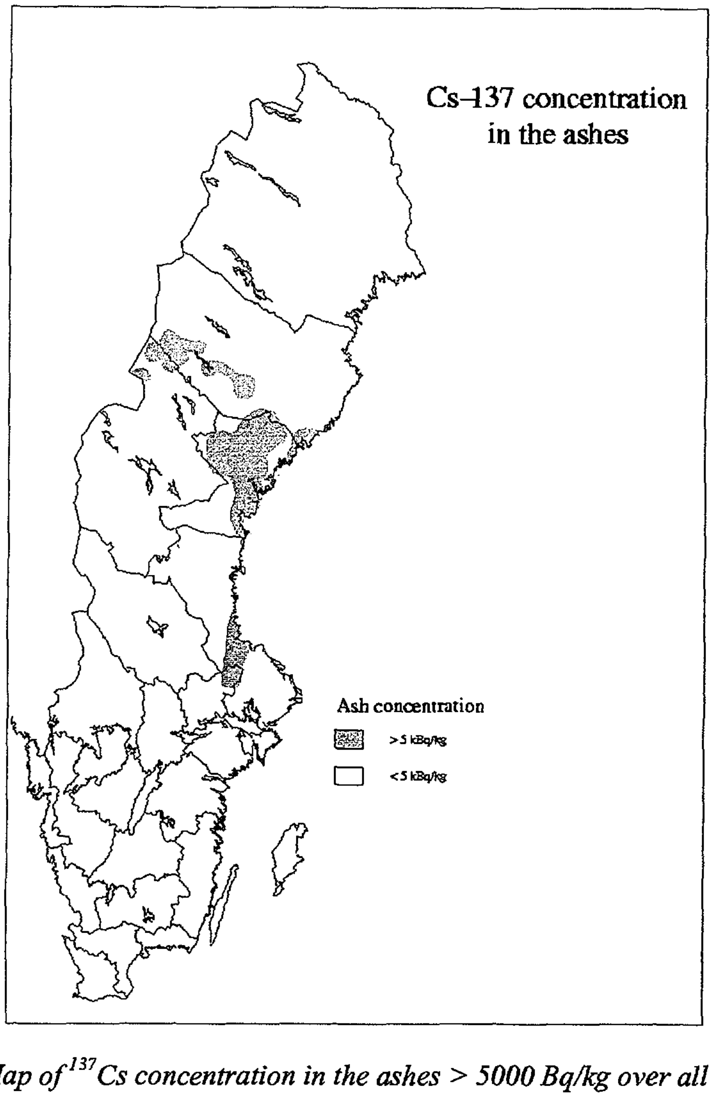

Figure 42. Map of ^{137}Cs concentration in the ashes > 5000 Bq/kg over all of Sweden.

LANDSCAPE – SUMMARY AND CONCLUSIONS

Main achievements and implications

The overall aim of the LANDSCAPE project has been to obtain a basis for forming reliable assessments of the radiation exposure to man under different time scales from radionuclides in plant and animal products of some representative semi-natural ecosystems in Europe. This aim was achieved by using both models and experiments. The main achievements and implications of LANDSCAPE, in terms of scientific findings, are in summarized below under the five groups of activities described in the Introduction of this report.

1. To review and evaluate selected data, and define data sets for use in the development of dynamic ecosystem and landscape models which describe the behaviour of radionuclides at different levels of ecological resolution.

- Three ecosystem models have been developed based on data sets from northern Sweden, north-eastern Italy and the Bryansk region in Russia. All models are based on compartment model principles and first order kinetics for the turnover of radioactive caesium in the forest ecosystem. However, different strategies have been chosen for describing the ecosystem and emphasis have been put on different transfer processes and different levels of ecological resolution.
- As part of the model development work, a literature review was performed in order to identify transfer processes which were included in existing models as well as factors not included.
- A system utilizing "interaction matrices" has been developed and used to describe the migration of radiocaesium, and the transfer processes, in the ecosystem, and to fascilitate the construction of conceptual models.
- All three ecosystem models (the FOA-model, the LOGNAT-model and FORESTLAND) calculate concentrations of radiocaesium in different compartments as endpoints, while FORESTLAND also includes a possibility to calculate internal and external radiation doses.
- Geographical Information Systems (GIS) are software tools to represent and analyse ecological processes at a landscape scale. Two applications of GIS methodology have been produced in LANDSCAPE.
- In the first one, GIS methodology was used to estimate the ^{137}Cs circulation in a study area of the Tarvisio forest, Italy, with time. The estimate was based on vegetation and productivity maps produced from satellite data, and on the application of the LOGNAT model to identified forest types and biomass classes. The results include maps showing the amount of ^{137}Cs in vegetation and soil expressed as percentage of the initial deposition at 1, 2, 5, 10 and 50 years after the deposition. The results also made it possible to estimate the present distribution of ^{137}Cs between soil and vegetation.
- In the second application, GIS methodology has been used as one instrument in the work to formulate a Swedish policy regarding radiation protection in connection to the use of radioactively contaminated forest by-products for energy production, and the subsequent handling of the waste products. GIS maps and databases on land parameters and ^{137}Cs ground deposition were used together with an empirically obtained correlation between ^{137}Cs in ashes and on the ground to create a final map over expected ^{137}Cs concentrations in ashes from biofuels. A map of those areas which would lead to concentrations in ashes exceeding 5000 Bq/kg was produced. The policy states that ashes with ^{137}Cs concentrations above this value should not be brought back to the forest as fertiliser.

2. To test model behaviours with data on radiocaesium from selected forest ecosystems in order to better predict the long term consequences of contaminated forest products, and the resulting radiation exposures, including estimates of the uncertainties involved.

- The three ecosystem models of LANDSCAPE were tested in a model-model comparison based on a real forest data from the Bryansk region in Russia. For some of the predicted end-points there were experimental data available for a model-data comparison.
- For the studied scenario, the model predictions were in a relatively good agreement (within a factor of 2.9) for all calculated endpoints. There was also good agreement (within a factor of 3.2) between the model predictions and the ten year experimental data series. There were no significant differences between the studied models in how they reproduced the experimental data. Thus, at least for the studied scenario and for the first ten years after the deposition, each one of the models can be equally well used to estimate concentrations in different forest components. The agreement between the models, however, decreases with time and there were differences in the form of the time dependencies predicted by the models. This may lead to larger differences, although probably within a factor of 10, between the model predictions and the experimental data for times beyond the period for which data were available for comparison. Due to the high variability in the experimental data, it was not possible to discern which model predicts the kinetics best. Further examination of the models performance for predictions of the radiocaesium kinetic in different forest components and especially in trees is thus still needed.

3. To measure relevant biological processes and links in selected semi-natural ecosystems in order to quantify the redistribution and reduction in time of radionuclides in plants and animals that constitute important exposure pathways to man.

- The dynamics of radioactive caesium in vegetation has been studied in field experiments carried out on sites in Finland, Sweden, and Belgium and include studies of ground vegetation, fungi, mosses and lichens, trees, as well as of litterfall and decomposition. The vertical and horizontal redistribution of caesium in soil has also been studied.
- Based on measurements of soil profiles and exposure measurements *in situ*, it can be concluded that there is a successive redistribution of ^{137}Cs in podzol soils slowly leading to an increased amount of cesium in the mineral soil horizon. The situation on peat soils is less clear and there is probably larger variations between sites. The change in vertical distribution of Cs-137 in soil have implications for plant root uptake, which depending on species can lead to either increased or decreased caesium concentrations.
- More than a decade after the Chernobyl deposition, the amount of ^{137}Cs leaving the boreal ecosystem by runoff is negligable except from the wetter fraction of mires. ^{137}Cs has now reached the ground water at 1 m depth, but this can be neglected as a discharge pathway.
- Even though the ^{137}Cs transfer from soil to plant differs in absolute values, in some cases by more than an order of magnitude, the temporal changes for some studied species are very similar at the boreal and temperate sites.
- Lichens and mosses are known to be suitable bioindicators for radioactive contamination. An interesting result of this study is the similarity in the long term ecological half-lives in boreal and temperate areas. The halflife is approximately 5 years.
- With a few exceptions, there was an overall reduction in ^{137}Cs concentrations in mushrooms between 1991 and 1998. When comparing the average activity in species usually characterised by a deep mycelium with those which have a more superficial mycelium, the results indicate a reduction of 60% and 80%, respectively, during this period. A rough assessment of the proportion of caesium retained in fruitbodies in their growing area shows that the average values in most mushrooms varied from 1.3 to 1.7%.
- The concentrations of radiocaesium in different parts of trees vary, and are changing with time. Ten years after the Chernobyl accident the total inventory of ^{137}Cs in Scots pine (*Pinus sylvestris*) on mineral soils are increasing. At one boreal site the inventory of ^{137}Cs has almost doubled from 1991 to 1997.
- The foliar ^{137}Cs concentration is highest in current needles at the beginning of growing season, and decreases along the ageing of the needles. The radiocaesium concentration follows inversely

the changes in needle dry mass. There was no significant temporal variation either in radiocaesium concentration or in content after the growing season for any of the age classes of needles. Sampling of tree fractions in radioecological studies should therefore be carried out during dormancy, when the concentration of the determined substance is not fluctuating.

- Of particular interest is the increase of ^{137}Cs in pine shoots. The increasing Tag values in young pines may be of significance for external doses; internal doses via meat from moose, for forestry in general (incl. biofuels) and the steady state levels for the different compounds in the forest ecosystem.
- The annual contribution from litterfall to the ground deposition is only a few tens of a percent of the total deposition. In the high deposition area, there seems to be an initial release (mobilisation) of ^{137}Cs, but then a substantial increase of the content of ^{137}Cs in the litter. This increase is probably caused by the interaction with fungi.

4. To quantify the influence of forest management, primarily fertilization, on the radiocaesium distribution, and to assess forest management as a possible countermeasure, or restoration technique

- When assessing the influence of the different forest ecosystems on the exposure of the human population to radiation, it is necessary to know how radionuclides cycle also in forest ecosystems that have been subjected to various management measures. The most common management measures are fertilisation, liming and site preparation. Potassium is probably the most interesting fertiliser affecting the distribution of radiocaesium in forest ecosystems. Generally, peat soils contain quite small amounts of potassium compared to the other main nutrients and compared to the amounts bound in the tree stands.
- The results show that dry mass accumulation can be increased by fertilisation, especially in stemwood. The results on a mineral soil site clearly show, that the reduction in caesium uptake cannot be explained with growth response alone, but that it is also related to root uptake. The growth response of trees to the fertilisation is normally observed in the growing season after the year of treatment. As to the timing of treatments in relation to Chernobyl fallout, the observed reduction in uptake of ^{137}Cs on the mineral soil may represent near to the maximum reduction that can be achieved with the amount of fertiliser used. However, the effect of fertilisation on the dry mass accumulation of vegetation will last only some years on sub-dry mineral soil sites and 15-20 years on peatland sites poor in mineral nutrients.
- The results indicate the benefits of fertilisation for restoration of contaminated forests in a severe fallout situation. The availability of timber to forest industry can be essentially increased with long term treatments of forests. Through multiple use of forests, pickers of wild berries and mushrooms, and hunters receive less radiocaesium through foodstuffs from fertilised forests than otherwise.

5. To quantify the impact of feed intake on temporal and spatial variation in animal populations and reduction in time of radionuclides in plants and free-ranging animals.

- It is expected that the radionuclide intake by herbivores would depend critically on the composition of the diet in terms of plant parts and plant species and the overall levels of environmental contamination as it varies over a range of spatial scales. The need to quantify diet composition is especially acute where herbivores ingest fungal fruiting bodies, which can contain very high concentrations of radiocaesium and other pollutants. The LANDSCAPE project has developed and applied methods to (i) measure the intake and diet composition of moose in the Boreal forest, and (ii) assess the use of space by moose and considers these in relation to likely patterns of deposition and distribution of radionuclides in vegetation.
- The application of controlled release devices to wild ruminants provides considerable opportunity, where the species composition of the diet is relatively simple, to measure intake of total dry matter

of food and hence intake of any of its chemical constituents, including radionuclides. The method has the advantage over previously used marker methods, that the animals need not be captured more than once for the dosing of the markers. Previous results have suggested that the behaviour of the dosed hydrocarbons in the digestive tract, is similar for all species of ruminant, and is similar to that of natural plant wax constituents.

- The validation experiments conducted with red deer and captive moose suggest that the methodology is equally applicable to ruminants ingesting diets of woody browse plants, in both summer and winter seasons. There was good quantitative agreement between the results of the hydrocarbon marker method, and measurements of diet composition by weighing, in the case of moose. There was also good agreement between the hydrocarbon marker method of intake estimation with detailed calibration of mass intake via browsing damage, in the case of red deer.
- N-alkanes are suitable dietary markers for birch in leaf, and although the concentrations in twig material are low, they are sufficient to be used as markers in winter, when leaves are absent. Scots pine did not contain significant amounts of n-alkanes; hence the use of long-chain fatty alcohols is suggested when this is a likely dietary constituent.
- Because of the negligible recovery of ergosterol itself in the faeces of goats, it is unlikely that it can be used as a suitable marker of dietary fungi in ruminants. The ergosterol is presumably being absorbed or transformed biochemically, in the rumen or elsewhere in the digestive tract. The possibility of using metabolites of ergosterol in faeces as an alternative qualitative marker for dietary fungi in ruminants, requires further investigation
- On the basis of the results, the winter diet of both moose bulls and cows is dominated by Scots pine. In contrast the summer diet was dominated by broadleaved tree species, primarily birch (*Betula spp.*). Some samples showed evidence that heather (*Calluna vulgaris*) was present in the diet, but the levels were such that it was unlikely to have significant radioecological consequences at the population level.
- Overall, it is unlikely that variation in diet composition of moose in late summer and autumn, prior to and during the hunting season, is a major determinant of variation in radionuclide uptake. The exception to this remains the ingestion of fungal fruiting bodies, which can lead to significant uptake of radionuclides, but the ingestion of which is unpredictable due to their ephemeral temporal and spatial distribution.
- It seems likely that the greatest variability in radionuclide uptake by moose is brought about by spatial variation in deposition, in conjunction with normal movements of the animals rather than being due to variations in diet selection or intake *per se.*
- The results show that while it is possible to make useful predictions about radionuclide exposure for regional areas, great care should be taken before interpolating the results. The factors that operate at the population level are largely irrelevant when discussing the exposure at an individual level. To understand this one must also begin to explore individual-based models of intake. A limited step has been taken in this direction, but it is believed that an integrated approach, combining basic ecological facts with geostatistical and individual based models give new insights to radioecology.

Exposure to man

The experience after the Chernobyl accident has shown that the forest can give a substantial part of the total radiation exposure to humans.

The LANDSCAPE project has been focussed on creating a basis for forming reliable assessments of the radiation exposure to humans. Less effort has been put into estimating radiation doses. In principle, knowing the concentrations of ^{137}Cs in relevant compartments of the ecosystem and the changes with time, it is fairly straightforward to calculate the internal doses. However, assumptions have to be made concerning consumption habits. In a similar way, knowing the radiation field in the forest, the external doses can be calculated with assumptions on times spent in the forest and the time of the year.

In LANDSCAPE dose estimates have been made in connection with the study of biofuels. The time variation of the radiation field in forests, expressed as photon fluence, have also been estimated and can be used to estimate external doses.

Future research

A research project normally generates new questions and proposals for future research. Some examples resulting from the LANDSCAPE project are listed below. A general comment is, that the huge amount of information and data collected in LANDSCAPE is not yet fully analysed. It is the intention to further analyse these data and to publish the findings in scientific journals.

1. Vegetation and forest management

- The dynamics of the ^{137}Cs circulation in the forest ecosystem emphasis the need for a continued study of the long term changes in the the transfer of ^{137}Cs, especially for the exposure pathways that include man.
- The soil processes influencing the radionuclide uptake should be examined in a systematic way to be able to model the mechanism of radionuclide uptake from forest soil to plants, and particularly to trees.
- A system for defining and experimentally measure the bioavailability of ^{137}Cs in plants would improve the understanding of the transfer processes.
- There is a need to continue the investigations on the role of mycorrhizae in the radiocaesium transfer from the soil to host plants.
- The results of the fertilisation study motivate further research on the practicability of forest management methods as countermeasures in a fallout situation. Such methods open possibilities to follow the principles of sustainable forestry, as the treatments do not change ecosystem in a radical way.
- The influence of forest management on the radiocaesium distribution in the forest should be tested for other representative sites and thereby show the practicability of such methods in European forests.
- A systematic expert evaluation of all experimental data from the radioecological forest research after 1986 could give an input to further work.

2. Herbivores

Further research needs to concentrate effort on the period immediately prior to and during the hunting seasons of large ruminants, usually in the late summer and autumn. This period is the most relevant to the transfer of radionuclides to meat.

- Further research needs to be undertaken on the calibration and testing of the marker methods for quantifying diet composition and intake, in the crucial period.
- Fungi are important contributors of radionuclides to the diet of free ranging ruminants, in this crucial period. Further work is required on the identification and development of fungal markers to determine their contribution to diet composition and intake. Because of the concentrations of ergosterol in fungi, investigation of its degradation products may represent a suitable starting point in the search for suitable markers.
- Further research should focus on the movements and distribution of moose immediately prior to and during the hunting season, in relation to likely spatial patterns of deposition and distribution of contamination.
- Experimental determination of transfer coefficients of radionuclides from natural forage to the meat and other tissues, and their half life in the wild ruminants being studied, is required. This would improve the accuracy of prediction of contamination, over estimates used in current models which are derived from studies of domestic ruminants.

- The approach to measure the food intake of wild ruminants developed and applied in the LANDSCAPE project could be applied to study transfer to man of other pollutants such as less common radionuclides or heavy metals.

3. Modelling

- Even though the results of the testing of ecosystem models in LANDSCAPE were encouraging, there is a definite need to further test the models against real data in the long term phase. For example, the prediction of the three models in LANDSCAPE diverged with time.
- A search for independent data sets available for model testing should continue.
- The use of GIS methodology to improve the ability to vizualize predictions in time and space need further work.
- The possibility and need to model other contamination pathways and contaminants should be investigated.

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ANNEX 1

Table 1. Some characteristics of experimental forests, and information on sampling of different tree fractions.

Site	Juupajoki	Ruovesi	Hille	Långsjön	Prylen	Siikakangas	Liesineva
Country	Finland	Finland	Sweden	Sweden	Sweden	Finland	Finland
Location	61°51'N, 24°19'E	61°52'N, 24°13'E	60° 48 "N, 13°E	60° 48 "N, 17°E	60° 48 "N, 17°E	61°52'N, 24°13'E	61°59'N, 23°15'E
Dominating tree species	Picea abies	Pinus sylvestris	Pinus sylvestris	Pinus sylvestris 58% Picea abies 38%	Pinus sylvestris	Pinus sylvestris	Pinus sylvestris
Altitude, m above sea level	177	154	25	75	70	154	152
Geobotanical region	Southern boreal	Southern boreal	Southern boreal	Southern boreal	Southern boreal	Southern boreal	Southern boreal
Temperature sum above 5°C, d.d.	1106 (1996)	1106 (1996)	1250	1250	1250	1396 (1997)	1304 (1997)
Growth period (mean dayly temperature above 5°C)	1.5.-12.10. (1996)	1.5.-12.10. (1996)	15.5.-15.10.	15.5.-15.10.	15.5.-15.10.	1.5.-12.10. (1996)	1.5.-12.10. (1996)
Annual precipitation, mm	615	615	617	617	617	615	700
Precipitation June-September, mm	243 (1996)	243 (1996)	200-400 June-August	200-400 June-August	200-240 June-August	274 (1997)	293 (1997)
Stand age, year (1999)	53	68	79	22	11	63	58
Tree density, trees ha^{-1}	893	381	940	2100	2300	488	797
Dominant tree height, m	21.9	22.0	20.6 (1997)	5.0 (1996)	2.59 (1996)	21.3 (1997)	15 (1997)
Diameter at brest height, cm	22.8	24.8	23.0 (1997)	9. 1 (1996)	3.7 (1996)	22.6 (1997)	14.7 (1997)
Soil type	Ferric podsol	Ferric podsol	Ferric Podsol	Ferric Podsol	Podsol	Ferric podsol	Peatland
Clay fraction, %	n.d.	n.d.	1.9	4.0	3.4	2.6	n.d.
Thickness of humus layer, cm			4.8 (+- 1.03)	6.1 (+- 1.7)	4.5 (+- 2.9)	2.9-5.8	7.2-8.6
Type of humus	Mor	Mor	Mor	Mor	Mor	Mor	Raw humus
Sampling period	1.4. 1996-28.2.1997	1.4.1996-28.2.1997	29.10.1990-30.5.1999	10.8.1991-30.5.1999	10.8.1991-9.11.1998	8.10.-19.11.1997	8.10.-19.11.1997
Number of sampling occasions	18	18	20	20	21	1 per fraction	1 per fraction
Sample types analysed	Needles	Needles	Bark, Wood, Branches, Needles	Bark, Wood, Branches, Needles	Bark, Wood, Branches, Needles	Bark, Wood, Branches, Needles, Stumps, Roots	Bark, Wood, Branches, Needles, Stumps, Roots

Table 1. Some characteristics of experimental forests, and information on sampling of different tree fractions..continued

Site	Vindeln	KD981, Vindeln young forest	KD984, Vindeln young forest	Kd985, Vindeln mature forest	Kd988, Vindeln mature forest
Country	Sweden	Sweden	Sweden	Sweden	Sweden
Location	64°14'N, 19°46'E	64°14'N, 19°46'E	64°14'N, 19°46'E	64°14'N, 19°46'E	64°14'N, 19°46'E
Dominating tree species	Pinus sylvestris	Pinus sylvestris, Betula spp and Salix spp	Pinus sylvestris, Betula spp Salix spp	Pinus sylvestris 50% Picea abies 50%	Pinus sylvestris
Altitude, m above sea level	175	200-260	270-290	220-225	215-220
Geobotanical region	Boreal	Boreal	Boreal	Boreal	Boreal
Temperature sum above 5°C, d.d.	868 (1995)	868 (1995)	868 (1995)	868 (1995)	868 (1995)
Growth period (mean dayly temperature above 5°C)	20.5.-26.9. (1995)	20.5.-26.9. (1995)	20.5.-26.9. (1995)	20.5.-26.9. (1995)	20.5.-26.9. (1995)
Annual precipitation, mm	552 (1995)	552 (1995)	552 (1995)	552 (1995)	552 (1995)
Precipitation June-September, mm	200 (1995)	200 (1995)	200 (1995)	200 (1995)	200 (1995)
Stand age, year (1999)	40-70	<15, (1998)	<15, (1998)	>70, (1998)	>70, (1998)
Tree density, trees ha^{-1}	1250			Ca 1000 (1998)	Ca 600 (1998)
Dominant tree height, m	8.4 (1986)	Ca 15 (1998)	Ca 15 (1998)	Ca 20 (1998)	Ca 20 (1998)
Diameter at brest height, cm	15 (1986)				
Soil type	Regosol	Podsol	Regosol	Podsol	Podsol
Clay fraction, %	<11	<9	<7	<5	<4
Thickness of humus layer, cm	2	4	2	5	5
Type of humus	Mor	Moder	Mor	Mor	Mor
Sampling period	1.6.1986-25.8.1996	06.86-08.98	06.86-08.98	06.86-08.98	06.86-08.98
Number of sampling occasions	15 (needles), 2 (other fractions)	12	12	12	12
Sample types analysed	Bark, Wood, Branches, Needles	Current shoots and needles	Current shoots and needles	Current shoots and needles	Current shoots and needles

Table 2. Soil properties on experimental sites in Sweden and Finland

Site	Material	Treatment	Sampl. time	AVERAGE org %	Clay Content, %	bulk density g/cm3	pH CaCl2	N tot g/kg	P extr. mg/kg	K extr. mg/kg	Ca extr. mg/kg	Mg extr. mg/kg
Hille	org-layer	0	-98	58,21	n.d. [a]		3,53	9,51	175	710	1890	356
	mineral soil 0-20 cm	0	-98	1,96	1,9		4,22	0,47	6	18	64	13
Långsjön	org-layer	0	-96	43,75	n.d.		3,63	7,37	132	382	1904	254
	mineral soil 0-20 cm	0	-96	5,02	3,7		4,07	0,95	15	23	210	17
	mineral soil 0-20 cm	0	-91	5,21	4,3		3,92	0,99	16	28	239	19
Prylen	org-layer	0	-96	53,62	n.d.		3,71	9,64	210	536	3539	342
	mineral soil 0-20 cm	0	-96	5,13	3,4		3,86	1,03	10	34	329	37
Liesineva	humus	0	-97	95,59	n.d.	0,073	3,20	16,30	351	853	2381	533
	peat 0-20 cm	0	-97	94,71	n.d.	0,134	3,27	22,58	38	126	1228	135
	humus	PK+K	-97	95,61	n.d.	0,085	3,18	15,83	328	872	2956	457
	peat 0-20 cm	PK+K	-97	95,69	n.d.	0,127	3,19	24,16	62	172	1280	174
Siikakangas	humus	0	-97	59,12	n.d.	0,168	3,08	8,00	106	433	969	148
	mineral soil 0-20 cm	0	-97	6,82	2,4	0,995	4,06	1,32	14	31	38	9
	humus	NPK+NPK	-97	44,50	n.d.	0,284	3,30	7,76	111	347	1004	290
	mineral soil 0-20 cm	NPK+NPK	-97	6,90	2,7	0,976	3,93	1,64	21	53	75	30
Umeå	humus	0	-98	66,83	n.d.		3,41	9,03	236	741	2240	327
(mean of 4 sites)	mineral soil	0	-98	5,22	missing		4,36	0,96	9	25	234	42

a) n.d. means: not defined

SSI-rapporter 1999

SSI reports 1999

99:01 Publikationer 1998
Statens strålskyddsinstitut

99:02 SSI:s projekt avseende avveckling av kärntekniska anläggningar – en förstudie
Avdelningen för avfall och miljö, Avdelningen för personal- och patientstrålskydd, Administrativa staben.
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99:03 Föreskrifter om skydd av människors hälsa och miljön vid slutligt omhändertagande av använt kärnbränsle och kärnavfall -bakgrund och kommentarer
Avdelningen för avfall och miljö

99:04 Calibration in Medical Diagnostic Beams at the Swedish Secondary Standard Dosimetry Laboratory
Avdelningen för miljöövervakning och mätning.
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Department of Waste and Environmental Protection
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99:06 Personalstrålskydd inom kärnkraftindustrin
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