nks

NKS-B FOREST Seminar, Helsinki, 7-8 October 2008. Proceedings.

Virve Vetikko (1), Aino Rantavaara (1), Lasse Aro (2),
Agneta H. Plamboeck (3) and Elisabeth Strålberg (4) (editors)

(1) Radiation and Nuclear Safety Authority (STUK), Finland
(2) Finnish Forest Research Institute (METLA), Finland
(3) The Swedish Defence Research Agency (FOI), Sweden
(4) Institute for Energy Technology (IFE), Norway

nks

Abstract

The seminar *"Towards improved understanding of radionuclide transfer in forests and preparedness to handle contaminated forests"* was carried out within the framework of the NKS-B Forest project in Helsinki, 7–8 October, 2008. The seminar was planned and arranged by four Nordic organisations and provided a forum for exchange of information for Nordic scientists currently working in the field of forest radioecology or using the data. Presentations of research on nutrient cycling and radionuclide distribution in boreal forests, discussion on the needs for future research and attendance of experts on forestry, forest research and radioecology offered a unique opportunity to disseminate and receive information. The seminar programme was composed of topics of radioecology and forest research, assessment of radionuclide contamination and management of contaminated forests. Also sampling in forests, monitoring and modelling of environmental impact of disposal of spent nuclear fuel, and recent radioecological studies on forests were handled. Future research was emphasised in discussions. Below is a short compilation of these discussions:

– Comprehensive planning of research projects contributing to a programme with long-term aims is possible in broadly-based multidisciplinary collaboration. Thereby independent initiatives and less coherent plans can be replaced.

– Clear definition of hypotheses, planning and improving the specific methods for sampling and laboratory analyses were found crucial.

– Effects of intensive biofuel harvesting on the nutrient and radionuclide flows in forests are major issues in the next decades.

– Gaps in knowledge, such as lack of data on processes contributing to radionuclide distributions in forests, and on certain long-lived radionuclides (those of Cl, Tc, Np, etc.) contributing to the environmental impact of final disposal of spent nuclear fuel.

Topical seminars like the one accomplished are welcome in the future; compiling acute issues of multidisciplinary nature for focussed expert review and discussion can be very rewarding in many ways. To support communication and future collaboration, an informal forest network was launched in the seminar. Information is available via e-mail (forestnetwork@stuk.fi).

Key words

Caesium, element distribution, forest management, mineral nutrients, monitoring, modelling, mushrooms, nuclear waste, radionuclides, transuranium elements, wild berries

NKS-B FOREST Seminar

Towards improved understanding of radionuclide transfer in forests and preparedness to handle contaminated forests

7–8 October 2008
STUK, Helsinki

PROCEEDINGS

Edited by

Virve Vetikko and Aino Rantavaara
Radiation and Nuclear Safety Authority (STUK), Finland

Lasse Aro
Finnish Forest Research Institute (Metla), Finland

Agneta H. Plamboeck
The Swedish Defence Research Agency (FOI), Sweden

Elisabeth Strålberg
Institute for Energy Technology (IFE), Norway

Foreword

NKS-B Forest seminar titled *Towards improved understanding of radionuclide transfer in forests and preparedness to handle contaminated forests* was held on 7–8 October 2008 at STUK, in Helsinki, Finland. The seminar was organised and funded by the Nordic Nuclear Safety Research (NKS), Radiation and Nuclear Safety Authority (STUK), Finland, Finnish Forest Research Institute (Metla), Institute for Energy Technology (IFE), Norway and the Swedish Defence Research Agency (FOI).

Members of the Programme Committee were Lasse Aro (Metla), Gordon Christensen (IFE), Agneta H. Plamboeck (FOI), Torbjörn Nylén (FOI), Aino Rantavaara (STUK), Elisabeth Strålberg (IFE) and Virve Vetikko (STUK).

Chairpersons of the seminar sessions were Professor Leena Finér (Metla), Professor Hannu Ilvesniemi (Metla), Professor Jukka Lehto (Laboratory of Radiochemistry, University of Helsinki), Associate Professor Klas Rosén (Swedish University of Agricultural Sciences – SLU) and Dr. Dina Solatie, Head of Laboratory (STUK).

The submission of papers from the speakers to this summary report was voluntary, and therefore not all presentations in the programme (Appendix 1) are included. There were 25 oral presentations and the seminar involved in all 37 participants from Finland, Norway and Sweden, representing research institutes, universities, authorities and forest industry (Appendix 2).

The editors of the report acknowledge the support from the NKS-B programme manager, Dr. Justin Gwynn, to the seminar at all stages. We also acknowledge the funding provided by NKS and the Nordic institutes involved. We warmly thank all participants and chairpersons for their inspiring contribution and for giving their time.

Contents

1 Introduction

The NKS-B FOREST seminar was held on 7–8 October 2008 at STUK, in Helsinki, Finland. The seminar was arranged to provide a forum for discussions and sharing updated information on radionuclide transfer in forests and on methods for mitigation of harms from radionuclide contamination to forestry. Sampling in forests for radionuclide analyses was emphasised in the programme, as a new guidance was ready for presentation to the seminar audience as a deliverable of the NKS-B project FOREST. The seminar was aimed to strengthen the Nordic collaboration by offering a platform for scientists and other interest groups to meet and discuss radioecology. The organisers also wished to inspire participants from the different organisations to collaborate in the radioecological field. The programme of the seminar covered the following topics:

– Radioecology and forest research – basis for preparedness to manage contaminated forests
– Sampling in forests, presentation of the sampling guidance prepared under the NKS/FOREST project
– Management of forests contaminated by radionuclides
– Assessment of environmental impact of final disposal of spent nuclear fuel, and
– Recent and ongoing research on radionuclides in forests.

In the opening speech Professor *Sisko Salomaa,* the Research Director of Radiation and Nuclear Safety Authority (STUK), introduced the goal of sectoral collaboration between people in governmental organisations in Finland. Collaboration between scientists at the Forest Research Institute (Metla) and STUK provides a functioning example, a coherent radioecological research programme where two institutes have joined their specific fields of expertise and contributed to sectoral research for almost 20 years. Professor Salomaa's words supported the basic aims of the seminar, and confirmed the fact how complementary expertise can facilitate clarifying and reaching the research objectives.

Professor *Hannu Raitio*, Director General of the Finnish Forest Research Institute, underlined in his opening remarks the importance of forest radioecology research, and the need for practical instructions to forestry professionals and forest owners on radiation safety issues. Metla's long-term field experiments have provided valuable knowledge on the effects of forest management measures on radiocaesium contamination of forest vegetation and on the behaviour of radiocaesium and strontium in forest ecosystems. However, further research on radionuclide behaviour in forests, e.g. on the influence of nuclear waste disposal on forest ecosystems, is urgently needed to provide solid basis for practical measures.

In his introduction to NKS-B programme, Dr. *Justin Gwynn,* the NKS-B programme manager since the beginning of 2008, described various types of activities and the administration of the programme. He completed the information in the closing session of the seminar by giving detailed instructions for application of project funding from NKS.

This summary report includes the papers submitted by the speakers and conclusions made by chairpersons and the editors. The report gives a good summary of where we are at the moment and can serve as a base for future work in the area.

2 Radioecology and forest research – Basis for preparedness to manage contaminated forests

2.1 Uptake of radionuclides with focus on cesium

Agneta H. Plamboeck

FOI, Swedish Defence Research Agency, CBRN Defence and Security,
SE-901 82 Umeå, Sweden

Introduction

Sweden belongs to an exclusive group of nine countries in the world with a forest cover greater than 65% of the land area. Forest products make a large proportion of Sweden's net export income. Not only wood fibres but also berries and meat from the forests are important. Game hunting, reindeer farming, berries and mushroom gathering, as well as fresh water fishery, are all included in human food chains that are likely to be affected if the forest ecosystem is contaminated by radioactive nuclides. Another main reason for focusing on uptake and availability of radio nuclides in the forest ecosystems is the safe radiation environment, an environmental goal decided by the Swedish Government (Prop. 2000/01:130). This means that both people's health and the biological diversity should be protected against harmful effects from ionizing radiation.

The Chernobyl fallout provided an excellent possibility to study the uptake, redistribution and retention in conifer trees of ^{137}Cs, since the deposition lasted for only a few days. The average deposition of ^{137}Cs in the region that originates from the Chernobyl accident in 1986 was 20 ± 9 kBq m^{-2}. Also ^{137}Cs from the atmospheric nuclear weapons tests was present in the area and was only 3 ± 2 kBq m^{-2} (reference date 860429, Nylén and Grip 1997).

In the first study presented here was the ^{137}Cs distribution in a boreal forest studied using a traditional sampling method compared with the use of a portable NaI-detector connected to a geographical information system. Field-portable detectors have been used frequently in routine monitoring and in hazard assessment studies. However, there have been few attempts to carry out a careful evaluation of their potential as an alternative to sample collection and subsequent laboratory analysis. The objectives were i) to determine the ^{137}Cs activity in recharge areas, discharge areas, shrub mire and open mire, ii) to map the geographical distribution of ^{137}Cs by means of the portable NaI detector connected to a GIS system and iii) to identify ^{137}Cs anomalies in the catchment. The study used two different methods, the portable NaI detector (GDM-40) and soil samples measured by gamma-ray spectrometry at the laboratory.

In the second study was temporal changes in ^{137}Cs distribution in a Scots pine (*Pinus Sylvestris* L.) stand studied during 1986 to 2006 in Northern Sweden. In the third study was root uptake of Cs and Na studied under two different water regimes in the same Scots pine forest as the second study was carried out in. The aims of this study was to determine the uptake of cations under two different water conditions, irrigation and desiccation and to determine the distribution of these cations in trees and, thirdly, to compare Cs distribution in trees subjected to air deposited Cs with trees subjected to Cs added directly to the soil.

Site

The studies were conducted in the Svartberget Research Forest, 60 km northwest of Umeå, in Northern Sweden (64º14'N, 19º46'E, alt. 175 m a.s.l.). The first site was the Nyänget catchment (Nylén and Grip 1997) and the other site was a 30- to 60- year old Scots pine (*Pinus Sylvestris* L.) stand with site index T16 and a stand density of 1250 trees ha^{-1} in 1986 (Plamboeck et al. 1999).

Results

Our results assessed by GDM-40 survey (Plamboeck et al. 2006) show that the mean ^{137}Cs activity on the discharge areas (12 511 Bq/m^2) is higher than in the recharge areas and the mire. Also the mean ^{137}Cs activity for the recharge areas (10 539 Bq/m^2) is higher than the ^{137}Cs activity at the mire (8896 Bq/m^2). Furthermore, the ^{137}Cs values range from 3000 Bq/m^2 in the mire to 17 000 Bq/m^2 in discharge areas along the stream (Fig.1).

The number of soil samples collected in 2000 is not enough to tell a difference between the different hydrological areas because the small-scale variation is considerable and the total area sampled is small. The GDM-40 results, on the other hand, show statistically significant differences between the hydrological areas because the detector is capable of providing spatially integrated measurements of the ^{137}Cs activity over a large area in one single measurement, thereby reducing the statistical significance of small-scale variations. There is a deviation of 20% between the soil sample measurements in 2000 and the GDM- 40 measurements (Fig. 2). This is a very good agreement in the field where so many factors vary, e.g. amount of soil water, soil substrate, ^{137}Cs distribution, vegetation.

Nylén and Grip (1997) showed that the radioactive caesium in this catchment was leached from the mire to the stream and constituted a secondary source of contamination of lakes. They estimated that the yearly loss of caesium from the mire amounted to approximately 2%.
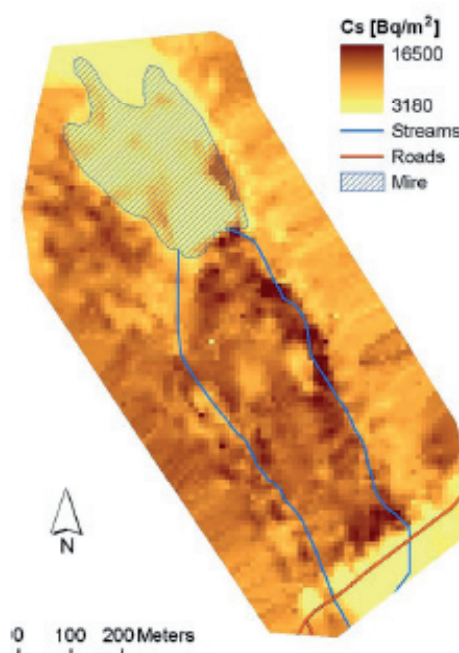

Figure 1. Interpolation of the GDM-40 measurements of the ^{137}Cs activity (Bq/m^2) at Nyänget.
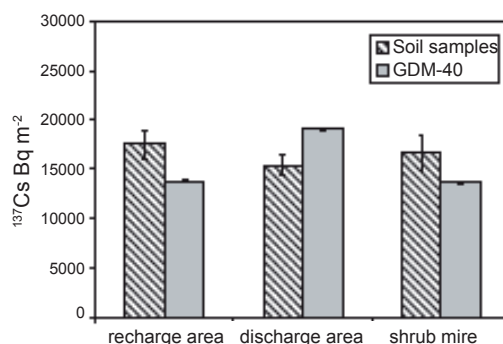
Figure 2. The ^{137}Cs activity (kBq/m^2), with standard errors, found in soil samples and GDM-40 measurements from the areas where the soil samples were collected at Nyänget.

However, Nylén and Grip found no evidence of any losses from the recharge areas during the period 1986 to 1989. Cumulative figures for the mire and the recharge area (6.58 ha) that supplies it with water indicate that the recharge area lost 0.35 GBq in the 11 years between 1989 and 2000, approximately the same amount of activity as the mire gained (0.43 GBq). The discrepancy in activity in the recharge area between 1989 and 2000 cannot be explained by the uptake in vegetation (Plamboeck et al. 2000). Our data indicate that the caesium lost from this catchment moves from the recharge area to the mire, from the mire out in the stream.

Results from the temporal variation study show that the main fraction of radioactive caesium in 1987 was retained in/on needles and bark that were directly exposed during the fallout in 1986. Ten years after in 1996 these needles had fallen as litter and the needles that had developed after 1990 only contributed with 15% to the total activity in trees (Table 1). The total amount of ^{137}Cs retained in trees have increased from 3% (1987) to 15% (2006) (Fig. 3).

These results show that the redistribution of radioactive caesium still contribute to high activity concentrations in some compartments of the ecosystem. The high activity concentration in branches and current needles during 2006 indicates that uptake of ^{137}Cs in Scots pine even 20 years after an accident may be important and has to be considered in forestry and other kind

Table 1. The mean biomass (kg d.w./m^2), ^{137}Cs (Bq/g) and retention (% of total activity in trees and soil) in different fractions of Scots pine in a stand in Northern Sweden through the years 1987, 1996 and 2006.

Year	Compartment	Biomass mean, kg/m^2	^{137}Cs mean, Bq/g	^{137}Cs in trees mean, %
1987	Stump	0.31	0.05	0.068
	Wood	0.88	0.05	0.19
	Bark	0.105	1.5	0.68
	Branch	0.145	0.41	0.26
	Needle	0.11	3.8	1.8
1996	Stump	1.49	0.063	0.55
	Wood	4.8	0.063	1.32
	Bark	0.5	0.36	0.8
	Branch	0.9	0.17	0.66
	Needle	0.4	0.36	0.63
2006	Stump	1.9	0.12	1.4
	Wood	5.0	0.12	3.8
	Bark	0.51	0.50	1.6
	Branch	0.78	1.2	6.5
	Needle	0.36	0.80	2.0
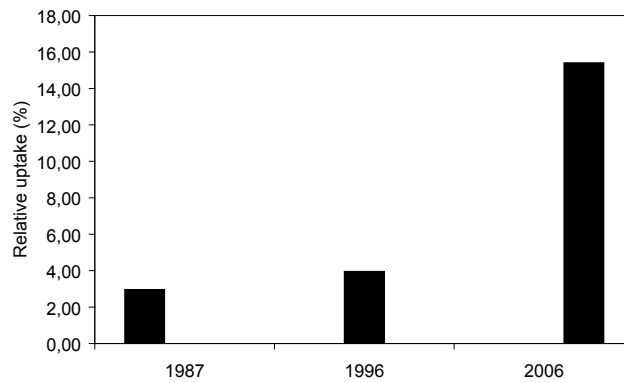
Figure 3. The relative amount of ^{137}Cs in Scots pine (Pinus Sylvestris L.) of the total load.

of utilisation of forest products. For example can this be of high importance in the bio fuel industry. Given an activity concentration of 1200 Bq/kg (d.w.) and a concentration factor of 10 during combustion the concentration in ashes would be 12000 Bq/kg. According to the recommendations from SSI (the Swedish Radiation Protection Authority) ashes that have concentrations higher than 10 kBq/kg must be stored in special deposits.

In the study (Plamboeck et al. 2000) where the uptake of Cs and Na were studied under two different water regimes results clearly show that uptake of injected Cs by the trees was higher if the upper soil horizon was moist. Plamboeck et al. (1999) calculated the water uptake in the pines on this site in 1995, and found that it was significantly higher from the humus layer on the irrigated plots than on the desiccated plots. This is consistent with the finding in this investigation that uptake of the injected ^{134}Cs was higher from the irrigated plots than from the desiccated plots. Also, more ^{22}Na seems to have been taken up from the irrigated plots than from the desiccated plots, which is consistent with the relative water uptake patterns from the 10–25 cm deep layer. Root uptake of injected ^{134}Cs from humus and the 0–10 cm deep mineral soil layer, and of ^{22}Na from 10–25 cm depth increased exponentially with relative water uptake (Fig. 4). The relative uptake of Na was greater than that of Cs, probably due to the low exchangeability of Cs in the soil.
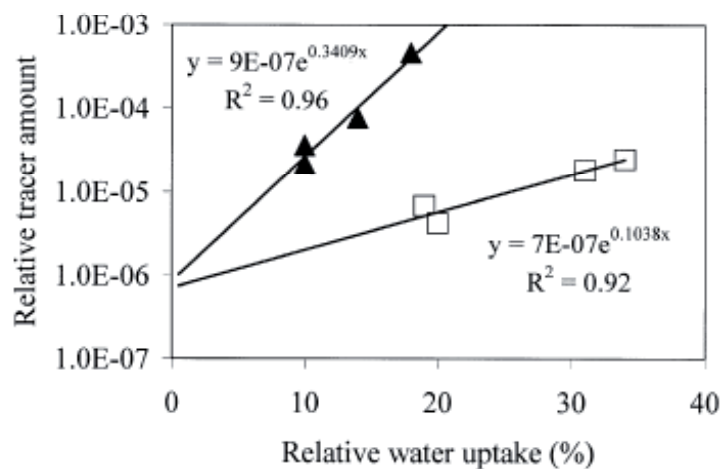
Figure 4. The relative water uptake from H–10 cm (Cs) and 10–25 cm (Na) as a function of the amount of ^{22}Na and ^{134}Cs in xylem sap relative to the amount injected, for the treatments irrigation and desiccation in July 1995 at Åheden.
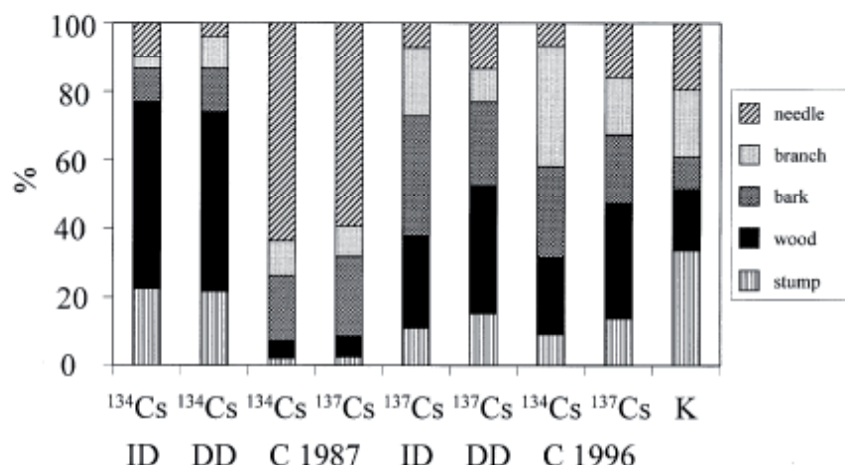
Figure 5. The distribution of injected ^{134}*Cs, Chernobyl* ^{137}*Cs and* ^{134}*Cs in different tree compartments in 1996 compared with the distribution of Chernobyl* ^{134}*Cs and* ^{137}*Cs in the same stand in 1987, 1 year after the deposition (Nylén 1996) and the K distribution in a Scots pine stand in Finland (Mälkönen 1974). Irrigated (ID) plots were only irrigated during 1995.*

The content of ^{134}Cs in stem wood and stumps amounted to nearly 80% of the total uptake in the trees a year after the injection. More ^{134}Cs was found in the needles on plots irrigated in 1995 (9.6%) than on desiccated plots (4.2%). A year after the Chernobyl accident, the Cs was predominantly located on or in the needles and bark. After 10 years of redistribution, the pattern of Cs contents in the different parts of the trees approached that of K. The major difference between the present distribution patterns of these two ions is that there is a higher proportion of K than of Cs in the needles (Fig. 5).

The conclusions from all the studies presented here are that caesium uptake in trees is correlated to water uptake and that caesium is available to uptake in vegetation and leakage out of the system for a long time after an accident. Those conclusions are based on studies at stands in one single area. It would of course be of interest to carry out similar investigations in different areas, stands (mature and newly established stands) and tree species to be able to generalize the above described patterns.

References

Mälkönen, E. 1974. Annual primary production and nutrient cycle in some Scots pine stands. Dissertation, Comminicationes Instituti Forestalis Fenniae 84.5, 87 pp.

Nylén, T. & Grip, H. 1997. The origin and dynamics of Cs-137 discharge from a coniferous forest catchment. J. Hydrology 192: 338–354.

Nylén, T. 1996. Uptake, turnover and transport of radiocaesium in boreal forest ecosystems. Thesis. Swedish University of Agricultural Sciences, Uppsala, Sweden.

Plamboeck, A. H., Grip, H. & Nygren, U. 1999. A hydrological tracer study of water uptake depth in a Scots pine forest under two different water regimes. Oecologia 119: 452–460.

Plamboeck, A.H., Nylén, T. & Grip, H. 2000. Uptake of cations under two different water regimes in a boreal Scots pine forest. Sci. Total Environ. 256: 175–183.

Plamboeck, A.H. Nylén, T. & Ågren, G. 2006. Comparative estimations of ^{137}Cs distribution in a boreal forest in northern Sweden using a traditional sampling approach and a portable NaI detector. J. Environ. Radioact. 90: 100–109.

2.2 ^{137}Cs deposition in peat profiles on a raised bog in central Sweden

Klas Rosén [1], M. Vinichuk [1, 2], P.R. Galán [1] and K.J. Johanson [1]

[1] Department of Soil Sciences, Swedish University of Agricultural Sciences, SLU, Box 7014, SE-75007, Uppsala, Sweden
[2] Department of Ecology, Zhytomyr State Technological University, 103 Cherniakhovsky Str., 10005, Zhytomyr, Ukraine

Abstract

Distribution of ^{137}Cs depositions within peat profiles in open bog and nearby (low pine) sites in raised bog are shown and discussed. A possible involvement of Sphagnum moss in radionuclide binding and retention in such nutrient poor ecosystem is suggested.

Introduction

Raised bog is an extremely nutrient-poor ecosystem with very low or non-existent clay content and little fungal mycelium present to which ^{137}Cs can be bound. This ecosystem is therefore different from both agricultural soils, where ^{137}Cs is presumably bound to certain clay minerals (Wauters et al. 1996) and forest soils, where extensive fungal mycelium counteracts the downward transport of ^{137}Cs by an upward translocation flux (Rafferty et al. 1997, 2000). In raised bog the vertical migration of ^{137}Cs is also rather slow but uptake is very high, particularly in some of the plants, which indicate binding of radionuclide by living matter. Thus, caesium uptake and binding in such nutrient-poor ecosystem is different from that in forest or on agricultural land and are not well understood. *Sphagnum* species are the key species in the radioecology of raised bog systems. This study was undertaken to investigate the distribution of ^{137}Cs within peat profile and its uptake by plants in raised bog in central Sweden and to determine the potential role of bog vegetation, including *Sphagnum* sp., in the biological retention of [137]Cs.

Material and methods

[137]Cs activity from Chernobyl fallout in 1986 in peat profiles and plants was studied in two sites on raised bog in central Sweden over the period from 1989 to 2004–2007. One site (open bog) was in an area with no trees and only a few sparsely growing plant species and the other (low pine) was less than 100 metres from the open bog site and had slowly growing Scots pine, a field layer dominated by some ericaceous plants. From each of these two ecosystems samples of peat profile and composite samples of plants were collected in 1989 and again during the period 2004–2007 and analysed.

Results and Discussion

Interception of ^{137}Cs fallout presumably differed at both sites: direct interception by *Sphagnum* moss at open bog and partly by the canopy of Scots pine on low pine site. Despite that the estimated mean ^{137}Cs deposition was very similar at both sites: 23.0 kBq m^{-2} in the open bog and

23.6 kBq m^{-2} at the low pine site. However, distribution of the ^{137}Cs activity within peat profile on the open bog site differed from that on the low pine (Fig. 1).

Thus, on the open bog site ^{137}Cs was mainly found in the uppermost a few cm of *Sphagnum* layers, whereas at the low pine site ^{137}Cs was predominantly located in deeper (10–12 cm) layers (Fig. 1). Correspondingly, the downward migration of the radionuclide within peat profiles was also found to be slightly different at two sites. At the open bog site ^{137}Cs migration rate was 0.57 cm a^{-1} and the migration centre of radionuclide was found at a depth of 10.7 cm, while the rate at the low pine site was 0.78 cm a^{-1} and the migration centre was at 14.9 cm.

These observations indicate possible involvement of different biota species, especially *Sphagnum* moss, in ^{137}Cs migration at two sites. Indeed, the most of the ^{137}Cs activity in the peat profiles from open bog was found in living moss plants, mainly in the apical few cm of the *Sphagnum* mosses, which has formed since the Chernobyl fallout occurred. Assuming the growth rate of *Sphagnum* mosses to be between 1 and 2 mm per year, the peat mat has increased by about 3 cm since the fallout. This indicates that there has been upward migration of ^{137}Cs within the *Sphagnum* plants. In might be plausible explanation of radionuclide distribution within peat profile, since internal translocation of major biogenic elements in *Sphagnum* mosses from the senescing plant tissue to the metabolically active growing apex has been suggested in several studies (Rydin and Clymo 1989, Aldous 2000). The density of *Sphagnum* moss is also higher in upper a few cm layer of bog carpet (acrotelm layer consists of living moss), which therefore resulted in higher activity concentration found here.

The ^{137}Cs activity concentrations in the few vascular plants growing on the open bog site, such as heather and cranberry, were usually higher than those in plants growing on the low pine site. It is likely that depth location of plant roots is one of the factors effecting radionuclide uptake. Shallow located roots on open bog site in such plants as sundew, cranberry and slightly deeper in heather seem promote ^{137}Cs uptake. Labrador tea, heather and crowberry plants that have relatively deep root location dominated in the field layer on low pine site. Heather was
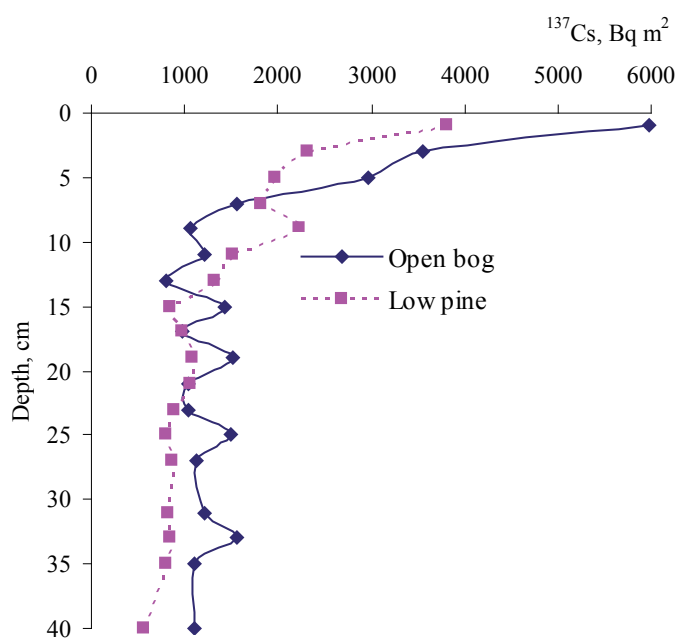
Figure 1. Vertical ^{137}Cs deposition in Bq m^2 on the open bog and low pine sites (mean values, 2005).

the greatest radiocaesium accumulator, consistently showing remarkably high ^{137}Cs activity concentration at both sites.

In general, ^{137}Cs activity concentrations in plants at both sites either decreased over the study period or remained unchanged. The ^{137}Cs peat-to-plant transfer factor (TFg) varied greatly between species but also between sites and years and generally reflected trends found in the retrospective study of ^{137}Cs activity concentrations in plants. The root distribution of the plants within the peat profile seemed to be the most appropriate explanation for the differences in ^{137}Cs uptake over time.

Conclusions

^{137}Cs activities in peat profiles on raised bog indicate involvement of *Sphagnum* mosses in radionuclide retention and migration. Further studies, however, required to clarify the depth-dependent transfer parameters of caesium to bog plants, since additional features apart from the root distribution seem to be involved.

Acknowledgements

References

Aldous, A.R. 2000. Nitrogen translocation in *Sphagnum* mosses: Response to atmospheric nitrogen deposition. In: Crowe, A. & Rochefort, L. (eds.) Que´bec 2000. Millennium Wetland Event, Program with Abstracts.

Clymo, R.S. & Reddaway, E.J.F. 1974. Growth rate of *Sphagnum rubellum* Wils. on Pennine blanket bog. J. Ecology 62: 191–196.

Rafferty, B., Dawson, D. & Kliashtorin, A. 1997. Decomposition in two pine forests: the mobilisation of ^{137}Cs and K from forest litter. Soil Biol. Biochem. 29(11/12): 1673–1681.

Rafferty, B., Brennan, M., Dawson, D. & Dowding, D. 2000. Mechanisms of ^{137}Cs migration in coniferous forest soils. J. Environ. Radioactivity 48: 131–143.

Wauters, J., Vidal, M., Elsen, A. & Cremers, A. 1996. Prediction of solid/liquid distribution coefficients of radiocaesium in soils and sediments. Part two: A new procedure for solid phase speciation of radiocaesium, Applied Geochemistry 11: 595–599.

2.3 Human dose pathways of radionuclides in forests

Aino Rantavaara

Radiation and Nuclear Safety Authority, Research and Environmental Surveillance,
P. O. Box 14, FI-00881 Helsinki, Finland

Introduction

Forest soil, understorey vegetation and trees are all sources of radionuclides and human radiation doses after contaminating atmospheric deposition. People are exposed to radiation externally from sources outside the body and internally via ingestion and inhalation of radionuclides. Understorey vegetation contributes to ingestion doses through berries, herbs, wild honey, mushrooms and game meat; also trees provide feed to terrestrial birds and big game. During stay in forests people are subject to external radiation from forest floor and overstorey, and they may inhale airborne radioactive aerosol or gaseous radionuclides in ground level air. In the early phase of contamination also resuspended radionuclides may add to the internal dose of people via inhalation. People in Nordic countries are most exposed to radiation via ingestion of radionuclides in wild foods.

The distribution of radionuclides in forests is changed by environmental processes, and thereby also the significance of various dose pathways to humans will change with time. External exposure is received in living environment from contaminated stemwood used as building timber and for manufacturing of furniture and other wood products.

The aim of this paper is to outline the significance of various human dose pathways of radionuclides in forests considering the public and workers in forestry and production of bioenergy. Examples on effective doses are given based on two historical events, atmospheric nuclear weapon tests (mostly in 1950's and in 1960's) and the Chernobyl nuclear power plant accident in 1986.

Radionuclide dynamics and radionuclide distribution in forests

Certain environmental processes are continuously changing the distribution of radionuclides in forest ecosystem, the changes being most remarkable in early phase after primary deposition. Weathering removes activity from surfaces of vegetation most in the first weeks or months after contaminating deposition. Like weathering also through-fall and stem-flow take radionuclides from the crown of trees to the forest floor, or down the stem. Through foliar absorption part of the initially deposited radioactive material on the plant surfaces penetrates the surface of leaves or needles and shoots and enters the nutrient cycle of the plant. The stand characteristics as leaf area indices, biomass densities and species structure also have effect on distribution of radionuclides in a forest.

The surface runoff changes the horizontal distribution of radionuclides on the forest floor or transports part of deposited material to water systems, particularly in the first year. The intensity of runoff varies by rainfall and topography of the fallout area. Forest soil with humus and litter layers becomes the dominant source of radionuclides and external dose rate in a few weeks or months after primary deposition, depending on type of stand and contaminating event. Downward migration rate in mineral soil varies by radionuclide, soil type and the stand of trees.

In boreal forests the migration rate of radiocaesium, the dominant contributor to external dose, is mostly low.

The availability of deposited radionuclides to plants via root uptake will increase after dissolving and transfer of radionuclides from the surface of forest floor to the root zone. Hardly any fixation processes are involved in the change of availability of radioactive caesium in the root zone of soils in boreal forests, because of minimal fraction or lacking of clay. In the long term, the root uptake and translocation of elements in plants gradually change the distribution of long-lived radionuclides in forest vegetation. The residence times of radioactive caesium in the soil-plant system are long, controlled mostly by radioactive decay. Together with rather high soil to plant uptake rates to understorey vegetation, the increased contamination of edible forest products may prevail long periods of time (Calmon et al., in press). It is noteworthy that ^{137}Cs concentration in reindeer meat does not follow the dynamics of root uptake during winter feeding with lichen. Lichens retain radionuclides of the primary deposition for years, and the radionuclide intake by reindeer is significantly higher from lichen than from vascular plants in summer.

Radionuclide dynamics in trees is rather complicated. During growth of trees the allocation of certain nutrients and radionuclides can correlate, and also seasonal translocation of elements is involved in dynamic changes in radionuclicde distribution. ^{137}Cs concentration in stemwood is lower than in other aboveground parts of trees.

Human external doses received in forests are varying by the time spent in forests, and the activity distribution in ground and overstorey. A number of short or medium half-life nuclides contribute to external dose from tree canopies and ground during the first weeks or months (Turunen et al., this report). During the first weeks after dry deposition the canopies of young seedling stands can be significant sources of external doses to people working in forest. The dose rate from radioactive contamination intercepted in the canopy is highest when the distance from the source of radiation to the exposed person is shortest. Compared to seedlings the radionuclides in crowns of mature pines at longer distance from the subject cause a much lower dose rate to human subject (at 1 m reference height above the ground).

External doses via timber in living environment are related with activities in stemwood, and are received with delay between harvest of trees and marketing and use of timber. Similar delays relate with the fuel wood and human exposure to radiation of radionuclides in wood ash. The amounts of wild foods consumed and the origin of berries, mushrooms, game meat and herbs govern the ingestion dose due to ^{137}Cs and, to a minor extent, ^{90}Sr. Contribution of tree stands and pathways to human radiation doses differ at different stages of rotation (Fig. 1).

Workers in the field of forestry and production of bioenergy can be exposed to radiation from radionuclides in forests and timber through normal forest work, and to radiation of wood ash at energy production plants and during transport and distribution of ash to forests or dumping grounds.

Deposited radionuclides of concern to human doses

Radionuclides deposited on vegetation and ground cause external and internal exposure of people using forests. Most fission product nuclides decay via beta emission, and photon emission

Figure 1. Significance of various exposure pathways from forests to people are changing with time and depend on the stage of rotation. (Origin of picture: Aulikki Hamari, Metla, Finland).

(gamma radiation) is accompanied for instance with the decay of the nuclides ^{137}Cs ($T_{1/2}$ 30 a) and ^{134}Cs (2.1 a), whereas ^{90}Sr (28 a) is a pure beta emitter. Plutonium isotopes ^{238}Pu (88 a), ^{239}Pu (2.41 ×10^4 a), ^{240}Pu (6.5×10^3 a) decay through emission of alpha particles and contribute to the nuclear fallout together with the beta active ^{241}Pu (14 a). Radioactive caesium dominates both external and internal doses received by people from artificial radionuclides deposited to forests.

Radiation exposure from past events

Atmospheric testing of nuclear weapons in 1945–1980 and the Chernobyl nuclear power plant accident in spring 1986 caused widespread deposition of artificial radionuclides in the northern hemisphere. The committed effective doses via ingestion of long-lived radioactive caesium and strontium in food and water due to these events were assessed for Finnish people. Radionuclides of global fallout were detected in foodstuffs in Europe since 1955, and the effect of Chernobyl-derived fallout on dietary dose was assessed for the years 1986–2005. So far, ^{137}Cs has caused most of the internal and external exposure from deposited artificial radionuclides, and also most ingestion doses via wild foods of forest origin. In the years 1955–2005 people in Finland received through ingestion of food and water an average dose of 2.5 mSv due to ^{90}Sr, ^{134}Cs (from Chernobyl only) and ^{137}Cs. The dose received via food and water in 51 years is comparable to one year dose from indoor radon and natural radionuclides in human body, 2.4 mSv, estimated at STUK[1]. ^{137}Cs in wild foods of forest origin caused one fifth of the ingestion dose, and only one per cent of the dose via wild foods was due to ^{90}Sr (Rantavaara 2008).

The upper limit of adult ingestion dose due to Pu in wild foods of forest origin in the area contaminated by the Chernobyl fallout in Finland was very low, 4 nSv a^{-1} in the late 1990's (Rantavaara and Kostiainen 2002). The estimate was based on average consumption of wild foods in 1995–1996 (Markkula and Rantavaara 1996). During the period 1955–2005 an upper

---

1 Muikku et al. in the report STUK-A211, 2005 (available at www.stuk.fi).

limit of average effective dose due to plutonium in wild foods was 0.5 µSv, only 0.1 percent of the dose from isotopes of caesium and strontium. Doses due to ^{90}Sr and plutonium were considerably lower than the dose from ^{137}Cs. The reasons were lower deposition rates, very low uptake from soil to the edible fractions of wild foods, and, after ingestion, the lower gut uptake factor compared to ^{137}Cs.

The contamination context of foodstuffs after the two events differed substantially. The contribution of wild foods of forest origin to the intake of ^{137}Cs in the era of global fallout was only seven per cent, and after the Chernobyl accident 29%. The difference was derived mainly from the deposition pattern; global fallout was received in several years, particularly in 1950's and 1960's during summer months, whereas radionuclides from the Chernobyl accident caused a single peak in deposition of ^{137}Cs in early spring 1986, followed by a fast decline of annual deposition in the next few years. Because of differences in monthly deposition of radioactive caesium during growth period, the environmental processes governing the uptake of radionuclides by plants had a varying effect on activity concentrations in foodstuffs (Rantavaara 2008).

External doses followed the time pattern of accumulated deposition of ^{137}Cs in the period of global fallout and after the Chernobyl accident. Maximum effective dose, 0.15 mSv a^{-1} was measured in Finland in 1986. By the year 2005 de dose declined to 0.02 mSv a^{-1}, approximately the same level as ingestion dose from food and water. Also in the years preceding the Chernobyl fallout, during low deposition rates, the external dose and ingestion dose were almost equal, 0.007 mSv a^{-1} (Rantavaara 2008).

Doses via utilization of timber

Finnish saw wood was surveyed for ^{137}Cs in 1993. Activity concentrations in debarked stemwood of pine, spruce and birch were 20 Bq kg^{-1} (80% dry matter) on average. About 95% of timber contained less than 100 Bq ^{137}Cs per kg (Rantavaara 1996). A survey in 1996–1997 confirmed the slow increase in contamination of timber; the increase in activity concentration of ^{137}Cs in saw timber during three to four years was approximately compensated by radioactive decay. Further surveillance of ^{137}Cs in timber is still of interest to find out the later development of the activity concentration.

Using the results for ^{137}Cs in timber the effective dose was derived for all-year residents of a house constructed of domestic wood. The annual doses for a log house were only a few percent of the dose constraint of 1 mSv a^{-1} set for ^{137}Cs in building materials. No limitation for the use of domestic timber has been needed in the 1990's or later.

Regarding other domestic utilization of timber it is important to consider the lower radiation exposure from wood products containing smaller volumes of wood, and the occasional or less frequent use of products than the log house of the scenario. A systematic dose assessment published by IAEA (2003) suggests dose conversion factors for ^{137}Cs of wood products. This data refers to at least one decade lower doses from the wooden floors, sauna and furniture, compared to living in a wooden house, assuming the exposure times are equal.

Radiation risk related with handling of wood ash

Use of wood ash for fertilisation of forests

The ash of wood fuel and mixed fuels was studied for radionuclide concentrations in 22 plants used for production of energy in Finland in 1996–1997 (Rantavaara and Moring, 2001). The results for ^{137}Cs in wood ash were then generally lower than 10 000 Bq kg^{-1} (dry matter). This value was chosen for estimation of upper limit of external dose rate in forests after ash fertilisation. The application dose of 0.5 kg ash per square meter was assumed. The dose rate from ^{137}Cs in ash was at maximum just after the soil treatment, 7 nGy h^{-1}. Forest worker would receive a dose of less than 10 µSv a^{-1} in 1500 hours. Natural radionuclides in five tons of wood ash distributed per hectare caused approximately the same dose rate in air as 1000 Bq ^{137}Cs m^{-2}, i.e. less than 0.002 mSv a^{-1} for the forest worker. Members of the public, assumed to spend in fertilised forest 120 h a^{-1}, would receive from ^{137}Cs in ash an annual dose of 0.0006 mSv a^{-1}.

It is worth considering the roughness of forest floor and downward migration of radionuclides via convection and partial dissolving of ash. In addition, most forestry work today is not without shielding. Therefore, and due to lower mean concentration of ^{137}Cs in wood ash compared to value in the scenario, the actual dose rate in forests would be lower than the conservatively calculated estimate reported here.

In areas where wild berries and mushrooms are picked for own use the ingestion dose from ^{137}Cs dominates the radiation dose received from forests. After ash fertilisation of forests the ingestion dose from wild foods will be gradually reduced, as evidenced for lingonberry (Levula et al. 2000). The reduction of ingestion dose from ^{137}Cs in berries and mushrooms can be significantly higher than the external dose from ash to pickers.

Workers handling ash in energy industry and forestry

Three different wood energy plants and a mill granulating ash were surveyed for radiation exposure of workers handling ash in industry (Vetikko et al. 2004). Dose rates exceeding the background radiation at the plant were measured and exposure times approximated together with exposed workers, considering various tasks and target areas. Often the upper boundaries of doses were obtained through assuming higher than average exposure times and activity concentrations in ash. Exposure of drivers transporting ash with long-distance trucks was assessed for ash containing 10 000 Bq ^{137}Cs kg^{-1}, and considering the shielding provided by the truck. Doses from airborne ash dust were assessed assuming 1 mg m^{-3} air concentration of ash. For inhaled ^{137}Cs, ^{90}Sr, ^{40}K, ^{210}Pb, ^{232}Th, ^{235}U and ^{226}Ra the mean activity concentrations in wood ash from energy production were used (Rantavaara and Moring 2001). Dose estimates varied with task ranging from 0.03 to 0.45 mSv a^{-1}, being highest for drivers involved in long distance transport of ash and workers of the granulating mill due to rather long exposure times. Dose estimates from inhaled ash were very low at the plants, and they can be minimised in conditions of ash spreading to forest from ground level in modern vehicles where filtered air intake and overpressure of the cabin protect the worker. Intervention dose to workers was 1 mSv a^{-1} and to members of the public 0.1 mSv a^{-1} (STUK 2001).

Discussion and conclusions

Surveillance of radioactivity of wild foods and better comprehension of the radionuclide dynamics in forests convinced authorities of the continuance of the intake of ^{137}Cs via wild foods. It was supposed that in the most contaminated regions of Europe the ^{137}Cs contamination of wild foods would continue for decades. Therefore, the Commission Recommendation was given (2003) for protection of the public from high concentrations of radioactive caesium, and information of the public on the potential radiation exposure from wild foodstuffs (European Commission 2003). Mushrooms have been of concern in several countries. Advice based on household cooking methods is logical to consumers of mushrooms. Radiocaesium contamination of edible food can be reduced significantly, up to 95%, by parboiling and soaking of mushrooms.

Concerning timber, no intervention has been reported in Nordic countries. However, it is important to update the data on contamination of saw wood, to be able to provide specific advice to forest industry and other end users of wood.

Wood ash needs consideration because of increase of radionuclide concentrations in ash during combustion of wood. For debarked timber the concentration factor can be as high as 200. For radionuclides in ordinary fire wood the concentration factor of 80–100 is assumed, and even lower factor for bark due to its higher fraction of ash. The use of ash as a constituent of concrete, for landfill or construction of roads is regulated for radiation protection of workers and members of the public (STUK 2001).

Analysis of human dose pathways related to radionuclide contamination of wild foods, timber, and effects of ash fertilisation need radioecological background knowledge. The exposure conditions of all end users of forest products as well as workers in forest and energy industries have to be carefully examined for realistic identification of the need for intervention.

References

Calmon P, Thiry Y, Zibold G, Rantavaara A, Fesenko S. Transfer parameter values in temperate forest ecosystems: a review DOI: 10.1016/j. Journal of Environmental Radioactivity. 2008.11.005 (In Press.)

Commission recommendation of 14 April 2003 on the protection and informing of the public with regard to exposure resulting from the continued radioactive caesium contamination of certain wild food products as a consequence of the accident at the Chernobyl nuclear power station (2003/274/EC, OJ L 99, 17.4.2003. p. 55).

International Atomic energy Agency, Assessing radiation doses to the public from radionuclides in timber and wood products. IAEA-TECDOC-1376, IAEA, Vienna, 2003.

Levula T., A. Saarsalmi, A. Rantavaara, 2000. Effects of ash fertilization and prescribed burning on macronutrient, heavy metal, sulphur and ^{137}Cs concentrations in lingonberries (*Vaccinium vitis-idaea*). Forest Ecology and Management, 126(3): 267–277.

Markkula M-L, Rantavaara A. Consumption of mushrooms and other wild products in Finland. In: Walderhaug T, Gudlaugsson EP (ed). Proceedings of the 11th Meeting of the Nordic Society for Radiation Protection and the 7th Nordic Radioecology Seminar. 1996 Aug 26–29; Reykjavik, Iceland. ODDI Reykjavik, 1997. p. 371–376.

Rantavaara A. Ingestion doses in Finland due to ^{90}Sr, [134]Cs, and ^{137}Cs from nuclear weapons testing and the Chernobyl accident**.** Applied Radiation and Isotopes 2008; 66: 1768–1774.

Rantavaara, A. & Kostiainen, E. Plutonium in coniferous forests. In: Ilus, E. (ed.). Proceedings of the 8th Nordic Seminar on Radioecology, 25–28 February 2001, Rovaniemi, Finland NKS-70. Pitney Bowes Management Services Denmark A/S: 163–165, 2002.

Rantavaara, A. & Moring, M. Puun tuhkan radioaktiivisuus. STUK-A177. STUK, Helsinki 2001.

STUK. 2003. ST Guide 12.2. Radioactivity of building materials and ash. Radiation and Nuclear Safety Authority, October 8th, 2003.

Vetikko, V., Valmari, T., Oksanen, M., Rantavaara, A., Klemola, S. & Hänninen, R. Energia-teollisuudessa syntyvän puuntuhkan radioaktiivisuus ja sen säteilyvaikutukset. STUK-A200. STUK, Helsinki 2004.

2.4 Effect of industrial pollution on behaviour of radionuclides in forest ecosystems

Iisa Outola

STUK-Radiation and Nuclear Safety Authority, P.O. Box 14, FI-00881 Helsinki, Finland

Introduction

Forests are an important source of radioactivity through the consumption of berries, mushrooms and game. Anthropogenic stresses such as heavy metals, organic charges, acidification, chemical waste, etc. may affect the migration of radionuclides in soil and their biological uptake. Because of the harsh climate and nutrient poor soil, boreal forests are especially sensitive to chemical pollutants, which in turn can affect the behaviour of radionuclides and essential nutrients in these ecosystems. In the event of an accident at a nuclear facility where radionuclides were released to the environment, understanding of the effect of industrial pollution on the behaviour of radionuclides, and also on the radiation risk, would be crucial.

To investigate how and to what extent industrial pollution affects the behaviour of radionuclides in forest ecosystems, studies were conducted in the vicinity of two Cu-Ni smelters: one in a pine forest at Harjavalta, Finland, and the other in a spruce forest at Monchegorsk, Russia. The effects of industrial pollution on the distribution of radionuclides in soil and on the soil-to-plant transfer were evaluated.

Sampling sites and methods

Research was performed in the vicinity of two Cu-Ni smelters: one in Monchegorsk, Russia and the other in Harjavalta, Finland. The smelter in Monchegorsk started to operate in 1938 and that in Harjavalta in 1945. The sampling sites in Monchegorsk were in spruce forest and those in Harjavalta were in pine forest. The sampling sites are described in detail by Rahola et al. (1999) and by Outola (2002).

Pollution from Cu-Ni smelters has had diverse effects on the forest ecosystem: concentrations of exchangeable cations have changed (Thørring et al. 1999), the water flux has been drastically altered, total microbial mass and activity have decreased in the proximity of the smelter complex (Fritze et al. 1989), and the vegetation has been affected by industrial pollution. Only minor variations were observed in soil pH and exchangeable acidity in Monchegorsk (Thørring et al. 1999).

Litter and soil in Monchegorsk were sampled in August 1997. Samples were collected from five plots at each site. The size of the sampling plots was 1 m x 1.5 m, and the soil was sampled to a depth of 30 cm. In Harjavalta soil sampling was performed in September 2000 and soil profiles were taken with a corer having an inner diameter of 10.4 cm and length of 22 cm. Five soil profiles were taken at each site in Harjavalta. The soil was separated into different horizons: litter (L), organic (O) and mineral (E and B) layers. Vegetation, mushrooms and lichen were collected at the sampling sites at the same time as soil. Vegetation was not washed before the activity determinations.

Gamma activities were measured with a high purity Ge-detector. Radiochemical separation of Pu-isotopes and ^{241}Am is described in detailed by Outola (2003).

Results and discussions

Radionuclides in Soil

The distribution of ^{137}Cs among the different soil horizons in Monchegorsk and Harjavalta is shown in Fig. 1. The higher amount of ^{137}Cs in the mineral soil in Monchegorsk as compared to Harjavalta is due to the different origin of ^{137}Cs. In Harjavalta approximately 90% of the ^{137}Cs activity is from Chernobyl, while in Monchegorsk only 10% of it is Chernobyl-derived and most of ^{137}Cs is from global fallout. Because ^{137}Cs has not had sufficient time to penetrate into the mineral layers in Harjavalta in the same degree as it has done in Monchegorsk, the litter layer contains a higher percentage of ^{137}Cs. In both areas, more ^{137}Cs was bound to litter and less to the organic horizon when approaching the smelters.

Distribution of ^{239,240}Pu among the various soil horizons along the pollution gradients in Monchegorsk and Harjavalta is shown in Fig. 2. Pu-239,240 originates mainly from global fallout at both sites. Most of the plutonium was present in the organic layer and the litter layer. In both areas the amount of ^{239,240}Pu in litter increased towards the smelters, whereas the amount of plutonium in the organic layers decreased.

The reason for the increasing content of radionuclides in litter and the decreasing content of radionuclides in the organic layer as a function of pollution is the inhibited decomposition of litter. In their study of the soil microbial effects of the Cu-Ni smelter in Harjavalta Fritze et al. (1989) found that both the numbers of bacteria and soil respiration decreased towards the smelter. The low microbiological activity caused by high Cu and Ni concentrations and acid deposition suppress the decomposition of litter, increasing the volume of the litter and decreasing the thickness of the organic layer. The thick layer of litter at the most polluted sites retains radionuclides, preventing them from migrating downwards to deeper layers and causing a higher concentration in the litter itself.
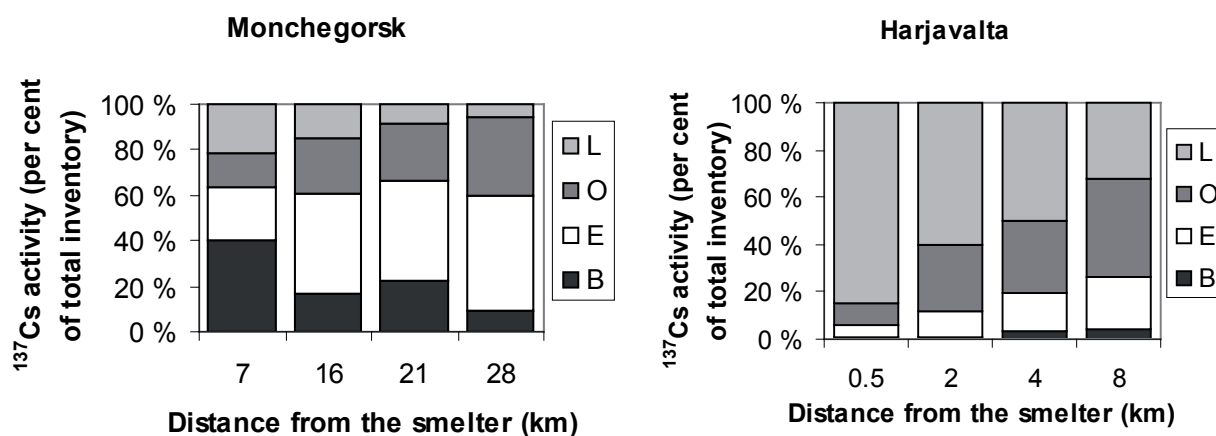
Figure 1. Distribution of ^{137}Cs among litter (L), organic (O) and mineral layers (E and B) along the pollution gradient in Monchegorsk (Bunzl et al. 2001) and Harjavalta (Outola et al. 2003).
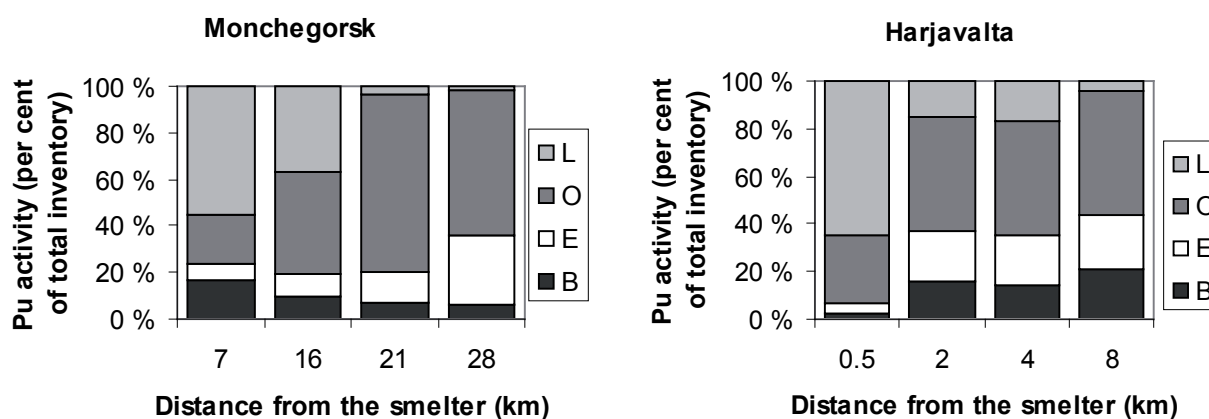
Figure 2. Distribution of ^{239,240}Pu among litter (L), organic (O) and mineral layers (E and B) along the pollution gradient in Monchegorsk (Bunzl et al. 2001) and in Harjavalta (Outola 2003).

Radionuclides in vegetation

In the absence of direct deposition from air the radionuclides in plants are due to root uptake and surface contamination with resuspended litter. Transfer of radionuclides from soil to plants can be described with the aggregated transfer factor that is defined as the activity in plants (Bq/kg) divided by the total deposition in soil (Bq/m^2). Table 1 and 2 presents the transfer factors of ^{239,240}Pu and ^{137}Cs for different vegetation along pollution gradients in Monchegorsk and Harjavalta.

Root uptake of ^{239,240}Pu from soil to plants is extremely low and surficial contamination of plants by resuspended soil particles is important. Transfer factors of ^{239,240}Pu were smaller in Harjavalta than in Monchegorsk. In Monchegorsk more ^{239,240}Pu was transferred from soil to *Empetrum nigrum* towards the smelter. A similar effect was observed for *Vaccinium myrtillus* and *Deschamsia flexuosa* (Outola 2003). This is probably due to contamination by resuspended soil particles since ^{239,240}Pu content of litter increased with pollution load as shown in Fig. 2.

Table 1. Transfer factors (T_{ag}) of plutonium for Vaccinium vitis-idaea and Empetrum nigrum at different distances from the smelters at Monchegorsk (Riekkinen and Jaakkola 2001) and at Harjavalta (Outola 2003). The uncertainties are given at the one sigma level.

Monchegorsk		Harjavalta	
Distance (km)	T_{ag} (10^{-5} m^2/kg)	Distance (km)	T_{ag} (10^{-5} m^2/kg)
Vaccinium vitis-idaea			
7	8.1±1.7	0.5	-
14	20±3	2	0.8±0.3
21	6.1±1.0	4	1.6±0.3
27	13±2	8	1.7±0.5
Empetrum nigrum			
7	51±4	0.5	-
14	60±6	2	5.1±0.7
21	26±3	4	3.7±0.5
27	7.3 ±1.0	8	3.2±0.7

Table 2. Transfer factors (T_{ag}) of ^{137}Cs for Vaccinium vitis-idaea and Empetrum nigrum at different distances from the smelter at Monchegorsk (Bunzl et al. 1999) and at Harjavalta (Outola et al. 2003). The uncertainties are given at the one sigma level.

Monchegorsk		Harjavalta	
Distance (km)	T_{ag} (10^{-3} m^2/kg)	Distance (km)	T_{ag} (10^{-3} m^2/kg)
Vaccinium vitis-idaea			
7	12	0.5	-
14	14	2	0.7±0.3
21	13	4	37±10
27	18	8	76±12
Empetrum nigrum			
7	7	0.5	-
14	11	2	0.8±0.3
21	10	4	9.2±2.5
27	15	8	16±3

Uptake of ^{137}Cs by plants was significantly altered as a result of industrial pollution, especially in pine forest at Harjavalta where T_{ag} of ^{137}Cs for all plants decreased significantly as the industrial pollution load increased (Table 2). One reason for the decreasing transfer factors with pollution is the lower content of ^{137}Cs in the organic layer at sites close to the smelter (Fig. 1). Secondly, due to the toxic effects of heavy metals on fine roots of the plants, the roots may penetrate deeper into the mineral soil, where ^{137}Cs concentration is lower. More nutrients were also leached into the deeper soil layers as a result of pollution. Cs-137 activity decreased also in mushrooms and lichens towards the smelter as shown in Fig. 3.

In Monchegorsk, the transfer factors of ^{137}Cs did not decrease as significantly as they did in Harjavalta (Table 2). The organic layer was thicker in Monchegorsk than in Harjavalta and possibly the roots did not penetrate into the mineral soil layer to the extent they did in Harjavalta. The mineral soil layer in Monchegorsk also exhibited a higher percentage of ^{137}Cs activity than the mineral soil layer in Harjavalta.
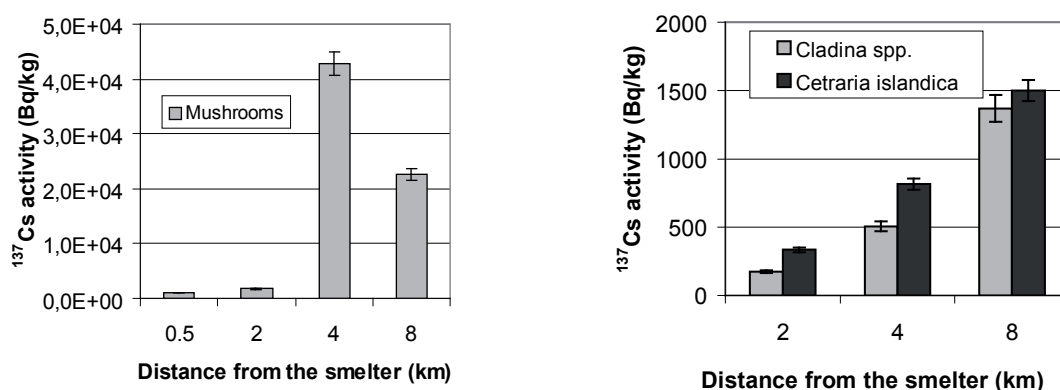

Figure 3. Activity of ^{137}Cs in lichens (Cladina spp., Cetraria islandica) and in mushrooms along the pollution gradient at Harjavalta (Outola et al. 2003). The uncertainties are given at the one sigma level.

Conclusions

Industrial pollution had significant effects on the distribution of radionuclides in soil horizons. With the increase in pollution towards the smelter, radionuclides were accumulated more in the litter layer because the conversion of litter into organic material was diminished due to inhibited microbial activity. As a result, the organic layer contained less radionuclides towards the smelter.

The effect of industrial pollution on soil-to-plant transfer was complex. The effect varied with radionuclide, plant species and also on forest type. For ^{137}Cs, soil-to-plant transfer decreased significantly as industrial pollution increased in pine forest, whereas the decrease was less pronounced in spruce forest. Root uptake of ^{239,240}Pu by plants is extremely small, and plant contamination by resuspended soil is an important factor in considering the soil-to-plant transfer of this radionuclide. In spruce forest, more plutonium was transferred into plants when pollution load increased due to resuspension of litter particles, which contained higher concentrations of plutonium in the vicinity of the smelter. Soil-to-plant transfer of plutonium was much less affected in pine forests contaminated with industrial pollution.

This research clearly indicates the sensitivity of the northern forest ecosystem to inorganic pollutants. Prediction of the soil-to-plant transfer of radionuclides in industrially polluted forest ecosystems requires detailed information on the total deposition, vertical distribution of radionuclides in soil, soil microbiological factors, other soil parameters as well as the rooting depths of the plants.

References

Bunzl, K., Puhakainen, M., Riekkinen, I., Karhu, P., Schimmack, W., Heikkinen, T., Jaakkola, T., Nikonov, V., Pavlov, V., Rahola, T., Rissanen, K., Suomela, M., Tillander, M. & Äyräs, M. 2001. Fallout ^{137}Cs, ^{90}Sr and $^{239+240}$Pu in soils polluted by heavy metals: Vertical distribution, residence half-times, and external gamma-dose rates. J Radioanal Nucl Chem 247: 15–24.

Bunzl, K., Albers, B.P., Schimmack, W., Rissanen, K., Suomela, M., Puhakainen, M., Rahola, T. & Steinnes, E. 1999. Soil to plant uptake of fallout ^{137}Cs by plants from boreal areas polluted by industrial emissions from smelters. Sci Total Environ 234: 213–221.

Fritze, H., Niini, S., Mikkola, K. & Mäkinen, A. 1989. Soil microbial effects of a Cu-Ni smelter in southwestern Finland. Biol Fert Soils 8: 87–94.

Outola, I., Pehrman, R. & Jaakkola, T. 2003. Effect of industrial pollution on distribution of ^{137}Cs in soil and its soil-to-plant transfer in a pine forest in SW Finland. Sci Total Environ 303: 221–230.

Outola, I. 2002. Effect of industrial pollution on the behaviour of ^{239,240}Pu, ^{241}Am and ^{137}Cs in forest ecosystems. Report Series in Radiochemistry 21/2002. Helsinki:Yliopistopaino, 2002. http://ethesis.helsinki.fi/julkaisut/mat/kemia/vk/outola/effectof.pdf

Outola, I. 2003. Effect of industrial pollution on distribution of Pu and Am between various soil horizons and on soil-to-plant transfer of Pu and Am in a pine forest in SW Finland. J Radioanal Nucl Chem 257(2): 267–274.

Rahola, T., Albers, B., Bergman, R., Bunzl, K., Jaakkola, T., Nikonov, V., Pavlov, V., Rissanen, K., Schimmack, W., Steinnes, E., Suomela, M., Tillander, M. & Äyräs, M. 1999. General characterisation of study area and definition of experimental protocols. Report STUK-A166, Radiation and Nuclear Safety Authority, Helsinki, 1999. 37 pp.

Riekkinen, I. & Jaakkola, T. 2001. Effect of industrial pollution on soil-to-plant transfer of plutonium in a Boreal forest. Sci Total Environ 278: 161–170.

Thørring, H., Steinnes, E., Nikonov, V., Rahola, T. & Rissanen, K. 1999. A summary of chemical data from the EPORA project. Report STUK-A167. Radiation and Nuclear Safety Authority. Helsinki 1999. 56 pp.

3 Management of forests contaminated by radionuclides

3.1 Field surveying of radionuclide contamination in forests

Jani Turunen, Aino Rantavaara and Michael Ammann

Radiation and Nuclear Safety Authority, Research and Environmental Surveillance
P.O. Box 14, FI-00881 Helsinki

Abstract

Field measurements of radionuclides after an accidental contamination of forests assume the capacity for identification of a number of nuclides in varying source geometries. The continuous redistribution of radionuclides in forests through natural processes implies a decrease of prevailing surface contamination of trees and an increase in activity density on the ground. Portable gamma spectrometers have long been based on Na(I) detectors which, due to their low energy resolution, are not the tool for analysis of contamination from accidental releases of fission and activation products in the first days or weeks after a deposition. Data of airborne radionuclides from the Chernobyl accident in April 1986 were used for demonstration of initial and later distribution of radionuclides as sources of air Kerma in forests. Forest model (FDMF, PV. 6.0) of the RODOS system was used for the assessment of time-dependent Kerma rate from different forest compartments. The results show the fast reduction of activities of short-lived nuclides and their contributions to the Kerma rate in the first weeks and months. The results also give an estimate for the time needed until a gamma spectrometer with a low energy resolution would give useful information about long-lived radioactivity on the forest floor. An example is given on a portable high resolution semiconductor spectrometer that has suitable characteristics for field surveying also during occurrence of a great number of radionuclides contributing to the gamma spectrum. The needs for further research of a recently deposited radionuclide contamination on forest vegetation and soil, and the efforts for improvement of portable radiation meters and their use in management planning and radioecological research on contaminated forests are discussed.

Introduction

The early warning system for accidental radioactive contamination of a large geographic area is in Finland based on a nationwide monitoring network including 275 radiation meters. Close to real-time dose rates are given in $\mu Sv\ h^{-1}$. Regional access restrictions that are potentially set for the protection of people naturally apply also to forests. During the first couple of weeks after a deposition, depending on the weather, a relatively fast removal of radionuclide contamination from the tree crowns through weathering also reduces the airborne contamination, caused by resuspension of deposited radioactive material.

Nuclide-specific field surveying of radioactivity in forests after accidental atmospheric contamination can provide information on immediate or delayed contamination risk of forest products. Particularly ^{137}Cs ($T_{1/2}$ = 30 years) can be a threat to the acceptability of wild foods and timber in the long term. Field measurements are important for ensuring a realistic view of the need for reduction of contamination in trees and for selection of optimal forest management methods (Rantavaara and Aro 2003). After the most short-lived radionuclides have decayed

to an insignificant level the need for intervention can well be assessed using a portable NaI spectrometer. About ten years after the contamination due to the Chernobyl accident, ^{137}Cs was surveyed in a forested runoff area in Northern Sweden by carrying out a systematic set of measurements with a portable NaI gamma spectrometer connected to a GPS. The results were connected with data from soil samples from the selected parts of the study area (Plamboeck et al. 2006). The study showed obvious advantages of field surveying with a portable spectrometer. High resolution spectrometers are useful for more accurate surveying of long-lived gamma active nuclides on forest floor, and for detection of the remains of nuclides of medium half-life, provided that radiation from the overstorey trees does not contribute to the measured gamma spectra. The need for the management of contaminated forest can be tentatively assessed using regional model assessment results and also targeted measurements with a portable gamma spectrometer. Soil samples give reliable information on activity density in the area, but the working time and laboratory costs for a representative study would be manifold compared to the use of a portable instrument.

The aim of this study was to demonstrate the significance of timing to obtain accurate nuclide identification and a rather constant level of ^{137}Cs activity on the ground, i.e. the stage after fast removal of deposited surface contamination from the tree crowns. A further aim was to present a portable high resolution semiconductor spectrometer, potentially useful in field surveying of forests.

Contamination of forests after atmospheric radionuclide deposition

The distribution of radioactive contamination in a forest and the contribution of various forest compartments to the Kerma rate at different times can be estimated with assessment models presenting regional outputs on top of maps, and local results in tables and time plots. The information is needed for targeting and timing of field measurements and for advising people involved in forestry. For this study, the forest model FDMF (PV 6.0) of the RODOS decision support system was used (Rantavaara et al. 2001, 2004). Contamination of an advanced thinning stand of pine was assessed using post-Chernobyl data from April-May 1986. Time integrated radionuclide concentrations from the Nurmijärvi air monitoring station in southern Finland (Aaltonen et al. 1987) included data for 21 airborne gamma active nuclides detected at the end of April and on the first days of May 1986, namely [95]Zr, [97]Zr, [99]Mo, [103]Ru, [106]Ru, [110m]Ag, [127]Sb, [129m]Te, [131m]Te, [132]Te, ^{131}I, [133]I, [134]Cs, [136]Cs, ^{137}Cs, [140]Ba, [141]Ce, [143]Ce, [144]Ce, [147]Nd, [239]Np. The assessment for deposition in dry conditions resulted in decreasing activity contents of short-lived nuclides and the Kerma rates from tree compartments (Fig. 1). The initial activity distribution depends on the interception capacity of the tree crowns. As to the isotopes of iodine, their physicochemical form should be known for the model input, as it greatly influences the interception of airborne iodine by the tree crowns. Radioactive decay will reduce the inventory of short-lived radionuclides in about two months to such a level that contribution from nuclides other than [134]Cs and ^{137}Cs to the Kerma rate is small and rather constant in the time scale of months (Fig. 1).

The activities distributed in a forest in our scenario-based assessment did not assume remedial actions. However, the results demonstrate the contamination context in a forest. In cases of manifold radionuclide activities compared to the model input, management or fertilisation of forest soil might be planned to achieve reduction of ^{137}Cs contamination of timber in the long term. Before implementing any remediation programme, reliable field survey measurements should be carried out for determination of radionuclide activities in the ground layer, and their
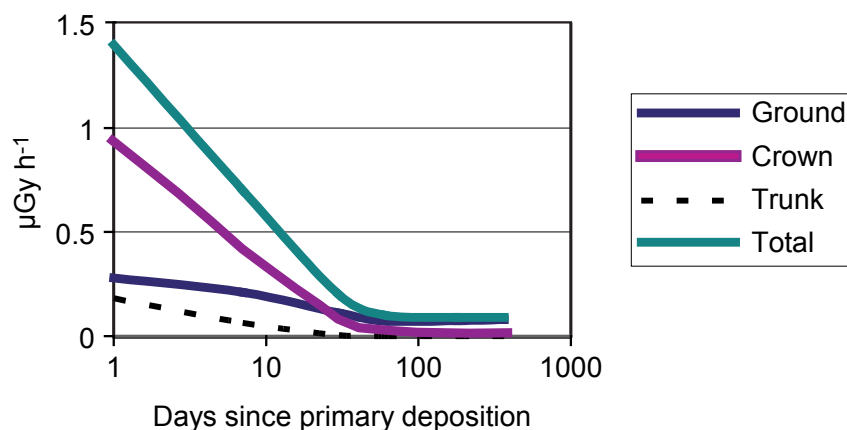
Figure 1. Kerma rate from the three compartments that comprise the radionuclide inventory in an advanced pine forest. Assessment was made with the food chain and dose model FDMF integrated in the RODOS PV 6.0 decision support system. The scenario was derived from post-Chernobyl measurements in Finland.

horizontal variation. Such data can be used for modifying forest management and to avoid poorly optimised measures.

Portable gamma spectrometer

Portable field survey instruments can be useful for the measurement of gamma active radionuclides in a forest area after radionuclide contamination of the forest. Their advantage is the fast collection of data, and the potential for estimation of radionuclide inventory of the forest floor, if the spectrometer is connected to a GPS. Furthermore, the spectrometers can show local variation of the long-lived ^{137}Cs, as shown by Plamboeck et al. (2006), who also reminded of the need to eliminate the contribution from radioactive contamination in trees to the detected radiation. They also referred to the effect of local variation in the radionuclide depth distribution in soil on the response of the detector, thus presenting a challenge to the calibration of the system and the necessity to base it on soil samples representing the true vertical distribution of [137]Cs.

After the initial deposition the radionuclide contamination in a certain area can be measured with gamma spectrometers developed for field use. The nuclide-specific activity density (in Bq m[-2]) in the ground layer of forests can be determined assuming that the radionuclides in tree canopies do not contribute to the detected activity. In the first couple of months after the primary deposition it may be necessary to use shielding of the spectrometer for elimination of radiation from the overstorey. However, there is no urgent need to survey forests for estimation of contamination risk of timber in the first days or weeks after the contaminating deposition when radioactive material intercepted by tree crowns is at maximum and short-lived radionuclides contribute to the Kerma rates and the doses received during stay in forests. For research a high resolution gamma spectrometer is an effective tool also on the first days after a radionuclide deposition. For instance, dynamics of the initially intercepted radionuclides could be studied. In this case a convenient shielding should be used to be able to measure separately the layer of overstorey vegetation and the ground.

Ortec's Detective-EX is a portable detector that can be used as a neutron and gamma nuclide

identifier. It is equipped with a 50 mm × 30 mm high purity germanium (HPGe) detector and a high efficiency neutron detector. A gamma-only version, that does not include a neutron detector, is also available. The Detective has its own user interface with a touch screen, but can also be connected to a laptop, and used with separate measuring and analysis software. The spectrometer is easy to operate, and the HPGe detector is electronically cooled, so no liquid nitrogen (LN_2) is needed. Its energy resolution is superior compared to other portable systems. The instrument is specially designed for identification of radioisotopes in illicit nuclear materials trafficking, for detection of hot particles and for searching of special nuclear material (search mode for ^{235}U, ^{239}Pu). The Detective is widely used by various public authorities in the field of nuclear security. It can identify and classify several nuclides originating from industrial, medical and nuclear facilities and also normally occurring radioactive material. The Detective could possibly be used to detect also atmospheric radionuclide contamination in forests.

Ongoing development work

Testing the Detective-EX in field conditions has been planned at STUK to take place in 2009. Therefore, a calibration of the spectrometer has to be performed, and the detection limits for different radionuclides determined. The results of the calibration and the field study will be published later.

In field conditions the quality of measurements with a portable instrument can be adequate for the primary purposes of surveying. However, a limiting factor may be the constancy of accurate moving of a rather heavy instrument (12 kg) in uneven terrain. A smaller and lighter version, the micro-Detective, with the same analytical performance and gamma radiation sensitivity is also available. The development towards even lighter portable instruments is going on. For instance, at STUK light and versatile instruments for field conditions are under development.

Discussion

After a large-scale atmospheric deposition the assessment of the radiation situation in forests using portable gamma spectrometers will develop towards more accurate results with time. In the first two months after the deposition most short-lived nuclides will decay and the intercepted radioactive material will mostly be removed from the tree crowns through natural weathering. However, the time-dependent distribution of the deposited nuclides in forest compartments has to be examined or assessed for protection of the public and people who are working in forests. The changes of radionuclide content in different compartments of forest, namely overstorey trees, understorey vegetation and soil, present a challenge to field surveying, particularly in the first weeks after the primary deposition. It is also important to base management of contaminated forests on site-specific data corresponding to the actual surface densities of long-lived radionuclides in the surface layer of the ground.

The ongoing instrument development by scientists in the field of radiation security can result in qualified radiation meters that are useful for surveying contaminated forests. Such meters would be useful not only for monitoring a site but also for supporting radioecological research. With relevant shielding such meters could be useful in studies of the dynamics of recently deposited radionuclides, not too well known. Further analysis of post-deposition processes in forests should be carried out to improve the overall picture of forest contamination. The commercially available portable semiconductor detectors corresponding the one presented in this paper could

work as an instrument for studies of forests. They could also serve as tools used in services provided for forestry experts planning remediation of contaminated forests.

References

Plamboek, A.H., Nylén, T. & Ågren, G. 2006. Comparative estimations of ^{137}Cs distribution in a boreal forest in northern Sweden using a traditional sampling approach and a portable NaI detector. J. Environ. Radioact. 90: 100–109.

Rantavaara, A. & Aro, L. 2003. Radioecology and management of contaminated forests. In: Paile, W. (ed.). Radiation Protection in the 2000s – Theory and Practice. Nordic Society for Radiation Protection. Proceedings of the XIII ordinary meeting, Turku/Åbo, Finland, August 25–29, 2002. STUK-A195. STUK, Helsinki. p. 443–449.

Rantavaara, A., Calmon, P., Wendt, J. & Vetikko, V. 2001. Forest food chain and dose model (FDMF) for RODOS. Model description. STUK-A178. Helsinki 2001. (The latest model description (PV 6.0) from 2004 is found in the internet address www.euranos.fzk.de)

Sinkko, K., Aaltonen, H., Mustonen, R., Taipale, T.K. & Juutilainen, J. 1987. Airborne Radioactivity in Finland after the Chernobyl accident in 1986. Report STUK-A56, STUK (previous name: Finnish Centre for Radiation and Nuclear Safety), Helsinki.

3.2 How to manage forests after radionuclide contamination

Aino Rantavaara

Radiation and Nuclear Safety Authority, Research and Environmental Surveillance,
P. O. Box 14, FI-00881 Helsinki, Finland

Introduction

Radionuclide contamination of forest compartments and various forest products depend on site conditions and time since the contaminating deposition. In recent decades the contamination patterns of different stands have been analysed for long-lived fallout radionuclides of caesium and strontium. Activity contents in forest understorey and overstorey vegetation, and in forest soils, are related to each other through nutrient and radionuclide flows of the forest ecosystem. Distribution of deposited radionuclides will change with time, and reaching an approximate 'steady state' can take tens of years. In fact, the integration of radionuclides in trees continues the whole period of rotation, although it becomes slower towards the end.

In boreal forests the uptake of several radionuclides from soil to vegetation is significantly higher than in agricultural systems. Wild foods are contaminated mostly by the radioactive caesium (^{134}Cs, ^{137}Cs), whereas the long-lived ^{90}Sr is of less importance to people's internal doses due to its low translocation from the green parts to the edible fraction of a plant. Also the content of strontium in muscle tissues of game animals is low. Woody parts of trees integrate both radioactive caesium and strontium slowly, while stem wood is the least contaminated part of a tree together with the wood of thick roots. However, after heavy radioactive fallout, remediation of forests is needed to avoid the contamination of wood. It is crucial to protect end users of forest products and to ensure regular delivery of acceptable saw timber and pulpwood.

Tools to construct strategies for remediation of contaminated forests in the Nordic countries were reviewed by Hubbard et al. (2002). The management options, or countermeasures, related to forest products and the use of forests have been compiled under the integrated project EURANOS of the 6th Framework Programme of the European Union (Beresford et al. 2006, Howard et al. 2005). Recently completed descriptions of management options are included in a compilation of rural countermeasures for Europe, in a logical format of fact sheets, available at http://www.strategy-ec.org.uk/EURANOS/euranos.htm. The compilation provides sufficient level of details to preliminarily evaluate the suitability of remedial measures in local conditions.

The aim of this paper is to introduce forest management options for forests contaminated by radioactive deposition. The evaluation of practicability of the measures is briefly outlined. Since the countermeasures related to wild foods are radioecologically simple and rather few options are available, they are commented only briefly. Main emphasis is on the contamination of trees and timber.

Management of the use of wild foods

Basic methods in reduction of radiation exposure of people after an atmospheric deposition of radionuclides include restriction of access to areas where external dose rates exceed the intervention value (100 µSv h^{-1}; STUK, 2001). During access restrictions forestry work, gathering of mushrooms and berries, and hunting are prohibited.

In the Nordic countries the consumption of wild foods is rather high compared to many other European countries. All types of wild foods gathered or hunted in the fallout area need therefore surveillance of radionuclide concentrations. Based on measurement results, the consumption may be locally restricted or certain types of wild foods, contaminated in excess of the set intervention values, banned for a period of time. Contamination of wild foods can be reduced by processing with normal household methods. If processing is considered as a means of intervention, the area for collection of wild foods is larger than it would otherwise be, and there is less need to change people's normal way of life.

Management of forests for optimal delivery of timber

Using contaminated stemwood as building timber and other uses of commercial timber can add to external radiation exposure of people. The dominant pathway of radiation dose is external radiation from isotopes of caesium in wood. The long-lived ^{90}Sr is a pure beta-emitter, and external doses received via wood are very low although the uptake of strontium from soil to trees can be considerable.

Three management options for forests, each to be adjusted to the prevailing site conditions, are included in the European compilation of rural countermeasures.

Selective harvesting of trees

The aim of the management option *Selective harvesting of trees* (described in the compilation of the European countermeasures for rural areas) is to optimise the purchase and use of acceptable wood available in a contaminated region. A dynamic assessment model for forests can facilitate the identification of the most contaminated production areas of stemwood, and prediction of the time period for banning harvesting of timber. Estimation of the area and duration of banning should be based on an intervention value for activity concentration of ^{137}Cs (in Bq kg^{-1}) in growing trees. Timber should be harvested through thinning or final felling at a time when the development stage of the stand is appropriate, prior to or after the assumed banning period. Selective harvesting is targeted to *stands* that are in the stage of harvesting in a shorter or longer term, not to individual trees.

Treatment of forest soil with potassium fertiliser or lime

The aim of fertilising forests contaminated by long-lived ^{137}Cs with potassium containing fertilisers is to achieve a long-term reduction in activity concentration of ^{137}Cs in timber (Rantavaara and Aro 2003). Liming has also been experimentally studied on pine saplings, and there is evidence of reduced strontium uptake by stemwood (Rantavaara and Raitio 2002).

Optimal management of contaminated forests

Intervention to avert human radiation doses due to contaminating radionuclides is at best optimised for timing, targeting and duration of each management option. For this procedure, the contamination on the site should be carefully estimated and the condition of forest examined. Forest management plans made earlier should be considered and revised to ensure that only acceptable wood for purposes defined in advance will be felled.

Profitable forestry can be carried on regardless of contamination, due to improved availability of timber through selective harvesting, or improved growth of trees after fertilisation or liming. Good planning, supported by assessments of future growth and contamination of trees, and adjustment of the remedial measures for local conditions are important. Informative instructions how to implement the measures are necessary for commitment of the forestry sector to intervention.

Practicability of management options

The management options chosen for remediation of contaminated forests must be practicable for the forests themselves, and their effects on the site should meet the following criteria, used also in other types of rural countermeasures (Nisbet and Mondon 2001):

– effectiveness in reduction of contamination of wood
– radiological impact, i.e. result in such averted doses to population where the benefit is higher than the costs of the implementation.
– costs of implementation (working time, use of machines, materials)
– technical feasibility; capacity (workers, machinery, time available etc.)
– environmental impact
– social and economic impact; acceptability
– landscape protection
– secondary costs (wastes etc.)

Stakeholders of forestry should be involved in evaluation of forest management options, for instance, to give them a chance to express their view and knowledge on practicability in local conditions. It is important to invite and activate the real stakeholders to contribute to planning of emergency preparedness of forestry in normal conditions. The continuance of evaluation process after activation should be ensured by authorities, and the motivation of the groups supported through coordination and adequate communication. During such a process the competence of the group in issues of forest radioactivity will be improved allowing further development of management options to national and local conditions.

Management options to be excluded

There are several options included in earlier suggestions for management of contaminated forests that are not relevant when sustainable management of forests and competence found in forestry sector are considered. There is no hurry to arrange other than earlier planned felling operations in the early phase of contamination. Contamination of stemwood through root uptake of radionuclides occurs slowly. There is time to carefully optimise the harvesting of trees. Clear cutting should be avoided to maintain the closed nutrient system in stands of advanced development stage. The forest is the safest in this state also as regards the runoff of radioactive material found in soil and vegetation.

Removal of contaminated biomass from forests after radionuclide deposition does not solve the contamination problems; it results huge quantities of biological waste to be managed in acceptable way, and is harmful to forest ecosystem. Methods developed for radical cleanup of industrial sites contaminated by radioactive materials, as removal of topsoil are not applicable to forests. This treatment of stands does not prove to be acceptable when sustainable forest

management is assumed (Rafferty and Synnott 1998). Defoliation of trees has also been discounted after the tests with non-lethal agents in early 1990s (Jouve et al. 1994).

Timber should not be harvested only to be disposed of as waste, but for planned commercial or industrial use or forest owners' own purposes. Intervention levels for radionuclide contamination of wood should not be set for raw wood but rather to various end products of forest industry, to avoid unnecessary wasting of forest resources.

Conclusions

Acceptability of management options applied and sustainability of management of forests during intervention are currently emphasised. Activation of evaluation processes in normal conditions is very useful to all actors involved in potential intervention. Predictive assessments can improve the comprehension on the consequences of fallout to forestry, although they do not replace field research and measurements of radioactivity. Dissemination of coherent and consistent advice to the users of forests is needed in an emergency situation.

References

Beresford, N. A., Barnett, C. L., Howard, B. J., Rantavaara, A., Rissanen, K., Reales, N., Gallay, F., Papachristodoulou, C., Ioannides, K., Nisbet, A. F., Brown, J., Hesketh, N., Hammond, D., Oatway, W., Oughton, D., Bay, I. & Smith, J. T. 2006. Compendium of countermeasures for the management of food production systems, drinking waters and forests. EURANOS (CAT1)-TN(06)-04 Deliverable D6C1R1.

Howard, B.J., Beresford, N.A., Barnett, C.L., Rantavaara, A., Rissanen, K., Reales, N., Gallay, F., Papachristodoulou, C., Ioannides, K., Nisbet, A.F., Hesketh, N., Oughton, D. & Bay. I. A revised compendium of countermeasures for the management of food production systems. In: Strand P, Børretzen P, Jølle T (eds). Proceedings from the 2nd International Conference on Radioactivity in the Environment. 2-6 October 2005, Nice, France. Østerås, Norway: Norwegian Radiation Protection Authority; 2005. p. 435–438.

Hubbard, L., Rantavaara, A., Andersson, K. & Roed J. 2002. Tools for forming strategies for remediation of forests and park areas in northern europe after radioactiv econtamination: background and techniques. Report NKS-52. NKS Secretariat, Roskilde, Denmark.

Jouve, A., Schulte, H., Maubert, H. & Bon, P. Non-lethal defoliation to impair the foliar uptake of fallout radionuclides by trees. The Scinece of the Total Environment 157 (1994): 407–410.

Nisbet, A. F. & Mondon, K.J., Stakeholder Pre-Involvement in the Post Accident Management of Rural Areas, (1) A Government Perspective. National Radiological Protection Board, Chilton, Didcot, Oxon OX11 0RQ, Food Standards Agency, 17 Smith Square, London SW1, 2001.

Rafferty, B. & Synnott, H. Countermeasures applied to forest ecosystems and their secondary effects: a review of literature. Serie Documenti 6/1998. Agenzia National per la Protezione dell'Ambiente, Roma (ANPA), Italia, 1998.

Rantavaara, A. & Aro, L. 2003. Radioecology and management of contaminated forests. In: Paile, W. (ed.). Radiation protection in the 2000s – Theory and Practice. Nordic Society for Radiation Protection. Proceedings of the XIII ordinary meeting, Turku/Åbo, Finland, August 25–29, 2002. STUK-A 195: 443–449.

Rantavaara A, Raitio H. ^{137}Cs and ^{90}Sr root uptake of pine saplings in a managed forest. In: Bréchignac, F. (ed.). Radioprotection. Proceedings Volume I of the International Congress: The Radioecology – Ecotoxicology of Continental and Estuarine Environments ECORAD 2001, Aix-en-Provence, France, 3–7 September, 2001. Radioprotection – colloques 2002; 37 (C1): CI-565–568.

STUK, Protective actions in nuclear or radiological emergency, VAL GUIDE 1.1. Helsinki: Radiation and Nuclear Safety Authority, STUK, 2001.

4 Sampling in forests

4.1. Introduction to the sampling guide prepared under the NKS project FOREST: Sampling in forests for radionuclide analysis – general and practical guidance

Elisabeth Strålberg[1], Lasse Aro[2], Agnetha H. Plamboeck[3], Aino Rantavaara[4] and Virve Vetikko[4]

[1] Institute for Energy Technology (IFE), P.O. Box 40, NO-2027 Kjeller, Norway

[2] The Finnish Forest Research Institute (Metla), Parkano Research Unit, Kaironiementie 54, FI-39700 Parkano, Finland

[3] The Swedish Defence Research Agency (FOI), Cementvägen 20, SE-90182 Umeå, Sweden

[4] Radiation and Nuclear Safety Authority (STUK), P.O. Box 14, FI-00881 Helsinki, Finland

Introduction

Guidance for sampling in forests for radionuclide analysis has been prepared for collecting samples of soil, soil solution, runoff water, fungi, understorey vegetation, trees, stand throughfall and litterfall. The guidance is based on standard, recommended or known, scientifically based sampling procedures. The motivation for preparing such a guide was the need for new data on radioactivity in forests to fulfill the requirements for environmental and dose assessments and modeling. Also, no project focusing on sampling methodology in forest ecosystems has earlier been carried out internationally within the Nordic Nuclear Safety Research (NKS) or the EC framework programmes.

The initiative to the NKS project FOREST was therefore taken in 2004 and the project initiated in 2005. In addition to preparing a guide for sampling in forest ecosystems for radionuclide analysis, the project aimed also to strengthen the Nordic collaboration in the field of forest radioecology by creating a network for radioecologists and forestry experts. A final draft of the sampling guide is currently under external review and the final guide will be submitted to the NKS by the end of 2008.

The aim of the sampling guide is to ensure improved reliability of new data sets through good documentation of field sampling and the following preparation of samples. This will improve
- the accuracy, comparability and representativeness of new data on forest radioactivity
- the reliability of model parameters derived from the data, and thereby also the assessment of radiation doses, and
- quality assurance of all stages in sampling.

The guide emphasizes sampling practices applicable in various types of boreal forests, good description of sampling site, and documentation of the origin and details of individual samples. It includes separate chapters with guidelines on sampling in the compartments soil, soil solution and runoff, fungi, understorey and overstorey vegetation, throughfall and litterfall. In annexes some examples on regional sampling, various types of radioactivity studies and monitoring procedures are referenced. The report is based on recently published general guidelines, and expertise and practical experience of the involved organizations.

The guide is intended for scientists, students, forestry experts and technicians who appreciate their radioecology projects in forests being based on sound sampling procedures. The guide hopefully encourages readers to get involved in field studies and sampling campaigns. Working in field conditions is significant for building own competence in sampling.

Some selected details from the different chapters are included below.

General aspects

Sampling is a necessary part of environmental research when new information is derived using laboratory instrumentation and methods, and has a crucial effect on the reliability of the results obtained through sample measurements. The preferred procedures/strategies for sampling will vary depending on the purpose of the study (monitoring, exposure assessments, research…).

There are several factors to consider when planning a sampling campaign, e.g. statistics, available resources, availability of sample object, timing and frequencies, detection limit, quantity of sample, composite vs. single samples, available analytical techniques, transport, storage, etc. Other factors that need to be included in a sampling plan are: what information is necessary to uniquely identify each sample, packaging (e.g. paper bags vs. plastic bags), how to collect and treat the samples to avoid cross contamination and how to secure that packaging, transportation and storage do not alter the original content of radionuclides and other substances to be analysed.

The guide also includes some references to methods for estimating sampling uncertainty and also gives some advice on special considerations for certain radionuclides or sample types (e.g. volatile compounds, adsorption of radionuclides to bottle surfaces for liquid samples etc.).

Forest definitions and guidelines for site description

The FAO (Food and Agriculture Organization of the United Nations) defines forest land as land spanning more than 0.5 hectares with trees higher than 5 meters and a canopy cover of more than 10 percent, or trees able to reach these thresholds in situ. In addition, each country has its own definition of forest land.

Site descriptions are very important when data sets from different areas are compared or when data from different areas are compelled and used in model predictions. Site descriptions should include information about the
- location (coordinates, map, elevation, topography, vegetation zone)
- site properties (soil type, hydrology, vegetation type, typical plant species and their cover percentage, site fertility)
- tree stand (tree species, main tree stand characteristics)
- land-use history if available (timing and descriptions of different forest management measures, former agricultural history, original site types of peatlands drained for agriculture or forestry)

Sampling methodology

Only sampling methodologies for soil, understorey and overstorey vegetation were presented at the seminar, although the sampling guide also includes sampling of soil solution and runoff, fungi, throughfall and litterfall. One chapter is dedicated to each compartment and the chapters include description of the compartment, sampling methodology and advice on sampling equipment.

Sampling of soil

The season to collect soil samples can vary a lot dependent on the aim of the study but usually are soil samples collected when there is no ground frost. If radioactivity in soil per ground area is required, the area and the depth of the sample must be defined. Soil density may already be well defined on the site and the area of the collected soil sample is then not necessary to estimate. Sampled aliquots from a chosen site usually need to be representative of that site at the time the samples were taken. The number of samples collected will depend on the size of the area, the choice of sampling design, but very often the financial situation will limit the number of samples and also timing and frequencies. One should not forget that the amount of a sample needed for radionuclide analysis depends on the detection limits of analytical techniques to be used. A compromise must often be established to find the best sampling strategy for the project/ programme in question.

Sampling procedure based on the use of a soil corer is described below. A soil corer can only be used if a sediment soil is sampled. When a till soil is sampled the soil corer is not useful and a spade must be used instead.

Sampling procedure:
1. Make sure the soil corer is assembled correct with the inner plastic film.
2. Drive the soil corer down in the ground to the required depth by hand or with a sledge.
3. Turn the soil corer counter-clockwise one lap and pull it out of the ground.
4. Open the corer carefully and section the soil core into shorter sections.
5. Put each section in separate plastic bags.
6. To avoid contamination between samples make sure the sampler is clean before next sample is collected.
7. Record all information for identification of the sample on the outside of the bag that contains the sample (e.g. date of sampling, sampling site, sample number, soil depths, the sampler used, and code identifying the sample in a set of samples).
8. Record all information of the sampling site needed for site description.

This method can be used if the objective is to determine total radionuclide content of the ground (in Bq/m^2 or Bq/kg). Soil samples can be pooled with other soil samples to decrease the number of samples and still get representative samples from a site.

Sampling of understorey vegetation

In boreal forests understorey vegetation consists of lichens, mosses and vascular plants such as herbs, grasses, ferns, dwarf shrubs, tree saplings and shrubs. Sources of variability

in radioactivity have to be considered in sampling vegetation and separating different parts of plants for analysis. These are spatial variation in environmental radioactivity, variation in radionuclide uptake from soil and accumulation in different plant parts, as well as seasonal and annual variation in activity concentrations in plants.

Usually the best season for collecting vegetation samples is when plants are fully developed but before senescence begins. If the aim is to determine radioactivity in vegetation per ground area, sampling should be carried out by using a frame. The frame defines the land area corresponding to ground vegetation to be sampled. The frame can be square or round in shape and made of metal, wood or plastic. Frames used for collecting samples of pasture or ground vegetation have varied in size from 25 x 25 cm to 1.5 m^2 (Isaksson 2000, Rahola et al. 1999, SSI 1999, Suomela et al. 1999). The amount of a sample needed for radionuclide analysis depends on the detection limit of an analytical technique to be used. The quantity itself will eventually depend on the treatment of samples, for instance, analysing bulk samples or sub-samples divided from bulk samples, and analysing single or composite samples. This should be taken into account when choosing a frame size or the size of (combined) vegetation sample for analysis.

The number and distribution of sampling points at the sampling site should be chosen so that to get a reliable estimate of radioactivity over the whole area. If the radioactivity distribution in vegetation is heterogeneous, then systematic sampling (grid pattern) is recommended. Random sampling can be used only for a homogeneous distribution. Guidance for designing sampling to estimate spatial or temporal variations can be found in ICRU (2006).

Sampling procedure based on the use of a frame is described below:
1. Set the frame on the ground at the selected sampling point. Avoid sampling points with large stones or roots, logs, holes, trails etc. If necessary, change the sampling point according to principles defined beforehand.
2. Collect all plants inside the frame by cutting the stems with scissors/pruning shears. Cutting height will depend on the aims of the study. Avoid contamination from soil particles.
3. Identify all plant species if needed (preferably in the field).
4. Put all plants in a bag if a bulk sample is needed, or sort them by species and put each sample in a separate bag if species-specific samples are needed.
5. Record all information of the sample needed for identification on the outside of the sample bag.
6. Record all information of the sampling site needed for the site description.

Samples (whole plants or certain parts of plants) can also be collected over the whole study area to get a representative pooled sample from a plot for determination of activity concentration in Bq kg^{-1}. In the laboratory, dead parts of plants are usually removed (and analysed if needed). Sub-samples can be made by separating different parts of plants, according to the aims of the study (e.g. leaves, stems and roots, if below-ground biomass was collected; different parts of plants of varying ages can also be separated). This will provide information on radionuclide distribution in plants.

If the objective is to determine total radionuclide content of the ground (in Bq m^{-2} or Bq kg^{-1}), soil and ground vegetation need to be collected from the same area to get total activity content at each sampling point. Associated soil and plant samples are needed also for determining radionuclide uptake from the soil. Then root zone of plants of interest define, in theory, the depth and horizontal area of soil samples. In practice, uptake has also been determined from

soil and plant samples representing the whole study area (instead of individual plants) and thus providing an estimate of an average uptake for that area.

If the objective is to examine atmospheric deposition using lichens or mosses as indicators, several methods have been used. A common criterion for selecting sampling points is that they should be located below openings in the tree canopy to avoid direct throughfall or stemflow water (ICP Vegetation Coordination Centre 2005). Bulk deposition (deposition via precipitation) in the open field can be collected in addition. Methods used within the ICP Vegetation programme (International Cooperative Programme on Effects of Air Pollution on Natural Vegetation and Crops) can be applied for sampling mosses (ICP Vegetation Coordination Centre 2005). For lichens there are no international standards or guidance for sampling available. Usually frames of various sizes are used for sampling. Lichen can be collected in one intact piece from a uniform lichen carpet. If the lichen is dry it should be watered before sampling to prevent crumbling. Soil below the lichen carpet can also be taken for a sample to get total activity content at the sampling point (e.g. Paatero et al. 1998, Puhakainen et al. 2007). In the laboratory, lichen can be divided into several horizontal fractions to get information on the vertical distribution of radionuclides (relevant for studies of intake of reindeers; see Tuominen and Jaakkola 1973).

Berries are collected over the whole study area. A representative sample can be obtained, for example, by collecting along lines set in W or X pattern across the study area (Environment Agency 2007, page 66) or along several parallel transect lines. Depending on the aims, it can be useful to collect berries from a known surface area. In that case, the number, size and distribution of sample units should be considered to get representative samples. Any extraneous material is removed from the sample when collecting. Results should be reported as Bq kg^{-1} dry weight (provide fresh:dry weight ratio). Samples should be stored in airtight containers until weighted to prevent changes in moisture content.

Sampling of overstorey vegetation

Foliage analysis provides information on the nutritional status of trees. However, the chemical analysis of materials on and in the foliage may under some circumstances provide information on the loading of different pollutants or radionuclides. Due to a fast mobility of some elements (i.e. internal translocation) it is essential to know from which part of the tree (e.g. vertical and horizontal position, needle age class) foliage samples have been collected before interpreting the results. Sampling time also affects the results of analysis.

Foliage sampling needs to be undertaken regularly and at the same phenological stage if used for monitoring purposes. This should be a requisite principle especially for monitoring changes in environmental radioactivity in forests. Usually deciduous species must be sampled in late summer (after the completion of growth but before the onset of senescence). Evergreen species are best sampled in the dormant season (i.e. from October–November to February–March depending on climatic and geographical conditions). Sampling branches conforms to foliage sampling when applicable. Normally foliage samples are collected with branches, and needles and leaves are being detached from branches in a laboratory.

At least 3–5 trees of each main species should be sampled annually for monitoring but for statistical reasons it is recommended to sample more trees (e.g. 10 trees). Due to heterogeneity in soil properties, sampling trees should be situated close to each other. A composite sample for each

tree species should be prepared in a laboratory by mixing equal quantities of foliage from each individual sample. However, if the aim of the study is to determine spatial variation in element concentrations in the study area, then each sample tree should be analysed individually.

For monitoring purposes the same trees should be sampled each year, and the trees must therefore be numbered. No trees should be felled for foliar sampling. The sampled leaves or needles should have been developed in full light. Usually, the current year needles of evergreen species are the most convenient for judging the nutrition level but, for a number of elements, comparing element concentrations in older needles with that in the current year needles may provide more useful results.

The foliage should be sampled from the upper third of the crown (Fig. 1), but not from the uppermost (1–4) whorls in conifers. For evergreen species, foliage should be sampled at least for the current year (c) and previous year (current+1, c+1). For all species, only mature leaves should be sampled. Samples should be taken from southern and western aspects of sample trees. If only one orientation will be sampled, or all orientations are represented in samples, it should always be documented in a proper way.

Finally, foliage samples can also be taken – and have been taken – in a very different way than described here. That is not restricted but it should be carefully documented when reporting the results. However, unusual sampling practices may be inconvenient when comparing results to other studies.

There is no need to give detailed guidelines on sampling devices. Any method of sampling is possible if it is suitable for planned sampling and reduces the possibility of sample contamination. Usually the needle/leaf and branch samples of living trees are collected using extendable branch cutters (up to 18 m). Samples should be taken avoiding contamination from soil surface, e.g.
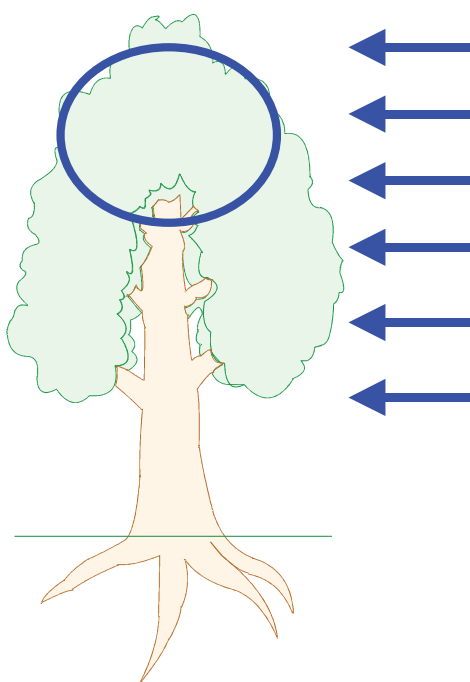
Figure 1. Foliage should generally be sampled from the upper third of the crown (circle). Other vertical positions (arrows) may be justified as well depending on study plan, but it should be documented with special care.

using plastic cover over the soil surface. Especially careful actions are needed when working with tree parts which are known to have highest concentrations of radionuclides, i.e. youngest shoots of trees and bark (for ^{137}Cs), to avoid cross contamination of samples.

Quantity of sampling material depends strongly on pre-treatment of samples and eventual analyses, and it should be estimated before starting sampling. Larger samples should be taken if samples are being archived for further analyses. The mass of 100 leaves or 1,000 needles should be determined before foliage analysis.

Concluding remarks

In the planning of the guide for sampling in forests the starting points were the research problem and the ecosystem studied. It was of particular importance in the guide project to understand the processes that change the radionuclide distribution. The natural variability in radionuclide concentrations in forest vegetation and soil makes representative sampling a challenge. The history of the site and presampling can give valuable background information in case the site has not been previously studied for radioecological purpose.

Good laboratory and sampling practices need consideration, expressly in advance of the sampling campaigns. As the aim was to prepare a guide for sampling in forests for *radionuclide analysis*, the authors emphasised preventing cross-contamination of the samples, losses of volatile or other radionuclides in samples, and also mass losses. The quantity of a sample to be analysed may vary depending on the radioanalytical methods used.

Documentation is essential at all stages, beginning from a plan for a preliminary study, description of the site, and the practical sampling instructions for a specific study in a certain type of forest. It is important to make systematic notes in the field and label the sample packages permanently. Organising the samples for storage in a practical way and in a convenient temperature to maintain the samples and analytes in their original form before the laboratory analyses is an important part of qualified research work.

Sampling is an essential part of a well-performed environmental study. Insufficient preparation of sampling will cost most to the responsible scientist who eventually will face many problems in the analysis of samples and data and in interpretation of the results. The whole chain of tasks and operations performed in the field, and later in the laboratory, until the work with the samples is completed, need careful consideration.

References

Environment Agency. 2007. Best practice techniques for environmental radiological monitoring. Science Report – SC030308/SR, Environment Agency, UK.

ICRU. 2006. The International Commission on Radiation Units and Measurements. Sampling for Radionuclides in the Environment. ICRU Report No 75. Journal of the ICRU Volume 6 No 1.

ICP Vegetation Coordination Centre. 2005. Monitoring of atmospheric heavy metal deposition in Europe using bryophytes. Monitoring manual, 2005/2006 survey. United Nations Economic Commission for Europe, International Cooperative Programme on Effects of Air Pollution

on Natural Vegetation and Crops. (monitoring manual can be obtained from http://www. icp-forests.org)

Isaksson, M. 2000. Sampling methods. A survey of methods in use in the Nordic countries. Report NKS-17.

Paatero, J., Jaakkola, T. & Kulmala, S. 1998. Lichen (sp. *Cladonia*) as a deposition indicator for transuranium elements investigated with the Chernobyl fallout. J. Environ. Radioactivity 38: 223–247.

Puhakainen, M., Rahola, T., Heikkinen, T. & Illukka, E. 2007. [134]Cs and ^{137}Cs in lichen (*Cladonia stellaris*) in southern Finland. Boreal Env. Res. 12: 29–35.

Rahola, T., Albers, B., Bergman, R., Bunzl, K., Jaakkola, T., Nikonov, V., Pavlov, V., Rissanen, K., Schimmack, W., Steinnes, E., Suomela, M., Tillander, M. & Äyräs, M. 1999. General characterisation of study area and definition of experimental protocols. WP 1 in the project 'Effect of industrial pollution on the distribution dynamics of radionuclides in boreal understorey ecosystems'. STUK-A166, Radiation and Nuclear Safety Authority, Helsinki. 37 pp.

SSI. 1999. Moberg, L., Hubbard, L., Avila, R., Wallberg, L., Feoli, E., Scimone, M., Milesi, C., Mayes, B., Iason, G., Rantavaara, A., Vetikko, V., Bergman, R., Nylén, T., Palo, T., White, N., Raitio, H., Aro, L., Kaunisto, S. & Guillitte, O. An integrated approach to radionuclide flow in seminatural ecosystems underlying exposure pathways to man (LANDSCAPE). Final report. Statens strålskyddsinstitut, SSI Rapport 99: 19. 104 pp.

Suomela, M., Bergman, R., Bunzl, K., Jaakkola, T., Rahola, T. & Steinnes, E. 1999. Effect of industrial pollution on the distribution dynamics of radionuclides in boreal understoreyecosystems (EPORA). Final report. STUK-A168, Radiation and Nuclear Safety Authority, Helsinki. 92 pp.

Tuominen, Y. & Jaakkola, T. 1973. Absorption and accumulation of mineral elements and radioactive nuclides. In: Ahmadjian, V. & Hale, M.E. (eds.). The Lichens. Academic Press, New York. pp. 185–223.

5 Environmental impact of final disposal of spent nuclear fuel

5.1 Estimates of naturally occurring pools of thorium, uranium and iodine in boreal forests of southeast Sweden

Anders Löfgren[1] and Ulrik Kautsky[2]

[1] EcoAnalytica , Haeffnersvägen 23, Hägersten 129 38, Sweden, ecoanalytica@gmail.com
[2] Swedish Nuclear Fuel and Waste Management Co (SKB), Box 250, 101 24 Stockholm, Sweden

Introduction

Long-lived radionuclides released from an underground nuclear waste repository after a failure may reach surface ecosystems and would thereby constitute a potential risk for humans and biota through different pathways. By describing the distribution of the elements in different ecosystems it is possible to describe the potential pathways to humans and biota. In most cases it is not possible to study the actual radionuclides themselves, so the distribution patterns of naturally occurring radionuclides or their stable isotopes have been used to study the long-term behaviour of the radionuclides that may originate from nuclear waste. The Swedish Nuclear Fuel and Waste Management Company is performing investigations at two potential sites for nuclear waste disposal in southern Sweden. Here is the distribution and total content of the three naturally occurring radionuclides/stable isotopes, thorium, uranium and iodine, described for six forest localities at those sites. Uranium and thorium have no known biological role, whereas iodine appears to be a trace element essential to biota. These two naturally occurring radionuclides and iodine were chosen to cover a broad span in regard to the radionuclide behaviour in the soil, i.e. having different soil solid/liquid partition coefficients (Kd), ^{232}Th and ^{229}Th (3200–89000 l kg^{-1}, Thibault et al. 1990), ^{2XX}U (15–1600 l kg^{-1}, Thibault et al. 1990) and ^{129}I (8–80 l kg^{-1}, Sheppard et al. 2006). A low Kd means a higher solubility in the soil water and is therefore potentially more bioavailable.

This study aimed at providing information concerning:
- Distribution of elements in different ecosystems
- Distribution patterns of naturally occurring radionuclides or their stable isotopes
- Potential pathways to humans and biota
- Landscape patterns – mass balances
- Transfer factors
- Dose model calibration/validation

This text will, however, only provide data and results for the first two points. For the remaining points the reader is referred to (Löfgren 2008).

Methods

The six investigated vegetation types and their location in Sweden are presented in figure 1. These localities were described in regard to a number of carbon pools (Fig. 2). For a more detailed description of the localities the reader is referred to (Löfgren 2008).
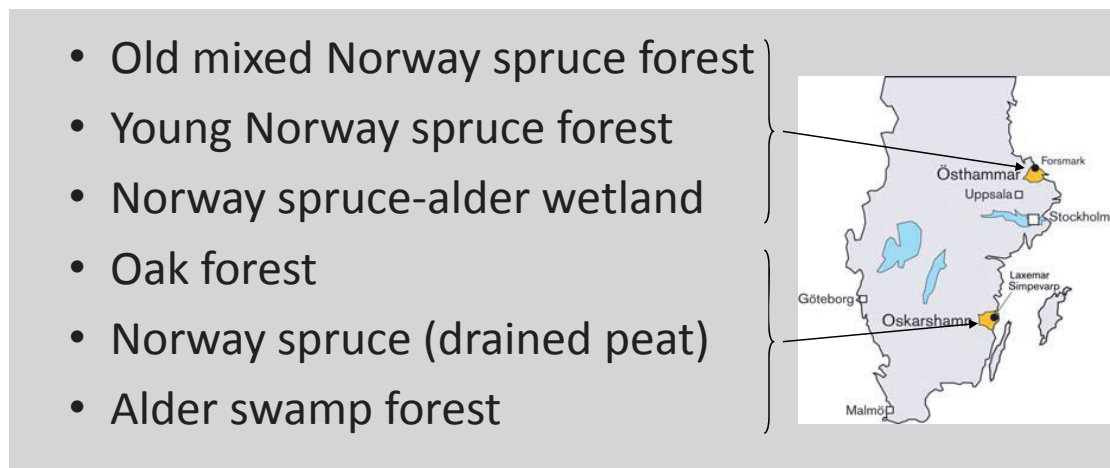
Figure 1. The six vegetation types and their location in south of Sweden.

- Old mixed Norway spruce forest
- Young Norway spruce forest
- Norway spruce-alder wetland
- Oak forest
- Norway spruce (drained peat)
- Alder swamp forest
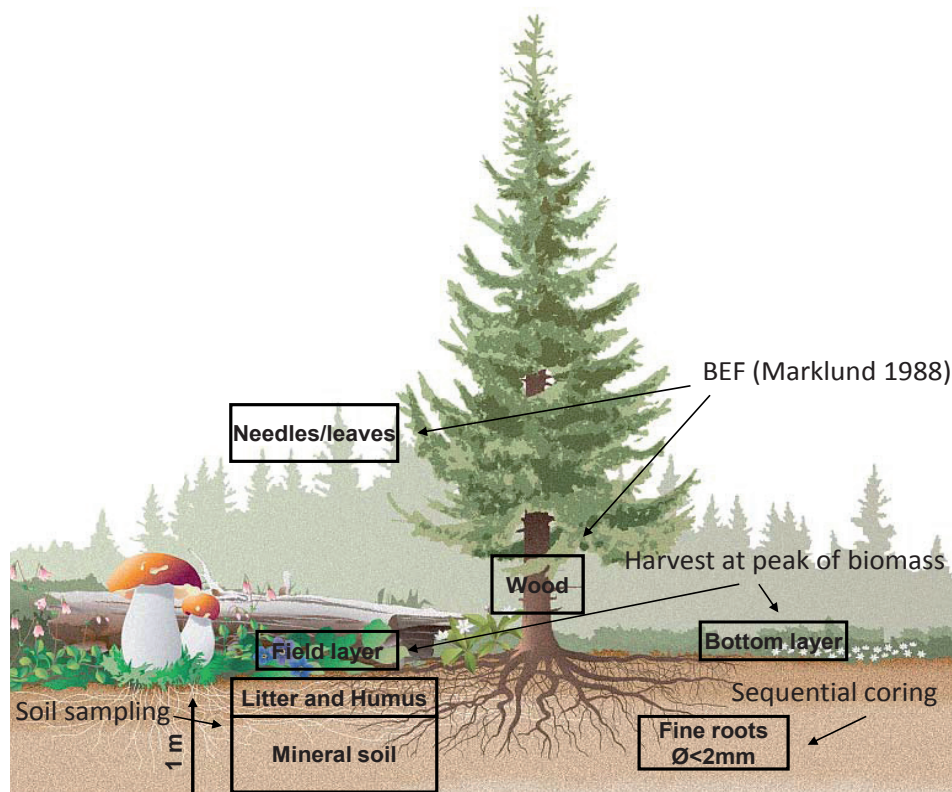
Figure 2. The ecosystem compartments that was used to contrast concentrations and content of thorium, uranium and iodine within six forest ecosystems of southeast Sweden.

Tree layer data for the different localities were derived from tree height and breast height diameter measured for ten representative trees at each locality (Tagesson 2006). For conifer trees (*Picea abies* and *Pinus sylvestris*) and birch *(Betula sp.)* Marklunds equations were used to calculate fractions of green tissue, stem and living branches (Marklund 1988). For alder (Alnus glutinosa) the equations presented in (Johansson 2000) were used. The stump, coarse roots and fine roots down to Ø 5mm and, between 5 mm and 2 mm were calculated using functions presented in Petersson and Ståhl (2006), where birch root functions also were used for alder. Fine root biomass estimates for diameters < 2mm were available for each locality (Persson

and Stadenberg 2007). These estimates were not corrected for stone and boulder content at the localities (Persson and Stadenberg 2007a). Deciduous leaf biomass was estimated using a function derived from Betula lenta (Martin et al. 1998). For oak (*Quercus robur*) equations presented in (Balboa 2005) were used to describe the oak above and below ground biomass. It was, however, unclear whether fine root biomass were included in derivation of the equations by (Balboa 2005) and the field estimates of fine root biomass for roots with the diameter less than 2 mm were added to the calculated estimates.

The derived average tree estimates were adjusted to the stand level of the locality by dividing with the average tree cross section area (m^2) and multiplying with the average basal area (m^2·ha^{-1}) resulting in a basal area weighted mean and standard deviation (Tagesson 2006).

The above-ground (AG) biomass for the field and bottom layer was investigated by collecting and measuring the biomass at the time of peak biomass. Bryophytes dominated the bottom layer were it was present (Löfgren 2005). AG biomass for the sixth locality, the young Norway spruce forest, was estimated with a regression equation between AG and below-ground (BG) biomass for the other 5 localities (Spearman r = 0.90, n=5, p=0.037) and applying that equation on the sixth locality using the BG biomass (below) as a predictor for the AG biomass.

BG biomass of fine roots was estimated by (Persson and Stadenberg 2007) who separated between tree and field layer roots in their estimations. The same location of plots was used as in the study of the field layer, hence making it possible to directly relate measured above- and below-ground biomasses of the field layer.

The litter layer thickness was investigated in five out of six localities (Löfgren 2005). The sixth locality, the Norway spruce forest, was assigned the same value as the old mixed Norway spruce forest, due to their similarities.

The soil organic carbon (SOC) pool was estimated for each locality making eight lateral transects with one humus and three mineral soil samples down to approximately one meter below the surface in each replicate, using the same methodology as the National Forest Soil Inventory (Lundin et al. 2004, Lundin et al. 2005).

The chemical sampling of the six ecosystems was described for Forsmark in Hannu and Karlsson (2006) and for Simpevarp in Engdahl et al. (2006).

The carbon content in different compartments was added together in accordance with the pools in figure 2. Element content for the pools in figure 2 was estimated using the ratio between the element and carbon concentrations from the chemical characterization of the localities representing different vegetation types (Fig. 1, Hannu and Karlsson 2006, Engdahl et al. 2006). This ratio was then multiplied with the carbon content of each compartment to give the element content of each compartment.

Results and discussion

The estimated concentrations of the three elements for the different compartment of the six ecosystems are presented in figure 3a–c. An increasing solubility was reflected as an increasing inventory in vegetation compartments (Th<U<I, Fig. 4a–c). There was no consistent pattern
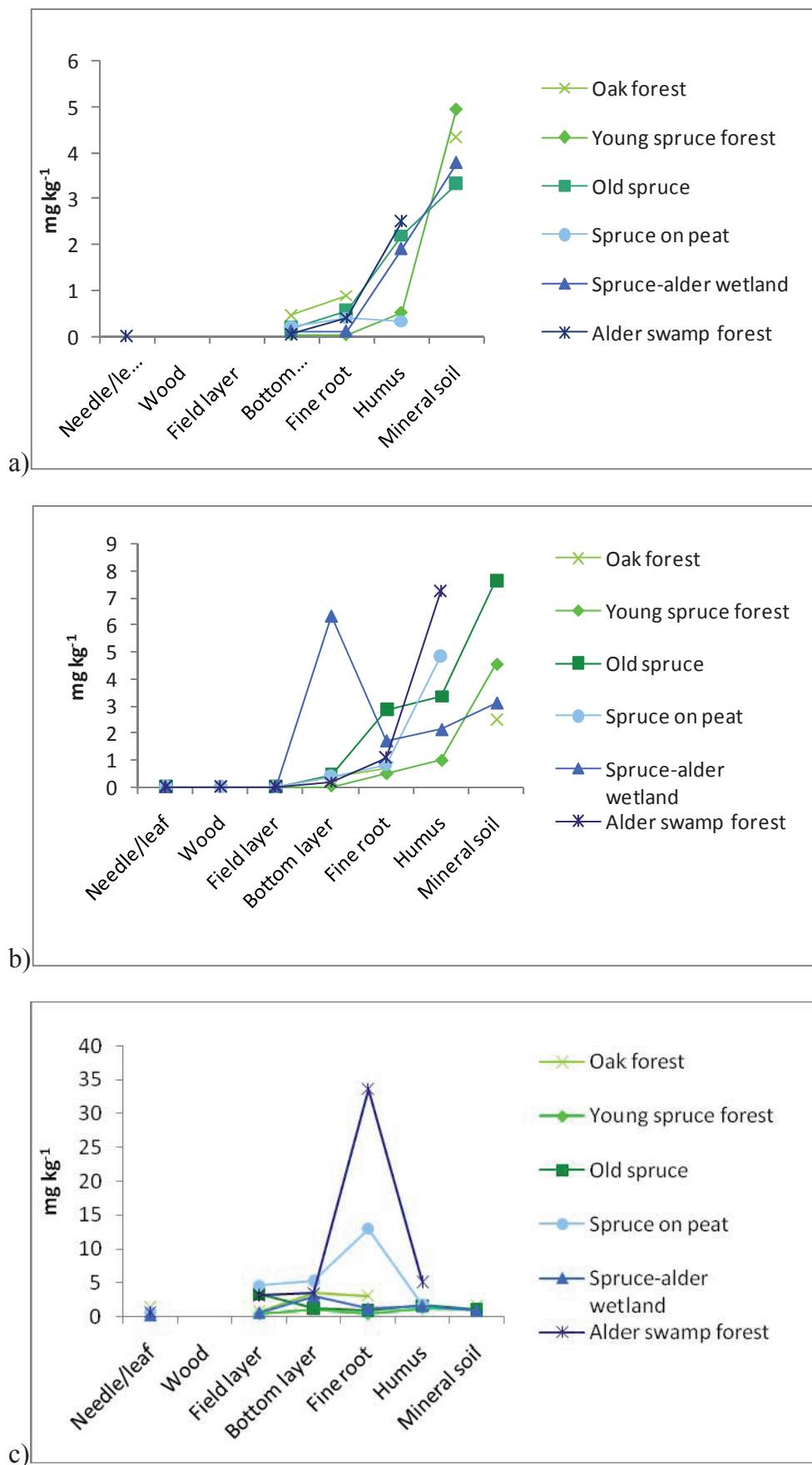
Figure 3. Measured concentrations of a) thorium, b) uranium and c) iodine in the different compartments of six forest ecosystems.
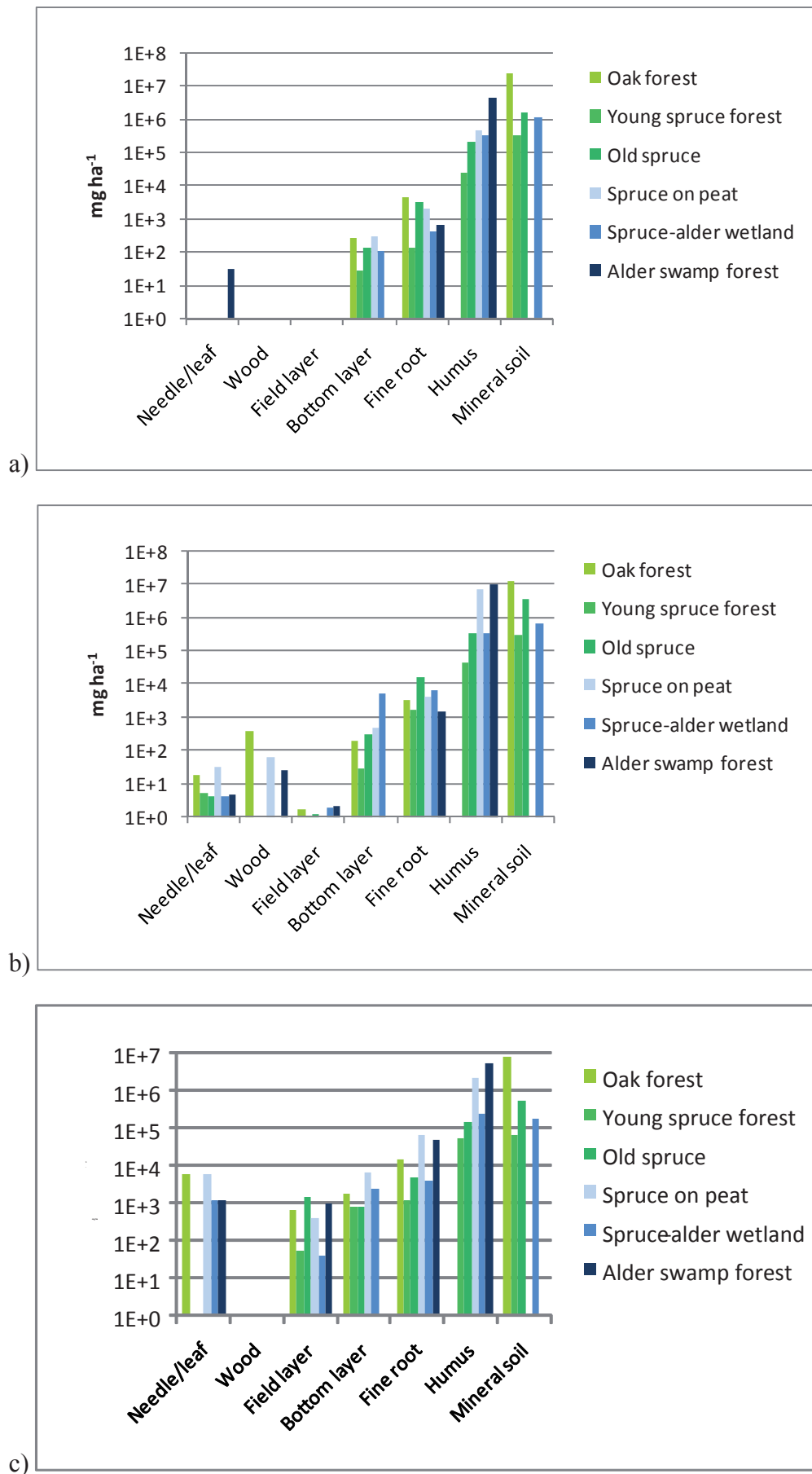
Figure 4. The estimated inventory of a) thorium, b) uranium and c) iodine in the different compartments of six forest ecosystems.

of differences between wetter and drier habitats. Both thorium and uranium are known to be associated and transported with dissolved organic carbon, which could suggest that wetter habitats rich in organic matter could act as traps for such elements.

The distribution pattern among the ecosystems did not suggest an accumulation of thorium and uranium in the humus layer. The iodine concentration was, however, higher in the humus layer for all three ecosystems where both a humus layer and a mineral soil layer were present (Fig. 3c, two of the ecosystems contained only peat soils down to 1 metre and one ecosystem lacked the humus layer). Regardless whether an element enters the ecosystem from below (e.g. weathering) or from above (e.g. atmospheric deposition) a certain bioavailability may cause a long-term accumulation of the element in the humus layer.

Fine roots had the largest inventory among the vegetation compartments for the heavier actinides, which are expected to enter the vegetation through root uptake (Fig. 4a and b). Iodine was, however, found in highest concentrations in above-ground compartments in four of the six ecosystems (Fig. 3c). Generally, the highest concentrations and content of uranium and thorium above-ground seems to be found in the bottom layer, which lack roots, suggesting a significant input via atmospheric deposition for these elements. This is in agreement with investigations and calculations of atmospheric deposition from the south of Sweden (Tyler and Olsson 2006). For iodine this was also evident, although similar concentrations and content was present in all above-ground compartments except for the stem. Iodine appears to be a trace element and it is often suggested that the iodine in plants largely originates from the atmosphere (Greger 2004).

Acknowledgement

References

Balboa, M.A. 2005. Biomasa arbórea y estabilidad nutricional de los sistemas forestales de Pinus pinaster Ait., Eucalyptus globulus Labill. y Quercus robur L. en Galicia. Doctoral Thesis. Univesidad de Santiago de Compostela. 244 pp.

Engdahl, A., Ternsell, A. & Hannu, S. 2006. Chemical characterisation of deposits and biota. Oskarshamn site investigation. SKB report P-06-320, Swedish Nuclear Waste and Management Co.

Greger, M. 2004. Uptake of nuclides by plants. SKB TR-04-14. Swedish Nuclear Fuel and Waste Management Co.

Lundin, L., Lode, E., Stendahl, J., Melkerud, P.A., Björkvald, L., Thorstensson, A. 2004. Soils and site types in the Forsmark area. SKB R-04-08. Swedish Nuclear Fuel and Waste Management Co.

Lundin, L., Lode, E., Stendahl, J., Björkvald, L. & Hansson, J. 2005a. Soils and site types in the Oskarshamn area. Oskarshamn site investigation. SKB R-05-15. Swedish Nuclear Fuel and Waste Management Co.

Hannu, S. & Karlsson, S. 2006. Chemical characterisation of deposits and biota. Forsmark site investigation. SKB report P-06-220, Swedish Nuclear Waste and Management Co.

Löfgren, A. (ed.). 2008. The terrestrial systems of Forsmark and Laxemar-Simpevarp SDM-

Site. SKB report R-08-01. In prep. Available in 2009 from the SKB web site (www.skb. se).

Löfgren, A. 2005. Estimation of biomass and net primary production in field and ground layer, and biomass in litter layer in different vegetation types in Forsmark and Oskarshamn. Oskarshamn/Forsmark site investigation. SKP P-05-80. Swedish Nuclear Fuel and Waste Management Co.

Marklund, L.G. 1988. Biomassafunktioner för tall, gran och björk i Sverige (In Swedish), Biomass functions for Scots pine, Norway spruce and Birch in Sweden. Rapport 45. Department of Forest Survey, Swedish University of Agricultural Sciences, Umeå.

Martin, J.G., Kloeppel, B.D., Schaefer, T.L., Kimbler, D.L. & McNuity, S.G. 1998. Aboveground biomass and nitrogen allocation of ten deciduoud southern Appalachian tree species. Canadian Journal of Forestry Research 28: 1648–1659.

Persson, H. & Stadenberg, I. 2007a. Distribution of fine roots in forest areas close to the Swedish Forsmark and Oskarshamn nuclear power plants. SKB report R-07-01, Svensk Kärnbränslehantering AB.

Petersson, H. & Ståhl, G. 2006. Functions for below-ground biomass of Pinus sylvestris, Picea Abies, Betula pendula and Betula pubescens in Sweden. Scandinavien Journal of Forest Research 21:7. p. 84–93.

Tagesson, T. 2006b. Indirect estimations and spatial variation in leaf area index of coniferous, deciduous and mixed forest stands in Forsmark and Simpevarp. SKB TR-06-29. Svensk Kärnbränslehantering. 36 pp.

Thibault, D.H., Sheppard, M.I. & Smith, P.A. 1990. A critical compilation and review of default soil solid/liquid partition coefficients, Kd, for use in environmental assessments. Atomic Energy of Canada Limited, AECL-10125.

Sheppard, S.C., Sheppard, M.I., Tait, J.C. & Sanipelli, B.L. 2006. Revision and meta analysis of selected biosphere parameter values for chlorine, iodine, neptunium, radium, radon and uranium. J. Environ. Radioactivity 89: 115–137.

Tyler, G. & Olsson, T. 2006. The importance of atmospheric deposition, charge and atomic mass to the dynamics of minor and rare elements in developing, ageing, and wilted leaves of beeh (Fagus sylvatica L.). Chemospere 65: 250–260.

5.2 Forest sampling and monitoring in Olkiluoto, Finland

Jani Helin

Posiva Oy, Olkiluoto, FI-27160 Eurajoki, Finland

Introduction

Olkiluoto Island has been selected as a repository site for spent nuclear fuel in Finland. In order to monitor possible environmental changes during the construction and operation of the repository and to provide radioecological models with site-specific empirical data, an extensive sampling and monitoring programme has been implemented. The forest ecosystem and its monitoring play a significant role in this multidisciplinary programme.

Characterization of Olkiluoto area

The island of Olkiluoto (ca. 12 km²) is situated off the Finnish west coast in the Bothnian Sea. The coast is characterised by shallow bays surrounded by small islands and skerries. The soil on this relatively flat island consists mainly of gravel, sand and fine-textured till. Some sedge and sphagnum peat soils are also found as well as exposed bedrock. The landscape in Olkiluoto is characterised by forests: pine, spruce, mixed coniferous, mixed deciduous/coniferous forests and deciduous forests (Fig. 1). There are some small mires and near the shore also meadows and shore scrubs. The whole local hydrogeochemical and biological system is affected by the postglacial land uplift (6 mm/y) typical of the Finnish west coast.

Summer 2008

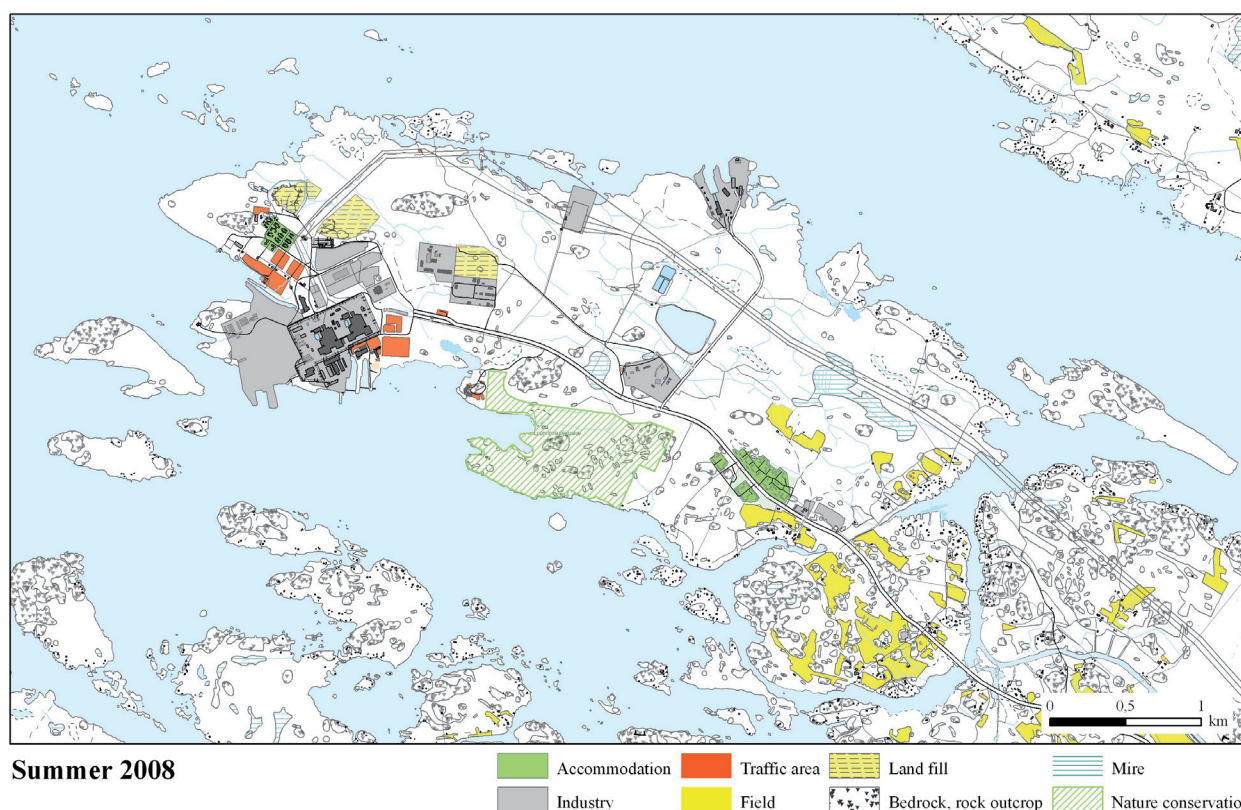|			
Accommodation	Traffic area	Land fill	Mire
Industry	Field	Bedrock, rock outcrop	Nature conservation

Figure 1. Land use on Olkiluoto Island. The white areas and the nature conservation area are mainly forested. © National Land Survey, Permission 41/MYY/08.

Environmental monitoring in Olkiluoto

On the basis of the type of the repository, including the underground rock characterisation facility ONKALO, the monitoring programme includes aspects of e.g. underground hydrogeological and geochemical parameters, while it should also reflect internationally prevailing conceptions of surface environment surveillance. Meteorological conditions, surface hydrology, the characterisation of natural habitats and the ecosystems and natural levels of radioactivity in the air, water, soil and plant life have been identified as central topics and have therefore been included in the monitoring programme (Ikonen and Rautio 2002, Posiva 2003)

Many of the studies, especially those concentrated on the marine ecosystem and radioactivity, have been conducted since the 1970s as part of the mandatory monitoring system of TVO, the company operating the two existing nuclear power plants on the island with a third power plant under construction. Posiva started studies on surface ecosystems in the late 1990s and an extensive monitoring system was established in the early 2000s. Some parts of the monitoring are continuous while some are performed annually or carried out in campaigns at intervals of e.g. 1–10 years (Haapanen 2008).

Characteristics of forest monitoring in Olkiluoto

Forest monitoring in Olkiluoto was launched in 2002. The monitoring system allows continuous follow-up of the vegetation and the soil, and the processes affecting them, as well as comparisons with areas further away from the island. The monitoring system aims at describing the present situation, as well as at detecting changes through repeated measurements, and at providing input for more advanced modelling purposes and on the other hand for e.g. environmental impact evaluations.

The monitoring system consists of several overlapping levels (Fig. 2). The first level is the assessment of the vegetation types and forest resources. The second level is the systematic forest monitoring network based on the vegetation and forest resources identified at the first level of the system. The third monitoring level comprises plots where observations are made daily or even hourly, whereas at the first and second monitoring levels samplings and measurements are carried out at intervals of several years. As the intensity of the sampling efforts increases
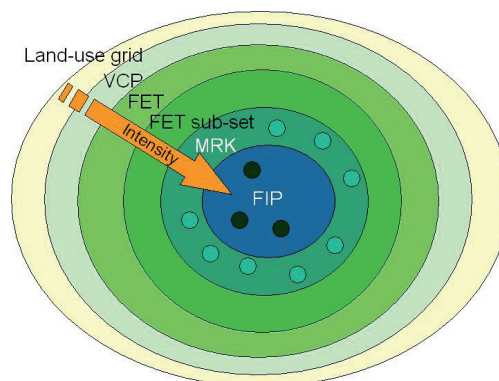
Figure 2. Forest monitoring levels. The outermost land use grid consists of plots located at 50 m intervals. These have been visually interpreted for land use. VCP contains the vegetation polygons, from which also the forest resources have been inventoried. The numbers of the established monitoring plots are 560 (FET), 94 (FET sub-set), 11 (MRK), and 3 (FIP) (Haapanen 2008).

towards the third monitoring level, the number of the monitored plots decreases and the spatial coverage of these observations is smaller than at the first and second levels, which cover most of the island.

Description of forest monitoring network

The outermost land use grid consists of plots located at 50 m intervals. These have been visually interpreted for land use. The grid is also used when tracking changes in the landscape utilising historical maps dating back to the 1600s and aerial images from 1940s onwards.

The first step in describing the forest ecosystem in Olkiluoto area to provide basics for further monitoring was Vegetation Classification and Mapping (VCP). It contains the vegetation polygons, from which also the forest resources have been inventoried. The mapping covered the main island of Olkiluoto and was performed in 2002 and the forest inventory in 2003 by the Finnish Forest Research Institute (FFRI), (Miettinen and Haapanen 2002, Rautio et al. 2004). The work was consisted of interpretation of aerial images and field work. The aim of the vegetation-type mapping was to classify the vegetation and its distribution for use as a basis in the monitoring of the primary plant succession caused by land uplift and the possible anthropogenic environmental impact. The forest inventory provided the basis for the division of the area into homogeneous parts and facilitated the optimal targeting of intensive monitoring measurements.

A permanent research frame was established as a grid network in the autumn of 2003 by selecting 560 plots around the main island from a regular 100x100-metre grid that covered the whole Olkiluoto and its surroundings (Fig. 3). The purpose of this FET (Forest Extensive Monitoring Plot) plot grid is to provide information about the volume and the quality of the tree stands in the area, as well as about the silvicultural state and occurrence of any damage. Furthermore, the FET grid provides a flexible framework for other studies, as its systematic structure is easy to expand. A comprehensive measurement of forest parameters is to be carried out at intervals of about 10 years, but a lighter inventory will be performed more frequently (e.g. Saramäki and Korhonen 2005).

A number of the FET plots have been selected for more detailed studies. Originally 94 such plots were established, but some have later been lost due to ongoing construction; this applies to the entire FET network. On these plots an inventory of the vegetation is carried out and the soil, the needles and the vegetation are sampled at intervals of 5–10 years to describe the soil properties and the nutrient balances. The first vegetation inventory and sampling was performed in 2005 by the FFRI (Huhta and Korpela 2006).

The functioning of the forest ecosystems on the island is studied on the FIP plots (Forest Intensive monitoring Plots). While studies are in the FET network carried out at specified intervals, the FIP plots include also constant measurements. The aim of the intensive monitoring activities is to continuously track changes taking place in the nutrient budgets and fluxes in the soil, tree stands and vegetation, at both the stand and the catchment level. A better overall understanding of the processes affecting the performance of the repository system and the consequences of potential releases from the repository is thus to be gained. Three plots have now been established in the Liiklansuo catchment area near the ONKALO construction site and there are plans to establish a fourth plot in the near future. One FIP plot is in a Scots pine forest, another in a Norway spruce

forest and the third in a young Norway spruce/birch forest.

The measurement activities performed on the FIP plots applying a recording or sampling period varying from one hour to several weeks include micrometeorology, stand throughfall and precipitation measurements, soil water sampling for the determination of the chemical composition and amount of the percolating water with plate and suction-cup lysimetes, litterfall and branch samplings, sap flow (evapotranspiration) measurements and diameter growth measurements with girth bands installed on two trees. Measurement activities with an observation period of one year or longer include e.g. defoliation degree, root tube investigations and nutrient status measured from needles.

The construction activities and rock crushing on the Olkiluoto Island are producing a potentially negative impact on forests, primarily in the form of stone dust. To monitor the effects on the forests, a bulk deposition and stand throughfall monitoring network of 11 plots with rainwater and snow collectors was established in 2003. The annual precipitation and interception by the tree canopies are also recorded on these plots. Three of the monitoring plots are within FIP plots and three in open areas. Rainwater is collected every two weeks and snow every four weeks and the deposition is analysed for the mean pH and the amounts of a range of anions, cations and other elements. In 2008 the network was reduced to four plots, three of them on FIP plots. Spruce and pine needles were also collected from the forested sample plots of the bulk deposition and stand throughfall monitoring network annually between 2003 and 2007 to follow the foliar element
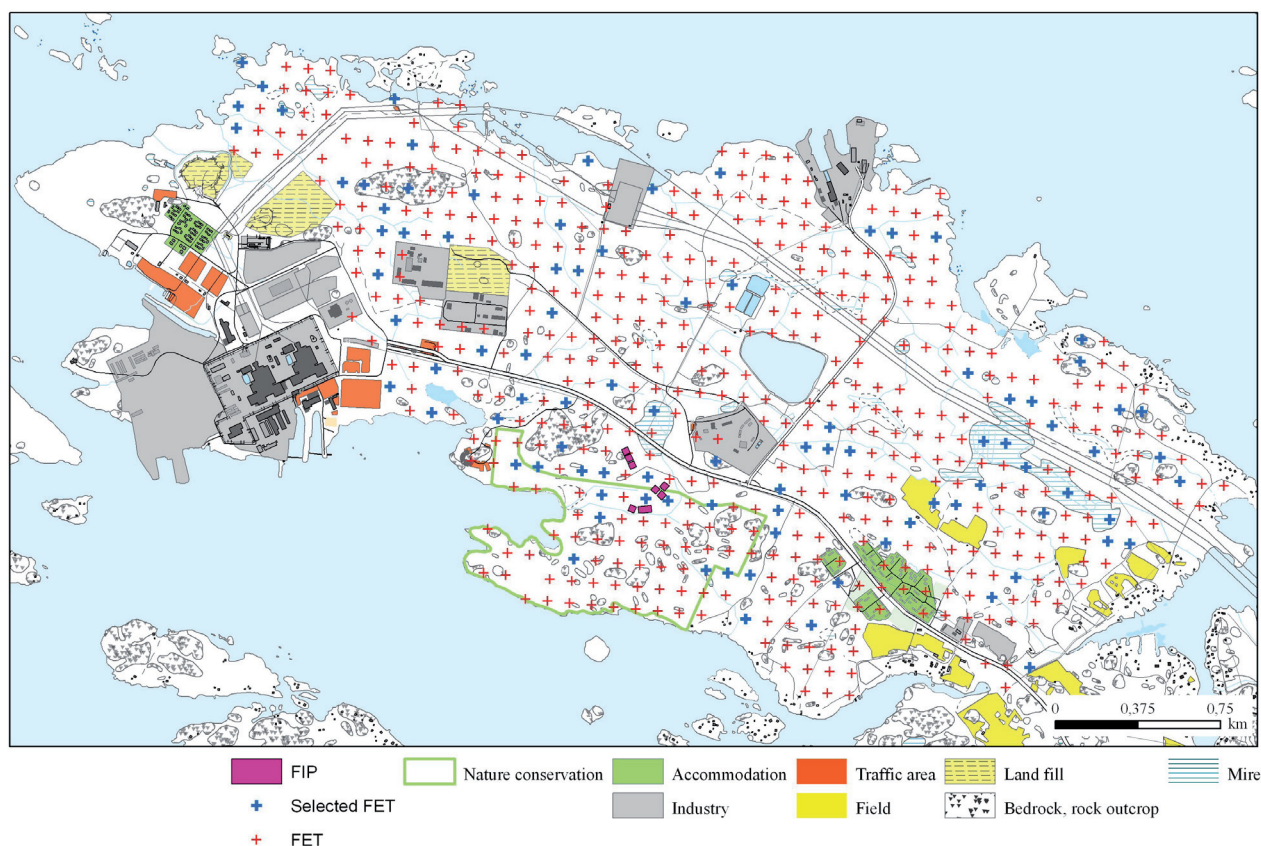
Figure 3. Forest monitoring plots on Olkiluoto Island. The map presents all the established 560 FET plots, of which 94 were selected for more detailed studies, including the plots already destroyed. Each of the 3 FIP plots is presented, consisting of 3 sub-plots. © National Land Survey, Permission 41/ MYY/08.

Figure 4. Samplers on a FIP plot. Litterfall samplers (green funnel), stand throughfall samplers (orange) and a bottle to collect soil water through lysimeters are shown.

concentrations. Special attention was paid to the assessment of the effects of particulate matter originating from the construction activities on the foliar concentrations by means of different washing procedures. Since 2007 the sampling has been carried out biannually.

Current monitoring of radioactivity in Olkiluoto

TVO runs an environmental radiation surveillance programme in Olkiluoto and in the surrounding land and sea areas to clarify possible exposure to radiation in the environment of the nuclear power plant. An extensive environmental monitoring programme was started soon after the construction works began in the late 1970s, and partly even before that. In the scope of the final disposal of spent nuclear fuel, these data constitute a part of the baseline database (i.e. conditions prior to any major facilities) and build up the basis for the environmental surveillance programme of the disposal facility. The information can also be used in the site-specific safety assessments (Roivainen 2005).

TVO's environmental radiation surveillance programme encompasses monitoring of aerosols and aerial deposition as well as the terrestrial environment, terrestrial foodstuff and the marine environment, including water, fish, fauna and sediment. Terrestrial environment samples are taken every four years from the soil profile and once or twice during each growing period from grazing crass, hair-moss, reindeer-lichen, birch leaves, fern and pine needles. The monitoring programme for terrestrial environment covers gamma nuclides, Strontium-89 and Strontium-90 (Roivainen 2005).

Radioecological studies in Olkiluoto

Before the operation of the repository can start in 2020s, a safety case has to be drawn up to substantiate the safety of the disposal solution. This safety case also involves the assessment of impacts on organisms other than human. Several radionuclides have a stable chemical analogue and analogues often display similar behaviour in the forest ecosystem and before more detailed

analyses of radionuclides are available, the non-radioactive elements will provide a sound basis for radioecological models, especially as data have already been collected from the forests on Olkiluoto. Using the data gathered from the monitoring plots, the migration of the elements in the food webs of the Olkiluoto forest ecosystem is studied. The objective of these studies is to produce concentration ratios, describing specific radionuclide activity in the plant tissue of interest and the corresponding activity in the soil between the various sub-components of the forest ecosystem and to utilise these results in modelling potential doses from the repository (e.g. Roivainen 2006).

In 2005 mineral soil, peat, needle and understorey vegetation samples were collected from 94 FET plots by the FFRI and analysed using ICP-AES (Tamminen et al. 2007). Aro et al. (2008) used these data to calculate concentration ratios for Al, B, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Ni, P, S and Zn between the plant and the organic layer or the uppermost (0–10 cm) peat layer. The total number of individual CR values (plant-soil sample pairs) in this study was 5783 with 1163 values for tree foliage, and 4620 for understorey vegetation.

Also in 2005 three alder stands on the shores of Olkiluoto were used as monitoring plots by Roivainen (2006). From the three monitoring plots humus, mineral soil, understorey, plant root and forest litter samples and samples of small mammals were taken. Earthworm samples were also to be collected from the monitoring plots, but none could be caught. All the collected samples were analysed for element concentrations using the ICP-AES method for Na, P, Ca and Cu. The soil and plant samples were also analysed with a gamma spectrometer for activity concentrations of gamma active radionuclides Cs-137, Cs-134, Be-7 and K-40.

Future development of forest monitoring

Large scale construction work related to the third power plant unit and to Posiva's underground research facility ONKALO is currently under way on Olkiluoto Island. A fourth power plant unit is planned near the existing units. The construction work is concentrated on the western part of the island but the eastern part of Olkiluoto is also affected with e.g. new roads planned for a harbour on the north side of the island and for the power plants further south. The construction activities compromise the integrity of the forest monitoring network when plots are destroyed. The replacement of the lost plots is also difficult as long-term certainty of future plans cannot be obtained when decisions for the studies should be made.

As in 2008, the biggest concern is the new road planned near and partly over the FIP plots and several intensively studied FET plots will also be lost. Replacements for some of these plots are already under consideration and a new FIP plot will probably be established in the western point of the island, which is protected from construction works by the land use plan. If established, this plot would provide much wanted data on alder stands, because littoral alder stands are an important biotope as recipients of the potential radionuclide contamination from the repository. All in all, new coniferous monitoring plots are needed in particular, but these are difficult to find in Olkiluoto. Excluding the new planned FIP plot, the monitoring network is most likely to be further developed into the eastern part of the island, where large-scale changes in the landscape are less probable.

New analyses will include several radionuclides that have proven to be of specific interest in recent modelling. These elements include e.g. I-129, Se-79 and Cl-36. New sampling

programmes on the existing and future monitoring plots are needed to provide radionuclide dose models with adequate input data for modelling potential doses from the repository.

References

Aro, L., Ikonen, A.T.K & Helin, J. 2008. Concentration ratios for chemical analogues of key nuclides for different vegetation types at the Olkiluoto site. In: Strand, P., Brown J. & Jølle, T. (eds.) Proceedings of the International Conference on Radioecology & Environmental Radioactivity, 15-20 June 2008, Bergen, Norway. Posters, Part 2: 31–34.

Haapanen, R. 2008. Results of monitoring at Olkiluoto in 2007: Environment. Posiva Oy, Posiva Working Report 2008-25.

Huhta, A-P. & Korpela, L. 2006. Permanent Vegetation Quadrats on Olkiluoto Island. Establishment and Results from the First Inventory. Posiva Oy, Posiva Working Report 2006-33.

Ikonen, A.T.K. 2002. Meteorological data of Olkiluoto in period of 1992–2001. Posiva Oy, Posiva Working Report 2002-44.

Ikonen, A.T.K. & H.J. Raitio. 2002. Deep crystalline rock repository of spent nuclear fuel and environmental monitoring at the Olkiluoto site, Finland. In: Børretzen, P., Jølle, T. & Strand, P. (eds.) 2002. Proceedings from the International Conference on Radioactivity in the Environment. 1-5 September 2002, Monaco. pp. 250–253.

Miettinen, T. & Haapanen, R. 2002. Vegetation Types on Olkiluoto Island. Posiva Oy, Posiva Working Report 2002-54.

Posiva 2003. Programme of Monitoring at Olkiluoto During Consturction and Operation of the Onkalo. Posiva Oy. Posiva Report POSIVA 2003-05.

Rautio, P., Latvajärvi, H., Jokela, A. & Kangas-Korhonen, P. 2004. Forest Resources on Olkiluoto Island. Posiva Oy, Posiva Working Report 2004-35.

Roivainen, P. 2005. Environmental Radioactivity Data of Olkiluoto in 1977–1983 and 2002–2003. Posiva Oy, Posiva Working Report 2005-26.

Roivainen, P. 2006. Stable Elements and Radionuclides in Shoreline Alder Stands at Olkiluoto in 2005. Posiva Oy, Posiva Working Report 2006-70.

Saramäki, J. & Korhonen, K. 2005. State of the Forests on Olkiluoto in 2004. Comparisons between Olkiluoto and the rest of Southwest Finland. Posiva Oy, Posiva Working Report 2005-39.

Tamminen, P., Aro, L. & Salemaa, M. 2007. Forest soil survey and mapping of vegetation nutrition on Olkiluoto Island. Results from the first inventory on FEH plots. Posiva Oy, Posiva Working Report 2007-78.

5.3 Present gaps in knowledge on fate and transport of radionuclides in forests - case of spent nuclear fuel repository

Ari T. K. Ikonen

Posiva Oy, Olkiluoto, 27160 Eurajoki, Finland

Introduction

Olkiluoto Island on the Finnish coast of Baltic Sea has been selected as a disposal site for spent nuclear fuel. Within the repository programme, extensive site investigations and biosphere assessments on the risk (dose) to the public from potential releases from the repository to the ecosystems have been carried out. In these assessments, forest ecosystems play a significant role as they can receive the releases directly or as inherited contamination from the former sea bottom and have a relatively high dose/release ratio (dose conversion factor). The models, data and assessment assumptions this far have been overly pessimistic, but there are reasons to believe that the significance of the forest objects in these assessments will prevail also after revisions into more realistic direction.

Material and methods

Modelling of changing landscape

Since the site is located in an area of significant land uplift, about 6 mm/y, the terrain and ecosystems will experience a major change in the forthcoming millennia. The change has been modelled by adjusting the topographic model (Ikonen et al. 2007) using the approach of Påsse (1996) with site-specific parameters from (Löfman 1999), where the apparent sea-level change has been described with two components: the isostatic uplift of the crust and the associated eustatic sea level rise due to the change in basin geometry. There are a number of versions of Påsse's model but those all give similar results to the relevant study area. The location of surface water bodies have been identified using conventional GIS (Geographic Information System) tools as further described in (Ikonen et al. 2007).

Biosphere descriptive model

Data on the overburden, flora and fauna have been collected into a Biosphere Description (Haapanen et al. 2006), which has been utilised in this study. Especially the surface sediment map of the sea bottom (Rantataro 2001) based on acoustic-seismic soundings conducted in the area has been used. The information on the forest types has been used more indirectly, and the forest monitoring network is presented in another contribution to the seminar (Helin, this publication). In addition to the Biosphere Description, the bedrock lineament data of (Korhonen et al. 2005) and the bedrock surface release locations of the groundwater visited the planned repository volume (Smith et al. 2007) support the reasoning presented below.

Radionuclide transport in the biosphere

For simulating the potential releases from the repository, a landscape model consisting of several

ecosystem-specific biosphere objects interconnected following the main water flows has been constructed (Broed et al. 2007). The biosphere objects are identified and delineated on the basis of reasonably homogeneous properties within each and the surface runoff estimates. To identify the forest objects, only the areas downhill from the assumed release locations were considered. The forest sub-model was as in (Avila 2006): the transfer between the single soil compartment to the tree wood, foliage and understorey is described using concentration ratios and growth rates, the transfer to litter and further to soil with loss rates, and the runoff from the soil the sorption properties and hydrological balance. To estimate the doses, the methods of (Avila and Bergström 2006) were applied.

Modelling results

Changes in the landscape and locations of releases to the biosphere

Figure 1 presents the predicted far-future landscape at the Olkiluoto site (after some millennia the coastline is out of the figure) with the potential surface release points of the groundwater visited the repository volume. There is a tendency for the releases to attach to the bedrock structures, or lineaments in this case, i.e. hypothetical bedrock structures, whose location or properties has not been confirmed by direct observations; although majority of the release points follow the surface waters or lowlands, some of the releases might occur also closer to the repository. In the figure the clay sediments are presented as potentially arable land. The other sediment types, mainly till or rock, have a low probability of being cultivated in the region. Thus, if no
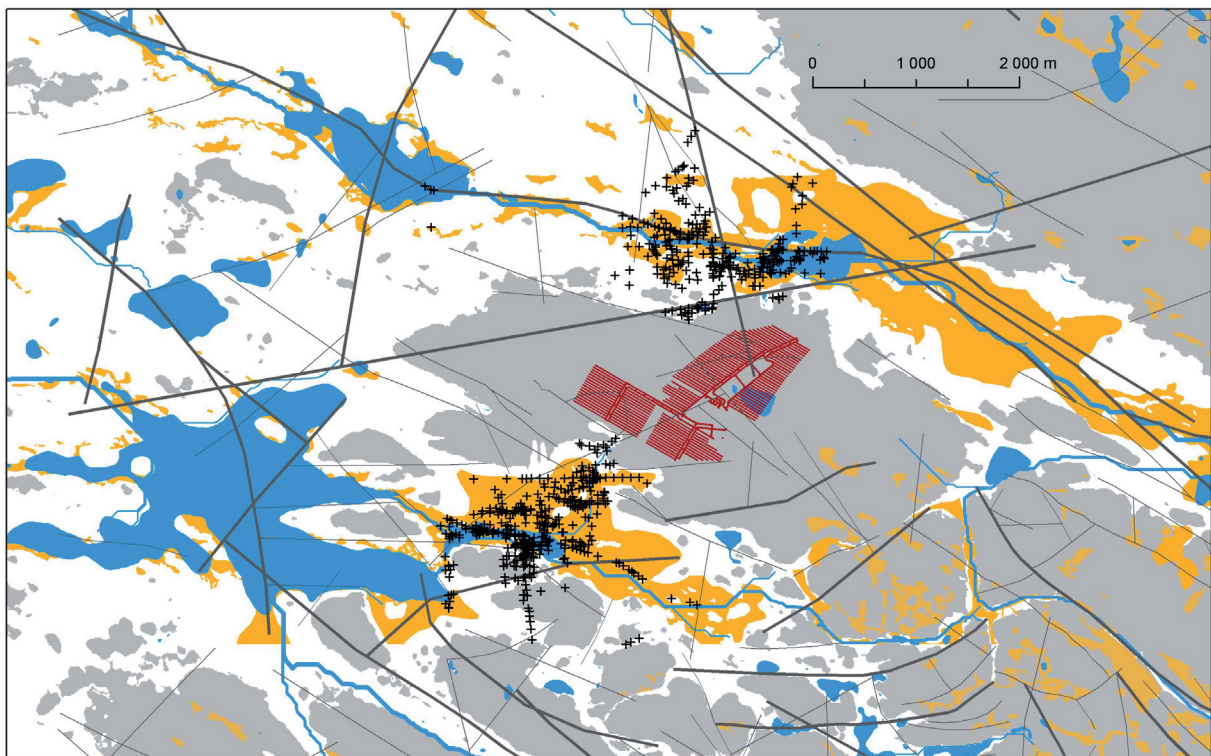
Figure 1. Potential release locations in a far-future landscape of Olkiluoto (+ signs). The gray area presents the land at present, present and predicted water courses are in blue and potential field areas in yellow. The gray lines show the bedrock lineaments and the red the repository layout used in the groundwater flow analysis.

other land use is assumed, at least the land area shown in white or gray would grow forest. The bottom sediment type of the lakes is not shown, but it is mainly clays as well. Furthermore, the figure describes the situation before the extensions of the repository to the east, which would likely result in increased number of release points out of the aquatic systems.

Radionuclide transport

In the cases analysed in the KBS-3H biosphere assessment (Broed et al. 2007), in nearly all cases the highest dose to an exposed individual are from few forest objects and from I-129, Cl-36 and C-14. In some cases also Mo-93 and Se-79 give significant doses. The situation does not change in practise even if different release patterns (combination of release locations) are assumed. In the average doses, estimated following the approach of (Broed and Ikonen 2008), the forests have a smaller role since only few persons can be exposed, limited by the food productivity in forests, and the average dose from the whole landscape in general is about five orders of magnitude smaller than from the forests alone.

Concerning the radionuclide inventories, for example of I-129, majority of the activity after 10 000 years of release is found in a forest, and in three times smaller extent in a wetland. Inventory in the other objects is marginal. Again, the situation is similar for all major nuclides and assessment cases. Additional calculations show that within the forest objects, after a chronic release of Cl-36, I-129, Nb-94 or Cs-135, the inventory retained in the area is mostly in the soil compartment (92–99.9%) for any of the nuclides, though most of the release is first flushed out of the object by runoff (98–99.9%) or decays (up to 64%). From the total release, only 0.02–6.7% is found in the organic compartments, of which the majority is in the litter layer and less than 20% in foliage and wood.

By comparing the ecosystem-specific dose conversion factors (EDF, the maximum dose in a case of direct release of 1 Bq/y to the object in question for 10 000 years), it can be concluded that for the most important nuclides listed above the terrestrial systems in general give significantly higher doses than the aquatic ones. On the other hand, doses from the forest, a wetland and a field are within one or two magnitude from each other and the forest and field give usually rather similar numbers. Depending on the nuclide, lakes can be close to the terrestrial systems, though, but especially rivers and coastal areas result in 2–5 orders of magnitude lower doses than the terrestrial systems.

Discussion

Forests as receiver of potential contamination

As presented above, the land uplift will cause a major change in the ecosystems at the Olkiluoto site within some millennia, which is less than the time frame when potential released from the repository at the depth of about 400 meters can reach the surface. In the bedrock the contaminated groundwater moves mainly along the fractures, and the extent of such flow is limited by major hydraulical features ("groundwater highways"). The potential release locations are covered by overburden, which has its role in the contaminants' way to the surface waters. This is often omitted in the biosphere assessment where the contaminants are assumed to arrive directly to the water body or a rather thin soil or sediment layer. In case of tight-bottomed lake or coastal area, the releases might take a longer but energetically shorter route around the bottom and end

up to the shoreline. Another mode of releases reaching forests directly is entering to the rooting zone. These forests would be rather moist as well since the unsaturated soil cannot be too thick. Drier conditions cannot be excluded either since the uplift causes former, possibly contaminated, sea bottom to emerge as dry land. Analysing these multiple pathways or release modes needs effort, but with the recently development modelling tools and approaches the uncertainties can be decreased with time and work. Apparently, forests will keep their role in the radionuclide transport and dose analysis anyway, only their properties might get better predicted.

Knowledge and data base

In the site-specific biosphere assessment the transport of contaminants from bedrock to the bioavailable region is a key feature that usually is ultimately simplified. With present surface hydrology modelling tools and approaches, it could be taken into account much better. Then also the types of forests likely receiving the potential releases could be predicted more accurately. The radionuclide model of forests itself has also potential for improvement by better representation of the soil-plant and internal plant cycling processes. This can be achieved gradually by utilising intensive monitoring of suitable forest systems (Helin, this publication), detailed process-level models and other information where valid, and possibly by experimental studies, although their results are often difficult to fully understand or scale to the desired level.

The data used in the assessments is often rather pessimistic, and the degree of conservativeness is hard to quantify. By defining the model parameters appropriately, acquisition of the data becomes rather straightforward. However, achieving valid time series and covering the spatial variability implies long-term extensive research. Using chemical and biological analogues and suitable reference locations seems thus reasonable provided that the parallels are adequately justified. In many occasions the exactly desired data, for example from the shoreline conditions (i.e. deciduous forests), is lacking, and weaker parallels have to be used in order to reach at least weaker conclusions.

Concluding remarks

The forest ecosystems are an important part for assessing the transport and fate of radionuclides potentially released from the spent nuclear fuel repository and the subsequent doses to the public inhabiting the site in far future. However, applications of the forest models to such assessments are rather new, and thus there is potential for improvements both in the knowledge and the data base. In the process understanding the focus should be on overall ecosystems: how the system of soil, biological production and storage, and consumer works together at mass balance level – this is the basic question in the ecosystem ecology, unfortunately seldom fully adopted in the radioecological research. Concerning the data basis, there are new good-quality data emerging both from the site, comparable reference sites and elsewhere, but gaining an adequate basis requires sufficiently long time series and coverage of spatial variability - such data is extensive and the effort of proper data analysis should not be underestimated. The stable elements can work as chemical and biological analogues and are readily available for sampling unlike most of the radionuclides of interest. All in all, introduction of site-specific landscape modelling has demonstrated the need for including forests in the biosphere assessments and also their potential importance. Due to the relative novelty of such considerations, the models and data applied for the forest objects is pessimistic, and further, well-focused, research is needed to reach the optimally conservative level.

References

Avila, R. 2006. Model of the long-term transfer of radionuclides in forests. Technical Report TR-06-08. Swedish Nuclear Fuel and Waste Management Co. (SKB), Stockholm, Sweden.

Avila, R. & Bergström, U. 2006. Methodology for calculation of doses to man and implementation in PANDORA. Posiva Working Report 2006-56 and Report SKB R-06-68. Posiva Oy, Olkiluoto, Finland and Swedish Nuclear Fuel and Waste Management Co. (SKB), Stockholm, Sweden.

Broed, R., Avila, R., Bergström, U., Hjerpe, T. & Ikonen, A. T. K. 2007. Biosphere analysis for selected cases in TILA-99 and in the KBS-3H safety evaluation, 2007. Posiva Working report 2007-109. Posiva Oy, Olkiluoto, Finland.

Broed, R. & Ikonen, A.T.K. 2008. Analysis of effect of groundwater discharge pattern assumptions and multiple canister failures to potential doses from a spent fuel repository. In: Strand, P., Brown, J. & Jølle, T. (eds.). Proceedings of the International Conference on Radioecology & Environmental Radioactivity, 15–20 June 2008, Bergen, Norway. Oral and Oral Poster Presentations, Part 1: 229–232.

Haapanen, R., Aro, L., Ilvesniemi, H., Kareinen, T., Kirkkala, T., Lahdenperä, A.-M., Mykrä, S., Turkki, H. & Ikonen, A. T. K. 2007. Olkiluoto Biosphere Description 2006. POSIVA 2007-02. Posiva Oy, Olkiluoto, Finland.

Ikonen, A.T.K., Hjerpe, T., Aro, L., & Leppänen, V. 2007. Terrain and ecosystems development model of Olkiluoto site, version 2006. Posiva Working report 2007-110. Posiva Oy, Olkiluoto, Finland.

Korhonen, K., Kuivamäki, A., Paananen, M. & Paulamäki, S. 2005. Lineament interpretation of the Olkiluoto area. Posiva Working report 2005-34. Posiva Oy, Olkiluoto, Finland.

Löfman, J. 1999. Site scale groundwater flow in Olkiluoto. POSIVA 99-03. Posiva Oy, Olkiluoto, Finland.

Påsse, T. 1996. A mathematical model of the shore level displacement in Fennoscandia. Technical Report TR-96-24. Swedish Nuclear Fuel and Waste Management Co. (SKB), Stockholm, Sweden.

Rantataro, J. 2001. Acoustic-seismic research in the sea area near Olkiluoto in the year 2000 (in Finnish with an English abstract). Posiva Working Report 2001-11. Posiva Oy, Olkiluoto, Finland.

Smith, P., Nordman, H., Pastina, B., Snellman, M., Hjerpe, T. & Johnson, L. 2007. Safety assessment for a KBS-3H spent nuclear fuel repository at Olkiluoto – Radionuclide transport report. POSIVA 2007-07 and SKB R-08-38. Posiva Oy, Olkiluoto, Finland and Swedish Nuclear Fuel and Waste Management Co. (SKB), Stockholm, Sweden.

6 Recent and ongoing studies on radionuclides in forests

6.1 [237]Np in peat and lichen in Finland

Susanna Salminen[1], Jussi Paatero[2], Per Roos[3] and Kerttuli Helariutta[1]

[1] Laboratory of Radiochemistry, P.O. Box 55, 00014 University of Helsinki, Finland
[2] Finnish Meteorological Institute, Arctic Research Programme, P.O. Box 503, FI-00101 Helsinki
[3] Risö National Laboratory, P.O. Box 49, DK-4000 Roskilde

Introduction

Atmospheric nuclear tests, nuclear fuel reprocessing and single nuclear events, such as the Chernobyl accident, have released [237]Np, [241]Am (mother nuclide) and [241]Pu (grandparent nuclide) to environment. Np has several, easily changeable valence states and due to its most typical pentavalent state, it is more mobile than other actinides in environment. Long half-life of [237]Np (2.14 Ma) and constantly increasing inventory from both direct [237]Np emissions and production by the decay chain [241]Pu → [241]Am → [237]Np lead [237]Np to be the dominating transuranium isotope in environment in the distant future. [237]Np has not been found to enrich in marine food chains, but marine and especially terrestrial food chains have not been studied extensively. Certain vegetable and grass have been found to uptake Np in greater extent than Pu or Am (Schreckhise and Cline 1980, Pavlotskaya et al. 1991), therefore it is possible that Np could enrich in some terrestrial food chain.

In this work, an analytical method was developed for separating Np from large sample masses. Activity concentration of [237]Np was determined from selected peat and lichen samples collected immediately after the Chernobyl accident in Southern and Central Finland (12.–14.5.1986). Furthermore, the contributions from the nuclear weapons testing in 1950–1960s and the Chernobyl accident to [237]Np inventory in Finland were evaluated. Activity concentrations of Pu isotopes, [241]Am, [244]Cm and various gamma emitters have been determined from the same peat samples previously (Paatero et al. 1994, Salminen et al. 2005, Jantunen et al. 1991, Paatero et al. 2007).

Material and methods

40-100 g of dried and homogenised peat and 20 g of lichen was used for analysis. Np was separated from sample matrix and disturbing radionuclides by ashing, wet ashing with concentrated acids, calcium oxalate co-precipitation and TEVA® column separation (Eichrom). Mass concentration of [237]Np in samples was determined by HR-ICP-MS (Micromass Plasma Trace 2). Chemical recovery of Np was determined by adding [235]Np tracer solution to samples before wet ashing. Low-energy x-rays and Auger-electrons from [235]Np were measured with the liquid scintillation counter Quantulus 1220. The counting efficiency of Quantulus 1220 was 33(±1)% for L-shell transitions (12–21 keV) of [235]Np.

Results and discussion

The recovery of Np was close to 100% in tracer experiments and 79 % (median value) with actual samples. A large sample mass and high uranium concentration in many samples complicated the analysis and another TEVA® column separation is recommended to reduce U concentration considering the ICP-MS measurement of ^{237}Np. In recovery determination, the major problem was relatively high background count rate of scintillation cocktail Opti Phase HiSafe 3. The analytical method needs further development to improve the separation efficiency of Np from U and reduce background count rate with LSC.

Activity concentration of ^{237}Np in peat was 0.48(±0.01)–3.84(±0.08) mBq/kg or 1.98(±0.05)–14.1(±0.03) mBq/m² (Table 1). For comparison, activity concentrations of $^{239+240}$Pu, ^{241}Am and ^{244}Cm in the same peat samples were 0.022(±0.003)–1.779(±0.087) Bq/kg (Reponen et al. 1993), 0.011(±0.002)–0.81(±0.03) Bq/kg and <0.0005–0.255(±0.017) Bq/kg (Salminen et al. 2005), respectively (reference date 1.5.1986). The activity concentration of ^{237}Np in peat is orders of magnitude lower compared to other transuranium nuclides. Activity concentration of ^{237}Np has been determined to be 0.08(±0.01)–2.08(±0.17) mBq/kg in Swedish lichens collected in 1986–1988 and in the most heavily ^{137}Cs-contaminated areas, the fraction of Chernobyl-originated ^{237}Np was 5–30% (Lindahl et al. 2004). The results of Finnish peats and Swedish lichens are in agreement.

^{237}Np/^{239,240}Pu activity ratios in global nuclear test fallout and Chernobyl fallout can be calculated to be 0.0037 and 0.000125, respectively (Beasley et al. (1998), UNSCEAR (2000)). In peat samples the ratio was 0.00145(±0.00009)–0.00286(±0.00010). The calculated Chernobyl-derived ^{237}Np fraction of the total ^{237}Np deposition in peat samples was 0.5–13%. The fraction of Chernobyl-derived deposition of ^{237}Np was low compared to other transuranium nuclides: 0–100% of

Table 1. Activity concentration of ^{237}Np and ^{237}Np/$^{239+240}$Pu activity ratio (reference date 1st May 1986) in peat and lichen samples. * = Activity of ^{237}Np in sample was below the detection limit.

Peat bog	Location	A ^{237}Np (mBq/m²)	A ^{237}Np (mBq/kg)	^{237}Np/$^{239+240}$Pu activity ratio
21 Läyniönsuo, Hankasalmi	62.2 °N, 26.3 °E	5.6±0.1	0.73±0.02	0.00255±0.00011
11 Viheräperä, Kankaanpää	61.7 °N, 22.8 °E	1.98±0.05	0.48±0.01	0.00145±0.00009
95 Korpisalonneva, Vimpeli	63.1 °N, 24 °E	3.70±0.08	1.97±0.04	0.00286±0.00010
144 Kulvesuo, Rautavaara	63.5 °N, 27.6 °E	8.5±0.2	3.37±0.08	0.00211±0.00040
148 Kumpusensuo, Pielavesi	63 °N, 26.8 °E	14.1±0.3	3.84±0.08	0.00263±0.00012
99 Korvaneva, Jalasjärvi	62.3 °N, 22.9 °E	5.7±0.1	1.81±0.04	0.00226±0.00010
L1 Vanttauskoski, Rovaniemi *	66.4 °N, 26.7 °E	<3.4	<0.76	-
L2 Rajajoosepintie, Inari *	68.6 °N, 28 °E	<7.5	<0.76	-
18 Keinusuo, Loimaa *	60.7 °N, 23 °E	<1.7	<0.33	-
19 Roitonsuo, Janakkala *	60.8 °N, 24.7 °E	<1.7	<0.62	-
42 Raatesuo, Punkaharju*	61.5 °N, 29 °E	<0.91	<0.21	-
37 Huppionsuo, Juva*	61.8 °N, 27.2°E	<0.75	<0.38	-

^{239,240}Pu, 10–99% of ^{241}Pu and 0.9–100% of ^{241}Am in the same peat samples originated from the Chernobyl accident fallout (Paatero et al. 1994, Reponen et al. 1993, Salminen et al. 2005).

It is estimated that the activity of ^{237}Np deposited in Finland from the Chernobyl accident was $1.1 \cdot 10^7$ Bq, based on $1 \cdot 10^{11}$ Bq of ^{239,240}Pu from Chernobyl to Finland (Reponen et al. 1993) and inventories of ^{237}Np and ^{239,240}Pu in the reactor core at the time of the accident (UNSCEAR 2000). From global nuclear test fallout, $7.4 \cdot 10^{10}$ Bq of ^{237}Np has been deposited to Finland (Hardy et al. 1973, UNSCEAR 2000, Beasley et al. 1998). It can be concluded that ^{237}Np originates mainly from global nuclear test fallout in Finnish peat and the concentration of ^{237}Np, both Chernobyl-derived and older global fallout, varies depending on location.

Acknowledgements

References

Beasley, T. M., Kelley, J. M., Maiti, T. C. & Bond, L. A. 1998. ^{237}Np/^{239}Pu atom ratios in integrated global fallout. Journal of Environmental Radioactivity 38, p. 133-146

Hardy, E.P., Krey, P.W. & Volchok, H.L. 1973. Global inventory and distribution of fallout plutonium. Nature 241: 444–445.

Jantunen, M., Reponen, A., Kauranen, P. & Vartiainen, M. 1991. Chernobyl fallout in southern and central Finland. Health Phys. 60: 427-434.

Lindahl, P., Roos, P., Eriksson, M. & Holm, E. 2004. Distribution of Np and Pu in Swedish lichen samples (Cladonia stellaris) contaminated by atmospheric fallout. Journal of Environmental Radioactivity 73: 73-85.

Paatero, J., Jaakkola, T. & Reponen, A. 1994. Determination of the ^{241}Pu deposition in Finland after the Chernobyl accident. Radiochimica Acta 64: 139-144.

Paatero, J., Kulmala, S., Jaakkola, T., Saxén, R. & Buyukay, M. 2007. Deposition of ^{125}Sb, ^{106}Ru, ^{144}Ce, ^{134}Cs and ^{137}Cs in Finland after the Chernobyl accident. Boreal Environment Research 12: 43-54.

Pavlotskaya, F.I., Goryachenkova, T.A. & Myasoedov, B.F. 1991. Behaviour of transuranium elements in the environment. Journal of Radioanalytical and Nuclear Chemistry 147: 133-140.

Reponen, A., Jantunen, M., Paatero, J. & Jaakkola, T. 1993. Chernobyl-Pu fallout in southern Finland. Journal of Environmental Radioactivity 21: 119-130

Salminen, S., Paatero, J., Jaakkola, T. & Lehto, J. 2005. Americium and curium deposition in Finland from the Chernobyl accident. Radiochimica Acta 93: 771-779.

Schreckhise, R.G. & Cline, J.F. 1980. Comparative Uptake and Distribution of Plutonium, Americium, Curium and Neptunium in Four Plant Species. Health Physics 38(5): 817-824.

UNSCEAR 2000 Annexes C and J.

http://www.unscear.org/docs/reports/annexc.pdf

http://www.unscear.org/docs/reports/2000/Volume%20II_Effects/AnnexJ_pages%20451-566.pdf

6.2 ^{210}Po and ^{210}Pb in Forest Soil and in Wild Berries

Kaisa Vaaramaa[1], Lasse Aro[2], Dina Solatie[3] and Jukka Lehto[1]

[1] Department of Chemistry, Laboratory of Radiochemistry, University of Helsinki**,** Finland
[2] Finnish Forest Research Institute (Metla), Parkano Research Unit, Parkano, Finland
[3] STUK-Radiation and Nuclear Safety Authority, Regional Laboratory in Northern Finland, Rovaniemi, Finland
E-mail address of main author: Kaisa.Vaaramaa@Helsinki.fi

The general goal in this project is to obtain overview on ^{210}Po and ^{210}Pb behaviour and mobility in forest environment. In this study binding and mobility of ^{210}Po and ^{210}Pb in soil was studied, as well as transfer of polonium and lead from soil to plants. Contribution of surface fallout in plants was discussed.

The soil samples were collected from seven different locations in Finland. All soil profiles were taken from Scots pine forests. Two of them were the main study areas, one in Southern Finland (62º9'N, 22º52'E) and the other in Northern Finland (66º21'N, 26º44'E). The soil profiles were divided into horizontal layers: litter, organic and mineral soil layers. Blueberry (*Vaccinium myrtillus*), lingonberry (*Vaccinium vitis-idaea*) and mushroom samples were also collected from the Southern Finland site and from the Northern Finland site. Activity concentrations of ^{210}Po and ^{210}Pb were analysed from rhizomes of the berry samples as well as from berries, leaves and stems separately.

It was concluded that
a) Maximum activities of ^{210}Pb and ^{210}Po were found in organic soil layers (Bq/m^2) in the Southern Finland site. In contrast, the maximum activity of ^{210}Po was in litter in the Northern Finland site.
b) The activity concentrations of ^{210}Pb and ^{210}Po in wild berry plants increases in the order: berries (i.e.fruits) < leaves < stems
c) Po/Pb ratio in wild berry samples was mainly above one and the activity concentrations of ^{210}Po in different parts of the wild berry samples are not explained by the in-growth of ^{210}Pb.
d) Naturally occuring ^{210}Po and ^{210}Pb dominate the radiation dose from ingestion of blueberry and lingonberry compared to anthropogenic ^{137}Cs in the Northern Finland.

The research results gained in this project will enable an assessment of the mobility of ^{210}Po and ^{210}Pb in the environment and in the food chains and estimation of ensuing radiation doses to humans.

Acknowledgement

6.3 Soil-to-plant transfer of uranium and its distribution between plant parts in boreal forest

Päivi Roivainen, Sari Makkonen, Toini Holopainen and Jukka Juutilainen

University of Kuopio, Department of Environmental Science, P.O. Box 1627, 70211 Kuopio, Finland

Introduction

Uranium can contaminate environment through the whole nuclear fuel cycle from mines to the spent nuclear fuel disposal. Chemical toxicity of uranium is related to compounds containing natural uranium while enriched uranium presents radiotoxicological problems (Ribera et al. 1996). The uptake of uranium by plants is a key process for possible adverse effects in the ecosystem. However, this process is not sufficiently well known at present. Concentration ratios between soil and plant are often used to describe this transfer when the exposure of plants is modelled. Natural uranium occurrences are good places to study the behaviour of this element in real conditions. The results can be used as an analogue to anthropogenically introduced radioactivity.

This study focused on investigating the soil-to-plant transfers of natural uranium and its distribution in the vegetative parts of four plant species representing different plant types common in boreal forest: May lily (*Maianthemum bifolium*), narrow buckler fern (*Dryopteris carthusiana*), rowan (*Sorbus aucuparia*) and spruce (*Picea abies*). The results were also used to assess concentration ratios between soil and different plant parts.

Material and methods

The study was conducted in a natural uranium occurrence located in Nilsiä, Eastern Finland (N63°04', E27°54') in July 2007. The site is a small herb-rich forest fully surrounded by an agricultural area. During 1960s small-scale ore prospecting was carried out in the area and the ore was found to be too poor for commercial purposes. A 100 m long excavation pit with a small pond at the other end exists in the area as a result of the prospecting. The dominant tree species at the site are spruce (*Picea abies*), rowan (*Sorbus aucuparia*) and common aspen (*Populus tremula*). Understorey species include e.g. common wood sorrel (*Oxalis acetosella*), May lily (*Maianthemum bifolium*), narrow buckler fern (*Dryopteris carthusiana*) and oak fern (*Gymnocarpium dryopteris*).

Topsoil (0–10 cm) and litter samples were collected from 29 sampling sites. Plant samples were collected from 19 to 28 sampling sites depending on the abundance of each species. The species collected were May lily, narrow buckler fern, rowan and spruce. May lily and fern were divided into root, stem and leaf. Leaf/needle, root and fine root (diameter < 2 mm) were the parts collected from the trees.

Uranium concentrations were analysed by inductively coupled plasma- mass spectrometer (ICP-MS) after HNO_3 digestion in microwave oven (EPA 3051). ICP-MS analysis was also performed on soil samples after 1 M ammonium acetate (buffered at pH 4.5) leach to obtain estimates of mobile fraction of uranium in soil. The detection limit for uranium was 0.01 mg/kg.

Organic matter (OM) content, pH, silt content and clay content were also analysed from soil samples.

Results and discussion

Topsoil uranium concentrations (total and mobile) and litter uranium concentrations are presented in Table 1. Table 1 also includes data of the measured soil properties. The total U concentrations at the study site were mostly within the average U concentration in soil (1–10 mg kg^{-1} Koch-Steindl and Pröhl 2001), except one value exceeding 100 mg kg^{-1}. Mobile fraction of uranium varied from 7 to 86 % of the total concentration and there was a strong positive correlation between these two concentrations (Spearman ρ=0.898). There were no correlations found between soil uranium concentrations and any of the soil properties.

Uranium concentrations in different plant species and plant parts are presented in Table 2. There were no significant differences noticed in uranium concentrations between any of the studied plant species. Roots and fine roots had significantly higher uranium concentrations than stems and leaves. The accumulation of uranium to the below-ground parts of plants has also been noticed in other studies (e.g. Shahandeh and Hossner 2002, Shtangeeva 2008). Plant uranium concentrations correlated significantly better with soil mobile uranium concentration than with soil total uranium concentration.

Concentration ratios between soil and different plant parts are presented in Table 3. The only significant difference between plant species was noticed in soil-to-leaf ratios of May lily and rowan. The explanation for this might be that U concentrations in rowan were mostly below detection limit. The values are in accordance with the values found in literature (Sheppard et al. 2006). It seems that the use of same concentration ratios for different plant species does not introduce extra variability to the results. The concentration ratios obtained for Finnish boreal forest species are in accordance with the results for other types of plants.

Table 1. Mean, median and range of the soil uranium concentrations (total, mobile and litter mg kg^{-1} dw), pH, organic matter content, clay content and silt content (n=29).

Soil property	Mean	Median	Range
Total U conc. mg kg^{-1}	6.63	1.64	0.46-109.24
Mobile U conc. mg kg^{-1}	3.57	0.33	0.04-66.47
Litter U conc. mg kg^{-1}	0.48	0.17	0.03-2.61
pH	4.41	4.41	4.04-5.11
Organic matter %	13.11	11.27	2.67-36.87
Clay %	9.63	9.45	3.56-18.31
Silt %	9.77	8.50	1.91-33.71

Table 2. The range of uranium concentrations (mg kg^{-1} dw) in different plant species and plant parts.

Plant part	Plant species			
	May lily (n=19)	Fern (n=27)	Rowan (n=28)	Spruce (n=26)
Root	0.01-1.72	0.01-3.09	0.01-0.99	0.01-1.74
Fine root			0.02-7.17	0.01-12.24
Stem	<0.01-0.07	<0.01-0.02		
Leaf	<0.01-0.35	<0.01-0.03	<0.01-0.01	<0.01-0.11

Table 3. Geometric mean (and geometric standard deviation) of concentration ratios between soil and different plant parts.

Type of CR	Plant species			
	May lily (n=19)	Fern (n=27)	Rowan (n=28)	Spruce (n=26)
Soil-to-root	0.04 (2.36)	0.08 (3.86)	0.07 (3.77)	0.06 (3.88)
Soil-to- fine root			0.24 (3.47)	0.32 (5.23)
Soil-to-stem	0.004 (2.99)	0.004 (3.18)		
Soil-to-leaf	0.008 (2.71)	0.004 (3.15)	0.003 (3.54)	0.006 (3.68)

Acknowledgements

References

Koch-Steindl, H. & Pröhl G., 2001. Considerations on the behaviour of long-lived radionuclides in soil. Radiation and Environmental Biophysics 40: 93–104.

Ribera, D., Labrot, F., Tisnerat, G. & Narbonne, J.-F. 1996. Uranium in the Environment: Occurrence, Transfer, and Biological Effects. Reviews of Environmental Contamination and Toxicology 146: 53–89.

Shahandeh, H. & Hossner, L.R. 2002. Role of soil properties in phytoaccumulation of uranium. Water, Air and Soil Pollution 141: 165–180.

Sheppard, S.C., Sheppard, M.I., Tait, J.C. & Sanipelli, B.L. 2006. Revision and meta-analysis of selected biosphere parameter values for chlorine, iodine, neptunium, radium, radon and uranium. Journal of Environmental Radioactivity 89: 115–137.

Shtangeeva, I. 2008. Uptake of uranium and thorium by native and cultivated plants. Journal of Environmental Radioactivity doi 10.1016/j.jenvrad.2008.06.004.

6.4 Alkali metals in fungi of forest soil

M. Vinichuk[1,2], A. Taylor[1], K. Rosén[1], I. Nikolova[1] and K.J. Johanson[1]

[1] Swedish University of Agricultural Sciences, SLU, Box 7026, S-750 07 Uppsala, Sweden
[2] Zhytomyr State Technological University, 103 Cherniakhovsky Str., 10005, Zhytomyr, Ukraine

Abstract

The high affinity of forest soil fungi for alkali metals such as potassium, rubidium, caesium as well as radiocaesium is shown and discussed. Good positive correlation was found between K: Rb concentration ratios in soil and in fungi, when correlation between K:Cs concentration ratios was less pronounced.

Introduction

In addition to acquiring essential macronutrients, mycorrhizal fungi are very efficient at taking up and accumulating radionuclides. Thus, in forest ecosystems fungi are the major contributors to accumulation and cycling of radionuclides, especially radiocaesium. However, the mechanisms involved in caesium accumulation in fungi are principally not known. Relatively little is known about uptake and retention of ^{137}Cs by fungal mycelia. The aim of this study was to prepare the mycelia mainly of ectomycorrhizal fungi and to study their possible role in the retention, turnover and accumulation of radiocaesium in forest ecosystems. We also attempted to quantify the uptake and distribution of the alkali metals K, Rb, Cs in the soil-mycelium-fruit bodies compartments and to study the competition between K, Rb and Cs in the various transfer steps (Vinichuk et al. 2004).

Material and Methods

The studies were conducted in Swedish and Ukrainian forests during 1996–1998 and 2000–2003. The sites consisted of 50 to 100 year-old mixed stands of Scots pine (*Pinus sylvestris* L.) and Norway spruce (*Picea abies* [L.] Karst.) The upper 10 cm of the soil had rather high organic matter content and the ground deposition of ^{137}Cs in those upper layers was ca. between 20 to 100 kBq m^{-2}. In Ukraine the study area was located in a forest ecosystem about 70 km West of Chernobyl Nuclear Power Plant. The ground deposition of ^{137}Cs within the sampling area was 55–233 kBq m^{-2}. We have been collecting fruit bodies of fungi and soil samples to a depth of 10 cm within an area of about 0.25 m^2 around and directly underneath the fruit body by using a cylindrical steel bore. The soil cores were sectioned horizontally in layers of 1- or 2-cm thickness; the mycelia from each layer were prepared under microscopic examination. The ^{137}Cs activity concentrations in prepared samples were determined using well-calibrated HP Ge and NaI(Tl) detectors. A part of the samples were analyzed using ICP-AES or ICP-SFMS. In the total about 200 samples of fungal mycelia were prepared and analyzed for ^{137}Cs, K, Rb and Cs.

Results

The fungal biomass was either calculated as mg per g dry weight of the soil, from which the mycelium was extracted, or as mg per each layer of the soil profile. The estimated fungal biomass in Ukrainian forest soils varied from 0.07 to 70.4 mg g^{-1} soil, in Swedish forests between 3.6 and 19. 4 mg g^{-1} soil (Vinichuk 2003).

The ratios between the ^{137}Cs activity concentration in the fungal mycelium (kBq kg^{-1}) and ^{137}Cs activity concentration in the soil (kBq kg^{-1}) were rather high and varied for each site. On the average, those ratios in corresponding soil layers for the most of analyzed sites of fungi were found between 20 to ca. 100.

The ratios between the ^{137}Cs activity concentration (kBq kg^{-1}) in fruit bodies and ^{137}Cs activity concentration (kBq kg^{-1}) in mycelium for 18 analyzed species of fungi also varied considerably from 0.1 to 65.8. However, for most of the analyzed sites (15) those ratios were > 1. The ^{137}Cs activity concentration in mycelia was thus higher than that found in soil, and ^{137}Cs activity concentrations in the fruit bodies was higher than that in the mycelium.

The amount of the total radiocaesium activity incorporated into the fungal mycelium, expressed as percentages of total ^{137}Cs activity found in soil to a depth of 10 cm in the Ukrainian forest also was found to be considerably variable. On the average, from 0.6 to 50% of the total ^{137}Cs activity was located within the fungal mycelia. For the Swedish forest we found that between 0.7 and 2.5% of the total ^{137}Cs activity (0–10 cm) was located within the fungal mycelium.

It is appeared, that fruit bodies of fungi accumulate greater amounts of K, Rb and Cs compared to their concentrations in mycelium. Thus, potassium concentrations in fruit bodies of fungi, collected at the same plots, where soil samples where taken and mycelium was extracted was about 15 times higher to concentrations of the element found in mycelium. The concentration of rubidium in fruit bodies of fungi was about 18 times and caesium about 9 times higher to that in fungal mycelium. There was a close positive correlation (r=0.95, p<0.01) between K:Rb in soil and in fungal mycelium (Fig. 1). This relationship was also apparent between soil and fruit bodies but was less strong (r=0.85, p=0.05).
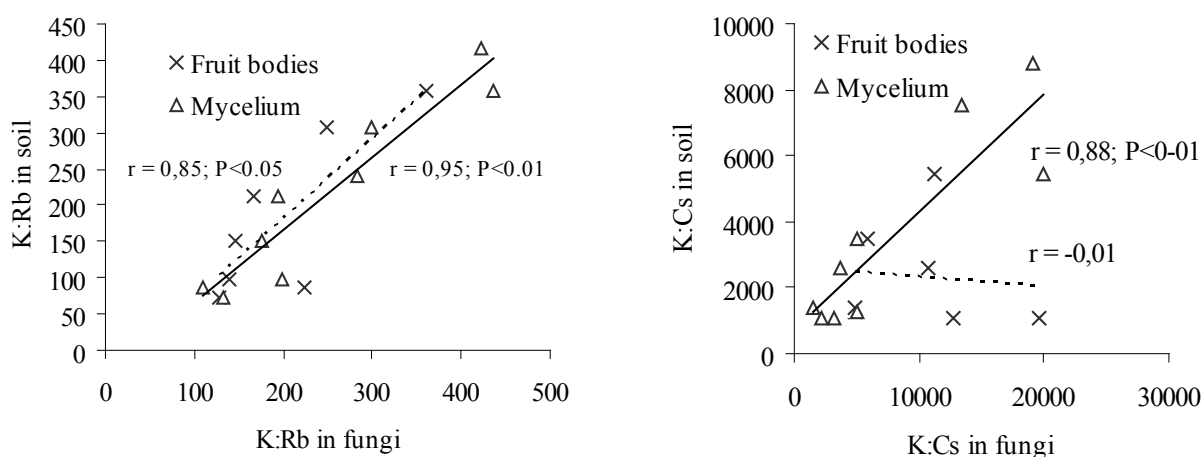

Figure 1. Relationships between K, Rb and Cs concentration, mg per kg d.w. in fungal mycelium and fruit bodies of fungi.

The K:Cs in soil and fungal components showed a different picture, with only the K:Cs in mycelium being positively correlated (r=0.88, p=0.01) with soil K:Cs. There was no correlation between soil K:Cs and that found in fruit bodies.

Discussion

It has been shown that fungi in forest soil have high capacity to accumulate alkali metals. Large fraction of ^{137}Cs, Cs, Rb and K is located within the mycelia of soil fungi. Indeed, fungi may form a major pool of those metals in the soil, because forest soils host an intense fungal life and fungal structures in the soil are often perennial. The exchange of mineral nutrients is the fundamental process of ectomycorrhizal fungi. Since fungi involved in the movement of nutrients through the hyphae, obviously fungi and fungal mycelia particularly will determine transfer of alkali metals and radiocaesium in those soils.

In order to fully understand the transfer mechanism from soil to plant of radiocesium, more information is needed on the mechanisms involved in transfer processes of alkali metals, particularly with regard to their competitive effects during uptake. There are systems for uptake of alkali metals from soil to mycelium. It seems that system for K nearly as effective as for Rb, when for ^{137}Cs is less efficient. Theoretically uptake process could be regulated by both fungus and also by the plant. Depending on the nutrient status of the environment it might be so that mycorrhizal fungi will have most of advantage of the symbiosis at least in normal forest nutrient situation. In nutrient rich locations plants can grow without fungi and on extremely nutrient poor locations there will be a little mycorrhizal fungi or not at all due to difficulties to find mineral nutrients.

The data provided above indicate that fungi are taking up more ^{137}Cs than they need. This is true at least for ^{137}Cs, nearly all fungi accumulate ^{137}Cs in levels between 10 to 100 times higher then plants do. Usually mycorrhizal fungi tend to accumulate more then saprotrophs do. The conclusion is therefore that fungi accumulate more ^{137}Cs then they actually need. There is also evidence that the chemical behaviors of caesium and potassium are not completely identical, when they are taking up by fungi. There was no correlation between soil K:Cs and that found in fruit bodies.

The models of ^{137}Cs distribution and uptake, which are important for prognosis in the radiation protection work usually, include ^{137}Cs migration, binding to some soil components, root uptake and some more parameters. Usually fungi are not mentioned here. It may be too complicated to make a realistic model for forest soil, but if we have not knowledge about the fungi and their role in ^{137}Cs transfer we need more research in this field.

Acknowledgement

References

Vinichuk, M. 2003. Radiocaesium in The Fungal Compartment of Forest Ecosystems. Doctoral dissertation. Uppsala.

Vinichuk, M., Johanson, K. J., Taylor, A. & Nikolova, I. 2004. Uptake of elements by fungi in the Forsmark area. Technical Rapport TR-04-26. 87 pp.

6.5 ^{137}Cs activity concentrations in mushrooms collected from two different types of habitats in Finnish Lapland

Jarkko Ylipieti, Virve Härkönen and Dina Solatie

STUK - Radiation and Nuclear Safety Authority, Research and Environmental Surveillance, Regional Laboratory in Northern Finland, Louhikkotie 28, 96500 Rovaniemi, Finland

Abstract

^{137}Cs activity concentrations from thirteen different mushroom species collected from the Kivalo Research Area in Northern Finland were analysed at the Radiation and Nuclear Safety Authority (STUK) in the Regional Laboratory in Northern Finland. The analyzed samples were collected from 1989 to 2008 from four types of forest, which were divided into two classes – fresh heath and dry heath – based on their properties as a habitat. The aim of this study was to compare the ^{137}Cs levels in the same mushroom species collected from the different habitats. In addition, the ecological half-lives for the different species were estimated.

Together 360 samples were analysed by gamma spectrometry. The results were processed statistically and the significance of the habitats was evaluated to nine different mushroom species. The impact of habitats could be observed inside the same species. ^{137}Cs activity concentrations in mushrooms were higher on the fresh heath than on the dry heath, with the exception of *Cortinarius armillatus* and *Suillus variegatus*. ^{137}Cs concentration levels in Bq/kg f.w. were under 50 in *Leccinum*, under 150 in *Russula*, under 300 in *Lactarius* and *Suillus*, under 400 in *Cortinarius armillatus* and under 700 in *Rozites caperatus*. The concentration was found to be over the EC recommendation of 600 Bg/kg in one sample which was *Cortinarius armillatus* in 1994.

7 Concluding remarks

Several issues dealing with the development of radioecological research towards better response to the needs of end users were raised during the seminar. Discussions were useful and new multidisciplinary seminars in the future with well-defined content were called for.

Sampling was one crucial topic during the seminar and a draft to a new guidance for sampling in forests for radionuclide analyses was presented and discussed. The guidance is now published and available as an NKS report no. 183 at www.nks.org.

In the seminar it was emphasised how important it is to have well-defined aims and hypotheses before starting a research project. It may also be useful to identify the end-users and involve them already in the project's planning process. Another issue brought up was that radioecologists, radiochemists, forest ecologists and forestry experts can learn from each other in research programmes with well-defined long-term aims. The quality of the research work of especially small projects with limited resources can be improved if the projects are connected to a bigger research programme. Research collaboration is needed, for instance, for the analysis of nutrient flows and radionuclide flows in forests. Data should be collected using a modelling approach, that is, dividing the forest soil and vegetation into compartments for sampling, and the analysis should ensure the usefulness of the resulting data in assessment models with a similar compartment structure. There is a continuous need for basic research about how northern conditions affect speciation and transfer of radionuclides in the forest ecosystem. This kind of research will demand for new sampling methods and laboratory analyses in the future.

Increasing production of bioenergy in the Nordic countries seems to call for intensive harvesting of forest biomass. Reduced availability of mineral nutrients need not follow increased bioenergy production, if compensating fertilisation is taken care for. This way also the radionuclide uptake from the soil to forest vegetation can be kept constant or even reduced. In opposite cases, however, the uptake of several radionuclides from the soil to forest vegetation will probably increase.

The interest in radionuclides from nuclear waste that could have very long-term environmental effect if migrated from the waste repositories to biosphere was noticeable during the seminar. Regarding statutory assessment of environmental impact of final disposal of spent nuclear fuel there is an obvious lack of data on several radionuclides, such as ^{59}Ni, ^{36}Cl and isotopes of technetium, thorium and uranium. On the basis of current knowledge, these nuclides may in theory contribute to the radiation doses to people using forests on the waste disposal site, close to the potential release locations. International projects with joint and external funding are important since special know-how in this field is needed and costs for experimental research are high.

In forestry, the organisations and mechanisms for expert communication and advice on forest management are well developed. They could be expressly involved in planning, evaluation, communication and training of sustainable mitigating measures in forestry. If a well functioning network of actors is established, the effect of implementation of protective measures in unexpected situations can be significantly more beneficial than otherwise. The multidisciplinary basis can be efficient in building this type of preparedness with key stakeholders. For instance, the delivery of timber and other forest products could be organised through systematic implementation of research-based management options in contaminated forests.

Finally, to support future collaboration and contacts between experts working in related fields, a proposal of an informal forest network was launched in the seminar. The network was activated two weeks later. The contact address for active and new members is forestnetwork@stuk.fi. The benefits of networking include, for instance, the possibility to initiate and develop broadly-based multidisciplinary projects with joint financing.

Appendix 1: Programme

Tuesday 7 October 2008

8.45   Registration at the reception of STUK

9.15   Opening of the seminar, *Sisko Salomaa, Research Director, STUK*

Opening remark of the Finnish Forest Research Institute, H*annu Raitio, Director General*

Introduction of the NKS-B programme, *Justin Gwynn, Programme Manager, NKS*

Practical announcements, Virve Vetikko, STUK

Session 1   radioecology and forest research – basis for preparedness to manage contaminated forests, hannu ilvesniemi, Metla

9.40   Uptake of radionuclides with focus on cesium, *Agneta H. Plamboeck, FOI*

10.30 ^{137}Cs deposition in peat profiles on a raised bog in central Sweden, *Klas Rosén, M. Vinichuk, P.R. Galán and K.J. Johanson, SLU*

11.00   Radionuclide contamination in forests, dose pathways and their significance to exposed people, *Aino Rantavaara, STUK*

11.30   Collecting accurate baseline forest data using airborne laser scanning and disseminating contamination guidelines over the internet – results from practical experience, *Tuomo Kauranne, Oy Arbonaut Ltd*

13.00   Nutrient, particularly potassium fluxes in boreal forest ecosystems, *Leena Finér, Metla*

13.30   Bioenergy concepts, soil processes etc., *Hannu Ilvesniemi, Metla*

14.00   Effect of forest management on ^{137}Cs activity of vegetation based on long-term field experiments, *Lasse Aro, Metla*

14.30   Effect of industrial pollution on behaviour of radionuclides in forest ecosystems, *Iisa Outola, STUK*

14.50   Discussion on session 1, defining the needs for future studies and the Chairperson's concluding remarks

Session 2   Management of forests contaminated by radionuclides, Dina Solatie

15.30   Field surveying of radionuclide contamination in forests, *Jani Turunen, STUK*

15.50   Introduction and demonstration of the RODOS forest models, *Michael Ammann, STUK*

16.10   Management of forests after radionuclide contamination and practicability of management options, *Aino Rantavaara, STUK*

16.30   Discussion on session 2, defining the needs for future studies and the Chairperson's concluding remarks

16.45   Ending of the programme of the day

19.00   Seminar Dinner

Wednesday 8 October 2008

SESSION 3   SAMPLING IN FORESTS, KLAS ROSÉN

9.00   Introduction to the Sampling guide, *Elisabeth Strålberg, IFE*
9.20   Sampling of forest soil and fungal fruit bodies, *Agneta H. Plamboeck, FOI*
9.40   Sampling of undestorey vegetation, *Virve Vetikko, STUK*
10.00   Different approaches to sampling trees, *Lasse Aro, METLA*
10.20   Discussion on session 3 and the sampling guide

SESSION 4   ENVIRONMENTAL IMPACT OF FINAL DISPOSAL OF HIGHLY RADIOACTIVE NUCLEAR WASTE, JUKKA LEHTO

11.00   Estimates of naturally occurring pools of Th, U and I in different forest types of the boreal region in the southeast of Sweden, *Anders Löfgren (EcoAnalytica) and Ulrik Kautsky (SKB), Sweden*
11.20   The forest monitoring programme in Olkiluoto spent nuclear fuel repository site *Jani Helin, Posiva Oy*
11.40   Present gaps in knowledge on fate and transport of radionuclides in forests – case of spent nuclear fuel repository, *Ari Ikonen, Posiva Oy*
12.00   Discussion on session 4, defining the needs for future studies and the Chairperson's concluding remarks
13.15   Discussion on possibilities for Nordic collaboration as a network or in a research project – opinions and suggestions from the participants and concluding remarks, *Justin Gwynn, NKS*

SESSION 5   RECENT AND ONGOING STUDIES ON RADIONUCLIDES IN FORESTS, LEENA FINÉR

13.40 ^{137}Cs, ^{239,249}Pu and ^{241}Am distribution in stratified podzolic soil in Finnish forests, *Jukka Lehto and Kaisa Vaaramaa, Laboratory of Radiochemistry, University of Helsinki*
14.00 ^{237}Np in peat and lichen in Finland, *Susanna Salminen, J. Paatero (FMI), P. Roos (Riso National Laboratory) and K. Helariutta, Laboratory of Radiochemistry, University of Helsinki*
14.40 ^{210}Po and ^{210}Pb in Forest Soil and in Wild Berries, *Kaisa Vaaramaa, Lasse Aro (METLA), Dina Solatie (STUK) and Jukka Lehto, Laboratory of Radiochemistry, University of Helsinki*
15.00   Soil-to-plant transfers of uranium and its distribution between plant parts in boreal forest, *Päivi Roivainen, Sari Makkonen, Toini Holopainen, Jukka Juutilainen, University of Kuopio*
15.20   Alkali metals in fungi of forest soil, *Mykhailo Vinichuk and Karl Johan Johanson, SLU*
15.40 ^{137}Cs activity concentrations in mycorrhizal macrofungi species in two different types of habitats after the Chernobyl accident, *Jarkko Ylipieti, Virve Härkönen and Dina Solatie, STUK*
16.00   Discussion on session 5, defining the needs for future studies and the Chairperson's concluding remarks
16.20   Closure of the seminar, *Justin Gwynn, Lasse Aro*

Appendix 2: Participants of the seminar

1. Ammann Michael   Radiation and Nuclear Safety Authority (STUK), Finland
2. Aro Lasse   Finnish Forest Research Institute (METLA)
3. Finér Leena   METLA, Finland
4. Gwynn Justin   Norwegian Radiation Protection Authority (NRPA) and NKS
5. Helin Jani   Posiva Oy, Finland
6. Huusela Kari   STUK, Finland
7. Hånell Plamboeck Agneta The Swedish Defence Research Agency (FOI)
8. Hämäläinen Kai   STUK, Finland
9. Härkönen Virve   STUK, Finland
10. Ikonen Ari   Posiva Oy, Finland
11. Ilvesniemi Hannu   METLA, Finland
12. Kauranne Tuomo   Oy Arbonaut Ltd, Finland
13. Kostiainen Eila   STUK, Finland
14. Kuru Hanna   UPM-Kymmene Corp, Finland
15. Lahtinen Juhani   STUK, Finland
16. Lehto Jukka   University of Helsinki, Laboratory of Radiochemistry, Finland
17. Lehtovaara Jaakko   Vapo Oy, Finland
18. Lusa Merja   University of Helsinki, Laboratory of Radiochemistry, Finland
19. Löfgren Anders   EcoAnalytica, Sweden
20. Muikku Maarit   STUK, Finland
21. Outola Iisa   STUK, Finland
22. Raitio Hannu   METLA, Finland
23. Raivio Suvi   Finnish Forest Industries Federation
24. Rantavaara Aino   STUK, Finland
25. Roivainen Päivi   University of Kuopio, Dep. of Environmental Science, Finland
26. Rosén Klas   Swedish University of Agricultural Sciences (SLU),
                             Dep. of Soil Sciences
27. Salminen Susanna   University of Helsinki, Laboratory of Radiochemistry, Finland
28. Salomaa Sisko   STUK, Finland
29. Solatie Dina   STUK, Finland
30. Strålberg Elisabeth   Institute for Energy Technology (IFE), Norway
31. Turtiainen Tuukka   STUK, Finland
32. Turunen Jani   STUK, Finland
33. Vaaramaa Kaisa   University of Helsinki, Laboratory of Radiochemistry, Finland
34. Vartti Vesa-Pekka   STUK, Finland
35. Vetikko Virve   STUK, Finland
36. Vinichuk Mykhailo   Swedish University of Agricultural Sciences (SLU) and
                             Zhytomyr State Technological University, Ukraine
37. Ylipieti Jarkko   STUK, Finland

Abstract	The seminar "Towards improved understanding of radionuclide transfer in forests and preparedness to handle contaminated forests" was carried out within the framework of the NKS-B Forest project in Helsinki, 7-8 October, 2008. The seminar was planned and arranged by four Nordic organisations and provided a forum for exchange of information for Nordic scientists currently working in the field of forest radioecology or using the data. Presentations of research on nutrient cycling and radionuclide distribution in boreal forests, discussion on the needs for future research and attendance of experts on forestry, forest research and radioecology offered a unique opportunity to disseminate and receive information. The seminar programme was composed of topics of radioecology and forest research, assessment of radionuclide contamination and management of contaminated forests. Also sampling in forests, monitoring and modelling of environmental impact of disposal of spent nuclear fuel, and recent radioecological studies on forests were handled. Future research was emphasised in discussions. Below is a short compilation of these discussions:

- Comprehensive planning of research projects contributing to a programme with long-term aims is possible in broadly-based multidisciplinary collaboration. Thereby independent initiatives and less coherent plans can be replaced.
- Clear definition of hypotheses, planning and improving the specific methods for sampling and laboratory analyses were found crucial.
- Effects of intensive biofuel harvesting on the nutrient and radionuclide flows in forests are major issues in the next decades.
- Gaps in knowledge, such as lack of data on processes contributing to radionuclide distributions in forests, and on certain long-lived radionuclides (those of Cl, Tc, Np, etc.) contributing to the environmental impact of final disposal of spent nuclear fuel. Topical seminars like the one accomplished are welcome in the future; compiling acute issues of multidisciplinary nature for focussed expert review and discussion can be very rewarding in many ways. To support communication and future collaboration, an informal forest network was launched in the seminar. Information is available via e-mail (forestnetwork@stuk.fi).