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Approaches to modelling radioactive contaminations in forests – Overview and guidance

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ABSTRACT

Modelling the radionuclide cycle in forests is important in case of contamination due to acute or chronic releases to the atmosphere and from underground waste repositories. This article describes the most important aspects to consider in forest model development. It intends to give an overview of the modelling approaches available and to provide guidance on how to address the quantification of radionuclide transport in forests. Furthermore, the most important gaps in modelling the radionuclide cycle in forests are discussed and suggestions are presented to address the variability of forest sites.

Contents

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1. Introduction

Contamination of forests can occur following a nuclear accident, as it happened in Chernobyl in 1986 and in Fukushima in 2011, where forests covered most of the contaminated terrestrial areas (e.g. Hashimoto et al., 2012; Kashparov et al., 2012; Yoschenko et al., 2017). Contamination may also originate from nuclear waste in host rocks at disposal sites or mining activities. Contamination events could lead to radionuclide migration in the soil column and accumulation of radionuclides in surface vegetation and local food sources (Goor and Thiry, 2004; Thiry et al., 2009). Increased external radiological exposure can affect humans who spend time in forests, such as hunters, foresters, mushroom and berry pickers. Internal exposure may result from ingestion of contaminated mushrooms or wild berries (Calmon et al., 2009; Carini, 1999; Gwynn et al., 2013; Steiner et al., 2000, 2002) and game, for example wild boar (Hartmann et al., 2016) or deer.

Forests provide wood for different domestic and industrial purposes, such as biofuel for heat and power plants, paper production, building material and firewood for private homes. The use of contaminated wood and the resulting residues may increase human radiation dose via external exposure and inhalation (Charro et al., 2013; Hubbard, 1997). Secondary contamination several years after the accidental release of radionuclides can be relevant in case of forest fires, which increase the inhalation dose of the exposed population (Yoschenko et al., 2006; Zhou et al., 2016). Secondary contamination may also result from flooding events, e.g. after the Chernobyl accident, when flooding led to the transport of the radionuclides from soil to surface waters (Laptev and Voitsekhovich, 1993).

Modelling the radionuclide circulation in a forest is challenging, because forests are by far more spatially heterogeneous than agricultural lands. Vegetation and soil have both a layered structure. The local soil profile and the contamination levels of the soil horizons can vary at a small scale, depending on the characteristics of the above ground biomass (e.g. type and density of trees) and the proximity to tree stems.

This article summarises the Forest Modelling Handbook which was developed by the Forest Modelling Group working under the EC FP7 project COMET (http://www.radioecology-exchange.org/content/comet). It gives an overview about forest modelling approaches and provides guidance for calculating the radionuclide fluxes between the different forest compartments. It is intended for modellers, experimentalists, experts of national authorities and radiation protection advisers.

2. Fundamentals of forest modelling

2.1. Pathways of radionuclides in forests

For modelling the fate of radionuclides in forest ecosystems, the source term must be known. A compilation of typical radionuclide releases from various sources to the atmosphere can be found in UNSCEAR (2000). After the atmospheric transport, radionuclides enter the forest via dry and wet deposition. They are partially intercepted by the canopy (Fig. 1). The amount of intercepted radionuclides depends on physico-chemical forms of the deposited radionuclides, the meteorological conditions (e.g. dry or wet deposition), forest type, tree population density and season (Shaw, 2007). Radionuclides deposited onto vegetation are partially taken up through stomata and cuticle. Then they are translocated from needles and leaves into branches and stems (Fig. 1). The period of 3–5 years after the deposition is characterised by a redistribution of the initial deposits through weathering of radionuclides from the canopy via throughfall, stemflow, litterfall and radionuclide migration in soil. After the initial rapid infiltration, radionuclides are partially fixed and immobilised through fungal or microbial activity or mineral constituents of forest soil. Correspondingly, the rate of downward migration is considerably reduced. In the organic horizons, this is determined mainly by the decomposition and litter accumulation rates. The downward migration of radionuclides is partially compensated by upward translocation by roots and, notably in the case of radiocaesium, by fungal mycelia (Rafferty et al., 2000). Fungal and microbiological activity are likely to contribute substantially to the long-term retention of radionuclides in organic layers of forest soil (Fig. 1). Bioturbation may also affect the transport of radionuclides in forest soil (Bunzl, 2002). Sorption and complexation with organic and mineral components within the soil are also relevant processes (Berkowitz et al., 2014). In the long-term, root uptake via symbiotic fungi (Smith and Read, 1997) is the dominating factor regarding tree contamination and the local soil is the major radionuclide reservoir (Calmon et al., 2009).

Another potential source of contamination could arise from a nuclear waste repository in host rocks. In case of groundwater infiltration into the nuclear waste followed by its dissolution, the contaminant transport is driven mainly by advection along the hydraulic gradient. In host rocks with a low permeability like clay rocks, the contaminant transport is often dominated by diffusion and molecular dispersion processes (GRS, 2007; IAEA, 2011). When contaminated groundwater has reached the root zone of soil (e.g. via capillary rise), radionuclides can be taken up by plants through the interaction with fungal mycelia and directly from the contaminated soil solution. Gaseous transport can lead to inhalation of radioactive pollutants or, in the case of $^{14}CO_2$, to photosynthetic uptake by plants (Berkowitz et al., 2014).

2.2. How to design a forest model

A radioecological model for forests should be designed in a way that it is as simple as possible but fit for purpose. The purpose of the model and the desired endpoints should be identified at first. Typically, radioecological forest models are used to quantify one or several of the following endpoints:

1. Time-dependent activity concentrations in the affected ecosystem
2. Time-dependent ambient dose rates in the affected ecosystem
3. Doses to humans (equivalent dose to tissues and organs and/or effective dose)
4. Doses to biota (weighted absorbed dose rates)

In the two latter cases, the model should provide conservative dose estimates (e.g. for demonstrating compliance with dose limits), high percentiles of the endpoints (e.g. 95th percentile of dose to a representative person) or the best estimate of human or biota radiation exposure.

A structured approach for the development of radioecological models was published by IAEA (Biomass, 2003). It consists of three major steps:

1. Compilation of a list of Features, Events and Processes (FEPs list).
2. Development of an Interaction Matrix, representing the compartments of the ecosystem to be modelled and the dominant processes.
3. Implementation of the conceptual model, i.e. the Interaction Matrix, into mathematical models.

The FEPs list is a document that compiles all features, events and processes which could be of relevance for the fate, transport and distribution of radionuclides in an ecosystem. The FEPs list helps to