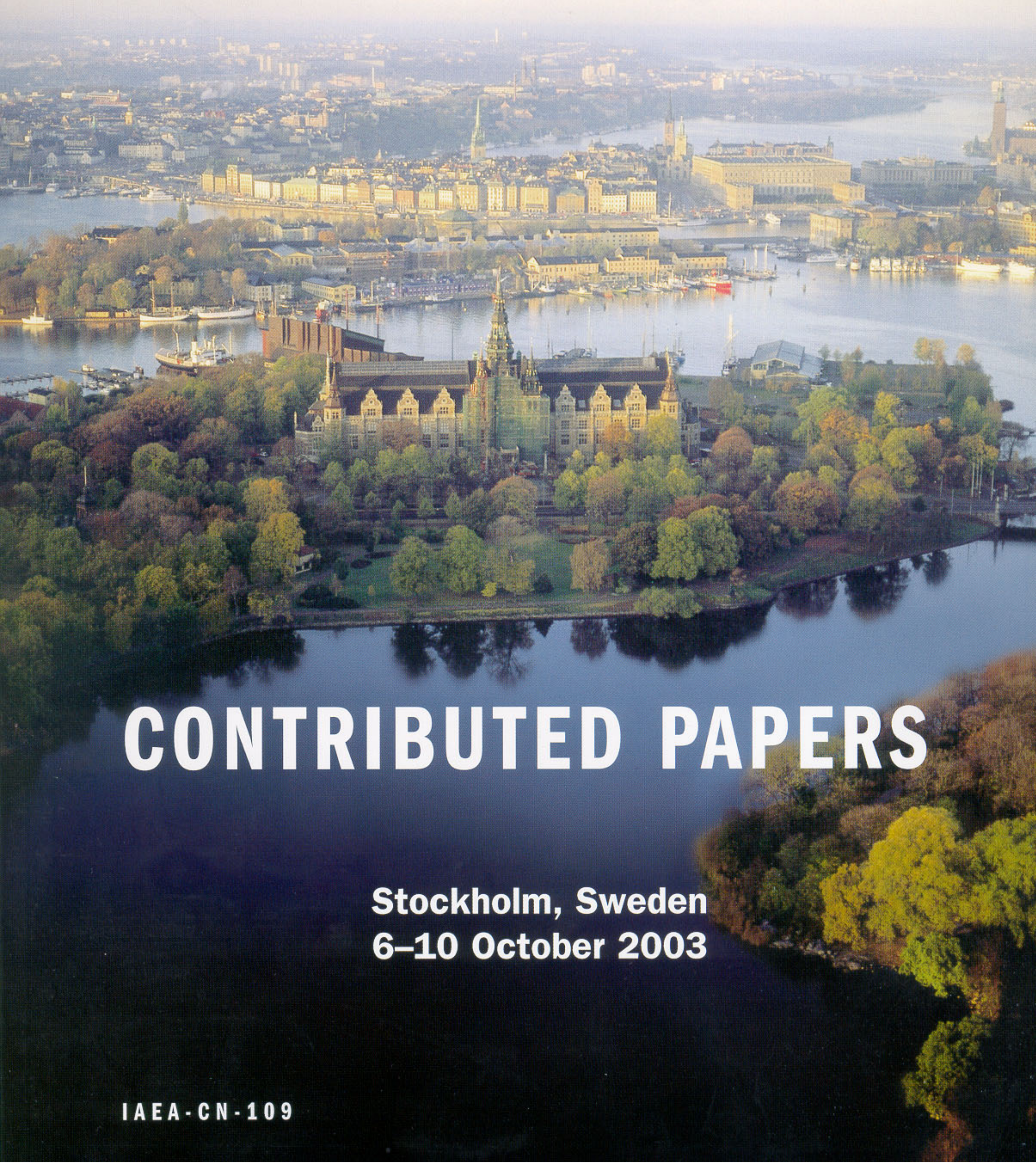
International Conference on the

PROTECTION OF THE ENVIRONMENT FROM THE EFFECTS OF IONIZING RADIATION



CONTRIBUTED PAPERS

Stockholm, Sweden
6–10 October 2003

IAEA-CN-109

Organized by the

IAEA International Atomic Energy Agency

In co-operation with the

UNSCEAR United Nations Scientific Committee on the Effects of Atomic Radiation

European Commission

International Union of Radioecology

Hosted by the

Government of Sweden

Through the

Statens strålskyddsinstitut Swedish Radiation Protection Authority Swedish Radiation Protection Authority

The photograph, on which the cover is based, was taken by Richard Ryan.

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(Version 3, 1 September 2003)

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FOREWORD

An International Conference on the Protection of the Environment from the Effects of Ionizing Radiation, organized by the International Atomic Energy Agency (IAEA) in co-operation with the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), the European Commission (EC) and the International Union of Radioecology (IUR), will be held in Stockholm, Sweden, from 6–10 October 2003. This Conference will be hosted by the Government of Sweden through the Swedish Radiation Protection Authority (SSI).

This publication contains contributed papers submitted on issues within the scope of the conference, which were accepted following a review by the Conference Programme Committee. These papers have not been edited, although some modifications have been made to ensure a unified format and minor corrections to the text have been made where required. It is intended that, after the conference, the contents of this publication will be made available in the form of a CD ROM as part of the proceedings of the conference. Authors wishing to make slight modifications or corrections to their papers are encouraged to contact the Conference Secretariat.

The primary objective of this Conference is to foster information exchange, with the aim of promoting the development of a coherent international policy on the protection of the environment from effects attributable to ionizing radiation. This Conference is one in a series of meetings organized by, or held in co-operation with, the IAEA on this subject. It will include a review of recent developments in this area, and consideration of their implications for future work at national and international levels. The topics on which contributed papers were requested are as follows:

— Existing environmental protection approaches;
— Development of an international assessment framework;
— The scientific basis for environmental radiation assessment;
— Development of management approaches.

This publication has been arranged in the order in which related Topical Sessions appear in the Conference Programme. Under each Session heading, papers are arranged in alphabetical order of the Member State of the first author.

The views expressed in the papers are the responsibility of the named authors; they do not necessarily represent the views of the Governments of Member States, the IAEA or co-operating organizations.

CONTENTS

1. EXISTING ENVIRONMENTAL PROTECTION APPROACHES – CASE STUDIES

A radiological case study of quantitative, probabilistic ecological risk assessment, using recently developed assessment tools

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Abstract. A case study of tritium release into a marine ecosystem demonstrates the feasibility of combining recent dose-estimation software (radiological impact assessment for coastal aquatic ecosystems, Version 1.15) with dose-response data selected from the FASSET Radiation Effects Database (FRED) to provide a basis for quantitative and probabilistic ecological risk assessment. The ecological risk assessment software, AQUARISK, fits probability distributions to both 'exposure' and 'response' data (matched as either dose or dose-rate) and uses a convolution algorithm to evaluate the overlap between the two distributions. Probabilistic estimates of exposure criteria are derived from the distribution fitted to the selected response data. Using only 'lowest observed effect' and 'highest no effect' dose-rates, a quite low criterion for the protection of 95% of species (15 μGy hr^{-1}, with a 95% lower uncertainty limit of 3 μGy hr^{-1}) is estimated. This compares with the proposed Canadian criteria for aquatic biota, with the 1/10th safety factor applied (i.e. 5 to 10 μGy hr^{-1}). A conservative scenario, assuming constant exposure to undiluted effluent, estimated a very low likelihood of environmental damage in this case study.

1. INTRODUCTION

Quantitative ecological risk assessment, based on comparison of the probabilities of exposure and response, has an increasingly accepted place in management of the non-radiological, ecological impact of human activity. This approach to ecological risk assessment is premised on the existence of many, usually laboratory-based, studies of the responses of various biota when exposed to pollutants. The publication of databases, such as the recently completed FASSET Radiation Effects Database (FRED; [1]) that brings together large amounts of published information on the effects of ionizing radiation on different biota, potentially facilitates the application of quantitative and probabilistic ecological risk assessment to assess the effect of radioactive pollutants in ecosystems. This study reports an example of the combined use of three pieces of software that a) estimate the exposure (dose or dose-rate) of ecosystem components based on available activity concentration monitoring data, b) provide response data for potentially affected organisms and c) estimate a probability of affect in the receiving ecosystem.

2. METHODS

2.1. Calculation of exposure

The 'Radiological impact assessment for coastal aquatic ecosystems' software (Version 1.15; [2]) uses concentration factors (CFs) to calculate both external and internal dose to a series of ellipsoidal reference organisms – weighting factors for habitat and the primary radiation types are incorporated. Average monthly tritium concentrations for liquid effluent from a research reactor facility [3] ranged from 1.23×10^6 to 9.9×10^7 Bq m^{-3} (mean ± Standard Deviation, SD, = $1.62 \pm 2.80 \times 10^7$; n = 12) and were used for calculation of exposure data. Default CFs and weighting factors for tritium (Low energy beta = 3) and habitat (proportion of time spent in contact with water or sediment) were used. Weighted dose-rates (external plus internal in μGy hr^{-1}) for six relevant groups of biota (i.e. zooplankton, macrophytes, fish eggs, benthic molluscs, small benthic crustacea and pelagic fish) were calculated. Calculated dose-rates for pelagic fish ranged from 1.2×10^{-2} to 9.7×10^{-1} μGy hr^{-1} (mean ± SD = 1.60

$\pm 2.75 \times 10^{-1}$; n = 12) and for fish eggs from 1.50×10^{-5} to 1.20×10^{-3} µGy hr^{-1} (mean ± SD = 1.93 $\pm 3.39 \times 10^{-4}$; n = 12). The dose-rates for other groups were equivalent to those for the fish. Exposure data were transferred to a spreadsheet for input to the ecological risk assessment software.

2.2.1. Assumptions

(1) The assumptions made in the Assessment Spreadsheet 1.15 are detailed in [2].

(2) For this study, organisms are continuously exposed to the average concentration (monthly) of tritium in undiluted liquid effluent (i.e. a worst case scenario). This conservative assumption implies that the receiving ecosystem is captive at the beginning of the liquid effluent pipeline. The exposure scenario therefore ignores the anticipated dilution of effluent in its passage from the liquid waste storage facility to the sewage treatment plant (typically ~25 times; [3]), then on to the cliff-based outfall and, after release, in the near-shore marine environment. The scenario also ignores any behavioural aspects (e.g. movement) of many marine organisms that would limit their exposure to the effluent.

(3) Estimated exposure, expressed as a dose-rate, can be reasonably converted to a monthly dose according to the formula dose (Gy) = dose-rate (µGy hr^{-1}) × 24 × 30.4 / 1000000.

(4) Calculated dose-rates (and doses) from exposure to tritium are for inorganic tritiated water [2] and are additional to the background from the range of radionuclides naturally found in seawater.

2.2. Selection of response data

The FASSET Radiation Effects Database (FRED) comprises published information from laboratory and field studies describing the exposure of various organisms to radiation. In the present context, the challenge was to readily select from this large database information specifically useful in quantitative risk assessment. Both acute and chronic response data for relevant wildlife groups (aquatic invertebrates, aquatic plants, crustaceans, fish, molluscs and zooplankton) were copied into a spreadsheet using the data transfer option available within the database [1]. A simple data filtration, based on classificatory information provided within the FRED, was applied to exclude data with doses/dose-rates identified as 'background' and where both the standardized dose and dose-rate equalled zero. Data that were not either Lowest Observed Effect Dose-rates (LOEDR) or Highest No Effect Dose-rates (HNEDR) were then excluded, leaving a total of 554 HNEDR and LOEDR response entries from some 123 references. Of these, some 83% were for fish. It should be noted that identification of LOEDR and HNEDR in the FRED is only based on references specifying these values. As a result, other relevant references will have been excluded.

2.2.1. Assumptions

(5) After conversion to standardised units (µGy hr^{-1}), all response data are comparable regardless of the radiation type to which the biota were exposed.

(6) Responses identified with HNEDR or LOEDR within the FRED are the most relevant for risk assessment. This represents a conservative data selection because it includes dose-rates (or doses) for which no effect of radiation was measured and excludes dose-rates (or doses) for which more serious effects were observed.

(7) All response effects are equally valid. These include, for example, feeding behaviour, effects on weight and/or size, changes in blood haemoglobin, chromosomal abnormality, sterilization of adults, percent survival of different life stages and after various time periods, and lethal doses to 50% of populations. This is also conservative, insofar as some of the biochemical changes, though measurable, may not translate to real population effects.

2.3. Ecological Risk Assessment

AQUARISK [4] fits probability distributions, separately, to both exposure and response data and uses a convolution algorithm to evaluate the overlap between the two. Probabilistic estimates of exposure criteria (dose or dose-rate) are derived from the distribution fitted to the response data selected from the FRED. Five combinations of exposure and response data were run. Exposure data included either all six groups of biota or just dose or dose-rate to fish. Response data were varied by a) selecting either chronic or acute data or using both together and b) selecting data regardless of radiation type (e.g. alpha, beta, gamma) or using only data for published studies that used tritium specifically.

2.3.1. Assumptions

(8) The assumptions made within AQUARISK are detailed in [4]. These always err on the precautionary side, i.e. towards realistic but slight overestimates of impact.

(9) Only response data with non-zero dose or dose-rate values are accepted for distribution-fitting within AQUARISK (i.e. zeroes are ignored).

(10) Chronic response data are most relevant to the chosen exposure scenario. Acute exposure and response data are best compared on a dose basis.

3. RESULTS

When both chronic and acute response data are included (columns 1, 2 and 3 in Table 1), the estimated dose criteria for 5 and 10% effect levels (i.e. protection targets of 95 and 90% of species, respectively) range from 15 to 1300 μGy hr^{-1}. This range is comparable with the various guidelines given in Table 3.1 of [2] which range from 50 to 1000 μGy hr^{-1} for aquatic organisms. If a 95% species protection target is adopted, the estimated dose criteria range from about 15 to 30 μGy hr^{-1}. A more precautionary approach is to use the 95% lower uncertainty limit of this distribution, giving criteria ranging from about 0.1 to 10 μGy hr^{-1}, significantly less than any given for aquatic systems in [2]. Note, however, that the comparatively few response data available for studies specifically using tritium (column 3, Table 1) increases the uncertainty, leading to a very low $HC_{5,95}$.

TABLE 1. RESULTS OF ECOLOGICAL RISK ASSESSMENT, USING AQUARISK

DATA		(1)	(2)	(3)	(4)	(5)
INPUT	EXPOSURE	[a] 6 groups	Fish only	Fish only	6 groups	6 groups
	RESPONSE (acute/chronic?)	Acute and Chronic	Acute and Chronic	Acute and Chronic	Chronic only	Acute only
	RESPONSE (radiation type?)	[b] All radiation types	All radiation types	Tritium only	All radiation types	All radiation types
	EXPOSURE data: n =	72	12	12	72	72
	RESPONSE data: n =	422	347	16	113	387
	Units:	μGy hr^{-1}	μGy hr^{-1}	μGy hr^{-1}	μGy hr^{-1}	Gy
DOSE	[c] $HC_{5,50}$	15	33	18	0.65	0.3
CRITERIA	[d] 95% lower limit ($HC_{5,95}$) =	3.1	8.0	0.08	0.10	0.23
	[c] $HC_{10,50}$	860	1290	150	8.3	0.62
	[d] 95% lower limit ($HC_{10,95}$) =	290	420	2.1	2.3	0.50
ECOLOGICAL	Likelihood of exceeding $HC_{5,50}$ =	0.001	0.001	0.002	0.05	0.000
RISK	Likelihood of exceeding $HC_{5,95}$ =	0.007	0.003	0.32	0.29	0.000
	Species Affected (% ± SD) =	0.2 ± 0.4	0.1 ± 0.1	0.2 ± 0.3	0.8 ± 2.2	0.0 ± 0.3

[a] Doses from exposure to tritium were calculated for zooplankton, macrophytes, fish eggs, benthic molluscs, small benthic crustacea and pelagic fish.

[b] Radiation types are listed in the FRED as alpha, beta, gamma, mixed and x-ray.

[c] $HC_{5,50}$ and $HC_{10,50}$ are Hazardous Concentrations (HC) estimated to affect 5 and 10% of species, i.e. to protect 95 and 90% of species, respectively.

[d] $HC_{5,95}$ and $HC_{10,95}$ are lower bounds for the corresponding HCs, determined with 95% confidence.

Selection of chronic response data (column 4, Table 1), gives an estimate of $HC_{5,50}$ of about 1 µGy hr^{-1}, with a 95% lower uncertainty limit of 0.1 µGy hr^{-1}. Hence, using chronic data only, which most closely represents the exposure scenario that assumes continuous monthly exposure, reduces the adverse effect criteria by more than an order of magnitude. Selection of acute response data (column 5, Table 1) gives an estimate of $HC_{5,50}$ of 0.3 Gy, with a 95% lower uncertainty limit of 0.23 Gy.

From 0.0 ± 0.3 to 0.8 ± 2.2% percent of species are estimated as potentially affected (%SA ± 1 Standard Deviation) in the conservative exposure scenario adopted in this study – in all cases, the error includes a zero effect. The first three combinations of exposure and response data (columns 1 to 3, Table 1) all estimate a similarly low 0.2% SA. The greatest % SA is estimated when chronic response data alone are selected from the FRED and zero % SA is estimated when acute response data are selected. The likelihood of exceeding the $HC_{5,50}$ (Table 1) is estimated to be 5% or less, and the likelihood of exceeding the more stringent $HC_{5,95}$ (Table 1) is at most about 30%.

4. DISCUSSION

Given the conservative selection of response data (i.e. using only HNEDR and LOEDR data), it is perhaps not surprising that AQUARISK estimates a quite low criterion for the protection of 95% of species (15 µGy hr^{-1}, with a 95% lower uncertainty limit of 3 µGy hr^{-1}). This estimation is most closely aligned with the Canadian criteria for aquatic biota, with the $1/10^{th}$ safety factor applied (i.e. 5 to 10 µGy hr^{-1}; [2].

The methods used here were chosen to make the integration of the three pieces of software as straightforward as possible and represent very much a ‘first pass’. A very wasteful selection process was applied to the FASSET Radiation Effects Database as a result, with only 10% of the available data entries retained. Nevertheless, the response data can be seen as appropriately precautionary and in sufficient quantity for statistical purposes. It should be noted that the FRED is limited by the quality of the information existing in the literature and interpretation of data in the FRED should be cautious, guiding its users to read the original literature. LOEDR and HNEDR values are determined purely according to statistical assessment of their data by the experimenters, and a review of LOEDR and HNEDR entries within the FRED may be valuable. Ideally, a consistent reporting approach should be adopted in future experimental work to ensure that response curves can be generated, and that all basic data are reported - this is often not the case at present. Nonetheless, the FRED provides a very useful resource.

We have demonstrated the potential for integrating available software packages to produce a quantitative, probabilistic, ecological risk assessment relevant to the release of radioactive liquid effluent into a marine ecosystem. A very low likelihood of environmental damage is estimated for this case study, even when based on overestimates of the likely real dose to biota in the receiving ecosystem.

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Obtaining and development of an environmental impact assessment Methodology for isolation hazardous wastes

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Abstract. Taking into account, the experience of the practical applications of the safety assessment in the radioactive waste management, the International Atomic Energy Agency (IAEA) recommendations in the topic of radioactive waste isolation, the standards and national and international legislation about noxious substances to the environment and their restriction limits, the best international practices and accepted and adopted approaches of isolation hazardous wastes sites, under Cuba particular conditions, a methodology is developed to obtain and/or confirm the hazardous wastes safety isolation, as a tool able to carry out the assessment of facilities to build and/or place where hazardous wastes isolated from the environment, for the risks of its contamination. The methodology, embraces the evaluation of technical, economic and social topics, allowing development of a process of documented site and a safety assessment that allows estimation of the environment possible impact for hazardous waste isolation. Their main contribution resides in the creation of a scientific-technique necessary guide for the evident demand of carrying out the most organised, effective and hazardous waste safety management. The obtained instrument, keeps in mind the analysis of all hazardous pollutants (radioactive and non radioactive), being alone shown in this work the approaches selection for the obtaining and/or evaluation of the best site and the steps description to continue for the definition of the main scenarios and models to take into account in the valuation of the possible liberation and incorporation roads to the environment of the non radioactive pollutants in the safety assessment of hazardous wastes isolation.

1. INTRODUCTION

The growing concern to evaluate the degradations in the environment and the human, has created the necessity of development different methodologies which allow to determine the possible risk associated to anyone practice. The methodologies existent, with the objective of carry out the environment impact assessment (E.I.A) for radioactive and not radioactive waste isolation, are not adjusted to the Cuban particular conditions [3, 4]. The wastes, that can not be recycled or reused by their characteristics, will require of the search of appropriate solutions for its isolation, guaranteeing the total protection. Based on the experience obtained in the radioactive waste final management, has been development a Methodology for evaluate the most feasible and safety hazardous wastes isolation. The developed Methodology is conformed by two stage, first one, the site selection and evaluation where the hazardous waste will be isolated, several criteria are taking into account (scientific-technicians and economic-social) in order to evaluate the site useful; the second stage embrace the safety assessment which allows to evaluate the possible behaviour of the hazardous waste isolation system for the current and future conditions. [3, 4]. The safety assessment includes the E.I.A of the facility or site for the hazardous waste isolation. There are different listings about toxic substances, this Methodology use the list accord to the Basel Convention classification.[5].

2. CREATED METHODOLOGY FOR THE CONFIRMATION OF THE SURE ISOLATION OF HAZARDOUS WASTE

2.1. Site selection criteria

The site selection process is defined taking account the following approaches:

(1) *Geology:* The geologic conditions should be stable, with rocks of high physical-mechanics resistance, the land should not have flaws neither important fractures which affect the stability of the host rocks.

(2) *Seismology:* The permissible values for maximum earthquakes of project (100 years) and maximum of calculation (for 10 000 years) are respectively of V and VII (according to the MSK scale).

(3) *Topography:* The characteristics of the relief should be not very abrupt which to facilitate the subsequent erosion neither it should be low that it facilitates a possible flood due to torrential rains.

(4) *Hydrology and hydro geology:* The site should be sufficiently far from bodies of superficial waters (not less than 1Km.). The aquifer should not possess waters aggressive and drinkable to the consumption. The aquifer regime and the movement of the underground waters should be minimum.

(5) *Social-economical condition:* Low population density and to be relatively far from important facilities and little economic and social development actual and perspective, not important use of the land and the possible affectation to cultivation and mineral locations of economic importance.

(6) *Natural extreme events or due to human:* The process should be preceded of the occurrence probabilities analysis and consequences of natural ends events or due to the man intrusion. Such are: weather events ends, flaws of near basin to the location, possible sabotages, aeroplane fall, direct impacts of explosive, etc. The probabilities of occurrence of such events should be minimum.

(7) *Environmental impact:* The site selection should be endorsed of a study which shows the environmental baseline of the region selected, as well as the evaluation of the most relevant environmental impacts and factors inside the facility to locate. The results should justify the technical and socio-economic favourable of the site proposed and the importance of the facility in the general development of the country with the indispensable minimum affectations to the global environment.

2.2. Develop of the methodology

It has been created a Methodology as the Guide for the gradual definition of the site [3, 4]. This methodology includes since a national study to regional scale, until the selection and characterisation of the most favourable sectors. The Figure 1, show the Methodology scheme.

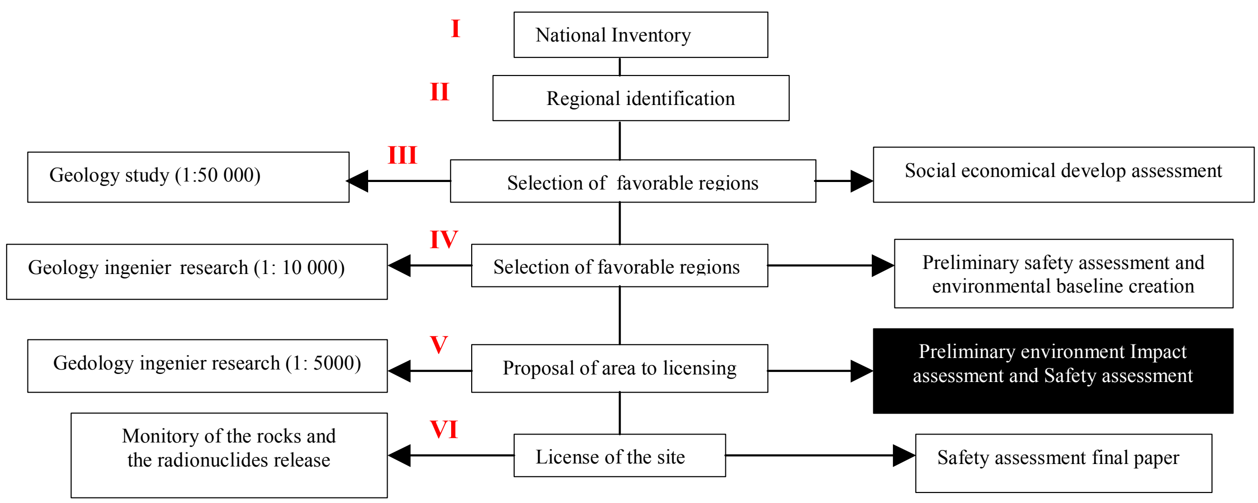

FIG. 1. Methodology scheme by the process of site selection.

2.3. Site selection process

- *National inventory (I):* Taking into account the requirement, the best international practices, and the Cuban particular conditions, is necessary a national search of the most favourable geologic formations.
- *Regional identification (II):* The geologic formations are technically evaluated to regional scale E-1:100 000 and according to the acceptance criteria adopted for the site, the most favourable regions are selected.
- *Selection of favourable regions (III):* There are an evaluate of their socio-economic development and in their environmental and geologic state to scale (E-1: 50 000) and the most favourable regions are selected.
- *Selection of favourable sectors (IV):* In the areas selected, are carried out a series of complex geologic investigations (geologic, engineer-geologic, geomorphology, hydrological, geophysical, etc.) to detailed scale (E-1: 10 000). The probability of extreme events mentioned above is also evaluated.
- *Proposal of the area to licensing (V):* There are the same investigations to Scales: 1-2 000 and 1-5 000, after of confirms the proposal site with investigations to scale 1: 2 000 it can pass to the stage VI ("License of the site"). The safety assessment is developed in the area of site isolation, in the case of facilities already existent, its should be initiate the investigations and safety assessment, starting from the stage V.

2.4. Evaluation process of the environment impact due to the hazard waste isolation

The fundamental objective is evaluate the role of each one from different pathway to the environment and the human for several pollutants that could be released from hazardous waste facility isolation. Being adopted a conservative scenario, the failure of different barriers of the facility is supposed in the time and the later transport of the toxic pollutants by environmental geologic, until the biosphere for different pathway.

2.4.1. Safety assessment process

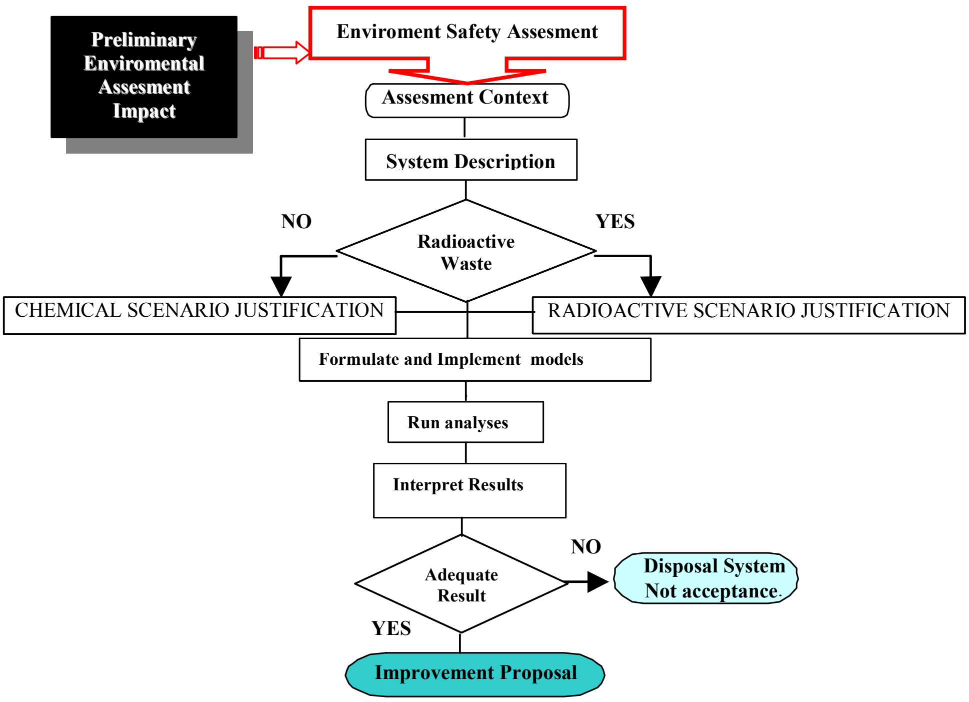

FIG. 2. Scheme of the safety assessment process.

The adopted methodology [3, 4], establish the assessment context, identify the scenarios to evaluate, define the models to use and finally evaluate the results (impacts, etc.), comparing it with the defined limits (endpoints). The safety assessment process includes activities such as:

— *Assessment context:* It is the initial stage where are included the fundamental elements of the analysis which are: *Assessment purpose* (the analysis objective is included, what will be analysed), *Assessment time schedule* (period of time evaluated), *Safety indicators* (impact, risk, environmental concentrations, etc.) and *Types of waste to assess* (hazardous wastes and physical form).

— *System description:* It requires the information about hazardous wastes characteristics, conceptual design and the characteristics which allow to establish the conceptual model of the disposed wastes release.

— *Obtaining, generation and scenarios justification:* Taking into account the possible pathways release, is possible to evaluate as scenario the not controlled release (bad practical isolation). For no-radioactive wastes assessment, the identification and justification scenarios process will be carried out through an E.I.A. methodology, its main steps are:

(1) Identification of the Project actions, which cause impact in the physical environment (initial stage, construction stage, operation stage, closure and restoration stages).

(2) Definition of impacted environmental factors (natural and socio-economic) by the Project actions. In the Physical environment (biotic, not biotic and perceptual environment) and the Socio-economic environment (social cultural and economical).

(3) Determination and classification of the environment impact.

(4) Importance calculation of the classified impacts, using the equation below:

$$Ip = \pm [\, 3\times I + 2\times EX + MO + PE + RV + SI + AC + EF + PR + MC]$$

where:

Ip: the importance impacts; I: Intensity; EX: Extension; MO: Moment; PE: Persistency; RV: Reversibility; SI: Synergy; AC: Accumulation; EF: Effect; PR: Periodicity; MC: Recoverability. The coefficients details are described in the environmental impact study [1]. Once obtained the Ip values for each impacts, a summa of results is carried out and according to the signs and the amount, the project acceptance are evaluated. Then, using matrixes approach, the importance and impact evaluation are defined.

(5) Proposal of preventives, mitigates and correctives Measures. Monitoring plan definition.

The measures, can be preventives, mitigates and correctives according to the importance of the impact and the Monitoring Plan includes the sequence of actions and the fulfilment of the proposal measures. In the Identification of the exposure pathways, according to the results obtained in the preliminary E.I.A, the relevant pathways of the release of no-radioactive pollutants from the isolation systems are selected. The scenarios, take into account all physical environments where occur the release, discharge, transport and the final incorporation of the no-radioactive pollutants to the environment and the human.

— ***Formulate and implement models:*** The conceptual model is became to mathematical models which describe the different processes, allowing to find a quantitative solution of the problem. The environmental models are used for the relative concentration determination of the pollutant in each compartments (water, soil, air, biota, etc) and to know the degradation process that can influence the behaviour of the pollutant [2, 3].

3. RESULTS INTERPRETATION AND ANALYSIS

The final step is the comparison of the results with the established approaches. The objective is to evaluate if the installation or the activity are sufficiently safe in relation with the risk. The effect of the toxic substances is associated to their final concentrations and the pollutant specific toxicity, therefore, the possible negative impacts should take into account the standards, requirements or accomplished studies which allow take an idea of their toxicity. In the comparison process, appear several uncertainties, this situation makes that the methodological approach of the safety assessment requires of uncertainties and sensibility analysis.

4. CONCLUSIONS

Have been created and developed a Methodology for the obtaining and confirmation of the hazard waste safety isolation, as tool able to carry out the evaluations of planned and current facilities or sites used to isolate hazardous wastes from the environment.

5. RECOMMENDATIONS

The Methodology should be generalised during the hazard waste isolation process.

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Radioactive waste management and environment protection in Latvia

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Abstract. The report provides and overview of radiation safety and environmental protection legal framework applicable to radioactive waste management in Latvia. There are some details about waste generations and safety assessments of radioactive waste repository, which were used to draft recommendations for modernisation of national legal system and enhancements of infrastructure. The ideology of the report is based on a widely used approach for protection of the environment from potential negative impacts from radiation – insurances of protection for public should ensure adequate protection for the entire environment (the assumption is that mankind is the weakest part of the environment and if we could adequately protect the general public then we are able to protect all other parts of the environment).

1. INTRODUCTION

Satversme (the Constitution of Republic of Latvia) stated in Article 111 - "The State shall protect *human health* and guarantee a basic level of medical assistance for everyone." and in Article 115 - "The State shall protect the right of everyone to *live in a benevolent environment* by providing information about environmental conditions and by promoting the preservation and improvement of the environment."

The Parliament and the Government of Latvia have devoted many efforts to establish the framework legislation and the system of supervision and control for protecting humans and the environment from ionising radiation.

2. LAWS AND REGULATIONS

The basic law dealing with the protection of people and environment against the hazards of ionising radiation in the Republic of Latvia is the Law on Radiation Safety and Nuclear Safety (2000). Latvia had the similar law since 1994, which now recently was modernised and updated. The principles of radiological protection includes the well know ICRP protection principles of justification, optimisation of protection and exposure limitation.

The EU and IAEA requirements and ICRP recommendations in the field of the radioactive waste are included in the Cabinet of Ministers Regulations – procedure for licensing, protection against ionising radiation, protection against ionising radiation transporting radioactive materials and practices involving radioactive waste and related materials. More over, Latvia has acceded to Joint Convention on the Safety of Spent Fuel Management and on the Safety of Radioactive Waste Management (2 February 2000) and to Convention on Nuclear Safety (1 November 1995).

The requirements for process of the environmental impact assessment (thereof and forward – EIA) are determined in Law "On Environmental Impact Assessment" and the Cabinet of Ministers Regulations "On Procedures for Environmental Impact Assessment" (No.213, 15.06.1998). The Law and Regulations, mentioned above, implement EC Directive 85/337/EEC, which was amended by the Directive 97/11/EC, that require assessment procedure and planning (building) permission in Latvia.

3. REGULATORY BODY AND RELATED AUTHORITIES

A basic principle for regulatory framework is - only the Parliament or the Cabinet is entitled to issue legally binding regulations to be applicable for any legal or natural person. Therefore all legal acts and regulations in this field had been issued by them.

The Ministry of the Environment conducts the governmental policy for the radioactive waste management and the Radiation Safety Centre (established in July 2001, being under the supervision of the Ministry of the Environment) provides supervision and control for all practices with radiation sources.

In according with Latvian legislation the State EIA Bureau co-ordinate questions for EIA procedure. The aim of the State EIA Bureau is to prevent or to reduce adverse impact on the environment caused by activities of natural persons and legal entities.

4. CURRENT STATUS, PLANS AND PRACTICES

The activities with radioactive waste in Latvia are managed by the limited liability non-profit state company “RAPA” Ltd. (Salaspils Nuclear Reactor (thereof and forward – Reactor) and Near-surface Radioactive Waste Repository in Baldone (thereof and forward – Repository)). The main tasks for RAPA Ltd. are to provide radioactive waste collection, transportation, storage and disposal with no risk to the people health and for environmental and to safe maintenance of Reactor. The company is fully financed by the State, and receives supplementary funding from the import tax on radioactive substances.

Latvia has neither any nuclear power plant nor any nuclear fuel cycle facility. The Reactor - Soviet designed water pool type research reactor (IRT) – with the nominal thermal power 5 MW located in Salaspils (25 km distance from Riga). This Reactor in June 1998 was finally shut down. In addition to the other producers of radioactive waste from the medicine, industry and science, the Reactor is the major operator in Latvia that have significantly increase the yearly amount of radioactive waste for the Repository in recent years and will do so also in nearest future.

The concept for decommissioning of the Reactor was approved by the Cabinet of Ministers on 26 October 1999 - the decommissioning of the Reactor shall be carried out until full dismantling of facility till “green field” status. Up to now for these purposes the Reactor received financing from Latvia’s Environmental Protection Fund and international assistance. There are allocated also funds from the State budget.

Radioactive waste from decommissioning of the Reactor is proposed to be disposed in the Repository. The free space of the Repository is currently around 600 m^3 (in vault No.7; vaults No.1-6, with the total volume 480 m^3, are now filled and closed), which is not sufficient for purposes of decommissioning of the Reactor and it is necessary to build a new radioactive waste vault. The main problem related to spent fuel - there is no any repository for storage and dispose of long-lived and high-level activity radioactive waste in Latvia. Spent fuel management considers several options – to send reprocessing or disposing to the state-origin or another country.

5. SAFETY ANALYSIS OF THE RADIOACTIVE WASTE MANAGEMENT

During 2000-2001 a consortium of EU radioactive waste management agencies “CASSIOPEE”, within the framework of European Union financed programme, performed a “Long Term Safety Analysis of Baldone Radioactive Waste Repository and Updating of Waste Acceptance Criteria” (forward – Safety Analysis). The main objective of that project was to provide advice to the Latvian authorities on the safety enhancements and recommendations for updating of waste acceptance criteria for disposal facility and has included the following main activities:

— analysis of the current status of radioactive management waste and at the Repository in particular;
— development of the short and long-term safety analysis, including the planned increasing of capacity for disposal and long term storage and the analysis for the post-closure period;

- development of the EIA, for the foreseen installations, considering the non radiological components;
- proposal of recommendations for future updating of radioactive waste acceptance criteria;
- proposal of recommendations for safety upgrades to the facility.

The recommendations of Safety Analysis deal with general aspects of the management (mainly storage versus disposal of long-lived sources), site and environmental surveillance, packaging (qualification of containers, waste characterisation requirements), the design of an engineered cap and strategies for capping. These recommendations are reflected in radioactive waste management regulations and the draft of Concept of the radioactive waste management (forward – the Concept).

The main practical conclusions from long-term safety analysis are included in the Concept. The prior aim of the Concept is to stimulate the development of the environmentally sound and population friendly system of the radioactive waste management, ensuring sustainable economic development in accord with the fair social system for the protection of human health and the environment.

The Concept extra aims:

- to construct a new radioactive waste storage vault in the Repository - to ensure the disposal of all the radioactive waste from the decommissioning of the Reactor;
- to construct a long-term storage for spent sealed sources that must not be stored in a near-surface repository and to investigate the possibilities for construction of a geological depository in Latvia;
- to improve the radiation safety in the Repository:
  (1) to enhance the long-term safety and decrease to an acceptable level the potential influence to the population, it is necessary to erect additional barriers above the closed vaults of waste;
  (2) to improve the system of radiation monitoring in the Repository;
- to implement compensation mechanism to the local government for the risk associated by the radioactive waste repository.

6. ENVIRONMENTAL IMPACT ASSESSMENT

EIA is a multi-stage procedure, which is required prior for construction of significant facilities that may leave harmful impacts on the environment. This procedure includes a set of measures, which envisages review of the state of the environment at the given territory, review of the environmental impact of the facility, preparation of proposals for reduction or prevention of negative impacts, as well as development of necessary monitoring requirements for monitoring of remaining impacts.

According to the Regulations of the Cabinet of Ministers No.301 “On the Procedure for the Issue of a Special Permit (Licence) or Permit for Activities involving Ionising Radiation Sources and Procedure for Public Dispute on the Establishment of Ionising Radiation Facilities of State Significance or on Essential Modifications thereto” (3 July 2001) EIA process will be perform before requesting a Special Permit (Licence) for the establishment of ionising radiation facility of national significance or the performance of essential changes therein.

The EIA procedure starts with a sending of notification by the proponent to the State EIA Bureau. After that the State EIA Bureau makes the decision about the necessity of EIA, requiring of additional information from the proponent, if necessary, and should send a written notification of the decision within 2 weeks from the date of registration of the notification to the proponent.

In case of EIA is considered necessary for a project the State EIA Bureau prepares and sends to the proponent an EIA programme on which the EIA report is based. The proponent is responsible for the realization of the EIA according to the EIA programme and should submit the EIA report to the State EIA Bureau for evaluation. The State EIA Bureau in the evaluation of the EIA report should take relevant authorities’ conclusions and the result of the public hearing as well as expert’s opinion into account. After including of possible amendments the proponent should submit the final EIA report to the State EIA Bureau. The final EIA report should also include the evaluation of the impacts of the proposed activity and its alternatives including alternative zero. EIA procedure is over when the State EIA Bureau publishes a notice that it has given its opinion on the final report, and notifies to the opportunity to examine the aforesaid opinion and the final report.

The time of action of the final report is three years. And the final the proponent shall submit to the relevant public or municipal institution the final report and opinion of the State EIA Bureau on the final report together with documents required by other normative acts.

The process of the EIA in the field of radiation safety and nuclear safety has not been performed in Latvia yet. According to the law “On Environment Impact Assessment” the EIA is necessary for:

(1) Decommissioning of the Reactor. This process is taking place during the year 2002–2003 by the Ministry of the Environment.

According to the Programme of the EIA it is necessary to indicate such questions in the final EIA report for the decommissioning of the Reactor:

— analysis of legislation requirements and international commitment;
— description of the Reactor and this locality;
— description of the Concept for decommissioning of the Reactor;
— location of the radioactive and other hazardous materials and waste;
— environmental and public impacts;
— measures for reducing environmental impacts;
— radiological and other type control in the object;
— measures for late decommissioning stage;
— possible problems regarding with other measures, deficiencies and other aspects which interrupt the project;
— the summary and evaluation of the results of the Public Hearing;
— a non-technical information summary.

(2) Expansion of the Repository to hold of radioactive waste from decommissioning of the Reactor. The Ministry of the Environment started the EIA process for expansion of the Repository in 2003.

The Safety Analysis presents the results of the preliminary EIA of constructing, operating and restoring the new vault and information from this project will be used in further EIA process. According to Safety Analysis it is necessary to analyse and describe:

— environmental effects of construction, operational and institutional (a period of institutional control for 300 years) phase of project – noise and vibration, traffic, land-take, air quality-dust, water pollution, visual effects, flora and fauna, labour, socio-economic benefits, resource usage, environmental monitoring;
— description of local environment (existing site, locality, population, flora, fauna and landscape, topography, geomorphology and geology, water resources, air quality, climate, recreational use, industry, transport);
— non-radiological impact (safety of workforce, safety of the public);
— radiological impact (to analyse pathways and scenarios, main hypotheses).

7. CONCLUSION

The legislation in the field of radioactive waste management and environmental impact assessment are developed according to EU, IAEA requirements and ICRP recommendations. The aim of the field of the radioactive waste management is to stimulate the development of the environmentally sound and population friendly system of the radioactive waste management, ensuring sustainable economic development in accord with the fair social system for the protection of human health and the environment.

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Emission of the radioactive substances into the atmosphere and Druksiai Lake from Ignalina Nuclear Power Plant in 2002

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Abstract. The paper presents the overview of the emissions of the radioactive substances into the atmosphere and Druksiai lake from the Ignalina Nuclear Power Plant (INPP) (Lithuania) during 1984-2002. The paper also covers the estimated annual doses to the critical group of population in 2002.

According to the legal acts of Lithuania, the public exposure from all controlled practices (excluding natural background radiation and medical exposure) shall not exceed 1 mSv per year and the annual dose constraint to the general public due to operation and decommissioning of nuclear facilities - 0.2 mSv.

The results of the paper reflect the situation which shows that public annual exposure due to operation of INPP in 2002 has not exceeded annual dose constraint.

1. INTRODUCTION

The legal basis for radiation protection allowing to protect people and the environment from the harmful effects of ionizing radiation is established by the Law on Radiation Protection of the Republic of Lithuania (1999) and other legal acts. The basic radiation protection requirements are set by the Lithuanian Hygiene Standard HN 73:2001 „Basic Standards of Radiation Protection" (2001). The Lithuanian Hygiene Standard HN 87:2002 „Radiation Protection in Nuclear Facilities" (2002) sets out radiation protection requirements for workers working at nuclear facilities and for radiation protection of members of the public during the operation and decommissioning of nuclear facilities. Based on the requirements of the HN 87:2002, the annual dose constraint to the general public due to operation and decommissioning of nuclear facilities shall be 0.2 mSv. Limitation of discharges of radionuclides into the environment from operation of nuclear facilities is regulated by the Lithuanian Environment Normative Document LAND 42 – 2001 „Limitation of Radioactive Discharges from Nuclear Facilities, Permitting of Discharges and Radiological Monitoring" (2001), where the maximum permissible levels of radioactive discharges from the Ignalina NPP are given. Overview of the emissions of the radioactive substances into the atmosphere and Druksiai lake from the Ignalina NPP in period 1984-2002 and particularly, in 2002, estimated annual doses to the critical group of population is presented in the paper.

2. BACKGROUND INFORMATION

Ignalina Nuclear Power Plant (INPP) with 2 reactors of RBMK type is located close to the borders of Belarus and Latvia. The plant was built on the bank of Lake Druksiai. Visaginas is a satellite town situated about 6 km from the INPP and has about 33000 inhabitants. The first Unit of the INPP was put into operation on December 31, 1983, the second Unit - on August 30, 1987. The projected thermal power of a single Unit of the INPP is 4.8 GW. The electric power is equal 1.5 GW. The INPP uses water from Lake Druksiai for cooling of the reactor.

The radiation state in the environment objects of the sanitary zone and within a 30 km radius monitoring zone of the Ignalina NPP in 2002 was estimated according to the results of the samples which have been taken by the Ignalina NPP from sanitary and monitoring zone.

3. PRINCIPLES OF RADIOACTIVE RELEASES TO THE ENVIRONMENT ACTIVITY LIMITING

Public exposure from all controlled practices (excluding natural background radiation and medical exposure) shall not exceed 1 mSv per year [1]. Dose assessment shall be made for critical group members. Different critical groups can represent different release routes (e.g. into air or water) as well as different radionuclides. As members of the critical group can be irradiated by other controlled and non-controlled (exempted) [1] sources simultaneously the average annual dose to the critical group members due to operation of nuclear facility, including short-time anticipated operational transient, shall not exceed the dose constraint. The established dose constraint for releases from operating and decommissioned nuclear facilities, is 0.2 mSv/year [2].

Only those subjects may release radioactive substances into the environment (in liquid, gaseous or solid form) that have obtained respective permission in advance [3]. Permission for radioactive discharge is the legal document qualifying the discharge of radioactive materials into environment; this document contains radioactive discharge limits for atmosphere and water [3]. In Lithuania, the subjects who are willing to get the permit shall have to submit the Ministry of Environment an application for the permit, the plan of radioactive discharge as well as the plan of radiological monitoring.

For calculation of doses to the critical group members, site specific dose factors and discharge limits have been assessed, taking into account the peculiarities of radionuclide migration in Ignalina NPP environment and site-specific life style and nutrition features of critical group members. In cases if the number of data is not sufficient, the conservative approach has been used to aximise dose calculation factors. Critical group members dose assessment has been performed as follows [3]:

- in the case of farmers – external exposure from immersion in the cloud and radionuclides deposited on the ground as well as resuspension of deposited radionuclides and internal exposure due to inhalation and ingestion of radionuclides in the food stuffs;
- in the case of fishermen – the external dose, resulted by radionuclides in the lake water and in the coastal zone sediments as well as the internal dose resulted by the fish used for food;
- in the case of gardeners – external dose resulted by the exposure from radionuclides deposited in the irrigated soil as well internal dose due to consumption of food from irrigated garden and inhalation of resuspensed particles.

Calculated dose conversion factors and discharge limits for atmospheric and aquatic releases from Ignalina NPP are given in [3].

4. VARIATIONS OF RADIOACTIVE EFFLUENTS FROM IGNALINA NPP

The release of the radioactive substances into the atmosphere from the INPP during 2002, have been changing day-to-day: for the noble radioactive gases from 74 GBq/day to 5.9 TBq/day, for the radioactive aerosols from 0.7 MBq/day to 21.3 MBq/day, for ^{131}I - from 5.18 MBq/day to 62.9 MBq/day.

The maximal value of the noble radioactive gases release per month, from the INPP in 2002 was 19.95 TBq in June, the minimal value – 2.77 TBq in May. The maximal value of the radioactive aerosol releases per month from the INPP in 2002 was 151.7 MBq in August, the minimal value – 16.56 MBq in December. The maximal value of the ^{131}I release per month from the INPP in 2002 was 863.6 MBq in August, the minimal value – 65.51 MBq in March.

The total value of the noble radioactive gas release from the INPP was 100.8 TBq in 2002 (as compared to 96.4 TBq in 2001), the total value of the radioactive aerosol release was 0.91 GBq in 2002 (as compared to 1.34 GBq in 2001), the total value of the ^{131}I release was 2.49 GBq in 2002 (as compared to 1.95 GBq in 2001). The INPP during its all operation period (1984-2002) has released total activity of 22.97 PBq of noble radioactive gases, 101.15 GBq radioactive aerosols and 366.78 GBq of ^{131}I. All results are given in the Table 1.

TABLE 1. EMISSIONS FROM INPP DURING 1984–2002

Year	Noble gases, TBq	Radioactive aerosols, GBq	^{131}I, GBq
1984	2200.00	2.90	3.60
1985	4900.00	38.40	80.30
1986	3200.00	8.60	147.00
1987	1600.00	4.07	34.40
1988	2100.00	3.26	38.00
1989	2300.00	1.74	2.74
1990	2400.00	0.98	4.26
1991	1800.00	10.60	10.10
1992	703.00	2.15	1.18
1993	485.00	1.46	0.53
1994	290.00	8.23	2.93
1995	283.00	4.18	7.22
1996	159.00	7.79	11.50
1997	99.70	1.31	6.28
1998	123.00	0.84	6.94
1999	70.57	0.80	2.72
2000	61.29	1.59	2.64
2001	96.40	1.34	1.95
2002	100.80	0.91	2.49

The outage of Unit 1 in 2002 lasted from June 22 until September 29; the outage of Unit 2 lasted from April 6 until June 8, 2002.

The highest release of the radioactive aerosols from the INPP was recorded from April to October, during this period there were released about 76% (690.3 MBq) of all radioactive aerosols in 2002 (907.7 MBq). This period coincides with the outages Unit 1 and Unit 2. The releases from the INPP range all over the year from 65.5 MBq in July to 161.5 MBq in March, however two months – August and September - differs on the big amount of released radioactive substances. In August and September of the 2002 the discharged activities of ^{131}I were 863.6 MBq and 432.4 MBq.

The annual dose estimated due to noble gases released from the INPP for the critical group of the population was 0.067×10^{-6} Sv. The annual dose due to radioactive aerosols and ^{131}I was 0.159×10^{-6} Sv. All radioactive substances released to atmosphere from the INPP in 2002 contributed to 0.226×10^{-6} Sv annual dose for population (dose constraint value for atmospheric discharges is 0.1 mSv/year). Table 2 illustrates annual dose for the critical group of the population caused by releases to atmosphere from the INPP from 1991 to 2002.

Values of the radionuclides activity in the discharged water from the INPP during 1985-2002 is presented in Table 3.

TABLE 2. ANNUAL DOSE FOR THE CRITICAL GROUP OF THE POPULATION CAUSED BY EMISSIONS TO ATMOSPHERE FROM THE INPP DURING 1991–2002

Year	Annual dose for the critical group of the population, μSv		Total dose, μSv
	Noble gases	Radioactive aerosols + ^{131}I	
1991	1.706	0.967	2.673
1992	0.691	0.136	0.827
1993	0.477	0.088	0.565
1994	0.244	0.271	0.515
1995	0.204	0.593	0.797
1996	0.116	0.722	0.838
1997	0.081	0.385	0.466
1998	0.089	0.418	0.507
1999	0.053	0.179	0.232
2000	0.041	0.196	0.237
2001	0.065	0.154	0.219
2002	0.067	0.159	0.226

TABLE 3. VALUES OF THE RADIONUCLIDES ACTIVITY IN THE DISCHARGED WATER FROM THE INPP DURING 1985–2002

Year	Activity, MBq
1985	144.30
1986	3034.00
1987	96.20
1988	229.40
1989	15170.00
1990	25860.00
1991	3219.00
1992	22570.00
1993	4188.00
1994	7683.00
1995	1662.00
1996	5909.00
1997	6103.00
1998	3559.00
1999	2014.00
2000	1253.00
2001	3686.30
2002	1195.40

The maximal value of the radionuclides' activity in the discharged water from the INPP to the Lake Druksiai in 2002 was 671 MBq in December (without ^{40}K), the minimal activity was in November. There were not detected the discharged radionuclides such as ^{131}I, ^{134}Cs, ^{58}Co, ^{59}Fe, ^{51}Cr, ^{95}Zr, ^{95}Nb in the discharged water from INPP to Lake Druksiai in 2002. Still there were detected ^{137}Cs, ^{54}Mn, ^{60}Co and ^{40}K in the discharged water to lake Druksiai. Total activity of the radionuclides in the water discharged from the INPP to Lake Druksiai in 2002 was 1195.3 MBq.

In 2002 ^{137}Cs contributed the annual dose of 0.28×10^{-2} mSv, ^{54}Mn – 2.8×10^{-8} mSv and ^{60}Co – 1.02×10^{-4} mSv. The annual dose contributed by radionuclides in the water discharged from the INPP for the critical group of the population was estimated and was 0.291×10^{-5} Sv which was far below the dose constraint for discharges (0.1 mSv/year).

5. CONCLUSIONS

(1) The annual dose due to noble gases, radioactive aerosols and ^{131}I released from the INPP in 2002 for population contributed to $0.226 \cdot 10^{-6}$ Sv.

(2) The annual dose contributed by radionuclides in the water discharged from the INPP for the critical group of the population in 2002 was $0.291 \cdot 10^{-5}$ Sv.

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Radiological impact assessment of routine atmospheric releases of MAAMORA Nuclear Research Center (MNRC)

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Abstract. CNESTEN is the main national operator of nuclear facilities within the MAAMORA Nuclear Research Center (MNRC) in Morocco. MNRC is presently holding a radioisotopes production facility, radioactive waste treatment and storage facilities and several laboratories using nuclear techniques and radioactive sources. The construction of a TRIGA Mark II research reactor is still on going.

In compliance with national regulations with regard of the licensing process, CNESTEN has performed a radiological impact assessment for the routine atmospheric releases of Maamora Nuclear Research Center facilities in order to obtain the first licence for environmental discharge. The objective of the study is to propose to the national nuclear safety authority the atmospheric release limits and conduct the assessment of radiological consequences to the population reference groups which will be compared to the 1 mSv regulatory annual limit to the public.

A conservative estimation has been developed for: the source term of the annual atmospheric releases, the release conditions at the stacks, the local meteorological data, exposure pathways scenarios of population reference groups. The dose assessment has been performed using two different calculation codes: GASCON (France/Commissariat à l'Energie Atomique) and CAP88 (USA/Lawrence Livermore National Laboratory). Under the conditions and the assumptions related to the routine radioactive atmospheric releases of MNRC, both of the calculation codes had given comparable estimations of individual and collective effective doses to the members of the public. The highest individual effective dose is estimated to 6.80×10^{-4} mSv per year which represents 0.07% of the 1 mSv regulatory annual limit to the public.

1. BACKGROUND

CNESTEN (Centre National de l'Energie, des Sciences et des Techniques Nucléaires) is a governmental institution operating the nuclear facilities at MAAMORA Nuclear Research Centre (MNRC) which is the national infrastructure in Morocco dedicated to research and development using nuclear techniques and applications. MNRC includes a 2 MW TRIGA Mark II research reactor (under construction), a radioisotopes production facility, radioactive waste treatment and storage facilities and several laboratories using nuclear techniques and radioactive sources. MNRC is located at about 20 km North East far from Rabat in the MAAMORA forest and about 8 km East far from the Atlantic Ocean.

2. MOROCCO REGULATORY CONTEXT

The Authorisation, Licensing and Control of nuclear facilities are subject to the legal provisions of the decree n° 2-666-94 [1]. This decree is concerned primarily with the licensing process of nuclear facilities including successively the following stages: construction, radioactive releases to the environment, commissioning, operations and dismantlement. The decree is administered and enforced in Morocco by the Ministry of Energy and Mines (the national nuclear safety authority) with the contribution of other relevant ministries. In compliance with the radioactive releases provisions, CNESTEN has to perform a radiological impact assessment of routine radioactive releases into the environment.

3. MNRC NATURAL ENVIRONMENT

MNRC is located within the MAAMORA forest where the flora is largely dominated by cork plantations. The area is also characterized by intensive agricultural activities due to the soil quality, vegetation cover and favourable weather conditions [2].

The weather conditions are defined from the data collected by the MNRC meteorological station under operation since 1992 and the neighbouring meteorological stations of Rabat and Kénitra [3]. The meteorological data processing has been performed according to the atmospheric dispersion characterisation and provide the wind rose in Figure 1.

The population distribution around MNRC site is based on the data issued from the national census conducted on September 1994 and the data extrapolation to 2004 [4]. The global population living in the area within a radius of 20 km from MNRC site is about 1 611 304 inhabitants. Given the wind conditions prevailing in the MNRC site, the population reference groups have been determined as given in Table 1.

The radioactivity baseline of MNRC site was determined through an environmental monitoring programme based on IAEA technical recommendations [5]. This programme had been carried out since 1994 in the area within a radius of 10 km from MNRC site and the results are showing traces in air samples (Be-7= 10 to 190 Bq/m^3; K-40, U-235 <LD; β tot=120 Bq/m^3), in water samples (K-40, Bi-214, Pb-214 <LD), in grass (K-40=3 to 17 Bq/g; Be-7= 0.10 to 3.34 Bq/g; Bi-214= 0.021 to 0.27 Bq/g; Pb-214= 0.021 to 0.27 Bq/g), summarized in reference [3].

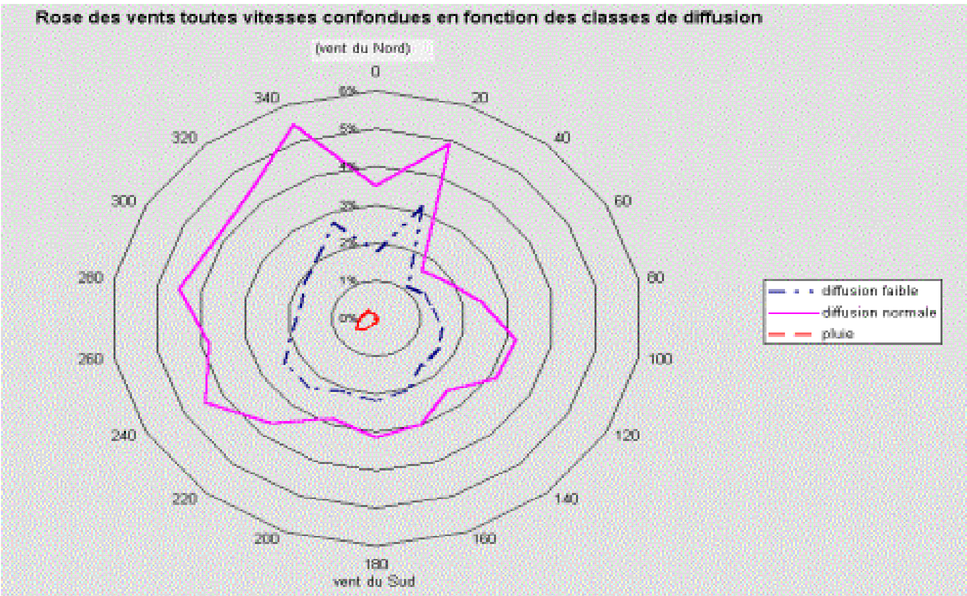

FIG. 1. Wind rose of MNRC site.

TABLE 1. IDENTIFICATION AND LOCALISATION OF POPULATION REFERENCE GROUPS

Reference group	Location Identification	Distance From MNRC site	Sector
1*	MNRC Fences		
2	Forest Guard	~ 1 km	NW
3	Military Base	~ 2 km	SE
4	Douar Nsar Zdagh	~ 5 km	SSW
5	Sidi Taibi	~ 6 km	SE
6	Ouled Taleb Machraa	~ 7 km	E

* Theoretical population group.

4. MNRC ATMOSPHERIC RELEASES

4.1. MNRC Atmospheric releases conditions

The main facilities of MNRC releasing to the atmosphere are: the reactor building (one stack), the radioisotope production facility (one stack) and the radioactive waste treatment facility (two stacks). All the routine atmospheric releases are driven by the nuclear ventilation systems which fulfils the functions of air inlet, extraction, filtration and evacuation through elevated stacks. The stacks parameters are given in Table 2.

TABLE 2. STACKS PARAMETERS OF MNRC NUCLEAR FACILITIES

Parameter	Reactor Facility	Radioisotope Production Facility	Waste Treatment Facility	
			Stack 1	Stack 2
Height (m)	28	18.3	13.6	11.3
Effective height (m)	35	27.1	19.7	17
Exit velocity (m/s)	11	8	8	2.55
Diameter (m)	1.2	0.95	0.55	1.53

4.2. MNRC Atmospheric source term

The reactor facility is a 2 MW TRIGA Mark II research reactor. The main radionuclide existing in the routine atmospheric releases are: Argon 41 (Ar-41) and Nitrogen 16 (N-16) produced during neutron interaction respectively with Argon-40 and Oxygen-16 dissolved in the reactor pool water [6]. The radioisotope production facility is processing radioisotopes for medical purposes and industrial applications such as: I-131, I-125, I-123, Ga-67, Tl-201, Mo-99, Au-198, In-111, Tc-99m and P-32 [4].The radioactive waste treatment facility is designed to take in charge the treatment of all solid and liquid radioactive wastes generated nationwide and containing radionuclides as: I-131, I-125, I-123, Mo-99, Au-198, Tl-201, Ga-67, P-32, Co-58, Co-60, Cr-51, Fe-55, Fe-59, In-111, La-140, Mn54, Mn-56, Tc-99m, C-14, Tritium, K-42, Na24, S-35 and Si-31 [4].

On the basis of the inventory of radioactive materials processed in the MNRC nuclear facilities, the total annual activity released to the atmosphere during routine operations is estimated to 3.83×10^{13} Bq with the spectrum given in Table 3.

TABLE 3. INVENTORY OF ANNUAL RADIOACTIVE ATMOSPHERIC ROUTINE RELEASES OF MNRC

Isotopes	Source Term of Annual Releases (Bq/an)
Noble Gases (Ar-41)	1.50×10^{13}
Iodine (I-131, I-125, I-129)	8.00×10^{8}
Tritium	5.00×10^{10}
Carbon 14	5.00×10^{8}
Gamma and Beta Emitters	2.5×10^{13}

5. ATMOSPHERIC DISPERSION AND DOSE CALCULATION MODELS

Atmospheric dispersion of radionuclides released into environment and dose assessment had been performed using two different calculation codes: GASCON developed by Commissariat à l'Energie Atomique of France [7] and CAP88 from Lawrence Livermore National Laboratory of USA [8].

The following assumptions have been considered and dealt with in dose calculations:

— atmospheric releases are supposed to be carried out at a constant rate;

- dose calculations have been made for radioactivity incorporation by inhalation and ingestion and for external exposure due to plume and deposits;
- dose conversion factors for incorporation are issued mainly from ICRP-67 [9] and BSS-115 [10] in compliance with ICRP-60 [11] methodology;
- dose conversions factors for external exposure are issued from [12];
- diet habits of population reference groups are representative of long term average behaviour in Maamora site.

In our case study, the relevant exposure pathways are:

- plume immersion which lead to external exposure and internal exposure by inhalation;
- soil deposit causing external exposure;
- transfer of radioactivity from atmosphere to man through food chain causing internal exposure by ingestion.

For each population reference group, Inputs data have been properly configured and exposure scenarios have been defined taking into account the different mechanisms and characteristics of radioactivity transfer from atmosphere to individuals. The calculations done by GASCON give for each reference group the following detailed results: radionuclide volume concentration in air, radionuclide surface concentration in soil, radionuclide concentration in agricultural products, annual committed effective dose for an adult, child and infant, annual effective dose distribution by exposure pathway and annual effective dose distribution by radionuclide.

The radiological impact of routine radioactive atmospheric releases was assessed over the different population reference groups. The radiological consequences given by GASCON code are summarised in Table 4.

CAP88 code have been used with the same input data to assess the radiological impact of radioactive routine releases of MNRC facilities. The comparison between the two code calculation results for an adult from the reference groups is illustrated in Table 5.

TABLE 4. RADIOLOGICAL CONSEQUENCES BY GASCON CODE

Ref. group	Committed Effective dose (mSv/y)			Dominant exposure pathway	Major radionuclide contribution
	Adult	Child	Infant		
1	1.03×10^{-3}	1.07×10^{-3}	2.2×10^{-3}	Plume external exposure (80%)	I-131 (61%), Ar-41 (37%)
2	6.80×10^{-4}	6.83×10^{-4}	7.12×10^{-4}	Plume external exposure (98%)	Ar-41 (92%), I-131 (7%)
3	1.38×10^{-4}	1.39×10^{-4}	1.55×10^{-4}	Plume external exposure (94%)	Ar-41 (84%), I-131 (15%)
4	9.98×10^{-5}	1.01×10^{-4}	1.18×10^{-4}	Plume external exposure (92%)	Ar-41 (77%), I-131 (22%)
5	5.49×10^{-5}	5.58×10^{-5}	7.14×10^{-5}	Plume external exposure (87%)	Ar-41 (67%), I-131 (31%)
6	5.82×10^{-5}	5.91×10^{-5}	7.49×10^{-5}	Plume external exposure (88%)	Ar-41 (68%), I-131 (30%)

TABLE 5. COMPARISON OF CAP88 AND GASCON DOSE CALCULATIONS

Reference group	Location Identification	Committed Effective Dose for an Adult (mSv/y)	
		GACSON Code	CAP88 Code
1	MNRC Fences	–	–
2	Forest Guard	6.80×10^{-4}	5.970×10^{-4}
3	Military Base	1.38×10^{-4}	3.80×10^{-4}
4	Douar Nsar Zdagh	9.98×10^{-5}	6.80×10^{-5}
5	Sidi Taibi	5.49×10^{-5}	2.2×10^{-5}
6	Ouled Taleb Machraa	5.82×10^{-5}	2.9×10^{-5}

6. CONCLUSIONS

The radiological impact assessment of routine radioactive releases of MNRC facilities has been carried out using two different calculation codes. Both of them have given comparable results in terms of individual committed effective dose for each population reference groups located around the site.

The highest dose value for an adult from the reference groups (except from the theoretical group) is estimated for an adult to 6.80×10^{-4} mSv per year which represents 0.07% of the 1 mSv annual regulatory limit for the public. Radiation exposure of this magnitude should put into context by comparing it with levels to which members of the public are exposed in their daily lives due to background sources.

Therefore, with respect of the source term and the associated release conditions, the radiological impact of routine radioactive atmospheric releases of MNRC is low and the overall risk to members of the public will be negligible.

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Environmental protection approaches in the uranium companies of Niger

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Abstract. In Niger, two companies, SOMAIR (Société des Mines de l'Air) and COMINAK (Compagnie Minière d'Akouta), produce Uranium and relevant radioactive and non-radioactive waste. The management of such waste has a big importance for both Government and leaders of these industries. Thus, policies and regulatory framework have been made to ensure human and environment protection against the effects of ionising radiation and industrial pollutants.

1. INTRODUCTION

Activities in uranium societies of Niger generate waste (liquid and solid wastes, radioactive and non-radioactive wastes), which must be managed to reduce negative impacts.

2. INSTITUTIONAL AND REGULATORY FRAMEWORK

2.1. Institutional framework

The two institutions in charge of control and follow of mining activities are:

— Ministry of Mine and Energy (MME);

— Centre National de Radioprotection (CNRP)

2.2. Regulatory framework

The management of radioactive waste is regulated by Niger mining law 003/MME/DM of 8th June 2001, relevant protection against danger of ionising radiation in mining sector. Thus, the section 6 concerns surveillance of ragiology in environment of research and exploitation or treatment of radioactive substances. The first chapter of this section is about the management of radioactive solid waste (article 40 gives definitions when the article 41 is about strategies of manage). The chapter 2 of this section concerns the following points:

— prevention;

— control;

— dosimeter (measurement).

3. WASTE MANAGEMENT AND RADIOACTIVE IMPACT ON ENVIRONMENT

3.1. Industrial pollutants

Two kinds of waste are produced in these industries: liquid and solid wastes. liquid wastes include wastewater and other liquid liquid effluents. COMINAK, for example, treats 3.4 millions m^3/year of wastewater and produces an annual volume of 2.18 millions m^3 of other liquid effluents [1].

The solid waste include barren overburden, low grade uranium ore and tailings. Table 1 gives COMINAK solid waste production in 1997.

TABLE 1. COMINAK SOLID PRODUCTION [1]

Waste nature	Waste quantity (year or grade)
Low grade uranium ore	423 561 tons (0.14%)
Heap-leach residues	401 894 tons (till 1990)
Mill solid waste	9 millions of tons

TABLE 2. SOMAIR AND COMINAK EVAPORATION BASINS AREAS

Companies	Evaporation basins areas (ha)
SOMAIR	10
COMINAK	65

The wastewater is decanted for reuse in the mills. COMINAK, which uses 16 basins that cover an area of 44 ha to a depth of 4 m , recycles 3.4 millions m3/year [2]. The others liquid effluents are stored in evaporation basins. Table 2 gives the evaporation basin areas [2]. The local desert climate produces appreciable evaporation rates.

Each basin is lined with an impervious PVC membrane. The basins are situated about 2 km from the mill, in a clayey zone which provides additional protection against any contamination of the underlying aquifer. the gradient is measured by piezometry.

3.2. Radioactive effects on environment

Tailings require safe management because they contain long lived uranium and its daughters, some of which, especially radium are toxic. Unless controlled, radium and its decay products may escape from the tailings and contribute to contamination and radiation exposure in the environment. The maximum values of radium emanation from the natural soils in SOMAIR and COMINAK are respectively 840 Bq/kg [3] and 6200 Bq/kg [4].

The emanation of radon and thoron together with their long lived daughter products is the basis for the problem posed by mill tailings. To find a method for reducing this emanation, a pilot project to cover mill tailings has been initiated. The project seeks to determine what materials can be used to reduce the radon emanation to acceptable levels.

The following interactive process will be applied:

— trial perimeter is defined;

— points of radon measurement are identified by co-ordinate;

— cover material is put in place to a know height;

— repeat of the first measurements at the same locations;

— comparisons are made with the first measurements to determine the degree of attenuation;

— the process is repeated until acceptable levels are overburden [1].

A 1500 m^2 test plot has been completed and some measurements have been made. The cover materials used for this test plot were SOMAIR and COMINAK barren overburden. [1]. The potential transmission vectors (water, food and air) have been controlled and the values measured are respectively 0.65 msv and 0.88 msv [3, 4] against 1 msv allowed by the Niger law.

4. CONCLUSIONS

Many efforts are made to ensure protection of human health and environment protection in Niger uranium mines. We can say that the aim of environment protection against the ionising radiation effects is achieved because th e maximum values of exposures are compliance with the standards the Niger law.

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Model used for assessing the environmental transfer of radionuclides from routine releases of IFIN-HH nuclear facilities

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Abstract. The paper presents the method developed in National Institute for Physics and Nuclear Engineering "Horia Hulubei" (IFIN-HH) to assess the activity limits of radioactive discharges for the existing practice. The dose assessment approach is based on simplified but conservative models for environmental processes that takes account of dilution and dispersion of discharges. Quantitative results based on available site-specific information have been obtained by means of a FORTRAN computer code, which implemented the model. The obtained activity levels, submitted to the Regulatory Authority are used as input to the review process for establishing the authorized discharge limits and optimizing protection against ionizing radiation in accordance with new demand on waste management.

1. INTRODUCTION

Radioactive discharges to environment, during normal operation of the IFIN-HH nuclear facilities (VVR-S research reactor, Waste Treatment Station and Radioisotope Production Station) have been done until now based on a previous standard, being controlled by a systematic effluents and environmental monitoring program. At the Regulatory Body requirement, a method of evaluating suitable discharge limits for the existing practice, in accordance with IAEA recommendations [1] adopted by Romania [2] has been necessary to be developed. The work has been performed in accordance with IAEA guidance [3] and is based on application at the specific case of methods and models recommended by IAEA [4, 5] and the Canadian standard [6]. To implement the requirements for discharge control [3] a reference level ($D_{lim}= 10^{-4}$ Sv y^{-1}) consistent with one tenth of the set dose constraint has been used.

2. ASSESSMENT APPROACH

The developed assessment approach consists in: (i) determination of the nature and magnitude of the radioactive discharges into the environment over the period of occurrence; (ii) modeling of the transport of materials discharged into the atmosphere and surface water and assessing the concentration of the radionuclides at critical group exposure locations; (iii) estimation of maximum annual doses arising from all exposure pathways during the period of practice; (iv) derivation of the maximum radioactive discharges to the environment based on the annual dose estimates. Significant exposure pathways given in Figure 1 by which discharged radionuclides can deliver public exposure are: (i) external exposure: (plume immersion; from ground deposition); (ii) internal exposure: (ingestion; inhalation). In order to estimate maximum annual dose received a discharge period of 30 years has been assumed, i.e. the dose is obtained for the 30th year of discharge and include contribution to dose from all material discharged in the previous 29 years. Combined generic input parameter (default values taken from [5-7]) and site specific data [9] have been used. The main assumptions applied are taken from [5] but have been adapted to the specific case.

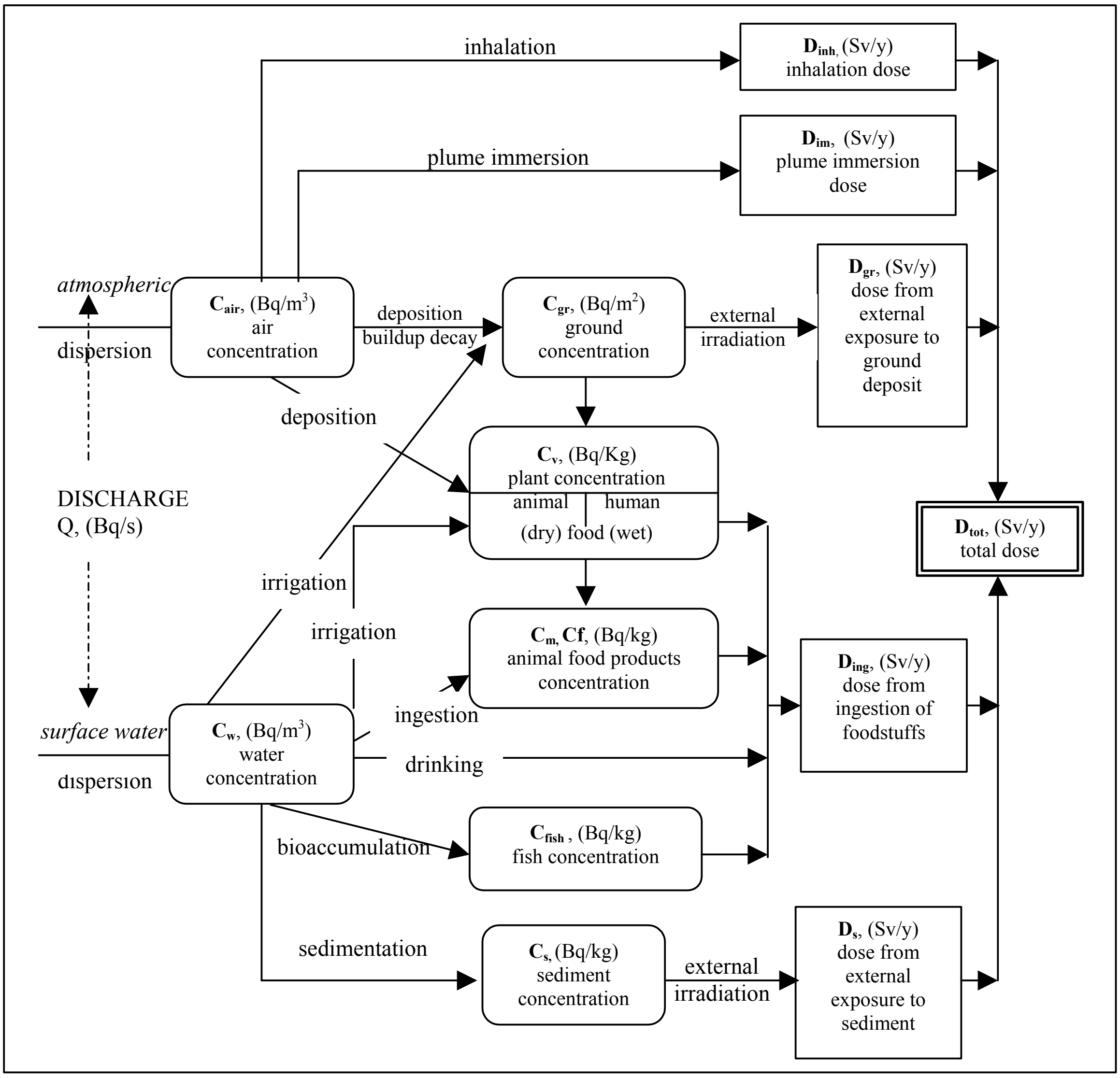

FIG. 1. Model used for assessing the environmental transfer of radionuclide.

3. CALCULATION PROCEDURE

3.1. Atmospheric discharges

Gaussian plume model [8] has been applied to assess the dispersion of long term atmospheric releases. This model has been considered appropriate for the site (flat terrain, 20Km short-range transport, approximate steady state meteorological conditions). Because no building wake effects have been included, same simplifications have been assumed: (i) a single wind direction for each air concentration calculation; (ii) a single long term average wind speed; (iii) a neutral atmospheric stability class (Pasquil stability class D). Ground deposition has been accounted by means of deposition coefficients [5].

3.2. Radionuclide transport in surface water

The model used to estimate the radionuclide transport in the river has been based on simplifying assumptions regarding the river geometry, flow conditions and dispersion processes in order to obtain an analytical solution [8] of advection-diffusion equation. Radionuclide concentrations in water have been estimated for the discharge point, considered the location where members of the critical group use this water for drinking, fishing, irrigation, swimming and the sediment for recreational activities. Calculations have been performed for the lowest annual “Ciorogarla” river flow rate 0.2 m^3 s^{-1} [9] recorded in the last 30 years. Sediment effects due to the beach deposition have been also estimated.

3.3. Terrestrial and aquatic food chains

The terrestrial food chain models accept input of radionuclides from both atmosphere and hydrosphere. The method used calculates the concentration of radionuclides in human food crops and animal produce, for milk and for meat [5]. The transfer of radionuclides from water to various trophic levels of aquatic life to fishes consumed by humans has been accounted by the bioaccumulation factor.

Necessary specific [9] and default [5] dosimetric data have been used to evaluate the effective dose based on calculated radionuclide concentrations in various environmental components. Summing the doses from all pathways of each particular transferred radionuclide has done estimation of total individual doses. A specific approach based on complete equilibrium between the environment and the exposed individuals has been adopted for assessment of transferred ^3H through its association with water molecules and discharges of ^{14}C associated with CO_2 molecules further fixed within the plants during photosynthesis.

4. DERIVATION OF THE MAXIMUM DISCHARGE LIMITS

Derived activity limit for each radionuclide i in case of both atmosphere and liquid discharges has been calculated using:

$$DL_i = \frac{D_{\lim}}{D_{i0}} \tag{1}$$

where:

DL_i is the discharge activity limit (Bq y^{-1}); ii) D_{lim} = the reference level criterion (Sv y^{-1});
D_{i0} is the total dose rate for unit discharges (Sv y^{-1} per Bq y^{-1})

5. RESULTS AND DISCUSSIONS

Annual average dose to a member of the critical group, corresponding to a unit discharge (1 Bq s^{-1}) over a period of 30 years are presented in Figures 2 and 3. Calculations have been performed both for adults and infants, but only the most conservative values are presented in the paper. The obtained values have been estimated based on the established assessment approach implemented in a FORTRAN computer code. The conceived code has been verified and tested by means of specific examples supplied in the literature [4, 5]. Code results has been verified by comparison with similar results obtained in Subsidiary for Nuclear Research (SCN) Pitesti, accounting for the differences in input specific parameters. Analysis of the obtained results leads to the following comments:

For atmospheric discharges:

— Ingestion of the foodstuffs is the major human exposure pathway (i.e. 98% for ^{95}Nb and ^{95}Zr for food crops and respectively 99% ^{22}Na and 98% ^{133}I for animal produces);

— Contribution of the inhalation pathway at the total doses is less then 25%;

— Except same iodine radioisotopes for which the external exposure is dominant, for the majority of the selected radionuclides the contribution of this pathway is null;

— The derived discharge activity limits belong to the range (10^8-10^{14}) Bq y^{-1}.

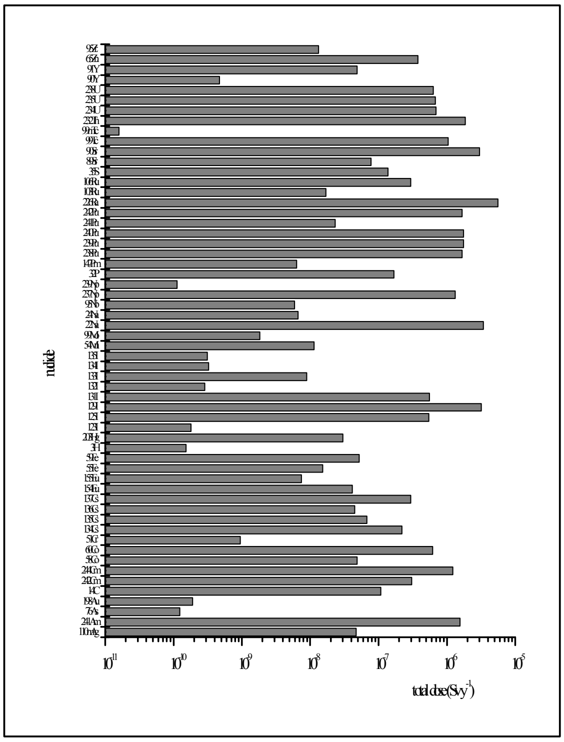

FIG. 2. Total dose per unit discharges for atmospheric release (infants).

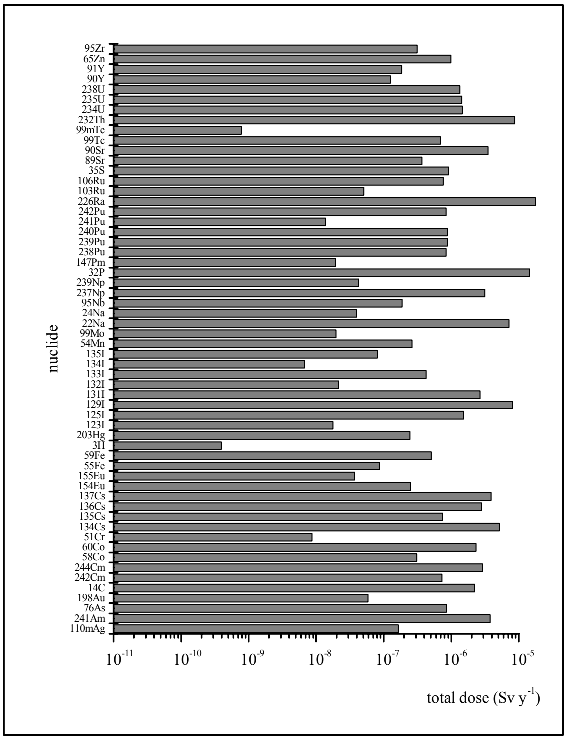

FIG. 3. Total dose per unit discharges for liquid release (infants).

For discharge into the river:

- The main contribution to the total doses is the fish ingestion (98% ^{32}P, 96% ^{136}Cs);
- For drinking contaminated water the relevant radionuclide is ^{99}Mo (53% contribution), followed by ^{90}Y (40%);
- Foodstuff ingestion has for the majority of the radionuclides a contribution less then 25%,
- except ^{22}Na and ^{90}Sr;
- External exposure from river beach sediments has a minor contribution to the total dose;
- The derived discharging activity limits belong to the range (10^8-10^{12}) Bq y^{-1}

6. CONCLUSIONS

A methodology approach (method and tools) has been achieved for dose rate assessment and further deriving of discharge activity limits for radioactive material in liquid effluents and airborne for normal operation of IFIN-HH nuclear facilities. To this end:

- A model used for assessing the environmental transfer of radionuclides has been developed;
- Conservative assumptions, input data, transfer coefficients and dose conversion factors have been selected to define a simple and conservative variant of the established calculation model;
- A FORTRAN computer code has been created based on the logical scheme of the model and testing of the code has been done by verifying the results for the examples supplied in the literature;
- Average total doses and contributions of each accounted exposure pathways per unit discharge for both adults and infants have been calculated;

- Dominant radionuclides and exposure pathways have been identified and evaluated;
- Corresponding activity limits of radioactive discharges have been derived;
- Confidence has been built by comparison with results obtained in SCN Pitesti and other data supplied by the literature.

ACKNOWLEDGEMENTS

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Doses due to naturally occurring radionuclides to wildlife terrestrial and freshwater species in South Africa

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Abstract. Contamination of soil and fresh water systems by naturally occurring radionuclides in South Africa have existed in mining and mineral processing areas for more than a hundred years. The gold deposits of the Witwatersrand Basin gave rise to huge quantities of historical residues, many of them exhibiting elevated concentration of natural radionuclides. This contamination has chronically exposed wildlife species and this paper attempts to present some results, based on available database of contaminated areas, regarding doses received by selected terrestrial and freshwater species.

The results obtained indicate that doses received by the majority of species considered in the paper are below the presently recommended guideline levels, however, doses to some species (mammalian herbivores, benthic molluscs) are above these levels and could be of concern. It should also be taken into consideration that the obtained results are likely underestimated due to limited data available.

1. INTRODUCTION

Mining and processing of minerals in South Africa has generated large volumes of mineral residues or waste, many of them exhibiting elevated concentration of natural radionuclides. The gold deposits of the Witwatersrand Basin have been exploited for more than a hundred years, giving rise to huge quantities of historical residues. Since 1952, some of the uranium in the Witwatersrand ores has been extracted as a by-product of the gold recovery process, although uranium production since 1980 has declined dramatically. This decline has resulted in the decommissioning of most of the uranium plants and associated sulphuric acid plants.

Other, more recently established mining and minerals processing activities, involving elevated levels of natural radioactivity, include the exploitation of the phosphate, copper and zirconium deposits and of coastal mineral sands.

A few mines were established in 1950 in Cape Province to exploit an ore-body of monazite containing up to 8% thorium oxide. The mine was abandoned in 1965, and attempts to restart operations have so far been unsuccessful. Large quantities of broken rocks and tailings have remained on the site resulting in spreading of radioactivity from the tailings into local watercourses and soil over the years.

Contamination of soil and fresh water systems by naturally occurring radionuclides have existed in mining and mineral processing areas for more than a hundred years. This contamination has chronically exposed wildlife species and this paper attempts to present some results, based on available database of contaminated areas, regarding doses received by selected terrestrial species.

Protection of the environment and wildlife species in South Africa relies on ICRP recommendation (issued in 1977 and modified in 1990), which effectively maintains that standards established for the protection of humans are sufficient to protect the environment. Despite that the South African Regulatory Authority is of the opinion that our knowledge on radiation impacts on the environment is insufficient to permit the introduction of the relevant regulatory requirements, it has also recognised the need to protect the environment on its own right.

2. SOURCES OF CONTAMINATION

Tailings account for most of the residues from the mining industry. The quantities of gold mine tailings, dating back from 1893, have been estimated to about 5 billion tuns. Most of the mine tailings, containing elevated concentrations of natural radionuclides, originate from the gold mining industry. The elevated radioactivity concentrations arise from elements in the uranium decay chain, and vary over the same range as those in the ore-body. Where uranium has been extracted as a by-product of the gold recovery process, the uranium content has become depleted in relation to the other decay chain elements, resulting thus in a moderate reduction in total activity and significant disturbance of the equilibrium between the radionuclides of the uranium series. Sampling in several tailings has revealed that generally ^{234}U and ^{230}Th are in equilibrium with ^{238}U, while ^{226}Ra is out of equilibrium.

Unless viable options for reuse of tailings can be developed in future, all mine tailings will eventually have to be finally disposed of, regardless of whether they are reprocessed in the meantime. The only feasible disposal route for the tailings would be their stabilisation in situ.

Waste rock accounts for the next largest category of mining residues, and again originates mostly from the gold mining industry. It contains the full uranium decay chain, but at lower activity concentrations than in tailings. The activity concentrations of uranium and its decay products in waste rocks are largely in secular equilibrium.

Mineral sands operations produce large quantities of tailings which may vary in activity concentration from background levels up to 2 orders of magnitude higher. Thorium is the main contributor to the elevated activity concentration.

Contaminated soil arises from the decommissioning of uranium and acid plants. Soil contamination has also occurred through the processing of contaminated mine scrap over periods of many years prior to the imposition of regulatory restrictions in 1993. Soil contamination has originated from:

— Ore dust (uranium series in secular equilibrium at low activity concentrations);
— Chemically leached uranium, e.g. scale from water pipes (uranium at moderate activity concentrations);
— Chemically separated uranium, i.e. yellow cake (uranium at high activity concentrations); and
— Radium sulphate, e.g. scales from acid plants (radium at high activity concentrations).
— Activity concentrations range from 10 to 5 000 times that of the ore-body.

The nature of the various ore processing operations is such that there is always the possibility of natural radionuclides in the ore becoming concentrated at certain points in the process, as constituents of unwanted **scales and residues**. In gold plants, the scope for radioactivity build-up is relatively small because of limited chemical driving forces, and any scales or residues would tend to be reprocessed to recover gold. The scope for radioactive scale formation is greater in uranium extraction plants and associated facilities. The main issue in this regard is the formation of radium sulphate in sulphuric acid and pyrite plants.

Activity concentrations range from 10 to 20 000 times that of the ore-body.

Phosphate rock is the starting material for the production of all phosphate products and is the main source of phosphorus for fertilisers. The principal residue from phosphate processing operations is phosphogypsum ($CaSO_4$), which is generated in large quantities. Phosphate rock exploited in South Africa, as in most other producing countries, contains elevated concentrations of uranium together with its decay products in secular equilibrium. The activity concentrations in the phosphogypsum appear to be elevated to levels approaching those in the raw material.

3. CALCULATION OF DOSES TO TERRESTRIAL SPECIES IN CONTAMINATED AREAS

Calculation methodology related to the assessment of doses to environmental species was based on the "Impact assessment of ionising radiation on wildlife" [1].

The proposed dosimetry model represented organism as ellipsoid and internal dose was calculated under the assumption that radionuclides were uniformly distributed throughout the organism. The other assumptions made in ref. [1] were also adopted, i.e.:

— Concentrations of radionuclides in biota were assumed to be in equilibrium with soil or water in which the organisms lived;

— Doses per unit concentration represented an average throughout the volume of the organism;

— Doses to micro-organisms were equal to the absorbed doses in the soil in which they were located;

— Absorbed fractions for α emissions were assumed to be zero for bacteria and unity for all other organisms.

Calculation of external doses took into account the statistical distribution of fractional occupancy (in the underground, in soil or water) of key organisms considered (Figure 1). Calculation of internal doses to organism was based on concentration factors specific to each radionuclide and organism and dose per unit concentration of internally incorporated radionuclides. Both the concentration factor and dose per unit concentration for various species and radionuclides were taken from the reference [1].

Only sparse data were unfortunately available for testing validity of the concentration factors. Mean values of these factors were compiled from ref. [1] and a statistical distribution was based on a few data available for South African ecosystems. External doses were evaluated using statistical distribution of radionuclide concentrations in soil and water, relevant to the selected contaminated sites in South Africa (see Figure 1 as an example).

4. RESULTS OF CALCULATIONS

Crystal Ball program implementing Monte Carlo method in Excel spread-sheet was used for the assessment of dose rates for a representative range of biota within terrestrial and freshwater ecosystems. Bacteria, lichen, tree, shrub, herb, seed, fungus, herbivorous mammal, rodent and bird were taken as representatives for the South African terrestrial ecosystem. Bacteria, phytoplankton, zooplankton, macrophyte, benthic mollusc, small and large benthic crustacea, benthic and pelagic fish, amphibian, small aquatic mammal and duck represented freshwater ecosystem.

Typical examples of statistical distribution of weighted dose rates (μGy/h) imparted to birds is shown in Figure 2.

The weighted dose rates, obtained for the selected representatives of terrestrial and freshwater ecosystems, are summarised in Figures 3 and 4, respectively. Contribution of radionuclides considered to the weighted doses imparted to terrestrial species illustrates Figure 5. It is apparent from this figure that ^{226}Ra contributes predominantly to the doses to terrestrial species.

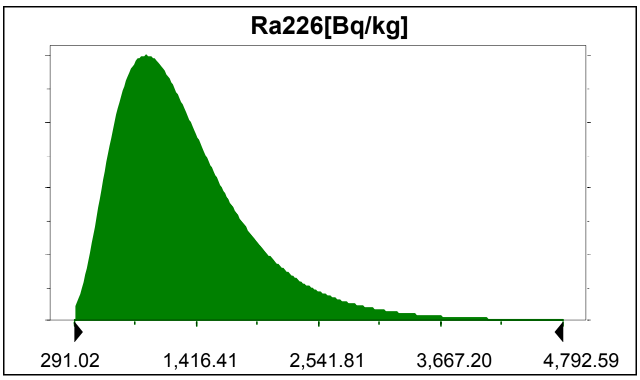

FIG. 1. Statistical distribution of ^{226}Ra in soil.

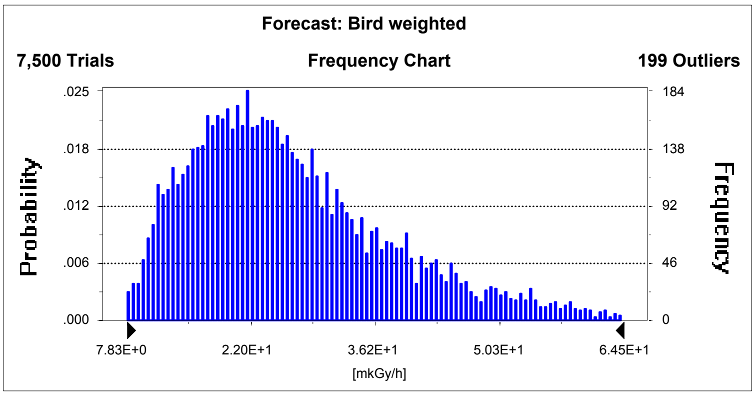

FIG. 2. Distribution of weighted dose to birds.

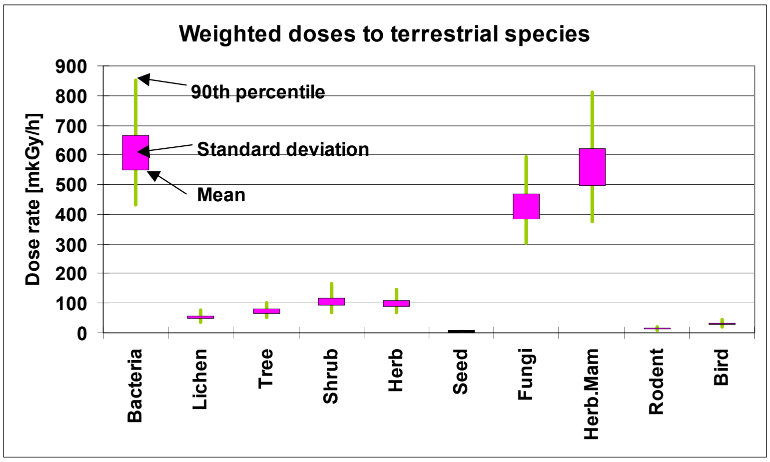

FIG. 3. The weighted dose rates, obtained for the selected representatives of terrestrial species in South Africa.

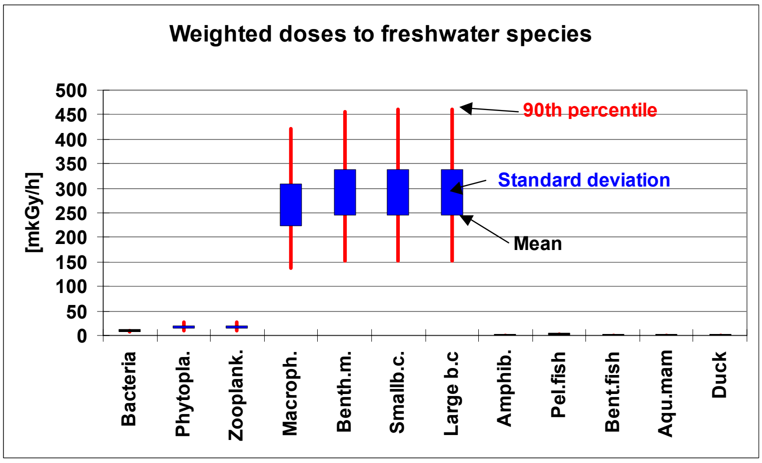

FIG. 4. The weighted dose rates, obtained for the selected representatives of freshwater species in South Africa.

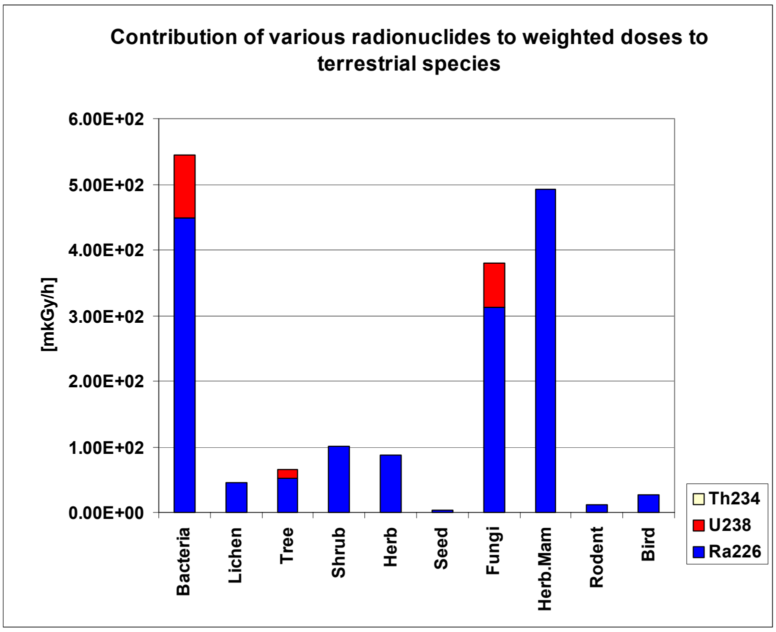

FIG. 5. Contribution of various radionuclides to weighted doses to terrestrial species.

5. CONCLUSIONS AND RECOMMENDATIONS

Guideline dose limits for biota, below which significant effects to the environment are unlikely, have been recommended in several countries (USA, Canada, etc.) and by IAEA. The dose limits for biota recommended by the IAEA (Table 1) have generally been well received.

Although the results obtained indicate that doses received by majority of species considered here are below the guideline given in Table 1, doses to some species are close to the recommended limits and doses to some other species (mammalian herbivores, benthic molluscs) are above the recommended limits. It should also be taken into consideration that the obtained results for biota exposed by elevated natural radionuclides are likely underestimated due to limited data available. Soil and water concentrations were not available for some other radionuclides of the uranium and thorium series as well as were not available the relevant concentration factors, doses per unit concentration, etc. It could therefore be concluded that areas in South Africa, contaminated by naturally occurring radionuclides due to past or present mining and mineral processing industry may pose a radiation hazard to some species in the relevant ecosystems.

More research is therefore required for developing a more comprehensive database with regard to radionuclide concentrations in the environment as well as for transfer and uptake of these radionuclides by various species in the terrestrial, freshwater and coastal water ecosystems. More information is also required for assessing the impact of ionising radiation on specific species. This should namely include rare and endangered species inhabiting contaminated areas, as well as long-live marine species.

Since ionising radiation is far not the only agent affecting the environment, more information is required on the impact of non-radioactive pollutants on biota and on the impact of ionising radiation in conjunction with conventional pollutants.

TABLE 1: RECOMMENDED DOSE LIMITS (μGy/h) TO BIOTA

Species	IAEA (1988, 1992) [2, 3]	Thompson (1999) [4]	DOE (USA) [5]
Terrestrial:			
Plants	400		400
Animals	40		40
Mammals		10	
Birds		50	
Amphibians/Reptiles		10	
Aquatic:			
Freshwater organisms	400		400
Benthic invertebrates			
Fish			
Deep ocean organisms	1000		

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Discharge policies for low level radioactive effluents in Turkey

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Abstract. The legal infrastructure established in Turkey covers the ways to manage low-level radioactive wastes including liquid, solid and gaseous wastes. The paper gives detailed information about the discharge policies and radioactive waste tank systems and dose assessment. Liquid Radioactive Waste tank systems, as best available technique, are used to collect and store the low level radioactive wastes and as a part of low-level radioactive effluent discharge policy, these decay and delay systems are effectively used for biomedical radioactive wastes generated in hospitals. The decayed waste is then discharged to sewage system in accordance with the discharge limit given in the legislation. Dose assessment studies were also completed and the annual effective dose that would be received by the workers of the Waste Treatment Facility was calculated and the results are presented.

1. INTRODUCTION

Radioactive substances are used in beneficial ways such as the generation of electricity, medical diagnosis and therapy, scientific research and specialized industrial applications. However, many of these activities generate radioactive waste, which occur either in gas, liquid or solid state, should be under an appropriate control program [1]. Airborne and liquid waste may be permitted for discharge into the environment. Unplanned and/or uncontrolled exposure to radiation can be detrimental to health that is why the regulatory system in any country should be sufficiently robust [2]. An essential requirement of any sound regulatory structure is to present a clear definition of its scope: certain sources or practices may be excluded from regulatory requirements or exempted from regulatory supervision. One reason for such exemption or clearance is when the radiological risk or detriment associated with the practice is so small as not to warrant the imposition of the system of reporting or prior authorization [3]. For the exemption of any source or practice from regulatory control, the general and widely accepted radiation safety requirements for a member of the public are as follows:

— The effective dose expected to be incurred by any member of the public due to the exempted practice or source is of the order of 10 μSv or less in a year;

— Either the collective effective dose committed by one year of performance of the practice is no more than about 1 man Sv, or an assessment for the optimization of protection shows that exemption is the optimum option [4].

The current global approach is toward decrease of radioactive discharges to the environment. The Joint Convention on the Safety of Spent Fuel Management and on the Safety of Radioactive Waste Management [5] assigns a system of regular peer reviews of the policies and practices of radioactive waste management including discharges to the environment in each Contracting Party. Moreover, the OSPAR Convention [6] states that Contracting Parties shall require adopting programmes and measures for the purpose of prevention and elimination of pollution from land-based sources, either individually or jointly, the use of:

— best available techniques for point sources;

— best environmental practice for point and diffuse sources, including, where appropriate, clean technology.

The limitation of the discharges of radioactive substances should be based upon the optimization of radiation protection, using best available technique.

2. LEGAL STRUCTURE

The Turkish Atomic Energy Authority has issued regulations and legislation on radiation protection including waste management; moreover legislation has been issued for the discharges of radioactive effluents from the licensed establishments. This legislation gives all the details for the disposal of short lived (up to 100 days half-life) low-level radioactive waste.

(a) **Solid radioactive waste:** Part of the legislation for the management of short-lived solid radioactive wastes with half-lives less than 100 days implies the disposal of these wastes as hazardous medical waste that is incinerated in the municipality authorized incineration facilities, after the decay of radionuclides with the activities that are statistically indistinguishable from the background radiation. This approach enhances a practical approach for the low level solid radioactive wastes. Sealed radioactive sources can not be disposed as the same way with the short lived low level solid radioactive waste, according to this legislation.

(b) **Liquid radioactive waste:** The liquid radioactive wastes generated can be discharged to the sewage system according to the limits set by the legislation, and there is an ongoing study to amend the discharge limit as 10 ALI_{min}/month for each establishment. Short-lived solid radionuclides with half-lives less than 100 days can be discharged to the sewage system regarding to the current legislation and also for the amended version.

(c) **Gaseous radioactive effluents:** The current studies for the amendment of the legislation covers also the release of gaseous effluents to the environment with the constraint of not exceeding the effective dose of 10 μSv that could be incurred by any member of the public due to the gaseous release during one year. Short-lived gaseous radionuclides with half-lives less than 100 days can be released to the atmosphere regarding to the current legislation and also for the amended version.

3. DELAY SYSTEMS

As the best available technique, waste tanks are used to collect and decay radioactive waste before the discharge of effluents into the sewage system. The single waste tank system makes use of the “decaying while filling” principle. Therefore, by the time that the tank is filled, the total activity in the waste tank is many times lower than the total input activity. The cascade tank system takes the advantage of physical decay without input so that the overall capacity requirement can be greatly reduced. The use of multiple waste tanks resolves most of the problem of single waste tank system. However, it is important to design waste tank system with optimum tank number and capacity. Detailed studies were done to end up with a procedure that gives optimum tank number and capacity. Design of waste tank system with different tank numbers and capacities is possible, however it is important to design a system with optimum tank number and capacity. In the past study [8], design parameters for waste tank system was studied to end up with a compact procedure that also includes economical valuation study and gives the optimum holding tank system arrangement. This optimization procedure has been used in the design of ten tank systems.

4. DOSE ASSESSMENT

Dose assessments due to the low level discharges to the sewage for the worker of the domestic waste facility was completed according to the results given in the IAEA-TECDOC-1000 [9], the individual dose assessment for the Domestic Waste Facility worker was done for the cities with populations of 10, 50, 200, 500 and 1500 thousands and the results are tabulated in Table 1.

TABLE 1. ANNUAL INDIVIDUAL DOSE TO THE DOMESTIC WASTE FACILITY WORKER

			Number of Establishment using ^{131}I			
			1	3	5	7
Activity Discharge - GBq [150 ALI $_{min}$ /(year x establishment)]			*0.12*	*0.36*	*0.60*	*0.84*
Dose Assessment Annual Dose to the Worker of Waste Facility ($\mu Sv\ a^{-1}$)	Population of the city	10 000	*45.1*	*135.4*	*225.7*	*316.0*
		50 000	*9.0*	*27.1*	*45.1*	*63.2*
		200 000	*2.3*	*6.8*	*11.3*	*15.8*
		500 000	*0.9*	*2.7*	*4.5*	*6.3*
		1 500 000	*0.3*	*0.9*	*1.5*	*2.1*

Figure 1 give the plot of Table 1 as dose vs. number of establishments.

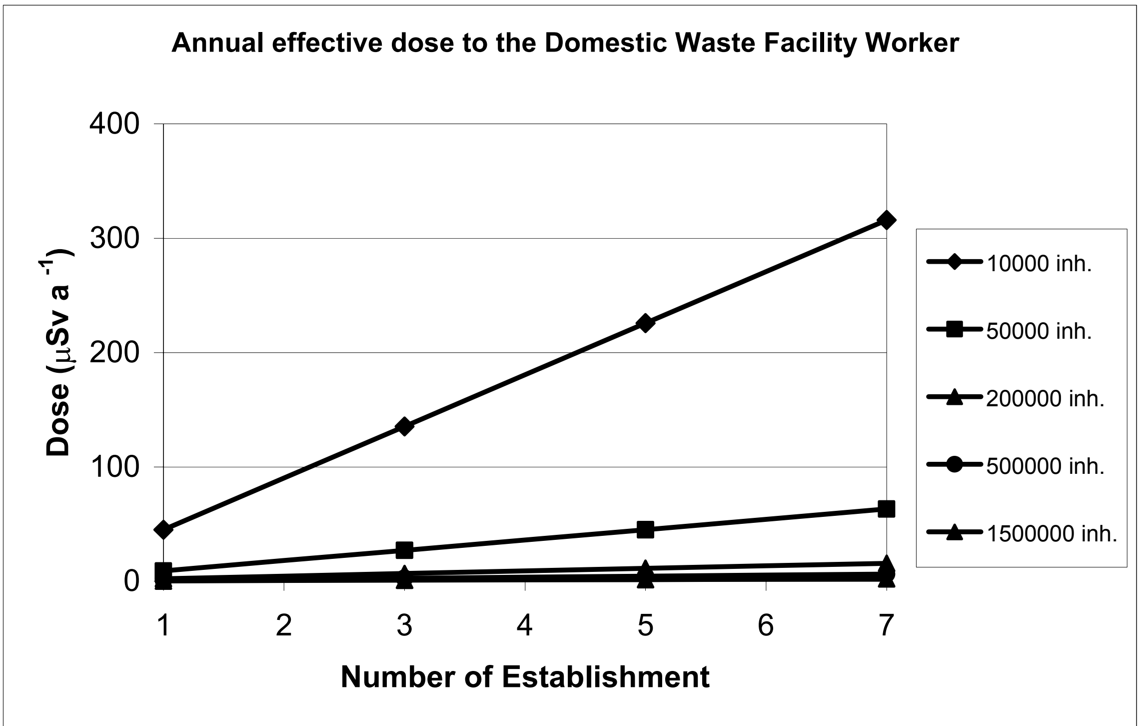

FIG. 1. Annual dose that would be received by the domestic waste facility worker.

In addition, it is worth to state that the use of large volume tanks reduces the radiation exposures around the system and it is also advantageous in the radiation protection point of view.

5. DOSE ASSESSMENT RESULTS

It can easily be concluded from Figure 1 that there is a inverse proportion between the dose to the waste facility worker and population of the city. It can also be concluded that annual dose that is received by the domestic waste processing facility is not high. Moreover decay and delay systems are easy to apply and very efficient to decrease the discharge activity of any establishment. As an important conclusion periodic monitoring of discharges should be properly done.

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Assessment of radionuclides in sediments proposed for dredging

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Abstract. Two screening models were used to assess the potential risk of radionuclides present in sediment proposed for dredging. The proposed dredging site is located in Bayou Rigaud, Louisiana, USA. The receptors of concern for the project include aquatic ecological receptors, and members of the public and the crew of the ship. Two screening models used in the assessment were the *RAD-BCG Calculator,* an assessment screening model developed by United States Department of Energy and the *Guidance on Radiological Assessment Procedures for the Protection of Human Health to Determine if Materials for Disposal at Sea are within the Scope of the London Convention 1972* currently being developed by the International Atomic Energy Agency. Both models were effectively used to assess the potential impact of radium-226, radium-228, and caesium-137 on aquatic biota and humans.

1. INTRODUCTION

Each year over 400 million cubic meters of sediments are dredged from waters of the United States to maintain navigational channels. Part of the process of removing the sediments is to ensure that the action and subsequent management options do not adversely impact the environment. Most environmental evaluations focus on non-radiological analysis such as measuring and assessing the risk of metals, polychlorinated biphenyls, polyaromatic hydrocarbons, and pesticides. However, there are projects that require a detailed analysis focusing on the risk of radionuclides present in the sediments proposed for dredging. Until recently there have been very few tools to assist in assessing the risk of radionuclides to ecological receptors and humans.

The scientific and regulatory communities have developed screening level approaches to address the lack of scientifically defensible tools for assessing the risk of radionuclides present in sediments. These approaches include the RAD-BCG Calculator developed by United States Department of Energy and the Guidance on Radiological Assessment Procedures for the Protection of Human Health to Determine if Materials for Disposal at Sea are within the Scope of the London Convention 1972 currently being developed by the International Atomic Energy Agency.

To assess the utility of these two screening models for evaluating the risk of radionuclides in sediments to ecological receptors and humans, they were field tested through a dredging project at Bayou Rigaud, Louisiana, USA. The sediments proposed for dredging were suspected to contain elevated levels of radioactivity as a result of the release of produced waters during petroleum drilling and production. Produced waters are a by product from oil production as a result of pumping an oil/water mixture from the ground. Produced waters containing radionuclides were released into the area of Bayou Rigaud for several years until the early 1990's. Around 382,280 cubic meters of sediment from Bayou Rigaud are proposed for dredging. Current research indicates that the mixture of barium and radium commonly found in produced waters precipitates to the sediment when it reaches sulfate-rich marine water. As a result, radium (Ra-226, Ra-228) may reach concentrations in Bayou Rigaud sediment that can pose adverse risk to human and ecological receptors during the process of dredging and placement of dredged material.

2. METHODS

2.1. Sample collection and analysis

Ten samples each of water, sediment and mussel tissue samples, from the Bayou Rigaud, were collected by the Army Corps of Engineers personnel and sent to Severn Trent Laboratories (STL) in Richland, WA, USA for radiological analysis. Nine of the ten samples of each type were collected from the Bayou Rigaud and one was collected from a relatively uncontaminated reference location. Radioisotopes of specific concern were caesium-137, radium-226 and radium-228. STL performed the analyses in substantial conformity with established test methods. Sufficient sample masses or volumes were provided, so alterations of the contractual minimum detectable concentration were unnecessary. Sample analysis results were reported electronically to Pacific Northwest National Laboratory staff, followed by hard copy transmittal of the data.

2.2. RAD-BCG Calculator

The RAD-BCG Calculator is an Excel spreadsheet based, Visual Basic® driven, computer program that uses a multi-tiered approach to determining compliance with radiological dose limits to aquatic biota set forth in DOE Order 5400.5. The radiological dose rates below which deleterious effects on populations of aquatic and terrestrial organisms have not been observed have been discussed by the International Atomic Energy Agency and the National Council on Radiation Protection and Measurements. Those dose rate limits are:

— Aquatic Animals – 10 mGy/d;

— Terrestrial Plants – 10 mGy/d;

— Terrestrial Animals – 1 mGy/d.

The data were perused, imported into Excel® spreadsheets and maximum radionuclide concentrations were determined for water and sediment samples. These maximum concentrations were entered into the RAD-BCG Calculator, and compared to initial Biota Concentration Guides (BCGs). When the sum of the ratios between maximum concentrations and associated BCG is below unity, no adverse effects on plant or wildlife populations are expected and compliance with current biota dose standards is determined to exist. In cases where radionuclide concentrations are measured in one environmental media, but not the other, a conservative default distribution coefficient is used to determine that radionuclide's concentration in the media where it was not detected, following the model's assumption of equilibrium.

2.3. IAEA *de minimus* screen

The Guidance on Radiological Assessment Procedures for the Protection of Human Health to Determine if Materials for Disposal at Sea are within the Scope of the London Convention 1972 developed by the IAEA is a screening approach for assessing radiation doses arising from sea disposal which incorporates two scenarios for human exposure. The two main receptors, disposal workers and non-workers, include the main exposure radionuclide pathways including exposure in vicinity of material (dredge and beach), contact, inhalation, waterborne, and consumption of seafood. Equations to calculate exposure include the several conservative default assumptions and it covers a comprehensive list of radionuclides included in the screen.

Exposure pathways for the screening model were different for the public and for crew on the dredge ship. Pathways for the members of the crew include external irradiation, inhalation of dust on the ship, and ingestion of sediment. Pathways for members of the public include external exposure to sediments on the beach, ingestion of seafood, inhalation of seaspray, and inhalation of dust. Using these exposure scenarios, doses to individuals and collective doses for both the crew and public were calculated. The radiological dose rates below which deleterious effects on members of the ship crew and public are:

— Individual doses – 10 μSv/yr ;

— Collective doses – 1 manSv/yr.

3. RESULTS

3.1. Radionuclide analysis

Radionuclides detected in samples submitted for analyses included potassium-40, caesium-137, radium-226 and radium-228. Potassium-40 (K-40), detected in some water and sediment samples, is a primordial radionuclide, about 0.01% abundant and is a constituent in living tissues. The RAD-BCG Calculator does not include this radionuclide in its list of isotopes for which dose conversion factors have been developed, because it is naturally occurring.

Caesium-137, an anthropogenic radionuclide produced in the fission process and present in world wide fallout, was detected in only two sediment samples collected from the Bayou Rigaud (BR), and was not detected in any reference sample. Caesium was not detected in any water or biota tissue samples.

Radium-226, a naturally occurring radionuclide from the uranium-238 decay series, was detected in seven of the nine sediment samples and in all nine water samples from the Bayou Rigaud. Likewise, it was detected in both the water and the sediment samples from the reference location. The reference value of radium-226 in sediments was between two and three standard deviations below the mean of the Bayou Rigaud data. The reference location's value for water approximately equaled the average value for the Bayou Rigaud data. Radium-226 was not detected in any biological samples submitted for analysis.

Radium-228, a naturally occurring radionuclide from the thorium-232 decay series, was detected in five of the nine biota samples from the Bayou Rigaud but not in the biota sample from the reference location. Six sediment samples contained this radionuclide, as did seven water samples. The sediment sample from the reference location contained a measurable amount of radium-228 which was between two and three standard deviations below the mean value of the Bayou Rigaud data. This radionuclide was not detected in the reference water sample.

3.2. Application of the RAD-BCG calculator and graded approach

The RAD-BCG Calculator and the Graded Approach were used to screen radionuclide concentrations in Bayou Rigaud employing maximum detected radionuclide concentrations listed in Table 1.

Table 2 contains the results of the initial screening calculations which indicate that Bayou Rigaud passes the screen analysis with a total sum of fractions of 0.347. Because caesium-137 was not detected in any of the water samples analyzed, the distribution coefficient of 500 L/kg was used to determine a conservative water concentration and the code also calculated internal and external doses from the estimated amount of caesium-137 in the water.

The 'Source of Calculation' column in Table 2 indicates three things. First is an abbreviation for the organism most likely to reach its limiting dose and here "RA" indicates that a riparian animal is most likely to reach the limit of 1 mGy/d. Next, the 'Lumped' statement indicates that simplified bio-concentration factors were used to determine the radionuclide concentration within the hypothetical organism. Finally, the 'Default' entry indicates whether or not any user-defined parameters were used instead of the RAD-BCG Calculator default parameters.

TABLE 1. MAXIMUM RADIONUCLIDE CONCENTRATIONS IN BR WATER, SEDIMENT AND CLAM SAMPLES

Radionuclide		Concentration (Bq/m^3 or Bq/kg)	Total Analytical Error	Minimum Detection Level
Water	Ra-226	5.7	1.3	0.66
	Ra-228	31.8	12.2	14.5
Sediment	Cs-137	4.1	2.5	3.8
	Ra-226	117.7	17.4	5.0
	Ra-228	59.2	13.7	8.6
Biota	Ra-228	17.5	6.7	7.8

TABLE 2.

Nuclide	Water Partial Fraction	Bq/m^3 Source of Calculation	Sediment Partial Fraction	Bq/kg Source of Calculation
Cs-137	5.21E-03	RA-Lumped, Default	3.55E-05	RA-Lumped, Default
Ra-226	3.78E-02	RA-Lumped, Default	3.15E-02	RA-Lumped, Default
Ra-228	2.54E-01	RA-Lumped, Default	1.83E-02	RA-Lumped, Default
Partial fractions	2.97E-01		4.99E-02	
Total sum of fractions (water and sediment):				3.47E-01
Result:	You have passed the site screen			

TABLE 3.

	Calculated Dose	Criteria Dose	Pass/Fail
Individual dose – Crew	7.09 uSv/yr	10 uSv/yr	Pass
Individual dose – Public	8.60 uSv/yr	10 uSv/yr	Pass
Collective dose – Crew	< 0.01 manSv/yr	1 manSv/yr	Pass
Collective dose – Public	0.04 manSv/yr	1 manSv/yr	Pass

3.3. Application of IAEA *de minimus* screen

The IAEA *de minimus* screen was used to assess the potential risk that the sediments proposed for dredging at Bayou Rigaud may pose to members of the dredging ship crew and public. Data from Table 1 were used as an input for this analysis. Results of the analysis are show in Table 3.

4. CONCLUSIONS

The two screening methods were effective at evaluating the potential impact of radionuclides at the proposed Bayou Rigaud dredging site. While these methodologies are relatively new or in draft form, the approaches used have been thoroughly reviewed and are technically valid.

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2. APPROACHES ADOPTED FOR NON-RADIOACTIVE POLLUTANTS – IMPLICATIONS FOR RADIATION PROTECTION

Environmental damage valuation as radiation protection tool

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Abstract. Environmental radiation protection procedures do not have global consensus. In researching mechanisms to guide environmental radiation protection procedures consensus searching, the approaches used by non-radioactive environmental protection are very promising. Among the approaches, environmental valuation procedures are commonly employed, and are very proper for environmental radiation protection.

1. INTRODUCTION

The environmental protection from ionizing radiation is based in a paradigm, proposed by International Commission on Radiological Protection - ICRP [1] “The commission therefore believes that if man is adequately protected then other living things are also likely to be sufficiently protected”. This paradigm was put into doubt by ICRP itself [2] and by many other authors (Amiro [3], Pentreath [4], Pentreath and Woodhead [5]). “Occasionally, individual members of non-human species might be harmed, but not to the extent of endangering hole species or creating imbalance between species” [2].

Actually, great progress are occurring in the ethic concepts of environmental radiation protection [6, 7]. Three ethic components have been identified: anthropocentric, bio-centric and eco-centric component [7]. The dose calculation methodology has shown significant improvement, also [3, 8, 9]. Beside this, we are very far away from environmental radiation protection consensus, like the one existing for human radiation protection.

The environmental damage valuation have been used as an important tool for environmental protection against damages caused by many different industries, mainly in electric energy generation sector [10–12].

2. ENVIRONMENTAL VARIABLES USED AS ECONOMIC DECISION TOOLS

Too little attention has been given to the economic conception of environmental radiation protection, differing from other pollutant problems. For these other pollutants, mainly after “Rio 92” conference, macro-economic concepts have been developed, as, for example, “Sustainable development, conservation, maintenance of bio-diversity, environmental justice and human dignity” [7].

In order to incorporate environmental variables into potentially pollutant industries decision processes, many principles have been developed for environmental protection, as, for example: the pollution prevention principle; the precautionary principle, the principle of using best available techniques, the substitution principle, the principle of informed consent and the polluter-pays principle.

This last principle, the polluter-pays principle, is based in the principle that “polluters are responsible for the economic and environmental consequences of their activities”. This principle lead us to the environmental question treatment by economic sight (environmental economy).

The environmental economy explains environmental questions as a relation between goods and their producers, guided by producers interests. In order to study the environmental economic question, one needs some concepts, such as the concept of natural resources and the concept of externality. Natural resources are goods not made by men. Externalities are the effects of people behaviours over the well-being of others.

One of the methods developed for support the polluter-pays principle is the valuation of environmental externalities. The valuation achieve importance not only to quantify the looses with environmental degradation, but also, to bring the degradation costs into the economic process, leading to a more adequate management of both environmental risks, and to allocate resources, in order to face possible environmental damages.

For the identification and valuation of environmental externalities, steps are determined, as follows: To collecting bibliography and the characterization of environmental impacts generating aspects; characterization of the respective environmental impacts; identification of the economic effects caused by environmental impacts, generally not included in the industry programs (externalities); to collecting available environmental valuation technique; identification of the most appropriate techniques; and the valuation of identified externalities.

3. ENVIRONMENTAL VALUATION

The economical value of environmental goods do not have a market to control its price, but as any good or service, its value comes from its attributes, that can or sometimes can not be linked to the good use. Therefor, one can divide economical value of environmental goods as a value of using (VU), and a value of not using (VNU).

The value of using (VU) is the value attributed to the environmental good by its present usage or by its future potential usage. The value of using (VU) can be sub-divided into three categories:

- Direct value of using (DVU) – The value attributed to the environmental good by its welfare by means of its direct usage.
- Indirect value of using (IVU) - The value attributed to the environmental good when the benefits of its usage is derived from eco-systemic functions.
- Option value (OV) – The value people are ready to pay for the maintenance of the option to make use someday, directly or indirectly, of the environmental good.

Yet, the value of not using, or the existence value (EV), is a value dissociated of using (though it represents environmental consumption) and is derived from a moral, cultural, ethic, or religious position in relation to the existence rights of non human species, or the preservation of other natural resources.

With the concepts of value of using and value of not using, the Economical Value of the Natural Resource (EVNR) can be defined as:

$$(EVNR)= ((DVU) + (IVU)+ (OV)) + (EV) \qquad (1)$$

4. METHODOLOGIES OF DAMAGE QUANTIFICATION

4.1. Damage quantification – damage response function

This method is based in the physical relation description between cause and effect of environmental damage, that is, the method relates the impelling activity to the respective environmental damage, for then, to give objective measures of the environmental damages.

This method uses damage functions, or dose response functions, that relate the impelling activity level (pollution type or concentration) with the degree of physical damage to the environmental goods, or with the degree of health damage. This function can be made by field studies or laboratory studies. This functions may be extremely complex and speculative.

In principle, the dose response function is not a valuation method or technique. This function gives a link between two events, one of them which affects, or cause damages, and the other, which is affected, or suffer the effects. With the dose response function it is possible to estimate the damage variation in terms of the variation in the environmental good or service. In the sequence, one can value, with appropriated techniques, the occurred damage.

4.2. Well-being variation quantification method

This method is based on the micro-economic concept of consumer excess of goods. Consumers achieve goods in order to improve their well-being. Different consumers impute different values to each good consumption. Therefore, the maximum values the consumers are ready to pay are also different for each one. The consumer excess of goods is the difference between what consumer was ready to pay and what he actually has paid, that is, it is the total benefit achieved by the consumption of the good, subtracted by the total cost of its acquisition.

5. ENVIRONMENTAL VALUATION – INDIRECT METHODS

The indirect valuation methods can be applied when the production or the consumption of a private good or service can be affected by the variation of quantity or quality of environmental goods or services.

In this method, the impact in changes of the quality of environmental goods or services production or consumption are analyzed in terms of their market prices.

5.1. Marginal productivity method

In this method, the environmental resource R is determined as function of its contribution as component of a product, that is, it is the production factor of a product P confection.

$$P= f\,(B,R) \tag{2}$$

where:

B is the entirety of components of a product formed by goods and services;
R are the utilized environmental resources.

This method is based in the premise that "P varies as function of R", and then, that a relation dose response between P and R must be established (in terms of quality /quantity). In this method, the product price is known and it has a market value.

5.2. Reposition expenses method

This method is based on the quantification of expenses by consumers to recover the good or service changed by the pollution or inadequate management. This method is very likely to the prevention/mitigation expenses method, with the exemption of this last method is based in terms of potential future damages, while reposition expenses works with real reposition damages costs in the present.

Three premises must be attempted for this method to be used:

(a) the damage extent must be measured;

(b) the reposition costs must be smaller than the production cost of the good, in order to the reposition be economically practicable;

(c) If preventive actions have smaller price than reposition actions, then the prevention actions must be executed.

5.3. Prevention/mitigation expenses method

This method is based in the costs for the industries to avoid future environmental damages.

5.4. Relocation expenses method

This is a variation of the reposition method. In this method there exist an evaluation of relocation of physical activities expenses, and the benefits of this relocation cause changes in the environmental quality of the industry.

5.5. Protection expenses method

By analyzing the environmental degradation question under the sight of environmental economy, the environmental costs must be considered in the planning processes of the industries.

6. VALUATION OF ENVIRONMENTAL DAMAGES – DIRECT METHODS

On the contrary of the indirect methods, the direct methods uses substitute markets or hypothetical markets to measure directly the needs for environmental quality.

7. CONCLUSION

The analysis of function dose response match with the type of impact caused by radioactive installations, and therefore this function is indicated as an environmental evaluation method. After the method of quantification of the damage being chosen, the valuation method should be chosen according to the specific site in question.

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Index for comparative evaluation of ecological effects between ionizing radiation and other toxic agents
Application to the model ecosystem data

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Abstract. The authors proposed an index for holistic evaluation of effects on various parameters in ecosystems exposed to ionizing radiation and other toxic agents. As the index, i.e., EEI (Ecological effect index), degrees of differences in parameter values between exposed and control ecosystems were expressed as the Euclidean distance weighted by ecological importance of each parameter. The authors applied the EEI to their ecotoxicological study, in which the aquatic model ecosystem (microcosm) consisting of three species of microorganisms was exposed to γ-rays, ultraviolet radiation, acids and some metals, and their effects on the cell densities were investigated. The EEI for the microcosm was positively correlated with log-transformed doses of each toxic agent, and the relationship between them could be fitted by a sigmoid curve. A 50% effect dose for the microcosm (ED_{M50}), at which the EEI became 50%, could be calculated in a similar manner to a 50% effect concentration (EC_{50}), which has been regarded as one of important toxicity data in conventional single-species tests for chemicals. The ED_{M50} made it possible to quantitatively compare effect doses for the microcosm between γ-rays and the other toxic agents concerned. It was therefore expected that the EEI would contribute to comparative evaluation of effects on natural ecosystems.

1. INTRODUCTION

Ecosystems are exposed to not only ionizing radiation but also other various toxic agents derived from human activities. Ecological effects of radiation should be therefore evaluated compared with those of other agents for a scientifically better understanding of radiation for the public. For this comparative study, ecological effects of radiation and other agents should be evaluated by a common index one another.

Ecosystems consist of various kinds of organisms, and each species has various parameters, e.g., population densities, reproductive rates, mutation frequency and so on, that can be used as endpoints for evaluation of ecological effects. Therefore, an index that holistically reflects effects on these various parameters is required for evaluation of ecological effects. For example, when radiation effects on an ecosystem consisting of 100 species are evaluated adopting their population densities as endpoints, one will require an index that holistically reflects the radiation-induced population changes in these 100 species, depending on the importance of each species for the ecosystem itself and/or man.

In this study, we proposed an index for comparative evaluation of ecological effects between ionizing radiation and other toxic agents, and applied this index to evaluation of effects on the experimental model ecosystem (microcosm) for validation of usefulness of the index.

2. EFFECT INDEX

As an index we propose, degrees of difference in values of concerned parameters between exposed and control ecosystems are expressed as the Euclidean distance weighted by ecological importance of each parameter. That is, the effect index on day *t* after exposure (EI(t)) is defined as follows:

$$\mathrm{EI}(t) = 100\sqrt{\sum_{i=1}^{n} W_i \left\{ \frac{P_{i,\mathrm{Con}}(t) - P_{i,\mathrm{Exp}}(t)}{P_{i,\mathrm{Con}}(t)} \right\}^2}\ [\%] \tag{1}$$

where:

n = the number of parameters concerned;
Wi = ecological weighting factors for parameter *i*. This value depends on importance of each parameter for the ecosystem itself in the ecocentric principle and for man in the anthropocentric principle. The more important the parameter *i* is, the larger the Wi value is. $\sum_{i=1}^{n} W_i = 1$;

$P_{i,Con}(t)$ = values of parameter *i* in the control ecosystem on day *t*;
$P_{i,Exp}(t)$ = values of parameter *i* in the ecosystem exposed to a toxic agent on day *t* after the exposure.

The EI(t) should be averaged for experimental periods, because the time-dependent changes in parameters are generally investigated for evaluation of ecological effects. As such an averaged index, the ecological effect index (EEI) is defined as follows:

$$\mathrm{EEI} = \frac{1}{T}\int_0^T \mathrm{EI}(t)\,dt\ [\%] \tag{2}$$

where:

T = days from the exposure to the end of the observation.

It is expected that the EEI is suitable for quantitative evaluation of effects of a toxic agent on entire ecosystems, and thus is useful for comparative evaluation of ecological effects between ionizing radiation and other toxic agents. This expectation is examined using the model ecosystem data as described in the following section.

3. CASE STUDY – APPLICATION TO THE MODEL ECOSYSTEM DATA

The authors have investigated ecological effects of γ-rays [1] compared with other various toxic agents such as ultraviolet radiation (UV) [2], acidification [3], aluminium [4], manganese [5], nickel [6], copper [4] and gadolinium [7] using the experimental model ecosystem, i.e., microcosm consisting of flagellate algae *Euglena gracilis* as a producer, ciliate protozoa *Tetrahymena thermophila* as a consumer and bacteria *Escherichia coli* as a decomposer [8]. This microcosm mimics essential processes in aquatic microbial communities [9]. That is, the microcosm is maintained with photoenergy which *Eu. gracilis* fixes by photosynthesis. Metabolites and breakdown products of one species contribute to growth of the other two species. *T. thermophila* grazes *E. coli* as staple food. As a result of these interspecies interactions, these three species can co-exist for more than one year without addition of any nutrients. The microcosm can be therefore regarded as a self-sustaining system. Since effects observed in the microcosm were not only direct effects of toxic agents but also indirect effects due to interspecies interactions, it is considered that this microcosm ecotoxicity test can evaluate community-level effects, which cannot be evaluated by conventional single-species tests.

Though effects observed in the microcosm were different in details among the toxic agents, the following dose-response pattern was commonly observed in most toxic agents we examined: (1) no effects; (2) recognizable effects, i.e., decrease or increase in the cell densities of at least one species; (3) severe effects, i.e., extinction of one or two species; and (4) destructive effects, i.e., extinction of all species [4]. For example, acute irradiation by 50 or 100 Gy γ-rays temporarily decreased the cell density of *E. coli*. At 500 or 1000 Gy, *E. coli* died out, and the cell densities of the other two species were decreased compared with controls. At 5000 Gy, all species died out [1]. For another example, at 1 or 10 μM nickel, the cell densities of all species were not affected. At 100 μM, *T. thermophila* and *E. coli* died out. At 1000 μM, all species died out [6].

The effect index we propose is applied to these microcosm data. It is thought that ecological importance of three species constituting the microcosm is the same one another, because each species plays one of ecologically important roles as a producer, consumer and decomposer, respectively. Thus, the value of the ecological weighting factor in Eq. (1) is estimated to be 1/3 for each species. The EI(t) and EEI for the microcosm ($EI_M(t)$ and EEI_M) can be expressed by the following equations:

$$\mathrm{EI_M(t)} = 100\sqrt{\frac{1}{3}\left\{\frac{N_{Eu,\,Con}(t) - N_{Eu,\,Exp}(t)}{N_{Eu,\,Con}(t)}\right\}^2 + \frac{1}{3}\left\{\frac{N_{T,\,Con}(t) - N_{T,\,Exp}(t)}{N_{T,\,Con}(t)}\right\}^2 + \frac{1}{3}\left\{\frac{N_{E,\,Con}(t) - N_{E,\,Exp}(t)}{N_{E,\,Con}(t)}\right\}^2}\ [\%] \quad (3)$$

$$\mathrm{EEI_M} = \frac{1}{T}\int_0^T \mathrm{EI_M}(t)\,dt\ [\%] \quad (4)$$

where:

$N_{X,Con}(t)$ = The log-transformed ($\log_{10}$ (N+1)) cell density of species X in the control microcosm on day *t*;

$N_{X,Exp}(t)$ = The log-transformed ($\log_{10}$ (N+1)) cell density of species X in the microcosm exposed to a toxic agent on day *t* after the exposure;

X = Eu (*Eu. gracilis*), T (*T. thermophila*) or E (*E. coli*).

If a toxic agent does not affect the cell densities of any species for the duration of the experiment, the EEI_M becomes 0**%**. If a toxic agent extinguishes one or two species just after exposure and does not affect the cell densities of the other species, the EEI_M becomes 58**%** (= 100 × $\sqrt{1/3}$) or 82**%** (= 100 × $\sqrt{2/3}$), respectively. If all species die out just after exposure, the EEI_M becomes 100**%**.

The EEI_M at each dose of toxic agents concerned was calculated by Eqs (3) and (4). Figure 1 shows dose-EEI_M relationships for γ-rays and nickel as typical examples. In general, the EEI_M was positively correlated with log-transformed doses of each toxic agent, and the relationship between them could be fitted by a sigmoid curve. This is similar to a concentration-effect relationship in conventional single-species toxicity tests for chemicals. In the single-species tests, a concentration-effect curve is determined by a probit method, and 50**%** lethal or effect concentrations (LC_{50} or EC_{50}) of the chemicals are calculated as one of important toxicity data [10]. Similarly, the dose-EEI_M curve in the microcosm test was determined by a probit method, and a 50**%** effect dose for the microcosm (ED_{M50}), at which the EEI_M became 50**%**, was calculated for each toxic agent. It is considered that the ED_{M50} corresponds to the EC_{50} in single-species tests. The resulting ED_{M50} of each toxic agent is shown in Table 1. It is considered that the ED_{M50} is useful for quantitative comparison of effects on the microcosm between γ-rays and the other toxic agents concerned. It is therefore expected that the EEI contributes to the comparative evaluation of effects on natural ecosystems. However, more case studies will be required to confirm this expectation.

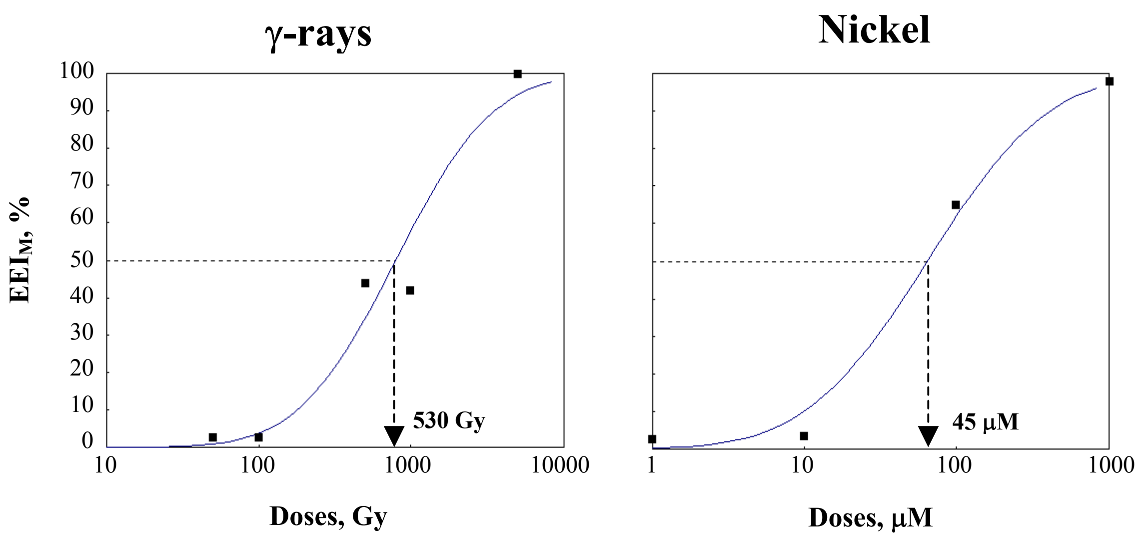

FIG. 1. The dose-EEI_M (Ecological effect index for the microcosm) relationships.

TABLE 1. ED_{M50} (50% EFFECT DOSE FOR THE MICROCOSM) OF Γ-RAYS AND OTHER TOXIC AGENTS

γ-rays	UV-C	Mn	Ni	Cu	Gd
530 Gy	2100 J/m^2	4100 μM	45 μM	110 μM	250 μM

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Environmental approaches adopted for the sound management of radioactive and non-radioactive pollutants in Zambia

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Abstract. This presentation gives an overview of the situation in Zambia with respect to the management of chemicals. A summary description of key pieces of legislation that deal with the protection of human beings and the environment and their objectives are discussed briefly. The paper gives also a summary description of key approaches and procedures for the management of chemicals.

1. INTRODUCTION

A strict control in the use of chemicals is one of the ways through which risks to human beings and the environment can be adequately managed. Controls are put in place through legislation, regulations, guidelines or codes of practice, as minimum requirements to be observed in handling, use, application, storage and disposal of chemicals. Legal instruments can contribute to a more efficient approach to the sound management of chemicals if they are adhered to and enforced.

The laws in place require that chemicals used should be registered. Before a chemical is registered for use, evidence is required to show that it has been adequately evaluated for toxicity, biodegradability, persistence and that the chemical will pose minimum adverse effects to the users and the environment. Registration of chemicals has come at a time when the trade in chemicals has been established for a long time. Nevertheless, it is envisaged that registration of chemicals will control and regulate the use of chemicals in the country.

2. OVERVIEW OF NATIONAL LEGAL INSTRUMENTS, WHICH ADDRESS THE MANAGEMENT OF CHEMICALS

The enactment of the Environmental Protection and Pollution Control Act (EPPA) No. 12, of 1990 was a means of consolidating various pieces of legislation into a single Act in order to cover all aspects of environmental protection and pollution control in an integrated way. The EPPA covers all aspects of air, noise pollution, waste management, water pollution control, pesticides and toxic substances. The Ionizing Radiation Act covers all aspects of importation, exportation, distribution, commissioning, possession, operation, maintenance, decommission, transportation, storage and disposal of radioactive substances.

The Food and Drugs Act, Chapter 303 and the Pharmacy and Poisons Act, Chapter 299 ensures that products meant for human and animal consumption meet acceptable standards of quality, while the Factories Act seeks to protect workers from any effects of chemicals at the workplace.

3. SUMMARY DESCRIPTION OF KEY LEGAL INSTRUMENTS RELATING TO CHEMICALS

The principle legal instrument governing chemicals is the EPPA. This Act regulates, inter alia:

— water pollution by ensuring the quality of water, determining the conditions of discharge of effluent, and determining standards and analytical methods;

— waste management through classification and/or analysis of waste and waste disposal methods, and monitoring and regulating the disposal sites; and

— pesticides and toxic substances through registration and requirements regarding labelling, packaging, transportation, general handling, use and safety, and storage and disposal of pesticides and toxic substances.

Registration under the EPPA covers all the classes of pesticides and industrial chemicals. The Pharmacy and Poisons Act covers Class II poisons and provides for their regulation with respect to acceptable levels in foods, while Fertilizer Act provides for the control of importation, use, storage and disposal of fertilizers.

Other aspects of environmental management are still under different authorities, such as the control of levels of pesticides residues and toxic contaminats in foods, and the transportation of inflammables, corrosives and chemical poisons. The Tsetse control Act provides for use of insecticides in the control of tsetse flies in infested areas. Occasionally, outbreaks of any worms and migratory locust entail the use of large quantities of insecticides.

4. IMPLEMENTATION OF LEGAL INSTRUMENTS

Legal instruments require that chemicals being used do not cause harm to the users and the environment under normal conditions of use. It is for this reason that the chemicals' behaviour is assessed at registration. However, registration only started in Zambia in 1993, and will take some time to be fully operational. Nevertheless, inspections and monitoring of operational facilities is on going and it is a requirement that accidents are reported to the relevant authorities.

5. SUMMARY DESCRIPTION OF KEY APPROACHES AND PROCEDURES FOR THE CONTROL OF CHEMICALS

The major approach to the control of chemicals is through product registration, which is the stage at which a chemical is assessed for its efficacy, safety in use, toxicity and eco- toxicity, persistence and behaviour in the environment. At the same time, the label is assessed for accuracy and completeness of information to the users.

However, the requirements for registration of chemicals came into force only in 1994, although the business in chemicals has been in existence for a long time. The registration provides for the issuing of a certificate of registration. Permits are issued for discharge of effluents, transportation of wastes and for operational disposal facilities.

6. NON- REGULATORY MECHANISMS FOR MANAGING CHEMICALS

Through the Zambia Agrochemicals Association (ZAA), the pesticides industry has a voluntary code of practice aimed at encouraging members to adhere to certain minimum standards of practice.

The Environmental Management System (EMS) encourages each industrial facility to have in place a corporate environmental policy, as a way of promoting environmental awareness on the part of management and workers. The environmental policy is a statement of observance of practices that do not endanger the environment. This is in the form of internal organizational controls put in place by the management, with the aim of monitoring other measures.

7. COMMENTS

Most of the overlaps and gaps in environmental management of chemicals have been taken care of by the enactment of the EPPA and also most of the issues related to radiation protection of human beings against the effects of ionizing radiation have been taken care of by the Ionizing Radiation Act.

However, available laws in the country do not cover all the foreseeable aspects and there is need for amendments so that gaps, overlaps and sometimes duplications are looked at so as to ensure adequate protection of the human beings and the protection of the environment.

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3. THE SCIENTIFIC BASIS FOR ENVIRONMENTAL RADIATION ASSESSMENT

3.1. BIOLOGICIAL EFFECTS

Biological response of *Tradescantia* stamen-hairs to high levels of natural radiation in the Poços de Caldas Plateau and in Brazilian radioactive waste deposits

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Abstract. The objective of the present study was to apply a highly sensitive botanical test of mutagenicity (the Tradescantia stamen-hair mutation bioassay), to assess in situ the biological responses induced by naturally occurring radiation in the Poços de Caldas Plateau and in Brazilian radioactive waste deposits. The mutagenesis was evaluated in environments presenting gamma radiation exposure rates ranging from 1.5 $\mu R.min^{-1}$ up to 750.0 $\mu R.min^{-1}$. The results consistently show only borderline increases in mutation frequencies in plants exposed to areas with high radiation background, as compared to non-exposed plants. It is concluded that the levels of natural radiation prevalent in the Poços de Caldas Plateau are not sufficient to induce significant increases in mutation rate, even in the extremely sensitive Tradescantia stamen hair mutation bioassay, and that this mutagenesis evaluation test can be a useful monitoring system for natural radiation exposure.

1. INTRODUCTION

The departure from linearity in the dose-response relationship of radiation effects has important consequences regarding the uncertainty related to the different parameters used in the evaluation of potential radiation hazards. Mutagenesis induction is one particularly valuable radiation assessment parameter, and plants are especially adequate experimental subjects for mutagenesis evaluation, not only for their amenability to *in situ* exposure, but also due to the high sensitivity of some plant test systems, such as the *Tradescantia* stamen hair mutation assay (Trad-SHM) [1]. The Trad-SHM assay is, thus, especially suited for the study of complex environmental situations, such as those found on the Poços de Caldas Plateau, which has been identified as amongst the most naturally radioactive locations on the Earth.

The Trad-SHM is a somatic mutation (mitotic) bioassay in which expression of the heterozygous dominant blue character of the stamen hair cell is prevented, resulting in the appearance of the recessive pink color [2]. The sensitivity of *Tradescantia* to the genetic effects of radiation and chemical agents is widely known [3, 4]. Studies on the effects of very low radiation levels with the Trad-SHM assay involve a series of exposure situations, from absorbed radioisotopes, radiation-contaminated substrates [5, 6], and high level background radiation from monazite sand [7]. The Trad-SHM assay showed to be an adequate genotoxicity bioindicator, both in terms of detecting radiation exposure, as well as in terms of sorting out the confounding environmental factors that interfere with biological responses to radiation. In the present study, the Trad-SHM assay was used to assess the mutagenicity induced by the high levels of natural radiation occurring on the Poços de Caldas Plateau and in Brazilian radioactive waste deposits.

2. MATERIALS AND METHODS

2.1. Exposure "in situ"

The mutagenesis evaluation was carried out in different environments, presenting gamma radiation exposure rates varying from 1.50 μR.min^{-1} to 100.0 μR.min^{-1}, as shown in Table 1. Groups of ten pots containing flowering *Tradescantia* plants (clone 4430) were kept in their respective exposure sites for 24 hours. In the mean time, for each exposed group there was one control group kept in controlled-environment greenhouses presenting a radioactivity background of 1.6 μR.min^{-1}. These *Tradescantia* stock plants maintained in the greenhouses were considered also as the reference to evaluate the spontaneous mutation frequency for clone BNL 4430. In order to evaluate possible greenhouse effects, and as a means of ascertaining a more stable set of controls, two *Tradescantia* stock populations were kept in two separate greenhouse spaces (the greenhouse itself, and its annex, set to the same environmental conditions). These plants were cultivated in 5-inch pots containing humus, supplemented with fertilizer each 15 days (nitrogen-phosphate-potassium), watered every other day and maintained clean and pest-free by manual scouting and pruning. The radiation level of each of the exposure sites was determined at the exact same position where the plants were placed, using a 1800 cc ionizing chamber and a radiation monitor controller, models Radcal 10x5 – 1800 and 9015, respectively. The measure was repeated 10 times for each exposure site.

TABLE 1. EXPOSURE SITES OF *TRADESCANTIA* PLANTS IN THE POÇOS DE CALDAS PLATEAU AND IN BRAZILIAN RADIOACTIVE WASTE DEPOSITS

Exposure site (Abbreviation)	Radiation exposure rate (μR.min^{-1})
Pit Mine (PM)	1.5
Greenhouse (GH)	1.6
Tailing Dam (TD)	6.0
Itataia Ore (Ita)	10.0
Morro do Ferro 1 (MF1)	21.0
Radioactive Waste Deposit (WDa)	33.0
Gallery of Morro do Ferro (Gal)	41.0
Morro do Ferro 2 (MF2)	50.0
Radioactive Waste Deposit (WDb)	100.0
Radioactive Waste Deposit (WDc)	750.0

2.2. Tradescantia bioassay

The Trad-SHM assay applied in the present experiments is a mutation (mitotic) assay in which expression of the heterozygous dominant blue character of the stamen hair cells is prevented, resulting in the appearance of the recessive pink color. Details of the experimental methods and a review of the results obtained with this bioassay are available in Rodrigues, et al., (1977) [4]. For each field experiment, twenty flowers were evaluated daily, being ten coming from exposed pots and other ten coming from control (greenhouse) pots. Mutation scoring was performed between the 7th and 13th days after exposure, in order to allow the exposed flower buds to open as mature flowers in which the stamen hairs can be observed (under X60 magnification). The number of stamen hairs per flower in each treatment group was estimated [8], and the number of mutation events per 1000 hairs was determined. On average, over 3000 hairs were scored for each treatment day. Statistical comparisons were carried out on the transformed data (y=[sqrt X]+[sqrt[X+1]) [9], by ANOVA ($p \leq 0.05$) for the days of largest mutation frequencies for all the treatments. Specific comparisons between each treatment and its specific control were carried out by unpaired t-Test ($p \leq 0.05$).

2.3. Results

Figure 3 shows the mutation frequencies for all the exposure sites and their corresponding controls, along with each site's gamma radiation exposure rate.

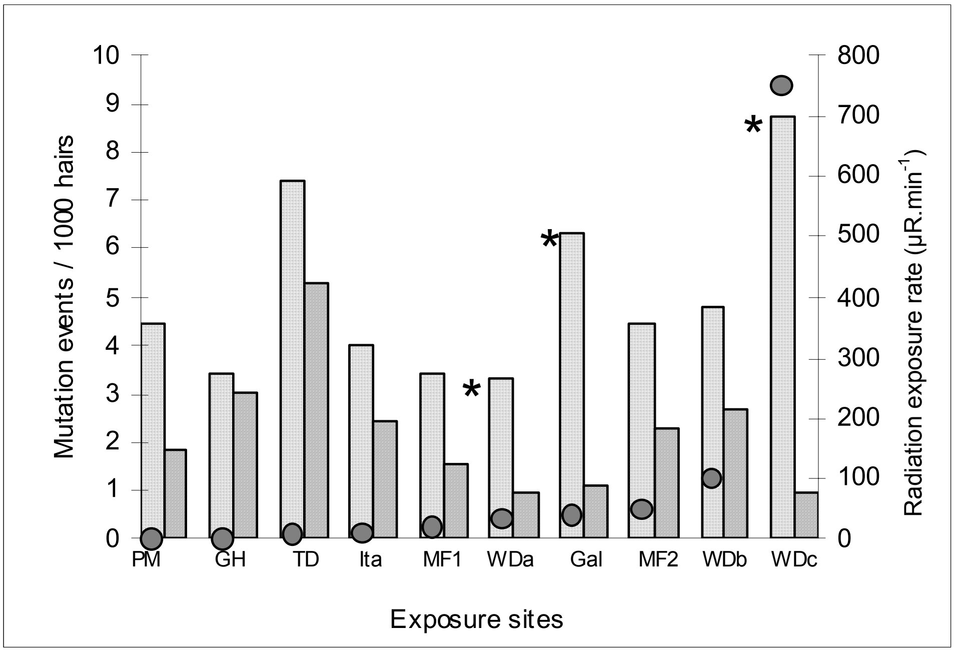

FIG. 3. Mutation frequency of the Trad-SHM assay after exposure to different sites of varying natural radiation levels on the Poços de Caldas Plateau. Each site is represented by the exposed plants (dotted bars) and their corresponding controls (slant lines bars), in addition to its gamma radiation exposure rate (full circles). The (*) symbol indicates statistical significance: $p < 0.05$. The exposure sites are as follows: PM - pit mine; TD - tailing dam; Ita - Itataia ore; MF - Morro do Ferro (sites 1 and 2); Gal – Gallery of Morro do Ferro; WD – radioactive waste deposit; GH - greenhouse.

4. DISCUSSION

Many studies have shown that a linear increase in mutation frequency occurs in *Tradescantia* stamen hairs exposed to increasing radiation doses [10]. The nonlinear relationship between gamma radiation exposure and mutation frequencies observed in the present study indicated that other interfering factors might be having a role in the exposure sites on the Poços de Caldas Plateau and in Brazilian radioactive waste deposits. The spontaneous mutation rate of *Tradescantia* can be affected by several environmental factors such as light, temperature, nutritional status, and air impurities [11, 12].

The Trad-SHM assay has been employed also to show that important synergistic (mostly additive) interactions occur between ionizing radiation and chemical agents [3, 13, 14]. In the present study, however, even though the plants were exposed *in situ* to environments presenting not only different gamma radiation exposure rates, but also a whole set of different environmental conditions, only three sites, the Gallery of the Morro do Ferro (gamma radiation exposure rate of 40 $\mu R.min^{-1}$), Radioactive Waste Deposit a (gamma radiation exposure rate of 33 $\mu R.min^{-1}$) and Radioactive Waste Deposit c (gamma radiation exposure rate of 750 $\mu R.min^{-1}$) showed a significant increase in mutation frequency relative to its corresponding control. The borderline response, showed for most of the exposure sites studied, indicates that the higher natural radiation levels occurring on the Poços de Caldas Plateau are not sufficient to induce significant increases in mutation frequency, even for a sensitive mutagenesis evaluation bioassay, such as the Trad-SHM.

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New Developments in our understanding of radiation effects in non-human biota; implications for radiation protection
Comparative radiobiology and radiation protection

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Abstract. Recent advances in our understanding of effects of radiation on living cells suggests that fundamentally different mechanisms are operating at low doses compared with high doses. Also, acute low doses appear to involve different response mechanisms compared with chronic low doses. At the biological level this is to be expected. Chemical toxicity has been known for many years to show these patterns of dose response. Cell signalling and coordinated stress responses appear to dominate acute low dose radiobiology while adaptive responses also become important when the dose is protracted. This means that in situations where cancer induction is not a problem, as is mainly true for non-human biota, the key questions to be addressed involve the effect radiation-induced responses might have on competitiveness or on ecosystem sustainability. Also important is information about the interaction between low level radiation and chemical exposures. This paper discusses our knowledge about low dose radiobiological effects in biota and draws attention to the problem of defining the important questions relevant for the level of organisation being studied.

1. INTRODUCTION

Actual studies of comparative radiobiology, (other than those relating to man or models used to study selected aspects of human radiobiology, for example, Nematode, fruit fly and yeast tend to be very limited. They are mainly found in old literature, and they use extremely high doses, which are irrelevant to environmental conditions. The endpoint is usually death of the irradiated animal which really precludes any mechanistic studies.

Where mechanistic studies have been done, these suggest extremely radioresistant responses (reviewed in [1]). Our group have also studied the radiation response of explants of salmonid skin and some established fish cell lines using in vitro techniques [2]. These tend to show radiosensitivities similar to mammalian systems. The crustaceans as a class seem to have been largely ignored in radiobiological studies Our group has recently commenced a study of crustacean radiobiology using cultures of haematopoietic tissue from *Nephrops norvegicus* [3].

The acute response of *Nephrops* cultures to radiation is interesting. The cultures are more sensitive than explant cultures obtained from human urothelium, or fish skin, both of which are cultured in the Dublin Laboratory under similar conditions. While it can be argued that in vitro responses do not necessarily correlate with in vivo sensitivities, it is likely that the comparative ranking of radiosensitivities seen in vitro would hold in vivo.

The bystander effect has also been found in explant cultures from crustaceans. To date this has only been seen in mammalian cells although our group has evidence of an effect in the salmonid (CHSE line). It is interesting also that vertebrate cells from two classes (Pisces and Mammalia) can respond to the signal produced by a completely unrelated species in a different phylum.

The data indicating a bystander effect similar in magnitude to the mammalian effect, point to common cellular mechanisms being involved in the response of these evolutionarily distant groups. The very sensitive response is unexpected and suggests that at least this species of crustacean may be more sensitive than expected to low acute doses of radiation. The implications of this for protection of the environment and species other than man, may need to be considered. There are no laboratory studies of chronic low dose effects in non-human (or rodent models) biota. No in vitro work has been done either. What data are available suggest that adaptive responses may dominate at low chronic doses (e.g. see [4]).

2. IMPLICATIONS FOR RADIATION RISK ASSESSMENT

Some of the implications of these considerations for risk assessment and for development of new protection policies include the following: At present we know that radiation induces the phenomena of genomic instability and bystander response, described above, in man, other mammals, fish and crustaceans. This covers all classes in which an effect has been sought. The induction dose required is low, the effect is fully expressed at acute doses of 3-5mSv for sparsely ionising radiation and one track trough a population of cells for densely ionising radiation. Action limits for workers in the radiation industry are in this range but this is for yearly exposures. It is important to realise that no one has tested lower and maximum expression of delayed effects is already seen at the relatively low doses tested. There does not appear to be an increasing effect with increasing dose so the effect is relatively more significant as a risk factor at low doses than at high doses. Delayed reproductive death (lethal mutations) is a common occurrence in progeny which survive irradiation. This cell loss is probably an important factor in determining long-term reproductive fitness at the population level. Immune system components are very sensitive to these delayed effects.We have strong evidence from human in vitro experiments and mouse in vivo and in vitro experiments that there is a genetic basis for instability and therefore we suspect that some species/individuals will be more likely to become genetically unstable after exposure to radiation than others.We do not know how radiation induces instability or what the mechanism of the bystander effect might be. Such knowledge might enable us to prevent it's induction. We do not know if there is a natural mechanism for controlling or preventing the establishment of cells carrying instability. Research is needed to investigate this and to determine mechanisms underlying the control of survival and death post irradiation. We do not know what underlies the apparent genetic or species basis for instability effects. It is particularly important to determine whether there are different subpopulations of humans and animals and whether these can be identified by simple screening tests. Again this information would provide possible avenues for protection of exposed populations such as the use of sensitivity scaling factors.We do not know whether there is a low dose threshold for genomic instability or the bystander effect. The lowest doses tested showed, in most cases, maximum expression of the effect. These doses are at the upper limit of environmental relevance. What happens at lower doses? The concept of "background levels of mutation" does not apply here since the radiation effect appears to be to raise the background or intrinsic mutation rate for the whole population of cells for as long as has been measured.We do not know how other pollutants such as mutagenic chemicals affect unstable cell populations. It is reasonable to expect higher levels of mutations following chemical exposure if the population already has a higher susceptibility to mutation induction. Research to clarify this might help to explain why studies in one area show evidence of "radiation" induced cancer while studies in another area do not. Radiation may just be facilitating the mutagenicity of another factor. Again knowledge of the mechanisms and interactions would aid development of logical and effective protection strategies.

3. THE FUTURE?

A new approach which might prove valuable would be to integrate environmental toxicology including radiotoxicology across a wider range from cell to ecosystem. At each level the questions which are important are different. For example among humans, the individual life is paramount, therefore survival of the organism for as long as possible is the goal but for other species, at least in our anthropocentric view, survival of the species, or sustainability of the ecosystem is sought. Thus the priotities and needs are very different. Table 1 is an attempt to identify some of the questions that might need consideration at different organisational levels. The planned departure from a human centred radiation protection system to one which is environment or ecosystem based will bring many challenges, not least of which is detaching from the deeply embedded paradigms which dominate human radiation protection at present.

TABLE 1. POSSIBLE QUESTIONS OF INTEREST AT DIFFERENT ORGANISATIONAL LEVELS

Cell- can live or die
Tissue-can function or not
Organism- can survive to reproduce or not
Species-can compete successfully or not
Ecosystem-is sustainable or not
There are different questions and priorities at each level.A sample of possible questions is presented below
Context of a cell in it's environment-cell's perspective
Spatial context What are the neighbouring cells doing? Cellular energy budgets and societal responsibilities
Temporal context How long have I got to live? Can I maintain the tissue structure pending repopulation, then die later? Have I the energy to repair before dividing?
Context of a cell in it's environment - tissue's perspective
Spatial context How widespread and dangerous is the damage? What is the minimum cell number necessary to retain function? What is the minimum to repopulate? Energy redistribution needed?
Temporal context What is the time scale for the above?
Context of a tissue in it's environment – organism perspective
Spatial context Is it possible to live with reduced function of this organ/tissue? Can a duplicate organ or different organ compensate Can diet or external factors such as environmental shelter or certain herbs/food help survival? Can society / the herd / the rest of the colony help or will they kill the weak?
Temporal context How long can I survive with this damage? How long will it take to recover from or repair the damage? Can I survive long enough to reproduce successfully? Can I hide well enough and for long enough from predators and not starve before I regain fitness?
Context of an organism in it's environment – Species perspective
Spatial context How many members of the species can die before its reproductive success is threatened ie what is a viable unit? What competition is there for territory and is fewer better? What preys on these organisms and will scarce prey and hungry predators further exercerbate the situation?
Temporal context Are mainly post reproductive members of the species at risk? How long will the population take to recover? Will reproduction and increased availability of territory lead to repopulation? Over what time scale? What other species are affected?
Context of a species in its ecosystem – ecosystem perspective
Spatial context How many species are affected? Are they critical or sentinal predators or prey? Is there a sustainable loss or a wipeout? Are other stressors present and do they interact synergistically? Is the ecosystem very vulnerable or biodiverse? ie is it worth saving What is man's role in this environment?- destructive or remedial?
Temporal context What are the food web dynamics? Is it reproductive time of year for critical groups? Has the ecosystem any resistance due to previous exposures or have previous/chronic exposures led to adaptation or loss of biodiversity already?

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Ecological propagations of radiological impacts in the microbial model ecosystem
A study with computational simulation

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Abstract. Well-designed experimental model ecosystem could be a simple reference of the actual environment with complexity. For ecotoxicological tests of radiation and other environmental toxicants, we made an aquatic microbial ecosystem in the test tube with autotroph flagellate algae, heterotroph ciliate protozoa and saprotroph bacteria. For theoretical analyses on the population and mass dynamics of these microbial species, an individual-based computer simulation model with dynamic energy mass budget theory was developed, and it showed the mechanisms of the synergistic impacts of acute gamma radiation exposures on their population dynamics and mass budgets.

1. INTRODUCTION

The ecosystem is a self-sustaining system of complexity. Its responses to the impacts are synergistic and subjected to the demographic stochasticity of the species, environmental stochasticity and randomness (catastrophes, etc.). Environmental fate and effects of radiation has ranged from observable DNA damage of the cell to the fare on tissues, individual(s), population (s), community and ecosystem(s). One of the key issues on the protection of environment from radiation is the ecological relevance of radiation effects at each endpoint.

Aim of this study is to develop some mathematical and computational ecosystem models to find advantages and limitations of ecotoxicity tests using the model microbial ecosystem (MICROCOSM) as a reference of radiation effects on the actual environment.

2. MATERIALS AND METHODS

The quantitative, systematic individual-based model, SIM-COSM was developed to simulate impacts of radiation exposure and other toxicants on an aquatic microbial ecosystem (microcosm) [1–3]. The microcosm consists of heterotroph ciliate protozoa, *Tetrahymena thermophila B* as a consumer, autotroph flagellate algae, *Euglena gracilis Z* as a producer and saprotroph bacteria, *Escherichia coli DH5* as a decomposer The culture medium is 10 ml of water with inorganic salt and initial organic nutrients, and held in the airtight test tube. It is cultured statically with fluorescent lamps under 2500 lx and 12 hrs light-dark cycle at 25°C (Figure 1).

The symbiosis among microbes is self-organized by realizing material cycle and sustained for more than 2 years after inoculation. The system can not afford to lose anyone of the microbes to maintain its sustainability. Experimental ecotoxicological tests for gamma radiation [2, 3] were conducted, and significant impacts on the population dynamics were observed with acute exposures of gamma radiation at several doses. To analyse these fndings, we developed an individual-based simulation model (Code: SIMCOSM) using StarlogoT, a computer language for object-based parallel model (developed and distributed by Center for Connected Learning and computer-based Modeling, North western University on the web site: http://ccl.northwestern.edu).

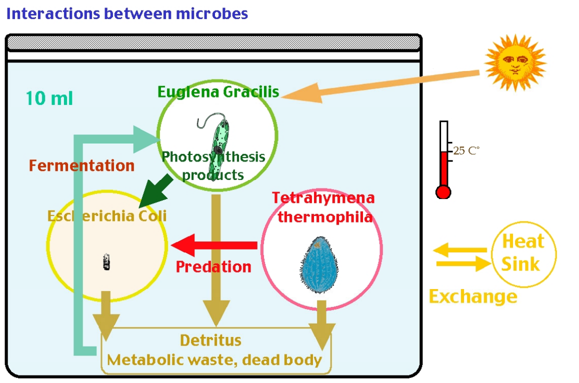

FIG. 1. Interactions between species in the microcosm and Interrelationships among constituent elements in the microcosm.

(1) The SIM-COSM has lattice of 10201 patches as spatial environments, each one of the patches has environmental attributes (*pH, O_2, CO_2 (HCO_3^-), NH_3 (NH_4^+), dissolved organics, etc.*).

(2) Each individual protozoa (*Tetrahymena* and *Euglena*) has its own physiological , structural and behavioural attributes (*heading direction, current patch address, velocity, structural biomass, reserve mass, age, cell cycle phase, maintenance rate, breathing rate, assimilation rate, etc.*)

(3) *Tetrahymena* behaves by following optimum foraging strategy to quest for *E.coli* and eat them (*predator-prey relationship*) [5].

(4) *Euglena* utilizes CO_2 in the patches and synthesize organic substrates in day time (*photosynthesis*).

(5) *E.coli* utilizes dead protoplasm and metabolic wastes in the patches and break them into inorganic substrates *(CO_2 etc.)* for the use of photosynthesis by *Euglena*.

(6) *Tetrahymena*, *Euglena* and *E.coli* respirate O_2 and release CO_2, utilise organic materials and excrete metabolic materials into the patches.

(7) *Tetrahymena*, *Euglena* and *E.coli* reproduce by cell division (cell cycle, structural biomass), and die by starvation (*reserves mass*).

(8) Environment and species are assumed to be verticality homogeneous. Gravity is not taken into consideration in the SIM-COSM.

Concept of dynamic energy mass budget [4] is illustrated in Figure 2, and its application for individual-based computer simulation model, SIMCOSM is illustrated in Figure 3 [6].

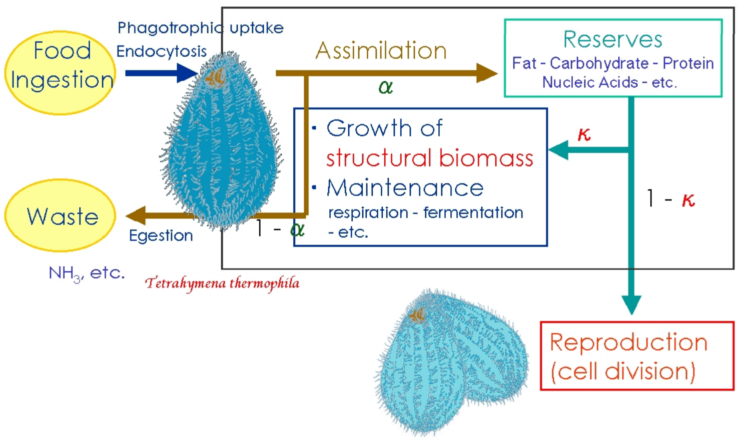

FIG. 2. Schemes of dynamic energy budget model in biochemical systems [4].

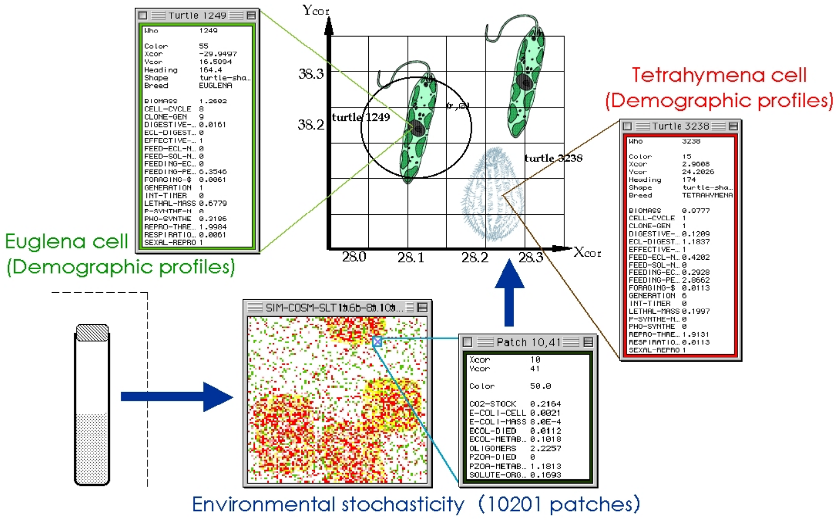

FIG. 3. Basic concept of the individual-based computer simulation model (SIM-COSM) [6] converted from microcosm [1–3, 7, 8].

3. RESULTS AND DISCUSSION

Population dynamics of microbes in Microcosm and its computer simulations by SIM-COSM are shown together in Figure 4 (left). To simulate the impacts of acute exposure of gamma radiation, acute lethal dose (LD50) are adopted to 330 - 170 Gy for Euglena gracilis, 4000 Gy for Tetrahymena thermophila, and 50 Gy for E. coli-DH5-alpha, which is a highly radiosensitive strain on the basis of experimental data in the references. For Tetrahymena, metabolism rate is regarded to reduce to 10-30% by 500 Gy exposure. Population dynamics in Microcosm and SIM-COSM exposed to 500 Gy of gamma-radiation at 50 days after inoculation are shown together in Figure 4 (right). Taking LD50 of Tetrahymena into account, extinction of Tetrahymena might be regarded as the secondary effect of the extinction of E.coli, that is a prey of Tetrahymena. As well, SIMCOSM found the reduction of metabolism rate of Tetrahymena originated its prolonged survival after exposure of 500 Gy gamma radiation.

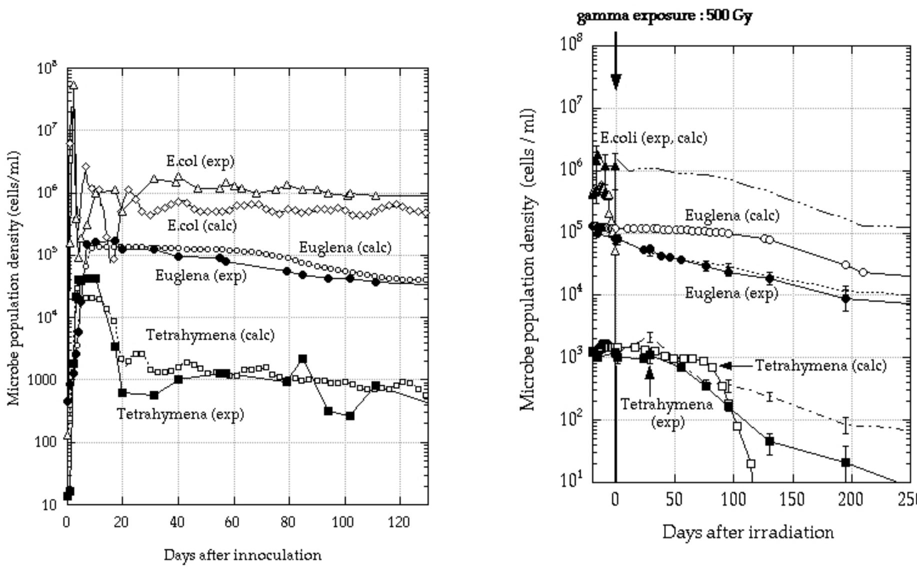

FIG. 4. Population dynamics of Microcosm [2, 3] and its computer simulations by SIM-COSM (in the left) and impacts of acute gamma exposure of 500 Gee on population dynamics of microbes.

4. CONCLUSIVE REMARKS

As to take the effects on the interactions between species and environment into account, ecotoxicity tests with microorganic ecosystem was recreated as a individual-based computer simulation model for the analysis of mathematical ecology. Management of environmental impacts of radiation should be discussed consistently with not only the field surveys but also an experimental model ecosystem studies and their mathematical analyses.

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Apoptosis-like cell death induced by ionizing irradiation in cultured conifer cells

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Abstract. Among plant species, coniferous plants are particularly sensitive to ionizing radiation. We examined the effects of X-ray irradiation on cultured cells of Japanese cedar. Cell death in the cultured cells was increased drastically by X-ray irradiation at 5 Gy, which is as low as the dose inducing radiosensitive programmed cell death, i.e. apoptosis, in mammalian cells. The advancement of cell death in the Japanese cedar cells was accompanied by nuclear DNA fragmentation, which is typically observed both in apoptosis of mammalian cells and in hypersensitive programmed cell death observed in plant cells exposed to various environmental stresses. The results suggested that radiosensitive cell death should be involved in the susceptibility of coniferous plants to ionizing irradiation, and furthermore, this death should be a kind of programmed cell death which has a similarity to apoptosis in mammalian cells.

1. INTRODUCTION

Plants are essential components in ecosystems, which could suffer from ionizing radiations due to nuclear accidents or radioactive wastes from nuclear facilities. Considerable information is already available about the relationship between radiation doses and damages in plants [1, 2]. On the other hand, only a little information is available about the cellular mechanisms relating to the damages. This is in striking contrast to mammals, in which cellular damages by ionizing irradiation have been extensively studied, mainly within the context of medical radiotherapy for humans.

The lack of information on plants is partly because many plant species, including agricultural plants, are so radio-resistant that they are unlikely to be damaged even in the extreme case of a nuclear accident. However, it is evident that plants have a wide range of radiosensitivities, and damages in radiosensitive wild plants such as conifers would induce serious effects on ecosystems [1, 2].

In this study, we used a cell culture of a Japanese native conifer, Japanese cedar, to elucidate cellular effects of ionizing radiation on radiosensitive plants. We show that the coniferous cells develop radiosensitive cell death from a low dose of X-ray irradiation. This radiosensitive cell death in the plant cells is associated with a subset of features characteristic of apoptosis, a radiosensitive cell death in mammalian cells.

2. MATERIALS AND METHODS

2.1. Plant materials

A suspension cell culture of Japanese cedar (*Cryptomeria japonica*), initiated from young embryo, was grown in modified CD medium supplemented with 7 μM 2,4-dichlorophenoxyacetic acid and 3 μM 6-benzylaminopurine, and subcultured at 3-week intervals [3]. A suspension cell line (BY-2) of tobacco (*Nicotiana tabacum*), initiated from a cotyledon, was grown in Murashige and Skoog's medium [4] supplemented with 0.9 μM 2,4-dichlorophenoxyacetic acid and 200 mg L^{-1} KH_2PO_4, and subcultured weekly [5]. Both cell cultures were grown in darkness at 23°C.

2.2. Ionizing irradiation

The cell culture of Japanese cedar was irradiated with X-rays at a dose rate of 0.9 Gy/min by an X-ray generator (Pantak H-320, Shimadzu Co., Tokyo) 3 days after the cells were subcultured. γ-Rays from a ^{137}Cs radio source were used for irradiation of the tobacco cell culture at a dose rate of 50 Gy/min. The irradiated cells were sampled 2 days after irradiation for analyses of cell viability and nuclear DNA fragmentation.

2.3. Measurement of cell viability

Cell viability was assessed based on the ability of cells to exclude the dye, Evans blue. The cells were incubated with 0.05% of Evans blue solution for 30 min. Dead cells were stained blue with Evans blue, whereas living cells remained unstained. The stained cells were viewed using a light microscope (Optiphot, Nikon Co., Tokyo).

2.4. In situ detection of DNA fragmentation

For detection of nuclear DNA fragmentation, a terminal deoxynucleotidiltransferase-mediated dUTP nick end labelling (TUNEL) assay was performed. The sample cells were fixed in a 4% (v/v) paraformaldehyde solution for 30 min, and an in situ cell death detection kit using fluorescein (Roche, Mannheim) was used for the TUNEL assay according to the protocol provided by the manufacture. The cells were counterstained with 4'-6-diamino-2-phenylindole dihydrochloride (DAPI) to allow nuclei to be visualized. Fluorescein-derived green fluorescence is produced in TUNEL-positive nuclei under blue light, whereas DAPI-derived blue fluorescence is produced in every nucleus under UV light. The labeled cells were viewed using a fluorescent microscope (BX-40, Olympus Optical Co., Ltd., Tokyo).

3. RESULTS AND DISCUSSION

Cell death in cultured cells of Japanese cedar and tobacco two days after ionizing irradiation was visualized by Evans blue stain (Figure 1). Most of the Japanese cedar cells irradiated with 5 Gy of X-rays were dead (Figure 1A), however cell death was seldom observed in the non-irradiated cells (Figure 1B). In the case of tobacco cells, on the other hand, no increase of cell death was observed even by irradiation with 50 Gy of γ-rays (Figures 1C and 1D). This indicated that the Japanese cedar cells were much more radiosensitive than the tobacco cells. We considered that such radiosensitive cell death as observed in the Japanese cedar cells should develop only in some cells, depending both on the tissues and the plant species the cells were derived from. This radiosensitive cell death may relate to the susceptibility to ionizing irradiation in radiosensitive plants such as conifers.

The Japanese cedar cells seemed to be as radiosensitive as some mammalian cells including lymphocytes, which are one type of radiosensitive cells in mammalian tissues [6]. In such types of mammalian cells, cell death by low dose ionizing radiations can occur by a gene-driven programmed cell death, apoptosis. To test whether apoptosis-like cell death is involved in the radiosensitive cell death of the Japanese cedar cells, we examined nuclear DNA fragmentation, which is a typical feature of apoptosis, using TUNEL analysis. In the control cells, no DNA fragmentation was found (Figure 2C) in any nuclei (Figure 2D). On the contrary, a drastic increase of DNA fragmentation was detected in the cells irradiated at 5 Gy (Figures 2A and 2B). The result showed that nuclear DNA fragmentation coincided with the cell death in the Japanese cedar cells. This indicated that the radiosensitive cell death in the plant was associated with a subset of features characteristic of apoptosis in mammalian cells.

Recent studies about stress physiology of plants suggest that nuclear DNA fragmentation is characteristic of programmed cell death, which occurs as a hypersensitive response to pathogen infection [7] and abiotic stresses such as ozone exposure [8]. It seems likely that a common mechanism should work between radiosensitive cell death and hypersensitive responses to other stresses in plant cells. Comparative analyses are necessary to elucidate the hidden mechanism relating to the radiosensitive cell death in plants, not only between plants and mammals but also between the effects of ionizing irradiation and other environmental stresses on plants.

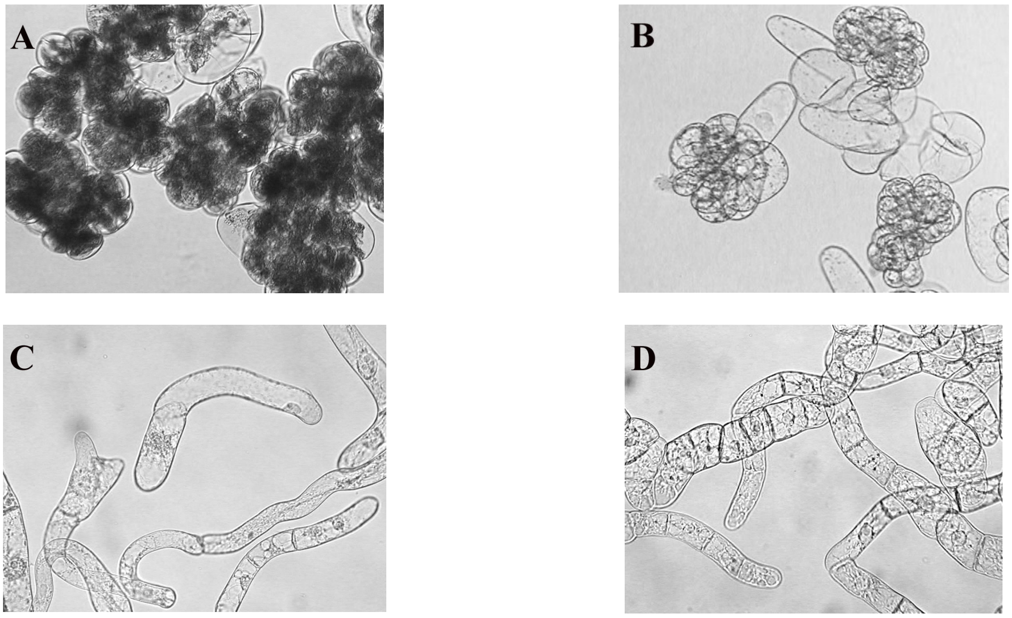

FIG. 1. Evans blue staining for in situ detection of cell death in cultured cells of Japanese cedar (A, B) and tobacco (C, D). The cells were irradiated with 5 Gy of X-rays for Japanese cedar (A) and 50 Gy of γ-rays for tobacco (C), and compared with non-irradiated cells (B, D). Dead cells were stained blue with Evans blue, whereas living cells remained unstained.

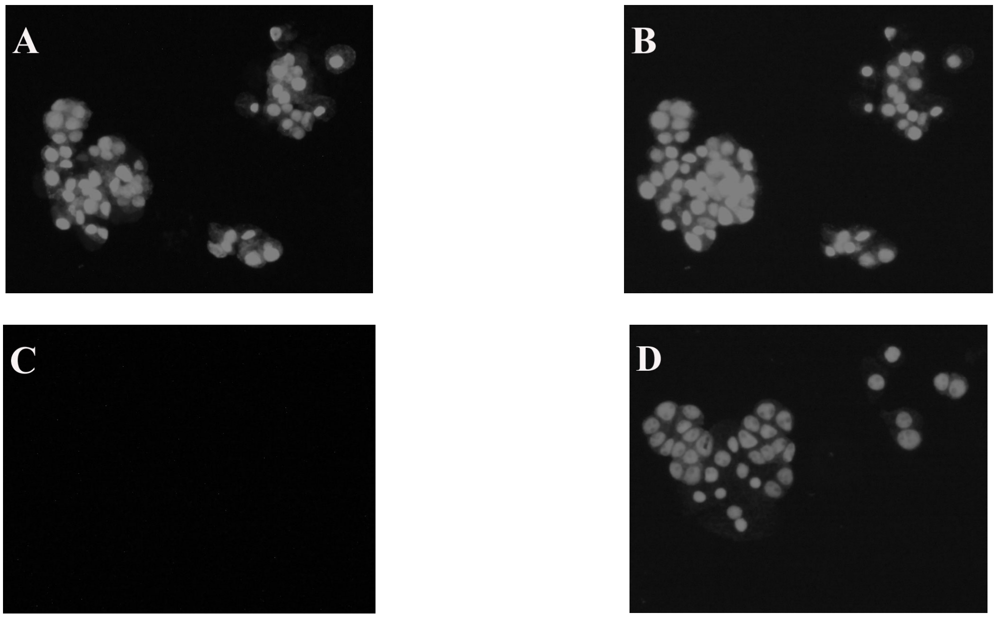

FIG. 2. TUNEL staining for in situ detection of nuclear DNA fragmentation in cultured cells of Japanese cedar. The cells were irradiated with 5 Gy of X-rays (A, B), or not irradiated (C, D). TUNEL-positive nuclei with DNA fragmentation produced green fluorescence under blue light (A, C), whereas every nucleus counterstained with DAPI produced blue fluorescence under UV light (B, D).

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Cytogenetic changes at the animals from the territories with the different concentration of Radon in the above-soil air

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Abstract. The article is devoted to the evaluation of the monitoring of the agroindustrial enterprises of the Ural regions which are situated on the territories, polluted with the radioactive and chemical substances, and ^{222}Rn among them. The cytogenetic analyses and the level of the incidence of the animals with the oncological diseases in these unfavorable zones are given.

The radiological situation in the Ural region is dissimilar and in some regions unfavorable. This trouble is caused not only by the natural environment, but by the activity of the enterprises of the Ministry of the Atomic Energy and other departments as well.

The natural radioactive background is characterized by the great mosaic caused by the inclusion of some natural radionuclides such as: ^{232}Th (Thorium), ^{40}K (Kalium), ^{238}U (Uranium), ^{226}Rd (Radium) and especially ^{222}Ra (Radon) to the geological complexes. The radiational loading from these radionuclides is high and, first of all, in the places of the granite intrusions. The great number of the local accumulations of the natural radioactive mineralization was revealed in the Sverdlovsk, Chelyabinsk and Orenburg regions [4].

At the present time the complex analysis and the summarizing of the ecological investigations are carried out on the territory of the Urals and it helps us to single out the groups of the ecologo-radiogeochemical zones. These zones are characterized by the increased level of the natural radioactivity of the upper part of the lithosphere, underground waters and by the concentration of ^{222}Rn in the above-soil air. Rn, a heavy gas without color and smell, is the product of the decay of Ra. It contributes to the natural radioactivity of the environment.

The significant irradiation of the living organisms is conditioned by gas itself and by the products of its decay. In the Ural regions the vast territories with the anomalous concentration of ^{222}Rn in the above-soil air have been revealed. Radon constantly gets to the environment from the lithological complexes from the earth's crust during the decay of the Uranium and Radium contained in the ore. The greatest amount of Rn is contained in the surface atmosphere, and the higher over the land the less is its concentration in the air.

The toxic activity is based on the good solubility in water and in the fluids of the organism. During inhalation ^{222}Rn is evenly distributed in the whole organism but mainly in the digestive system, fat tissues and in the brain tissues. In these tissues the maximum concentration advances in 10-15 minutes, but in 2 hours only traces of this concentration are left.[3] During such a short period of time and after disposable entering Rn practically doesn't effect ruinously the organism. But the constant introduction of it leads to the dystrophical and degenerative changes of the parenchymal organs.[2] Some investigations to reveal the radiodangerous areas on agricultural territories were carried out. The level of Rn was defined by the device "Ramon-01" in the closed and open cattle-raising and subsidiary placements. The results of the investigations showed that in some settlements of the Urals the level of the equivalent, volumetric activity of ^{222}Rn was substantially lower then the existing hygienisation and norms (1 Bk/m^3), but in the mountainous regions this level by 52 times exceeded the previous one. In the notventilated placements of the milk-producing firms the substantial exceeding of the norms was registered. These radonodangerous areas are hypothetically connected with the great break of the earth's crust in these zones.

One of the negative actions of the ^{222}Rn is its influence on the genetic apparatus of the living animals [1]. The cytogenetic monitoring of the medulla of the cattle, which spent a long period of time under the influence of these substances, confirmed the influence of the environment on the organism. The animals from the zones which were favorable concerning ^{222}Rn were used as the monitoring.

The analysis of these data revealed the reliable difference between animals from different territories regarding the quantity of anaphases with the reconstruction of chromosomes. It was noted that the animals from the territories with the increased concentration of ^{222}Rn during the investigation of the medulla showed the great amount of anaphases with bridges, while cows from the zones with the very low level of ^{222}Rn more often revealed the anaphases with fragments.

The results of our investigations are coordinated with the data of the genetic investigation on the rodents in the zone of The Eastern Urals radioactive trace as well as on the cattle from the firms situated in the river basin of Techa [1, 5].

The investigations carried out on different territories with different concentration of ^{222}Rn showed the different level of chromosome aberrations among animals. These investigations confirm the negative effect of influence of ^{222}Rn on the genetic apparatus of the animals. Thus the discovery of the territories with anomalous concentrations of Rn must become the important link in the system of the agroecological monitoring of the agricultural enterprises and the forecasting of the state of health of the productive animals in these zones.

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Investigation of the relationship of radio-induced apoptosis of *Drosophila* larvae nervous system and the ageing of imago nervous system

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Abstract. The analysis of age-dependent change of nervous system activity of *Drosophila melanogaster* imago (with use of the test on negative geotaxis) at wild type strain *Berlin* and strains with mutations of apoptosis genes апоптоза *reaper*, *grim*, *hid*, th^1, th^4, $Dcp\text{-}1^{02132}$, $Dcp\text{-}1^{k05606}$ and $dArk^{k11502}$ is carried out. It was used *Drosophila* individuals, developing in conditions of a chronic low dose irradiation. Besides was studied an apoptosis level in nervous ganglion of third instar larvae after irradiation and in the control with use of hardware-software complex DiaMorph Cito (Moscow). The exposition doze was 0.17 sGy/h. The absorbed doze for one generation (from an embrio stage up to imago start, 10-12 days) corresponded 60 sGy.

It was shown, that the irradiation of strains with the increased apoptosis sensitivity results in elevated nervous - muscular activity of imago during all experiment periods. At th^1 strain increase of activity in comparison with the control in the first week has made 41%, and in two subsequent - about 80%. Last week authentic increase did not observe. At th^4 strain statistically significant increase of activity in comparison with the control observed in the first week of experiment (18%), in the second (67%) and the fourth (88%).

The irradiation of a wild type strain *Berlin* has caused authentic increase of activity in comparison with the control for the thirteenth day (on 17%) and for the twentieth day (37%).

At the same time, at strains with the lowered sensitivity to apoptosis: at strain which individuals are heterozygous in deletion of proapoptosis genes *rpr*, *grim* and *hid*; at strains with dysfunction of proapoptosis gene *dArk*; at strains with dysfunction of caspase *Dcp-1* ($Dcp\text{-}1^{02132}$ and $Dcp\text{-}1^{k05606}$ *allels*) authentic changes of nervous - muscular activity it was not shown. Exception has made action of an irradiation on strain with *rpr* deletion, at which activity only in the first week after influence increased on 39%.

1. INTRODUCTION

The analysis of the modern literature shows, that *Drosophila melanogaster* is convenient object for research of apoptosis role in various natural and induced processes in an organism, both due to the high level of scrutiny, and due to high conservatism of apoptosis in evolution [1]. However attempts to carry out complex research of the role of apoptosis genes in life span and ageing regulation with use of this classical modelling object till now were not undertaken. Postmitotic condition of somatic imago tissues interferes with regeneration of cellular populations and evidently should show a prospective role of apoptosis in ageing on this laboratory object [2]. Researches on experimental ageing and life expectancy differ duration, therefore *Drosophila*, having short life cycle (2 weeks) and small life span (about 80 days) is the most convenient object for similar researches. As complex experiments with a role of apoptosis genes in ageing it was not carried out, therefore results will represent the high fundamental interest.

Researches of radioinduced life span alteration of *Drosophila* which is carried out in our laboratory in 1996-2001 years, have revealed interrelation between mutations of apoptosis genes (*reaper*, *th*, *Dcp-1*) with low doses ionizing irradiation and speed of ageing.

In the given work investigated the apoptosis level in nervous ganglion of the 3d instar larvae and age changes of nervous system activity under the test on negative geotaxis at laboratory strains of *Drosophila melanogaster* with mutations of apoptosis genes *reaper*, *grim*, *hid*, *th* and *dArk* after a chronic irradiation in low dozes (60 cGy for generation) on preimago stages.

2. MATERIALS AND METHODS

The following *Drosophila* laboratory strains were used:

(1) wild type strain *Berlin*;

(2) strains with increased sensitivity to apoptosis:

— strain 618 bears defect in a apoptosis inhibitor gene diap-1/th and has a genotype th1/th1;

— strain 5053 is characterized by loss of function of th, that results in reduction of ability to apoptosis induction and has a genotype $th^4/TM6C, Cu^1\ Sb^1\ Ca^1$;

(3) strains with decreased sensitivity to apoptosis:

— strain 1576 is characterized by loss of site H99 on the third chromosome, bearing proapoptotic genes reaper, grim and hid and has genotype Df(3L)H99,kni(ri1)p[p]/TM3,Sb[1]; this defect results in sharp reduction of the apoptosis induction ability;

— strain 11041 shows reduction of an apoptosis level because of loss the activity of proapoptotic gene dArk (dApaf-1); has a genotype y1 w67c23; P{w+mC=lacW}l(2)k11502k11502/CyO;

— 11179 differs by broken expression of gene Dcp-1 coding effector caspase and has a genotype cn1 P{ry+t7.2=PZ}Dcp-102132/CyO; ry506;

— 10390 bears the same changes in apoptosis regulation, as 11179 and has a genotype y1 w67c23; P{w+mC=lacW}Dcp-1k05606/CyO.

Assuming, that differences in life span after experimental influence is caused by distinctions in an nervous cells apoptosis induction, in the present work the level of apoptosis after an irradiation in wild type strain with use of luminescent microscopy of nervous ganglion after acridin orange staining is investigated. For analysis of results used hardware-software complex "DiaMorth" (Russia).

Age changes of nervous - muscular activity is applied by gerontologists as a parameter of speed of ageing (so-called behavioural ageing) on a level with the life span [3]. Its reduction in advanced ages is evidence of direct correlation of the given parameter with ageing. There is a set of versions of tests for Drosophila activity, successfully used by gerontologists: age dynamics of spontaneous locomotion activity, patterns of moving, the climbing assay, the maximal locomotion activity [4]. Delay of activity decrease with age in comparison with the control testifies to reduction of ageing rates. At the same time the acceleration of activity loss is evidence of increase in ageing speed.

With the purpose to study the ageing speed of strains under investigation after irradiation and in the control estimated age - dependent dynamics of nervous - muscular activity of imago, having applied the test on negative geotaxis [4]. Each variant included about 50 males at the beginning of the experiment. We used vertically located glass tube (50 sm length and 1 sm diameter).

The statistical significance of differences between irradiated samples and control was estimated with use independent t-test.

3. RESULTS

The accelerated ageing of nervous system, resulting in reduction of life span, should correlate with decrease of activity of nervous system. Using the test on geotaxis, we have shown correlation between differences of age dynamics of nervous system activity after irradiation and genotypes of the strains with apoptosis mutations.

It was shown, that the irradiation of strains with the increased apoptosis sensitivity results in elevated nervous - muscular activity of imago during all experiment periods (Figure 1). At *618* strain increase of activity in comparison with the control in the first week has made 41%, and in two subsequent - about 80%. Last week authentic increase did not observe. At *5053* strain statistically significant increase of activity in comparison with the control observed in the first week of experiment (18%), in the second (67%) and the fourth (88%).

The irradiation of a wild type strain *Berlin* has caused authentic increase of activity in comparison with the control for the thirteenth day (on 17**%**) and for the twentieth day (37**%**) (Figure 2).

The irradiation of strains with the lowered sensitivity to apoptosis induction has not resulted in statistically significant effect (Figures 3 and 4). Exception has made action of an irradiation on a line *1576*, at which activity only in the first week of influence increased on 39**%** (Figure 3).

It is important to note, that strains *618* and *5053* bear defect of the same gene *th* (allels th^1 and th^4 accordingly) and are characterized by the raised apoptosis induction. Probably it is the reason of their similar reaction in reply to an irradiation (substantial growth of nervous - muscular activity) during all experiment that the most expressed since the second week when processes of natural ageing presumably should become more active.

In spite of the fact that in wild type strain *Berlin* the authentic increase of the nervous - muscular activity also is marked, the given effect is less expressed, than at strains with the increased apoptosis sensitivity.

However, at strains with the lowered apoptosis induction *1576* (individuals is heterozigous on deletion of proapoptosis genes *rpr*, *grim* and *hid*), *11041* (loss of function of proapoptotic gene *dArk*), $Dcp\text{-}1^{02132}$ and $Dcp\text{-}1^{k05606}$ (caspase *Dcp-1* dysfunction) authentic changes of nervous - muscular activity was not revealed.

4. DISCUSSION

As we made new approach to studying a role of apoptosis in ageing - influence on preimago stages, we observed the delayed role of apoptosis induction on ageing speed. Basing on above stated results, and also on the fact of essential increases of an apoptosis level in the larvae nervous system after irradiation (Figure 5), we put forward the following assumption. It is possible, that any tissues include cells with diverse sensitivity to damage. One cells have more powerful reparation system and antioxidant protection, whereas others - much weaker (for example, owing to somatic mutations of the appropriate genes). It is probable, that cells with the weakened protection will accumulate damages and to be exposed to ageing with the greater speed, than steady cells. Plenty of such potentially dangerous cells in a tissue will determine the premature or accelerated course of age - dependent degenerative and atrophic changes. However in tissues with the raised sensitivity to apoptosis induction (strains *618* and *5053*) irradiation on preimago stages will result in elimination of cells with the weakened protection that delayed ageing changes at imago. At wild type strain *Berlin* we also observed reduction of ageing speed after irradiation, however considerably less expressed, than at strains with the raised apoptosis sensitivity. At the same time, at strains with the lowered sensitivity to apoptosis: at strain which individuals are heterozygous in deletion of proapoptosis genes *rpr*, *grim* and *hid*; at strains with dysfunction of proapoptosis gene *dArk*; at strains with dysfunction of caspase *Dcp-1* ($Dcp\text{-}1^{02132}$ and $Dcp\text{-}1^{k05606}$ *allels*) authentic changes of nervous - muscular activity it was not shown. That is it is possible to assume, that at apoptosis tolerant strains the offered mechanism does not function.

Thus, the received results possibly testify, that apoptosis induced on preimago stages can act in a role of the original mechanism of antiageing, eliminating potentially dangerous cells with the weakened system of a reparation or antioxidant protection.

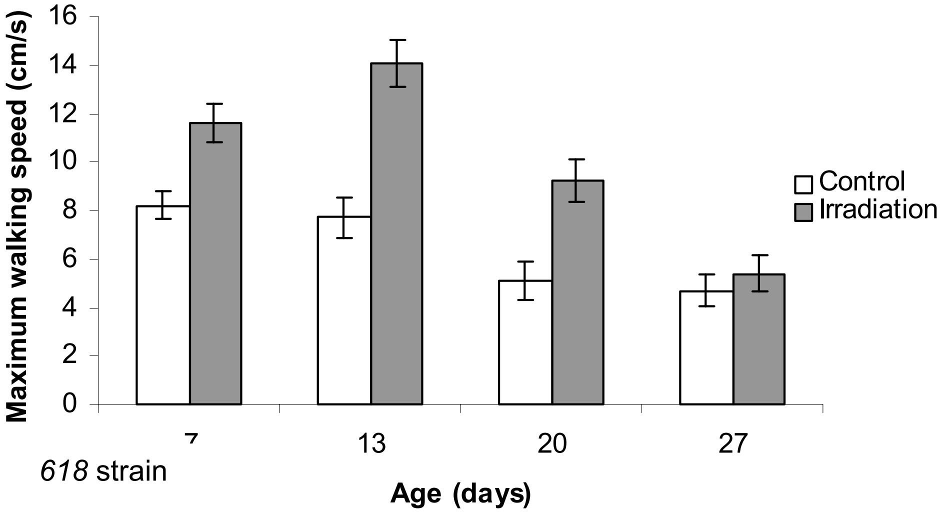
In connection with the received facts arise three problems, which are necessary for solving:

(1) Whether the marked effects are observed at irradiation in others (lower or higher) dozes or at treatment with chemical apoptosis inducers?

(2) Whether they are shown in others, not postmitotic, renewed tissues?

(3) Whether they are possible at not postmitotic organisms (for example, mammals)?

The described facts induce on interesting reflections about ways of therapeutic antiageing. Methods of antiageing existing nowadays are based on maintenance of stability and prolongation of weakened cells life, however indefinitely to stabilize the broken mechanism it is impossible. At the same time, the cells predisposed to damage accumulation, can not only have accelerated senescence, resulting to atrophic changes in tissues, but also can be blasttransformed. Being based on discussion of our data, we assume other two approaches to antiageing problem: the most simple and dangerous way of therapeutic antiageing - an induction of self-liquidation of the weakened cells at stages of organism development, previous to activation of ageing processes; the second, more reliable and difficult way - selective support of functioning of cells with powerful reparation and antioxidant system, their activation of proliferative potential (if it is not cells from postreplicative tissues and organisms) and suppression of proliferation potential of the weakened cells.



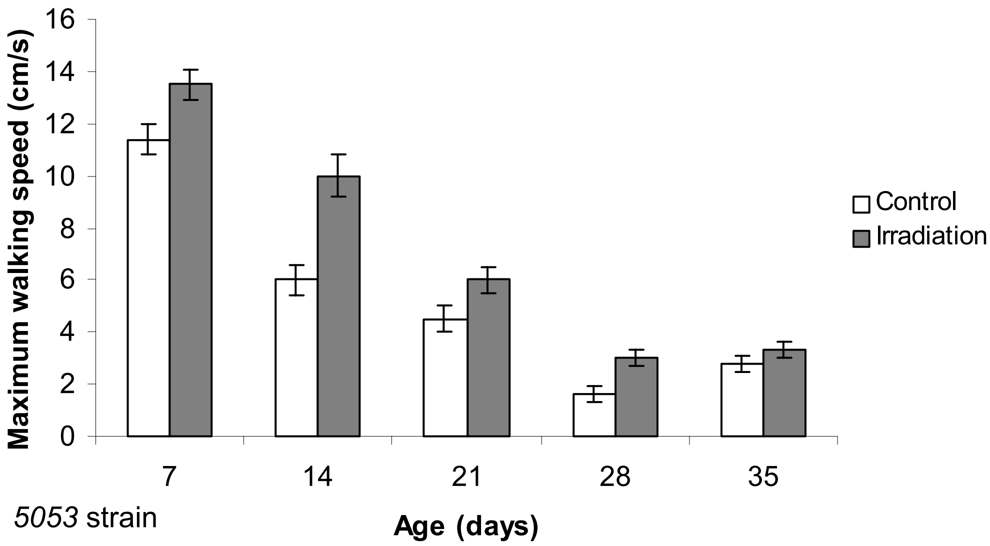


*FIG. 1. Age-dependent dynamics of nervou- muscular activity of imago after irradiation of strains 618 and 5053 with the increased apoptosis sensitivity. * p<0.05, ** p<0.01.*

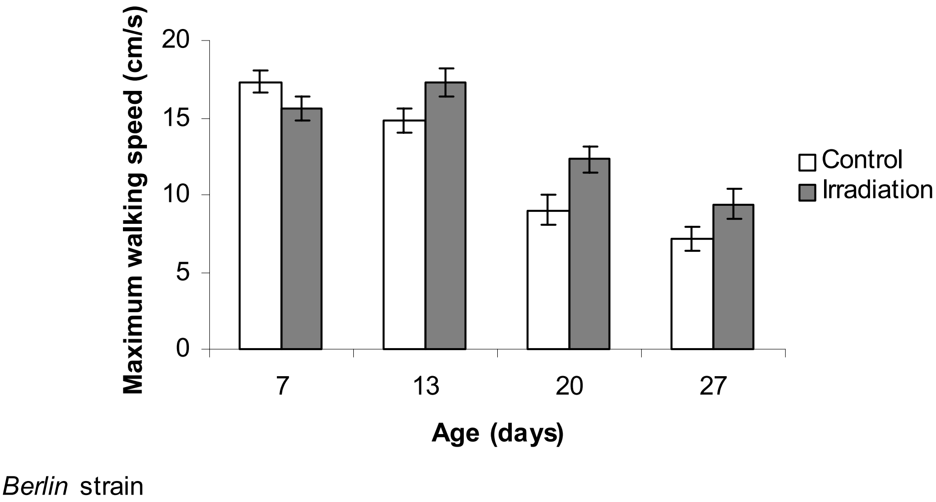


*FIG. 2. Age-dependent dynamics of nervous-muscular activity of imago after irradiation of wild type strain Berlin. * p<0.05.*

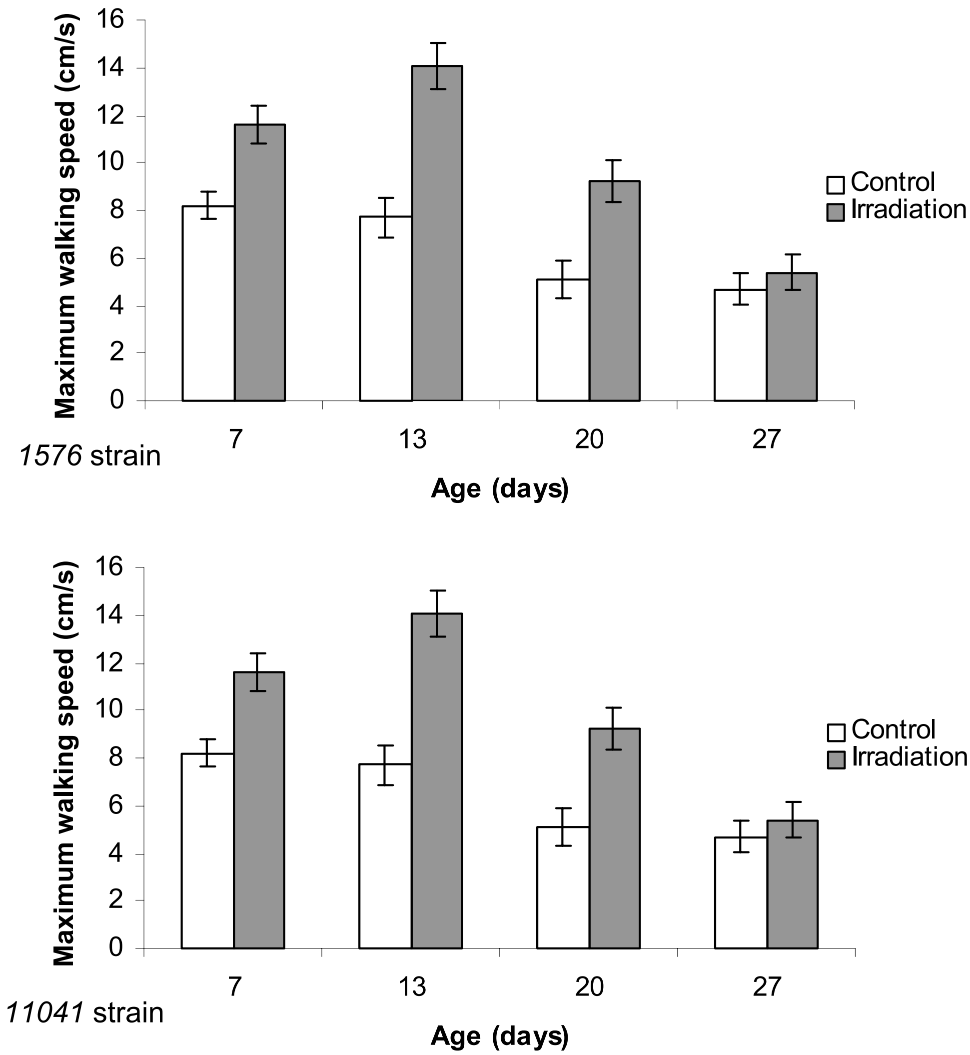


*FIG. 3. Age-dependent dynamics of nervous-muscular activity of imago after irradiation of strains 1576 and 11041 with the decreased apoptosis sensitivity. ** p<0.01.*

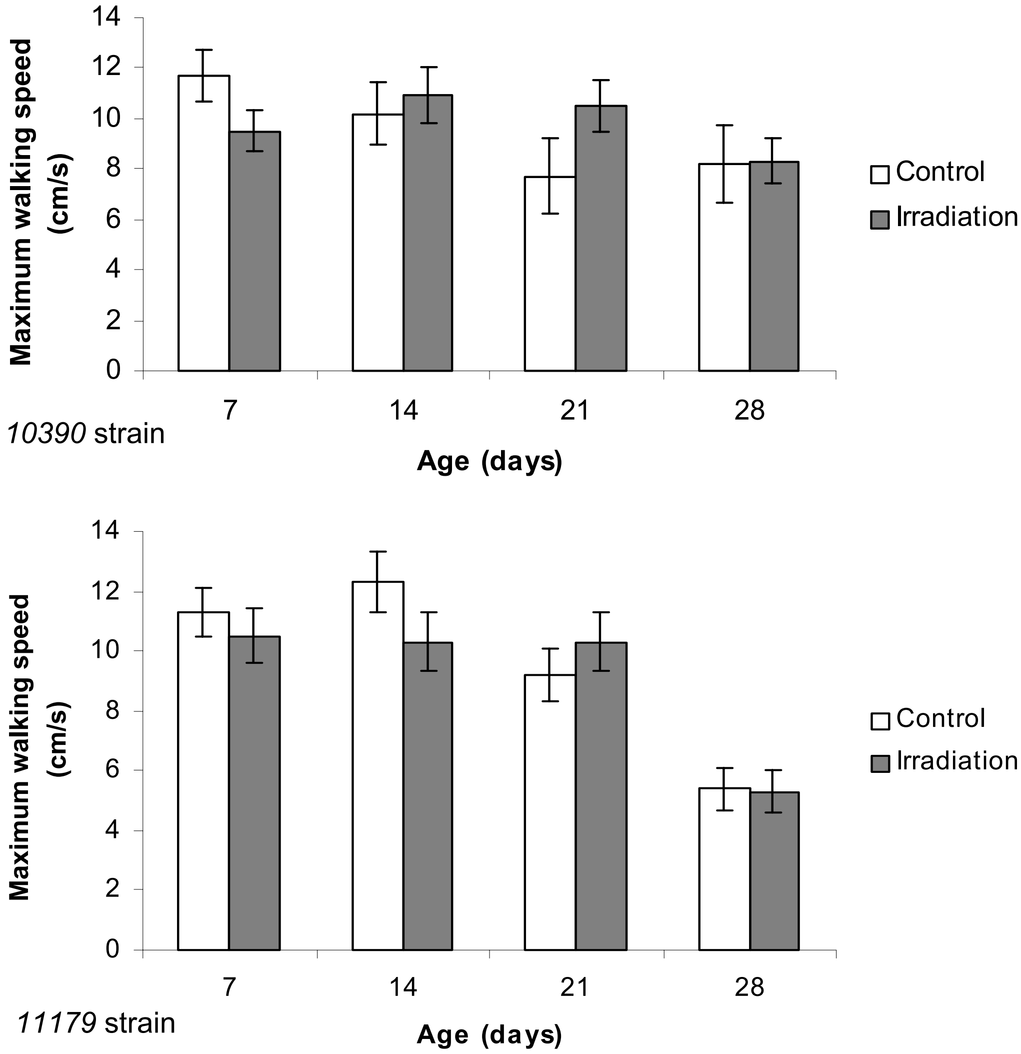


FIG. 4. Age-dependent dynamics of nervous-muscular activity of imago after irradiation of strains 10390 and 11179 with the decreased apoptosis sensitivity.

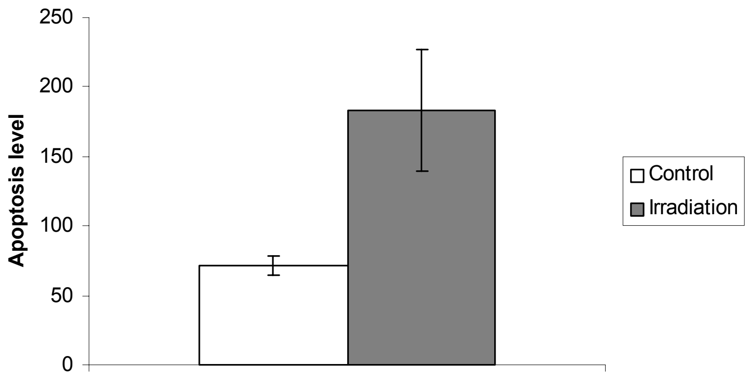


FIG. 5. Apoptosis level in nervous ganglion of Drosophila melanogaster third instar larvae at wild type strain Berlin in the control and after gamma-radiation (60 cGy for generation). The apoptosis level was estimated by quantity of points (pixels) with a yellow - red luminescence after acridin orange staining for each field of a video camera review. Difference in comparison with the control is statistically significant (p<0.05).

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Level of contamination and biological effects of natural heavy radionuclides in the ecosystems of Russia east region

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Abstract. A radioecological investigation of technogenic landscapes of Aldan Upland (east region of Russia) showed that the main factor responsible for their contamination with uranium and radium is radionuclide dispersal by air with products of rock weathering. Coefficients of biological absorption by plants decrease with an increase in contamination level. Radiation load is mainly determined by background gamma-radiation. The contribution of internal irradiation accounted for by incorporated radionuclides does not exceed 16%. Manchurian alder seeds (*Dushekia fruticosa* Rupr.) produced under conditions of contamination are more viable than those from the control plot. The seed generations of plants growing under condition of chronic irradiation are highly radioresistant.

1. INTRODUCTION

Prospecting and exploitation of uranium ore deposits can result in the formation of contaminated zones with an increased content of natural radionuclides. Such zones have been formed around numerous dumps of rocks enriched with uranium and products of its decay, which are found in mountain-taiga landscapes of east region of Russia within the Aldan Upland. The living components of ecosystems are exposed to chronic irradiation at small doses. The accumulation of natural heavy radionuclides (NHRN) by plants depends on their biological features and on the quota of movable forms of elements in the soils. Consequences of the effects of small doses are still debated. In this paper we submit the results of a complex radioecological investigation of technogenic landscapes, which allowed us to reveal specific peculiarities of ^{226}Ra and ^{238}U distribution in soil and plants and estimate their biological effect on plants.

2. MATERIAL AND METHODS

Studies were performed in the central Aldan Upland at elevations of 700 to 800 m above sea level, in the middle taiga zone. Uranium deposits located in this area are of the hydrothermal hydrosomatic type [1]. Geological prospecting for uranium ore deposits, which continued for 30 years, resulted in significant technogenic disturbances at numerous sites located mainly in mountain canyons drained by small rives. The slopes are covered with mixed forests consisting *Larix sibirica* L., *Picea obovata* Ledeb., *Pinus pumila* Regel., *Dushekia fruticosa* Rupr. *Betula lanata* Regel. *Chosenia macrolepis* Turcz. The slope of a hill located no less than 3 km away from disturbed sites was chosen as a control area. Soil sections were cut within and beyond contaminated zones and soil samples were taken from the major horizons. Plant samples were taken in two or three replications in the immediate vicinity of soil sections.

The content of ^{226}Ra was determined using multichannel gamma-analyzer with scintillation detector (measurement error below 30%). ^{238}U was determined photocolometrically using the reagent arsenazo-III (error 20%). Uranium and radium in different physicochemical forms were extracted out of soil by conventional methods. Background gamma- and beta-irradiation was measured with SRP-68-01 radiometer and PSP-101M “Poisk-Pripyat”. In calculating radiation loads on biota we distinguished contributions of internal and external irradiation and took in to account specific features of alpha- and beta-emitters.

The biological effect of enhanced background radiation was estimated using the seeds of *Dushekia fruticosa* growing at sites with different levels of gamma-radiation: 20-40, 100-150, 200-300, 400-500, 600-700, 800-900 and 100-1100 μR/h. At each site, seeds were collected from 5-12 plants. This allowed us to evaluate both average viability of the progeny and individual variation of several parameters of growth. Chromosome aberrations in root meristem cells were analyzed in preparations stained with acetoorcein (approximately 500 anaphases in each variant). The level of plant adaptation to radiation depending on its intensity during seed formation was studied using additional provocative gamma-irradiation (apparatus with ^{60}Co emitter, dose rate 15.7 R/s).

3. RESULTS AND DISCUSSION

In soil of control areas ^{226}Ra content varied from 4.9 to 5.2 × 10^{-7} mg/kg, and ^{238}U content, from 03. to 1.3 mg/kg. In the contaminated zone, ^{226}Ra and ^{238}U contents in the forest litter and the upper soil profile exceeded those concentrations by approximately one and two orders of magnitude respectively. At depths more than 30 cm, contents of these radionuclides in disturbed and control areas were equal. It is means that uranium and radium enter the soil mainly with dispersed products of rock weathering carried by the air from dumps.

Concentrations of ^{226}Ra and ^{238}U in the woody and herbaceous plants growing in the working areas exceed the control level by factors of 15-20 and 4-7 respectively (Table 1).

However, coefficients of biological absorption (CBA), calculated as ratio of radionuclide concentrations in plant ash and the substrate proved to be lower in the contaminated area than beyond them. Special study showed that radionuclides at concentrations equal to their clarke values are highly mobile and accessible for plants. Directly assessing the proportions of exchangeable and acid-soluble forms, it was found that the concentration of forms accessible for plants increases with distance from the dump [2].

It was important to estimate the roles of exposure to internal radiation (by incorporated radionuclides) and external radiation with regard to gamma- and beta –components separately. External gamma-radiation was measured using an SRP-68-101 radiometer. Contribution of external beta radiation was calculated by the formula $D=8.10\ E\ qg(h)$. E is average energy of beta-irradiation (1.3 MeV); q is density of beta particles; h is distance from the soil surface (100 cm); $g(h)$ is function reflecting the absorption of beta-radiation by the air [3].

Internal radiation dose was calculated by the formula $D=51.2\ E\ C\ Q$. E is mean energy of particles per decay, MeV; C is isotope concentration in tissue; Q is a coefficient of quality, which is 10 for alpha and 1 for beta particles [4]. The data shown that gamma-radiation accounted for 71-86.4% of total dose, the contribution of external beta-radiation was approximately 9.5% (Table 2). The contribution of incorporated ^{238}U, ^{226}Ra and ^{232}Th to the total radiation load varied from 0.01 to 16.2%. This proportion decreased with an increase in the level of contamination. Below we will only refer to the power of background gamma-radiation, as its effect is the greatest.

TABLE 1. CONTENTS OF ^{226}RA AND ^{238}U IN PLANT ASH

Species	Part of plant	Content, mg/kg		CBA	
		^{226}Ra, n × 10^{-7}	^{238}U	^{226}Ra	^{238}U
Control area					
Dushekia fruticosa	Leaves	20±4	0.9±0.2	4.5	0.7
	Branches	36±3	1.4±0.3	8.1	1.1
Herbaceous plant	Above ground parts	21±5	1.0±0.5	4.7	0.8
Technogenic zone					
Dushekia fruticosa	Leaves	429±52	5.0±0.2	0.4	0.05
	Branches	700±15	8.2±2.1	0.7	0.09
Herbaceous plant	Above ground parts	310±27	4.3±0.3	0.3	0.04

TABLE 2. RADIATION LOAD ON MANCHURIAN ALDER GROWING IN DIFFERENT SITES (100 CM ABOVE THE SOIL SURFACE)

Total dose, $n \times 10^{-2}$, μSv/h	Internal radiation, $n \times 10^{-2}$,μSv/h			External radiation, $n \times 10^{-2}$,μSv/h	
	^{238}U	^{226}Ra	^{232}Th	gamma	beta
21 (100%)	0.6 (2.9%)	3.4 (16.2%)	0.08 (0.4%)	15 (71%)	2 (9.5%)
1100 (100%)	12 (1.1%)	35 (3.1%)	0.08 (0.007%)	950 (86.4%)	103 (9.4%)

TABLE 3. VIABILITY OF *DUSHEKIA FRUTICOSA* SEEDS IN DEPENDENCE ON THE LEVEL OF BACKGROUND RADIATION IN THE AREA OF GROWTH

Gamma-background, $n \times 10^{-2}$, μSv/h	Germination energy, %	Survival, %	Leaf formation, %	Chromosome aberrations, %
20-40	19.6±4.2	38.2±5.3	23.0±3.7	2.5±1.7
100-150	19.0±4.1	31.8±3.6	19.0±2.8	7.5±2.1
200-300	32.3±2.3	53.7±2.7	33.7±2.4	3.6±2.1
400-500	32.0±3.8	53.4±2.9	32.2±3.1	3.3±2.6
600-700	30.6±4.0	55.8±4.3	41.3±3.2	-
800-900	41.5±3.6	58.3±3.5	37.0±4.8	4.9±2.3
1000-1100	37.1±4.6	52.5±3.5	43.7±4.7	2.3±1.3

The biological effect of enhanced radiation was studied using an example *Dushekia fruticosa*. It is the first species spontaneously occupying disturbed areas. To characterize seed progeny, we used a complex of morphological and cytological criteria (Table.3). The results showed that seeds collected from the control areas had lower germination energy and survival. This parameter did not correlate with seed weight. Analysis of variance confirmed significance of differences between seed progenies collected from different plants (5-12 from each cenopopulation) (F = 38.34 at $F_{0.01}$ = 6.98) and showed that the effect of radiation environment on the viability was also significant (F = 5.80 at $F_{0.01}$ = 4.82).

The number of plants with true leaves emerging one month after germination is the principal criterion characterizing the onset of apical meristem functioning in seedlings. The process of leaf formation developed more successfully in samples from contaminated area. And it was confirmed statistically (F = 29.44 at $F_{0.01}$ = 4.82).

The analysis of chromosome aberrations in root meristem is narrow and includes chromosome fragments. On the whole, no significant effect of radiation on the frequency of aberration in the seed progeny was revealed (F = 0.70 at $F_{0.05}$ = 2.4). We can establish only a tendency to the rise of chromosome damages. By all criteria, the linear dependence of the quality of seed progeny on background radiation level in the areas of maternal plants growth proved to be absent. This situation is typical for the effects of small radiation doses on living organisms.

Thus the results of our studies demonstrated that Manchurian alder seed formed under conditions of enhanced background radiation are more viable. We can conclude that chronic irradiation of seed progeny within maternal plants has a stimulating effect.

In the paper [5] the authors described the effects of radiation on vegetating Manchurian alder plants from the areas in which we collected seeds. They found that peroxidaze does not play any significant role in the system of antioxidant protection, and the main agents are low-molecular antioxidants. In areas with enhanced background radiation their content in leaves increased from 59 to 170 µg-equiv./g, and the protein content per gram leaf tissue also increased.

The data in Table 3 allow us to understand in more detail the enhanced viability of seeds formed under conditions of NHRN contamination.

It was interesting to assess the level of adaptation to radiation exposure in the seed progeny and its dependence on conditions of maternal growth and the general viability of seeds (Figure 1). The results indicated that the seeds collected in areas strongly contaminated with NHRN were more viable and radioresistant than seeds from the background area. Analysis of variance also confirmed that conditions of seed formation have a statistically significant effect on radioresistance (F = 8.18 at $F_{0.01}$ = 4.20). This may be evidence that these seeds have some specific physiological and biochemical features resulting from their preadaptation to radiation effect within maternal plants.

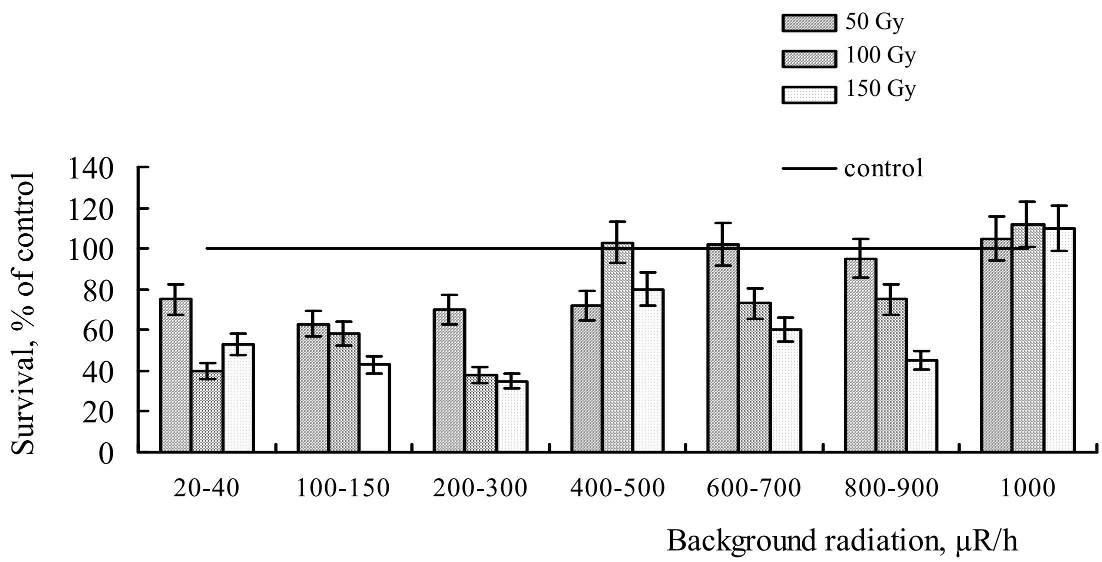

FIG. 1. Effect of provocative irradiation of Manchurian alder seeds at doses of 50, 100 and 150 Gy on the survival of seedlings depending on level of background radiation in areas of maternal plants growth.

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Effects of ionizing radiation to aquatic organisms
The EPIC Database

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Abstract. The paper presents overview of the data from the EPIC database "Radiation effects to biota", outlining the effects of radiation exposure in aquatic organisms. The EPIC database has been compiled as part of the current EC Project EPIC (Environmental Protection from Ionizing Contaminants in the Arctic). The EPIC database is based on information from publications in Russian (Russian/(Former Soviet Union) data). The effects are grouped by several key endpoints: morbidity, reproduction, mortality/life shortening, cytogenetic effects, stimulation, and adaptation. In general, data are focused on the effects in aquatic organisms at relatively low doses of chronic radiation exposure. A preliminary dose-effects relationships for aquatic organisms has been collected.

1. INTRODUCTION

The EPIC database "Radiation effects on aquatic biota" is compiled as part of the current EC Project EPIC (Environmental Protection from Ionizing Contaminants in the Arctic). The EPIC database is based on information from publications in Russian (Russian/(Former Soviet Union) data). The database is focused on the effects of chronic/lifetime radiation exposure; this information is of great importance for the purpose of establishing the permissible dose limits to biota. The EPIC database is to be completed by the end of 2003 [1].

2. DESCRIPTION OF THE EPIC DATABASE

The EPIC database is constructed in form of electronic tables Microsoft Excel, accompanied by extended text abstracts (Microsoft Word files with more detail description of each effect), which can be accessed using a hyperlink option. A special format of the database has been developed to provide a possibility to analyze the 'dose-effects' relationships for different types of biota, and impact scenarios. The information in the database is arranged in such a way, that one item of effect (one record in the base) corresponds to one case of exposure. The supplemental extended abstracts are prepared also in a unified format, each abstract describes one experiment or field study [2, 3]. The effects are grouped by several key endpoints: morbidity, reproduction, mortality/life shortening, cytogenetic effects, stimulation, and adaptation.

Estimations of dose rates and absorbed doses to aquatic organisms, presented in the database, follow in general the original publications, where data on radiation exposure of organisms were obtained from direct measurements or dose calculations. If the information on radiation exposure was not provided in the original publication, reconstructions of doses were made from data on radionuclide concentrations in aquatic organism and its environment, using appropriate dosimetric models [4–6].

At present, the EPIC database on the radiation effects to aquatic organisms (EXCEL table "EPIC_AQUATIC") includes 515 records from 121 publications. The effects on the following aquatic organisms are represented in the database:

(a) Fish and fish eggs – 332 records;

(b) Molluscs and other benthic invertebrates – 112 records;

(c) Zooplankton and microplankton – 71 records.

Most of records in the aquatic database refer to the laboratory experiments – 74%; data from the Chernobyl zone comprise about 11% of records; data from the Kyshtym contaminated area represent about 15% of records. Representation (in %) of different types of effects (umbrella endpoints) in the database "Radiation effects to aquatic organisms" is the following:

(a) Effects on morbidity – 5% of records;

(b) Effects on reproduction – 35%;

(c) Mortality/life shortening - 17%;

(d) Cytogenetic effects – 5%;

(e) Others (stimulation, adaptation, etc.) – 7%;

(f) No effects on exposure – 31%.

Among aquatic organisms the most extensive studies were carried out on fish and fish eggs. The usual organisms studied in the laboratory experiments were small aquarium fish or young fish of different species, which could survive in laboratory conditions. For a long time fish eggs were the most favorite objects of investigation, because the development of fish embryos could be easily observed. However, fish eggs were found to be sensitive to any defects in artificial incubation (oxygen and temperature conditions, etc.), which could result in misinterpretation of radiation effects in the early publications. The results of modern experiments with fish eggs are more reliable, because the technique of incubation was considerably improved. The long-term observations of radiobiological effects on fish have being carried out in the most contaminated lakes of the Kyshtym radioactive trace (Lakes Berdenish, Uruskul and others), and in the highly contaminated cooling pond of the Chernobyl NPP.

3. EFFECTS ON MORBIDITY OF AQUATIC ORGANISMS RESULING FROM RADIATION EXPOSURE

Effects of chronic radiation on morbidity of aquatic animals include the deterioration of various physiological and metabolic characteristics, which lead to a decline in health and well-being of the organisms. They represent early signs of the reduced fitness of organisms. The following specific effects were identified as effects on the morbidity of fish (effect code – "MB"): negative changes in blood composition; weakening and delay in immune response to bacterial/viral infection; weakening of the resistance to parasite infestation; negative changes in functioning of organs and tissues, etc.

The EPIC database contains the records, describing the radiobiological experiments on the effects of radioactivity on various parameters of immunity of aquatic organisms, as well as field observations of the health of fish and other organisms. In the natural conditions, the effects of radiation on fish immunity manifest themselves by an increased percentage of organisms in the population infested with parasites, and subjected to various infections; usually these effects are not linked with the radiation exposure.

4. RADIATION EFFECTS ON REPRODUCTIVE SUCCESS OF AQUATIC ORGANISMS

The following specific effects were identified as effects on the reproductive success of aquatic organisms (effect code – "R"): increased number of abnormalities and mortality in developing embryos of fish; morphological and functional abnormalities in gonads; sterility; teratogenic effects; and, decrease in the production of healthy progeny by irradiated organisms. The records in the EPIC database, demonstrating the effects of chronic radiation on reproduction of aquatic organisms refer mainly to lifetime experimental studies and to long-term observations of fish in natural water bodies highly contaminated with radionuclides. Numerous experiments on effects of acute exposure on development of fish eggs are also included in the database.

5. EFFECTS OF CHRONIC RADIATION ON MORTALITY AND LIFE SHORTENING

At low doses, an increase in mortality cannot be observed directly, but mortality usually manifests itself in the form of a reduction in age-dependent survival. The effect of life shortening may be a cumulative result of effects on morbidity, as well as abnormalities in reproduction, and cytogenetic damages. In the EPIC database, the records on life shortening include the results of lifetime experiments with aquarium fish, also observations of age-structure of fish populations dwelling for many generations in highly contaminated water bodies.

6. CYTOGENETIC EFFECTS OF RADIATION

Cytogenetic effects are known to be sensitive indicators of radiation damage in living cells. The EPIC database includes records of cytogenetic studies for fish eggs, fish, molluscs, and other organisms. The cytogenetic effects were detected at dose rates above 10^{-5} Gy d^{-1}, and at all higher doses.

7. PRELIMINARY SCALE OF DOSE-EFFECTS RELATIONSHIPS

The results of chronic experiments and observations, compiled in the EPIC database, provide a possibility to develop a preliminary scale of dose-effects relationships for fish from northern and temperate climatic zones. The approximate threshold levels of chronic exposure above which specific types of effects can be detected, are the following:

(a) At dose rates below 10-5 Gy d^{-1} no effects, or weak stimulation in aquatic organisms;

(b) Dose rates 0.5-1 mGy d^{-1} with accumulated doses above 0.05-0.2 Gy are threshold levels for appearance of first changes in fish blood, and early signs of decrease in immune system; at lower dose rates (less than 0.5 mGy d^{-1}) the organisms seemingly are able to adapt provisionally to radiation with gradual restoration of health parameters;

(c) Dose rates 2-5 mGy d^{-1} with accumulated doses above 1.5 Gy are threshold levels for appearance negative effects on the reproduction system of fish ;

(d) Dose rates 5-10 mGy d^{-1} of chronic lifetime exposure lead to life shortening of adult fish;

(e) At dose rates 10-2 – 10-1 Gy d^{-1} symptoms of radiation sickness are developed in fish, sterility of fish, considerable increase of abnormalities in fish eggs, increase of mortality;

(f) At dose rates 10-1 – 1 Gy d^{-1} high lethality of fish, decrease of lifetime of Daphnia, high mortality of fish eggs and pond snail’s eggs;

(g) Doses 5–10 Gy (acute exposure) results in high mortality of fish and fish eggs;

(h) 100–200 Gy (acute exposure) - mortality of some zooplankton species, decrease of biodiversity in zooplankton association;

(i) 200–500 Gy (acute exposure) - total mortality of zooplankton, mortality of some phytoplankton species; stimulation of bacterioplankton.

With the increase of radiation exposure, different groups of effects are summarized in organisms, i.e. several types of effects can be found in one organism. The evaluation of dose-effects relationships for aquatic organisms provides a basis for establishing permissible levels of radioactivity in aquatic environments, ensuring protection of even the most sensitive organisms and populations.

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Effects of ionizing radiation on terrestrial animals: *Dose-effects relationships*

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Abstract. The paper presents the general characteristics of the terrestrial part of the EPIC database “Radiation effects to biota”, outlining the effects of radiation exposure in terrestrial animals. The EPIC database has been compiled as part of the current EC Project EPIC (Environmental Protection from Ionizing Contaminants in the Arctic). The EPIC database is based on information from publications in Russian (Russian/(Former Soviet Union) data) on radiation effects for species inhabiting northern and temperate climatic zones. The effects are grouped by several key endpoints, which are important for well-being and survival of natural populations (morbidity, mortality/life shortening, etc.). Special attention is given to the effects in animals at relatively low doses of chronic radiation exposure. A preliminary dose-effects relationships for terrestrial animals has been developed.

1. INTRODUCTION

The EPIC database has been compiled as part of the current EC Project EPIC (Environmental Protection from Ionizing Contaminants in the Arctic), and will be finalized in 2003 [1]. The EPIC database is based on information from publications in Russian (Russian/(Former Soviet Union) data) on radiation effects for species inhabiting northern and temperate climatic zones.

The general database is subdivided into two parts [2]:

(a) Database on the radiation effects to terrestrial organisms with subdivisions, e.g. “terrestrial animals”, “soil fauna”, “plants”;

(b) Database on the radiation effects to aquatic organisms.

This paper outlines the characteristics of the terrestrial subdivision of the EPIC database, namely the collection of radiation effects to terrestrial animals.

2. DESCRIPTION OF THE EPIC DATABASE

The special format of the database has been developed to provide one a possibility of analyzing the “dose-effect” relationships by mapping the calculated dose onto a given effect. The EXCEL tables with the database “Radiation effects to biota” have a unified format, consisting of 12 vertical columns and horizontal rows. The following information is given in each record: type and name of organism; type of radiation impact; radionuclides with concentrations in organism and environment; dose rate and accumulated dose; radiobiological effect and its code in the base; reference; link to relevant extended abstract.

At present, the database on the radiation effects to terrestrial animals (EXCEL table “EPIC_TER_ANIMALS”) includes 428 records from 113 publications. Effects on the following types of terrestrial organisms are represented in the database:

(a) Mammals – 361 records;

(b) Birds – 20 records;

(c) Insects – 28 records;

(d) Amphibia and reptilia – 19 records.

The largest amount of data refers to the radiation effects from the Chernobyl accident (52% of records), 26% refers to the area contaminated as a result of the Kyshtym accident; 10% refers to the areas of high natural radioactivity in the Komi Republic in Russia; laboratory and other studies comprise 12% of the database "radiation effects to terrestrial animals". Proportions of different types of effects (umbrella endpoints) in the database "Radiation effects to terrestrial animals" are the following:

(a) Effects on morbidity – 13% of records;

(b) Effects on reproduction – 9%;

(c) Mortality/life shortening - 9%;

(d) Cytogenetic effects – 23%;

(e) Ecological effects – 7%;

(f) Others (stimulation, adaptation, etc.) – 2%;

(g) No effects on exposure – 37%.

There are several territories in Russia/FSU, which are characterized with high levels of radioactivivity. Among these territories are the following: territory contaminated in 1957 as a result of Kyshtym accident; Chernobyl zone, impacted by the Chernobyl accident in 1986; numerous local areas of high natural radioactivity were revealed in Komi Autonomic republic in Russia. Numerous radiobiological investigations have been made in these areas, these large territories constitute the unique grounds for studies of radiation effects on natural life, which is not separated from stresses and risks of natural habitats.

Among terrestrial animals, most extensive studies were performed on small mammals, such as mice, voles, and other mouse-like rodents. These species form numerous populations, they can be easily caught from wild populations. The lifespan of mice/voles is about 1-1.5 years, so the investigations can be carried out for many generations of organisms. Mice are settled animals, they do not accomplish any distant migrations from contaminated areas. Because of these features, mice are the favorite objects of radiobiological investigations in all areas with high levels of radioactivity. The radiobiological information of effects to mice is available for the Chernobyl 30-km zone, Kyshtym area, as well as for Komi radioactive Ra-U spots.

3. PRELIMINARY SCALE OF DOSE-EFFECTS RELATIONSHIPS

The records in the EPIC database collection represent the effects of chronic radiation exposure from very low doses above 10^{-5} – 10^{-4} Gy d^{-1} up to doses greater than 1 Gy d^{-1}. From the preliminary analysis of the dose-effects relationships for terrestrial animals, the following scale may be proposed, see Table 1.

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TABLE 1. THE RELATIONSHIPS BETWEEN THE DOSE RATES OF CHRONIC RADIATION EXPOSURE AND EFFECTS OF RADIATION ON TERRESTRIAL ANIMALS (BASED ON THE EPIC DATABASE "RADIATION EFFECTS TO TERRESTRIAL ANIMALS)

Dose rates of chronic radiation exposure	Radiation effects on terrestrial animals
$10^{-4} – 10^{-3}$ Gy d^{-1}	Some changes in blood (α,β exposure) After-effects on progeny born from exposed parents (doses to parents >1Gy) Increase in chromosome aberrations in cells
$10^{-3} – 10^{-2}$ Gy d^{-1}	Pathology in liver, kidney (radionuclide specific) Considerable decrease of reproduction potential (α+γ exposure), shortening of reproduction period Some mice species show compensatory increase of reproduction (reaction to decrease in population density) Some life shortening, also higher risk to be captured by predators Weakening of immune system, increase of infestation with parasites, increase of various infections (α,β,γ exposure) Changes in blood, chronic radiation disease (α,β,γ exposure) Cytogenetic effects, increase of embryonic losses
$10^{-2} – 10^{-1}$ Gy d^{-1}	Sterility, decrease of gonad's mass Strong infestation with parasites Osteosarcomes (Sr-90), anomalous teeth (mice, rats) Pathology in liver, kidney (radionuclide specific) Life shortening Changes in blood, chronic radiation disease (α,β,γ exposure) After-effects in progeny born from exposed parents Decrease of populations, replacement of some populations by those species, which received lower doses, or by more radioresistant species Cytogenetic effects, increase of embryonic losses
10^{-1} – 1 Gy d^{-1}	Acute radiation sickness Death of many organisms, decrease of populations
> 1 Gy d^{-1}	Acute radiation sickness In receiving lethal dose death within several days

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Development of resistance to pesticides in pests exposed to ionizing radiation

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Abstract. The development of resistance to pesticides in phytophages after ionizing irradiation was studied. Sensitivity of spring grain aphid (*Schizaphis gramina Rond*) and corn thrips (*Haplothrips tritici Kurd*.) collected from sites in the 30-km ChNPP zone to two groups of pesticides, though varying considerably, does not show any regularities in resistance changing depending on radioactive contamination density (33.9–3.3 MBq/m^2). By parameter such as 50% mortality, the most sensitive to Malathion was population of spring grain aphids sampled from the plots with minimum density of radionuclide contamination. Where the level of radioactive contamination was by factors 4 to 10 higher aphid individuals proved to be more resistant to the insecticide, with the resistance growing as the contamination level of experimental plots increased. As to another insecticide – Permethrin – the most sensitive on the contrary were insects sampled from the plot with the maximum contamination density. Results from a model experiment performed in laboratory conditions using spring grain aphids and red spider (*Tetranychus urticae Koch*.) irradiated at doses of 5, 25 and 50 and insect offsprings also provide evidence of different sensitivity of arthropods to insecticides (Pyrymyphos-methyl and Cypermethrin). The offsprings of irradiated aphids exhibited an enhanced resistance to Pyrymyphos-methyl. Particularly pronounce it was in offsprings of arthropods treated at 25 Gy: the population became almost 7 times more tolerant in terms of 50% death indicator; as to the indicator of 95% death, differences in sensitivity were 40–fold. In this case ionizing radiation can be considered as a selection factor where the most vital individuals survive. It may be assumed that aphid populations subject to irradiation can become more heterogeneous in sensitivity to insecticides of different chemical groups and to a certain extent more tolerant than untreated individuals.

1. INTRODUCTION

The basis of an adaptive potential of biological objects to adverse effects is an adaptive polymorphism which is characteristic of populations of all biological species. The development of resistance in destructive arthropods to insecticides mainly occurrs under the influence of drugs used for protection of plants. Resistance of living organisms to adverse environmental factors is one of the most crucial aspects of their existence as an integral system. The development of resistance is a complex genetic process during which under the action of environmental factors in populations occurs a selection of individuals with altered physico-biological properties assisting in their survival. However, the development of resistance in phytophages may be influenced by other stress agents that contribute to the selection in a population of the most vital individuals, e.g., ionizing radiation [1, 2].

2. DESCRIPTION OF METHODS

Studies on the evaluation of resistance in farm pests to pesticides were carried out on populations of spring grain aphids (*Schizaphis gramina Rond*) and corn thrips (*Haplothrips tritici Kurd*.) sampled from plots in the 30-km ChNPP zone with varying contamination density (stationary plots 1, 2 and 3 ^{137}Cs, contamination density – 33.9 ± 0.07; 15.9 ± 0.09; 3.30 ± 0.01 MBq/m^2, respectively). Sensitivity of insect populations to pesticides was estimated using insecticides belonging to two different chemical groups: phosphorous-organic and synthetic pyrethroids. CR_{50} and CR_{95} were taken as criteria of pesticides effects, i. e. drug concentrations rate (CR) at which death of individuals in the study populations amounted to 50% and 95%. For determining CR_{50} and CR_{95} values, the Miller – Tyter method was employed.

TABLE 1. SENSITIVITY OF SPRING GRAIN APHIDS AND CORN THRIPS FROM THE 30-KM CHNPP ZONE TO INSECTICIDES

Insects	Plot	Malathion, 30%			Permethrin, 25%		
		CR_{50}	CR_{95}	β	CR_{50}	CR_{95}	β
Spring grain	1	5.7	34.6	2.28	3.5	13.8	3.16
aphids	2	4.8	75.8	1.56	4.2	25.1	2.40
	3	3.6	31.0	2.08	3.8	18.1	2.96
Corn thrips	1	5.0	72.4	1.64	3.5	39.8	1.68
	2	2.1	22.9	2.08	3.5	22.9	2.36
	3	2.5	13.0	2.52	5.0	28.8	2.28

Plot location and ^{137}Cs contamination density:

1 – near "Red forest" – 33.9 ± 0.07 MBq/m^2;

2 – in front of settlement Chistogalovka – 15.9 ± 0.09 MBq/m^2;

3 – behind settlement Chistogalovka – 3.3 ± 0.01 MBq/m^2.

β- slope of line of death rate of individuals;

$CR_{50,95}$ - the indicator value (* 10^{-4}).

3. SENSIVITY OF INSECTS TO SOME OF PESTICIDES

3.1. Enviromental studies

It has been found that by CR_{50} indicator the most sensitive to Malathion was the population of spring grain aphids sampled from plots with the minimum contamination density (plot 3). Aphid individuals sampled from plot 2 and 1 where the contamination level was markedly higher (4 – 10 times) proved to be more resistant to the insecticide used and the resistance grew with increase in level of contamination of the experimental plots (Table 1).

As for Permethrin, the most sensitive on the contrary proved to be insects sampled from plot 1 (near "Red forest"), with CR_{50} for all the experimental plots varying between 0.000346 and 0.000416%.

Attention is drawn to the fact that populations of thrips sampled from different plots of the abandoned ChNPP zone were quite dissimilar in age which is explained not only by the radiological situation but also by likely active and passive migration of insects within the 30 km zone. All this couldn't but tell on the values of phytophages resistance to insecticides.

3.2. Laboratory test

Results from the model laboratory experiment with two pest species (spring grain aphids and red spider - *Tetranychus urticae Koch*.) treated at doses of 5, 25 and 50 Gy and their offsprings indicate various sensitivity of insects to insecticides (Pyrymyphos-methyl and Cypermethrin). Aphid offsprings exposed to ionizing radiation showed enhanced resistance to Pyrymyphos-methyl (Table 2). It is especially pronounced for arthropode groups treated at 25 Gy: the population became 6.9 times more tolerant in terms of CK_{50}. As for CR_{95}, differences in sensitivity amounts to 41.5 times. The slone of mortality line (β) in the experiment is more gently inclined (β=1,28) that in the control (β=3,30).

A diagnostic concentration of Cypermethrin, 0,0004%, allowed to survive 19,6% individuals in the population treated at 5 Gy, 50% individuals in the group treated at 25 Gy and 30% treated at 50 Gy.

So, exposure of aphid population to ionizing radiation results in changes of aphid sensitivity to pesticides and resistance in offsprings of irradiated aphids grows significantly. It should be noted that sensitivity of red spider to pesticides under the influence of ionizing radiation has lesser differences compared with the resistance of untreated insects.

TABLE 2. TOXICITY OF PYRYMYPHOS-METHYL FOR APHID OFFSPRINGS EXPOSED TO RADIATION

Parameter	Dose, Gy			
	0	5	25	50
CR_{50} (% a.s. , $*10^{-4}$)	0.13	0.44	0.90	0.29
CR_{95} (% a.s. , $*10^{-4}$)	0.41	0.87	17.00	1.20
β	3.30	line of inflection	1.28	2.92
CR_{50}/ CK_{50} irradiated/non-irradiated	–	3.40	6.90	2.20
CK_{95}/ CK_{95} irradiated/non-irradiated	–	2.10	41.50	2.90

a.s.- active substances;
β- slope of line of death rate of individuals;
$CK_{50,95}$ - the indicator value.

A single treatment didn't have effect on the response of red spider to Pyrymyphos-methyl – contact and intestinal acaricide with fumigation and deep activity. Neither red spiders developed from irradiated eggs nor offsprings of irradiated females changed a normal sensitivity to this drug (Table 3). In no case was there a significant difference in sensitivity values between the experiment and control. When using Chlorethanol (group of chlorine – organic acaricides), no differences were noted in the sensitivity of treated and untreated red spiders to the drug.

So, an external irradiation of agricultural crop pests belonging to two arthropode classes (spring grain aphids and red spider) quite differently influenced the development of resistance to pesticides. Acute irradiation of red spiders did not result in changes of the resistance to insecticides in their offsprings [3]. At the same time, in the case with spring grain aphids, we observed an increase in insect resistance in the absent of pronounced dependence of this parameter on the dose – 5, 25 or 50 Gy.

Worth noting that mechanisms for development of response to acute and chronic irradiation at low doses have their own peculiarities. Therefore, for populations of insects – phytophages and spiders exposed to ionizing radiation (ChNPP abandoned zone) and not treated by pesticides during several generations, both decrease and increase in resistance to insecticides can be expected. To conclude on how much ionizing radiation influences the gene pool of populations and, in particular, the number of mutations responsible for resistance of insects to pesticides is only possible after studying a number of indicators describing the state of pests.

TABLE 3. TOXICITY OF PYRYMYPHOS-METHYL FOR RED SPIDERS EXPOSED TO RADIATION

Dose (Gy)	CR_{50} (% a.s.)	CR_{95} (% a.s.)	β	CR_{50}/CR_{95} irradiated/sensitive
F_1 from irradiated eggs				
5	0.000076	0.000400	2.14	0.7
25	0.000083	0.000260	3.26	0.7
50	0.000090	0.000325	2.91	0.8
F_1 from irradiated female				
5	0.000125	0.000375	3.41	1.1
25	0.000110	0.000343	3.32	1.0
50	0.000073	0.000245	3.11	0.7
Non-irradiated red spiders, sensitive				
0	0.00011	0.00043	2.76	1.0
Non-irradiated red spiders, resistanse				
0	0.039	1.87	1.40	354

a.s.- active substances;
β- slope of line of death rate of individuals;
$CK_{50,95}$ - the indicator value.

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Are there ecological effects of ionizing radiation?

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Abstract. Concerns of environmental effects of ionizing radiation has increased, and has also shifted from being centred on only humans, towards including also effects on ecosystems. Knowledge of radiation effects on wild populations and ecosystems is however limited and methods for assessing ecological risks of such effects are still in their infancy. For example, development of better methods for extrapolating effects on individual organisms to estimates of negative effects on populations would greatly improve risk assessments, since populations often constitute what we wish to protect. In this paper, possible ecological effects of ionizing radiation from radionuclides are discussed and evaluated in terms of how individual level effects are translated to risks at the level of the population and higher. We conclude that ecological effects of ionizing radiation at the level of the population and above can be caused by two different mechanistic routes: Firstly, the detrimental effects of radiation to individuals such as increased mortality, morbidity, and decreased fertility and fecundity can, depending on their magnitude have implications on population dynamics and thereby also potentially on community and ecosystem properties. Secondly, ionizing radiation may cause genetic damage, which can have effects at the population level by affecting the genetic structure, and genetic diversity of populations.

1. INTRODUCTION

Traditionally, radiological protection has focused on protecting humans with the assumptions that this would also protect other components of the ecosystem. Now the concern of radioactive releases has expanded from being focused on human health effects only to also include consequences of ionizing radiation on the environment, or ecosystems. The requirement for assessments of the environmental effects of radiation is increasing due to public concern for environmental protection issues, new legislation in several countries and scientific reasons. The ultimate aim of environmental protection programs is to protect natural populations and ecosystems. The dilemma is that as ecological relevance in systems increases, so does the complexity of the system, and the difficulty in measuring the response to a stressor. Which is why effect investigations most often focus on simpler systems, or on lower levels of organisation such as individual level endpoints, which are by far the most common toxicity data available today. The effects are then extrapolated from individuals to populations and higher organisational levels. This applies also to understanding of effects of ionizing radiation, knowledge of radiation effects on wild populations are limited, the effects reported are mainly on the individual level or lower. In this paper, possible ecological effects of ionizing radiation from radionuclides are discussed. The purpose is not to cover all biological effects of ionizing radiation, but to discuss possible ecological effects, or effects at higher level of organisation than at the individual level.

2. ARE THERE ECOLOGICAL EFFECTS OF IONIZING RADIATION?

2.1. Impact of individual effects on populations

Ionizing radiation does not seem to have any effects directly at the level of population, community or ecosystem. Many studies have however documented effects of radiation on the individual level or lower, and they have been consistent in concluding that the likely consequences of radiation are: dose-dependent increases in morbidity and mortality, decrease in fertility and fecundity, and increase in mutation rate [1]. The question is, are those effects ecologically relevant, are there any effects of ionizing radiation above the individual level; at the level of population or higher?

Increased individual mortality and morbidity, and decreased fertility and fecundity can in fact have implications on the population, depending on the magnitude of the effect. To estimate such individual level endpoints to effects at higher organisational levels, extrapolations are commonly used in environmental risk assessments. Extrapolation is based on the assumption that effects on populations are first manifested at the level of the individual, and from there emerge at the level of population. The utility of such individual level endpoints in understanding toxicant effects at the population level are however associated with a number of assumptions and uncertainties, and there are several complicating issues that need to be considered, some of which are discussed below.

2.1.1. Different life stages

Life stages of species are differently sensitive, and it is often assumed that the population will be protected if the most sensitive stage of the life history is protected, which usually is the juvenile stage. This is true in most cases [2]. However, the most sensitive life stages might not be the most important stage for maintaining the population viability. For example, in species that produce very large number of offspring like bivalves, contaminant effects on later life stages will actually be more important for the viability of the population. For such populations, 10% increase in mortality of juveniles may thus have little effects, whereas 10% increase on adult stages can have great impact. Tests performed on sensitive life stages, extrapolated to effects on population levels thus tend to provide an added measure of protection.

2.1.2. Different life cycles

Different life cycles of species are also important to consider when extrapolating individual effects to populations. Species have different reproductive strategies and life cycles, and may, therefore respond differently to the same degree of radiation effect on survival and reproductive capacity. Forbes et al. [3], considered the life cycles of the most widely used ecotoxicological test species, and estimated the proportional decline in population growth rate resulting from a decline in juvenile survival using a simple two-stage demographic model. The reduction in juvenile survival turned out to have very different effects on the population growth (λ) dependent on life cycle. 10% reduction in juvenile survival would result in 10% reduction in λ for a benthic invertebrate life cycle, 5% reduction in λ for a green alga life cycle, 2% reduction in λ for fish life cycle, and only 0,6% reduction in λ in daphnid life cycle. Also, a 5% reduction juvenile survival of benthic invertebrate life cycle would have the same effect on population growth rate as 80% reduction in juvenile survival of the daphnid life cycle [2-3]. This shows the importance of taking the life cycle of species into account when extrapolating effects from individuals to populations.

2.1.3. Different growth strategies

In very general terms population growth strategy can be described by so called k, or r strategy, but species can also fit in somewhere in between these two described strategies. Typical k type species (k = carrying capacity) show rather small fluctuation around its carrying capacity. Density dependent processes (such as mortality caused by crowding) is common. These are typically long-lived organisms that produce few offspring, but invest much energy in each one of them, so they have a good chance of surviving. Typical r type species are opportunistic populations that have a rapid, or even exponential growth rate (r) during favourable times, which is followed by a “crash” in the population number during environmental changes or stress. The cost of this strategy is that these organisms cannot afford much resources per offspring, so they have small seeds or eggs and less developed care for their young. Increased mortality of individuals due to radionuclide exposure could actually benefit the population by removal of individuals which increases the fitness for the remaining individuals – depending on the population growth strategy and growth regulating factors.

2.1.4. Density dependent factors

Population growth can be regulated both by density-dependent factors and density independent factors. For a full understanding on how pollutants affect populations, we need to know not only how the pollutant – in this case radionuclides affects the population, but also how density-dependence affects it. Density-dependent factors are affected by the population density; and vary in their effect with the density of the population. Example: a) A predator encounters and captures more prey as the density of the prey increases. b) Intraspecific competition, where individuals of the same species rely on the same limited resources. As the population density increases, the competition becomes more intense, and growth rate declines in proportion to the intensity of the competition. c) nutrient availability decreases because of increasing population density and results in lower birth rate. Increased mortality of individuals due to radionuclide exposure could thus actually benefit the population by removal of individuals which increases the fitness for the remaining individuals – depending on the population growth strategy and growth regulating factors.

2.1.5. Indirect effects

Direct effects of pollutants, such as effects on survival, growth, or reproduction are commonly considered in environmental risk assessment. However, species may also be affected indirectly through their interactions with other species that are directly affected by toxicity, or by *indirect effects*. The main ecological vehicles for indirect effects are interspecific interactions, particularly predation and competition, although other interactions such as mutualism, commensalism or parasitism may also be important (Preston, 2002). The nature of the indirect effect may however be difficult to predict, and differ from one community to another. An example is that effects of toxicants on predators or competing species may have beneficial effect for a particular species. However, reduction in a key prey resource may result in a predator species decline despite it being unaffected by the toxicant directly [4].

2.1.6. Key species

Key species are species that are important for the structure or function of the ecosystem, and therefore more ecologically important than other species. If they are affected by contaminants or even lost, it could have serious effects for the ecosystem. It is therefore very useful to recognise and protect key species, but the problem is that in most cases the identity of them is not known.

2.2. Effects of mutations on genetic diversity of populations

At low doses, the main concern of ionizing radiation is perhaps that it may cause genetic effects, by inducing damage to the DNA. DNA damage, if fixed, may lead to mutations. Mutations occur at the molecular level, but effects are emergent at the level of the population. Mutations in somatic cells can possibly be detrimental to the exposed individual, whereas mutational events in germ cells may be passed to the offspring, thereby affecting subsequent generations. Heritable mutations are capable to affect the genetic diversity of populations, since they can lead to increased or decreased genetic diversity, as well as changes in gene expressions that affect Darwinian fitness. Increases in mutation rate can increase genetic diversity of the population by producing new alleles or genotypes, but it can also result in decreased genetic diversity, since the mutations could reduce the viability and fertility of the individuals [5].

Most mutations are only slightly deleterious. Because their individual effects are very slight, they have the potential to affect larger portions of the population and persist for longer period of time. However, in the longer term, their effects can substantially lower the viability and fertility of the population [6]. Genetic diversity is considered positive, even crucial to the long term viability of populations. Without variation, a population lacks the raw material with which to adapt to changes in its environment, and will eventually disappear when the environment changes. During hard environmental conditions or stress, genetic diversity seems to play a greater role in fitness differences among organisms. Genetic variation has also shown positive effects on stress resistance and numerous fitness indices (growth, fecundity) [7]. A decrease in genetic diversity has been associated with decreased growth rates, diseases and reduced fertility, and may therefore have ecological consequences.

Investigations on genetic effects in wild populations due to environmental radiation can be found in the literature. Studies from the Chernobyl area have for instance reported increased mutation rates, increased assymetry in organisms, and increased genetic diversity in populations after the nuclear reactor accident in the area in 1986. All believed to be caused by DNA damage due to environmental radiation. In some cases the effects observed have even affected the mating status of the organism, such as where it is the secondary sexual characters that show assymetry, as is the case of the barn swallows *Hirundo rustica* (males with assymetric tail feathers breed later) [8], and the stag beetle *Lucanus cervus*, where more assymetric males were found unmated [9].This type of effect may also be considered of ecological significance.

3. CONCLUSIONS

Ionizing radiation does not seem to have any effects directly at the level of population, community or ecosystem, but numerous papers have reported effects on individual level and lower. These are mainly increases in morbidity and mortality, decrease in fertility and fecundity, and increase in mutation rate. Increased individual mortality and morbidity, and decreased fertility and fecundity, depending on the magnitude of the impact, is believed to have implications on population dynamics and thereby also potentially on communities and ecosystems. To estimate effects of these individual level endpoints on populations (or higher), extrapolations may be used, such as is commonly done in environmental risk assessment. There are however a number of assumptions and uncertainities associated with extrapolations. Increased mutation rate is a radiation effect that can on the other hand have ecological effects through other mechanisms, namely by affecting the genetic structure, and genetic diversity of populations.

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Hydrobionts of the Chernobyl NPP exclusion zone
Radioactive contamination, doses and effects

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Abstract. The territories of the Chernobyl NPP exclusion zone are characterised by significant heterogeneity of radionuclide contamination, which is reflected by the radioactive substance contents in aquatic ecosystem components. Due to high water change rate the river bottom sediments have undergone decontamination processes and, over the years that have passed since the accident, have ceased to play an essential role as a secondary source of water contamination. On the other hand, the closed reservoirs, and in particular the lakes in the inner exclusion zone, have considerably higher levels of radioactive contamination caused by limited water change and by relatively high concentration of radionuclides deposited in the bottom sediments. In 1997–2002 the values of the absorbed dose for hydrobionts from reservoirs of the exclusion zone were found to be in the range from 1.6×10^{-3} to 3.5 Gy·year^{-1}. The highest value was found for hydrobionts from lakes within the embankment territory on the left-bank flood plain of Pripyat River, the lowest for specimens from the running water objects. The following are considered in the paper: (1) the latest data on content of the most biohazardous radionuclides (Sr-90, Cs-137, Pu-238, Pu-239+240 and ^{241}Am) in hydrobionts of the different trophic levels and ecological groups (higher aquatic plants, molluscs and fish); (2) the possibility to use of hydrobionts of different trophic levels as biological indicators of radioactive contamination of water objects; (3) the absorbed dose rate for hydrobionts from different water bodies; (4) some biological effects (somatic and citogenetic) of radiation exposure on hydrobionts living in water reservoirs with different levels of radioactive contamination.

1. RADIONUCLIDES IN HYDROBIONTS

The radioactive substances which have been released in an atmosphere as a result of the Chernobyl accident, contaminated the catchment territory of surface of water objects, and were partially transferred in solution or deposited in bottom sediments. More than 90% of radionuclides contaminating the aquatic ecosystem are deposited in bottom sediments. Other parts are distributed in water, suspensed matter and aquatic organisms.

The levels of radionuclide contamination of water objects within the Chernobyl NPP exclusion zone has stabilised. Riverbeds were washed out in flood periods and radioactive substances in bottom sediments ceased to play the major role as secondary contamination sources of water streams. Now, radioactive contamination of rivers is due mainly to other secondary processes: washout from water catchment areas and inflow of radionuclides from more heavily contaminated water bodies. At the same time, closed water bodies, in particular the lakes of the inner exclusion zone, are substantially more contaminated. This is due to limited water exchange and higher levels of radioactive contamination in bottom deposits (compared to the washed bottom sediments of riverbeds). Therefore, contemporary levels of radioactive contamination in the majority of closed water bodies are determined by rates of exchange of mobile radionuclide forms between water and bottom deposits, and by inflow of washed out radionuclides from water catchment areas.

Aquatic organisms have differing abilities to accumulate radionuclides that are connected with their various structures and ratio of one or another substances in organisms. The higher aquatic plants are one of the components that dominate the biomass in freshwater ecosystems. They have a high production potential and ability to accumulate radioactive substances. They occupy the littoral and partially sublittoral zone in fresh reservoirs. The vegetative communities play the important role in processes of self-purification of aquatic ecosystems, due to their ability to assimilate radioactive substances from water and bottom sediments, depending on biomass and structure of the plant communities. When radionuclides are washed out from the catchment territory the phytosenoce of aquatic plants carry out the function of the natural biofilter, by accumulating and depositing radionuclides on suspensed matter, thus taking them out of circulation from the water ecosystem, and interfering the further distribution.

The patterns of ^{90}Sr and ^{137}Cs accumulation have been shown to be species-specific. Among the species with relatively high ^{137}Cs content are the helophytes (air-water plants) of genus *Carex, Phragmites australis*, *Glyceria maxima*, *Typha angustifolia*, as well as strictly water plant species *Myriophyllum spicatum* and *Stratiotes aloides.* The low values of ^{137}Cs activity in all reservoirs were found in the representatives of family Nymphaeaceae – *Nuphar lutea* and *Nymphaea candida* as well as *Hydrocharis morsus-ranae.* Relatively high content of ^{90}Sr was shown by the species of genus *Potamogeton*. Obviously this is related to this plant's tendency to accumulate large quantities of calcium (which is not washed off during standard sampling) on its surface during photosynthesis. At the same time, calcium carbonate that is removed from the plant could contain 7–20 times more radioactive strontium than the plant tissue. Thus, *Potamogeton* species makes a good prospective radioecological monitoring object as a specific accumulator of ^{90}Sr.

The content of radionuclides $^{238+239+240}$Pu and ^{241}Am in higher aquatic plants of the left-bank flood plain of Pripyat River was found, respectively, in the ranges of 1–66 (11) Bq kg^{-1} with concentration factor (CF) of 24–4175 and 1–45 (11) Bq kg^{-1}, with CF of 83–7458. *Typha angustifolia* showed the highest CF, which was 5–7 times higher than the average CF values for other studied plant species. This indicates that this species may be considered as a specific accumulator of transuranic elements in reservoir conditions within the ChNPP exclusion zone.

Others important hydrobionts for radioecological studies are freshwater molluscs, which often considered as bio-indicators of radionuclide contamination in water objects. These invertebrates accumulate practically all the radionuclides found in water and, due to their high biomass, molluscs play an important part in bioaccumulation processes and radionuclide redistribution in aquatic ecosystems.

The highest CF for both ^{90}Sr and ^{137}Cs were found in bivalve molluscs *Dreissena polymorpha* and *Unio pictorum,* which are the most active filtrators. The highest CF for ^{90}Sr was noted in *Dreissena polymorpha* – in excess of 1100, while for ^{137}Cs the highest CF (about 500) was found in the tissues of *Unio pictorum*. Considerably lower CF were determined for the gastropod species *Lymnaea stagnalis*, *Planorbarius corneus* and *Viviparus viviparus*.

The average contents of transuranic elements ^{238}Pu and $^{239+240}$Pu in mollusc tissues in Glubokoye Lake and Dalekoye-1 Lake were as follows: the lowest value was determined for *Lymnaea stagnalis* – 0.1 and 0.2 Bq kg^{-1} respectively in Dalekoye-1 Lake, 2.7 and 6.4 in Glubokoye Lake. The highest content was determined for *Stagnicolia palustris* from Glubokoye Lake – 14 and 36 Bq kg^{-1} respectively. The highest activity among gastropods was shown in *Planorbarius corneus* – 1 and 2 Bq kg^{-1} respectively from Dalekoye-1 Lake; 25 and 53 Bq kg^{-1} in Glubokoye Lake. *Dreissena polymorpha* from the cooling pond of the ChNPP showed ^{238}Pu and $^{239+240}$Pu contents of 3 and 6 Bq kg^{-1} respectively.

The contents of ^{241}Am in *Lymnaea stagnalis* tissue was the lowest – in the range of 4–30 (15) Bq kg^{-1} in Dalekoye-1 Lake and 6–51 (27) Bq kg^{-1} in Glubokoye Lake. For *Stagnicolia palustris* from Glubokoye Lake the value was about about 75 Bq kg^{-1}. The highest value was found in *Planorbarius corneus* – 18–29 (24) Bq kg^{-1} in Dalekoye-1 Lake and 80–310 (170) Bq kg^{-1} in Glubokoye Lake. The content of ^{241}Am in *Dreissena polymorpha* tissue from the cooling pond of the ChNPP was at the level of 8 Bq kg^{-1}.

The fish species that are found at the upper levels of the food webs may also constitute a part of human diet and, therefore, are of a particular interest in radioecological research of water ecosystems.

The content of ^{90}Sr and ^{137}Cs radionuclides in lake fish of the left-bank flood plain of Pripyat River in all cases considerably exceeded maximum permissible level (MPL), according to the standards accepted in Ukraine for fish production: for ^{90}Sr on average 146 times higher (MPL – 35 Bq kg^{-1}), for ^{137}Cs – 134 times (MPL – 150 Bq kg^{-1}). The highest measured values were 373 and 180 times in excess of MPL. The content of ^{90}Sr in fish of the cooling pond practically in all caught specimens also exceeded MPL (on average 8 times higher), with the highest registered values being 43 times higher than MPL. The ^{137}Cs content in all cases also considerably exceeded MPL – on average 33 times higher, with highest registered values exceeding MPL 84 times.

The concentration of transuranic elements ^{238}Pu, $^{239+240}$Pu and ^{241}Am was measured in fish of Glubokoye Lake and Dalekoye-1 Lake. The activity of ^{238}Pu in fish tissue was found in the range of 0.4–0.5 (0.4) Bq kg^{-1} with CF of 72–98 (83), $^{239+240}$Pu – 0.7–0.9 (0.8) Bq kg^{-1} with CF of 68–87 (75) and ^{241}Am – 2.2–10.0 (6.2) Bq kg^{-1} with CF of 367–1667 (1028).

2. DOSES

The radioactive contamination of the environment resulting from the Chernobyl accident has resulted in substantial growth of a radiation background and, accordingly, of a radiation dose for hydrobionts in reservoirs. However, establishment of direct dependence of biological effects from impact of ionizing radiation from natural incidence is connected to the large difficulties. A number of processes and factors capable of impacting on features of radiobiological effects on animated bodies include: heterogeneity of radionuclide distribution in terrestrial and aquatic ecosystems; effects of different types of ionizing radiation on biosystems; wide spectrum of radiosensitivity, and mechanisms of organism restoration, and also the modifying impact of the natural and anthropogenous factors.

In 1997–2002 the values of the absorbed dose for hydrobionts from reservoirs of the exclusion zone were found to be in the range from 1.6×10^{-3} to 3.5 Gy·year^{-1}. The highest value was found for hydrobionts from lakes within the embankment territory on the left-bank flood plain of Pripyat River, the lowest – for specimens from the running water objects.

The ratio of external and internal doses varied considerably for hydrobionts from different reservoirs and depended on the contents of γ-emitting radionuclides in the littoral zone bottom sediment, as well as in soils close to the river bank. Thus, in Glubokoye Lake, which contains a so-called abnormal contamination strip at the shoreline border, about 95% of absorbed dose results from external exposure and only about 5% from internal exposure to radionuclides incorporated in tissue. The similar ratio is observed for the exclusion zone rivers – Uzh River and Pripyat River. However, in these objects, the ratio is related to the high flow rate as well as to the relatively low radionuclide content in water and, consequently, in hydrobiont tissues.

In Azbuchin Lake and Yanovsky Backwater, at a relatively low external radiation dose, the main contribution to the absorbed dose is made by radionuclides incorporated in hydrobiont tissues. It is linked to the high radionuclide content in water and at the same time to the low contamination level of the bottom sediment within littoral zone, and in the soils of nearby areas (with sandy soils showing low levels of radionuclide fixation). In this respect, the cooling pond of the Chernobyl NPP is in a mid-way position.

3. BIOLOGICAL EFFECTS

The numerous effects of irradiation of hydrobionts within the exclusion zone are revealed. Some of these effects required a short period of time for their formation, but it is supposed that an increasing importance will be given to longer-term consequences – genetic damage induced by a long-term irradiation. These consequences are realised over a longer time in which the initial molecular damage can be transferred through many generations of cells.

The proof of intensive impact of ionizing radiation on hydrobionts within the exclusion zone can be the results of cytogenetic research, which has been carried out from the first and early years after the accident for higher aquatic plants, bristle worms, molluscs and fish. In cells of apical meristems of plant roots and sexual products of hydrobionts, structural chromosome damage of practically of all types occurred. The chromosome aberration rate for cells of hydrobiont from the most contaminated reservoirs reached 10%.

The studies of the different species of plants within the exclusion zone has revealed numerous morphological anomalies as repeated organs, gigantism or dwarf, under-development or sterility of reproductive organs, excessive branching, growth inhibition of the secondary points of growth etc. All of this variety of plant anomalies of development testifies that the vegetation within the exclusion zone has undergone to strong damage of the genotype, with consequential genetic instability and thus increased variability of many species.

In populations of bur reed (*Phragmites australis*) from the cooling pond of the Chernobyl NPP and lakes of the left-bank flood-lands of Pripyat River during 2000–2002 we revealed the numerous morphological anomalies (sometimes observable at 6–8% of individuals in a population). The most frequently occurring anomalies are shown in various forms of branching of shoots – arrangement of ears not only on a top and along of the stalk (up to 7 lateral ears), formation on one knee the several lateral ears, branching of a top of ear up to two or three ears. In a number of cases, the similar branching had the expressed modified character. In most cases such anomalies occurred on plants with the slowed rate of growth.

It is necessary to emphasise the important circumstance – the process of abnormal forming of plants, to all appearances, proceeds after radiation doses on a population have substantially decreased, and in some cases practically do not differ from pre-accident ones. Obviously, it displays latent damage of the genome that is transferred from generation to generation and, actually, is the remote effect of irradiation.

The research of plant radiomorphs within the exclusion zone demonstrates that the organization of regular genetic monitoring of the contaminated territories is the important measure, extremely necessary for forecasting and the prevention of negative long-term consequences of irradiation. The profound study of morphological anomalies of plants will provide information on the nature of molecular-biological processes underlying radiobiological effects of a long-term irradiation of natural populations.

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Experimental long-term exposures of fish to low dose rate gamma- or alpha-radiation

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Abstract. While it is well known that α-radiation is much more damaging than γ-radiation (on an equal absorbed dose basis) to man and mammals, there are no data available to compare the effects of these radiations on fish. As many authorised disposals of radioactivity are made to sea or rivers and include α- and γ-emitters it is important that such data is obtained so that estimates of the combined biological damaging effects may be made. This paper describes experiments made to examine the comparative effects of chronic exposure to α- or γ-radiation on reproductive endpoints in a representative fish, the zebrafish *Danio rerio*. Fish were exposed to γ-radiation or α-radiation for approximately 1 year. The fish were allowed to breed once per week and for each breeding opportunity the total numbers of eggs, numbers of eggs viable at 24 hours and eggs hatching successfully were recorded. Dose rate groups were 300, 1000 or 7400μGy h^{-1} for γ-radiation and 9.6, 19, 84 and 214μGy h^{-1} for α-radiation. None of the α-radiation groups showed significant effects on any measure of egg production while among γ-radiation groups only the highest, 7400μGy h^{-1} group showed an effect. This was a rapid decrease in total number of eggs and viable eggs laid per opportunity, leading to a failure to lay any after 20 weekly opportunities. There was, however, no significant decrease in any group in the hatch rate of eggs which were viable at 24 hours. Comparison of the highest α-radiation dose rate which produced no effect (214μGy h^{-1}) and the γ-radiation dose rate which had a significant effect (7400μGy h^{-1}) gives a relative biological effect (RBE_{α}) of <35.

1. INTRODUCTION

The need for a comprehensive system to protect the environment from ionising radiation is now widely recognised and work towards this is being carried out at international (e.g. ICRP, OECD-NEA) and national (e.g. USA, Canada, UK) levels. In order to develop environmental protection standards, data on radiation dose rates received by environmental organisms and radiation dose rate-response relationships for damaging effects on them, are required. In the environment organisms receive absorbed doses from a range of radiation qualities (e.g. α-, β-, γ-radiation). It is often possible to estimate the physical dose rate of each radiation quality but determination of their combined biological damaging effect requires knowledge of the degree to which some radiation qualities are more damaging then others. It is well known that α-radiation is more damaging than γ-radiation to the tissues of man and mammals but data which allows comparison of the effects of these radiations is sparse for other wildlife species. There are no data which allow comparison of the damaging effects of α- and γ-radiation on fish and so the work described here was undertaken to examine the comparative effects of chronic exposure to γ-radiation or to α-radiation in a representative fish, the zebrafish *Danio rerio*. Reproductive endpoints were examined as they are of major importance in maintaining population levels and it is the population which is usually the main concern when impacts on the environment are assessed.

2. METHODS

The zebrafish used in the experiment were the offspring of stock wild type AB fish kindly supplied by the zebrafish facility at University College London. Eggs laid by stock fish were hatched and fry raised until they were 12 weeks old by which time they were taking adult fish flake and artemia larvae as food. At this time irradiation was started. Fish were placed in pairs in 1.5 litre plastic tanks which had been modified with a stainless steel grid in the bottom. This allowed eggs to pass through so that they could not be eaten by the parents.

TABLE 1. THE ACTIVITY CONCENTRATIONS OF ^{210}PO IN WHOLE BODY AND OVARIES OF ZEBRAFISH KILLED AT TERMINATION OF THE EXPERIMENT AFTER RECEIVING SPIKED MEALS FOR APPROXIMATELY 1 YEAR

α-radiation group	Nominal radiation dose rate µGy/h	No. whole fish analysed (No. ovaries)	Mean whole body activity (± *st.dev.*) Bq/g wet wt	Mean ovary activity (± *st.dev.*) Bq/g wet wt	Mean α-radiation dose rate (estimated from whole body activity)[b] µGy/h
1	8	9 (3)	3.11 ± 1.63	2.14 ± 0.12	9.6
2	25	8 (4)	6.21 ± 1.67	6.17 ± 3.25	19
3	185	13 (5)	27.08 ± 4.42	39.85 ± 16.85	84
4	740	7 (3)	68.9 ± 13.45	214.4[a] ± 110.36	214

[a]Actual values 336, 187 and 121 Bq/g

[b] These values are used to make comparisons with results for γ-irradiated zebrafish

Tanks containing pairs assigned to γ-radiation groups were placed in a radiation facility [1] where ^{137}Cs sources could be remotely moved from their lead-shielded housings to irradiate the fish. Three separate arrays of ^{137}Cs sources allowed three different γ-radiation dose groups to be irradiated simultaneously. Dosimetry was carried out by placing lithium fluoride thermoluminescent dosimeters at various positions in the tanks. Mean dose rates to the fish in the three groups were 7400, 1000 and 300 µGy/h.

Zebrafish in an unirradiated control group and in the four α-radiation groups were kept under the same conditions as the γ-radiation group but received no external radiation. Fish in the α-radiation group were given meals of artemia larvae spiked with the α-emitter ^{210}Po twice weekly. Pilot experiments had determined the feeding rate of ^{210}Po-spiked food required to achieve the equilibrium tissue concentrations of ^{210}Po necessary to deliver α-radiation at the chosen experimental dose rates. At termination of the experiment, after irradiation for approximately one year, several fish from each group were analysed for ^{210}Po and unfortunately the concentrations, and thus α-radiation dose rates, in fish from the higher dose rate groups were found to be lower than expected: 84 µGy/h instead of 185µGy/h; 214µGy/h instead of 740µGy/h. The tissue concentrations and estimated α-radiation dose rates are shown in Table 1, along with the nominal (intended) dose rates. α-radiation dose rates estimated from whole body activity at the end of the experiment were used to make comparisons with γ-irradiated fish.

It has been shown that approximately the same number of eggs per week per female zebrafish were produced when breeding was allowed daily or once per week [2] and so to keep the laboratory staff workload manageable fish were allowed to breed once per week and separated by a perspex divider at other times. At each breeding opportunity the number of eggs laid (all-eggs) and the number viable at 24 hours (viable eggs) were recorded for each pair. At approximately monthly intervals (every 4th lay opportunity) viable eggs were kept and the number of them hatching successfully was recorded. At termination of the experiment, after approximately one year of radiation exposure, fish were killed and samples taken for histology and alpha-autoradiography (results discussed elsewhere [3]), as well as for ^{210}Po analysis.

3. RESULTS

Each weekly occasion when fish were allowed to breed was characterised as a “lay opportunity” and counted whether or not eggs were actually produced. Those where eggs were produced were characterised as “*used* lay opportunities”. For each pair the mean number of all-eggs per lay opportunity (total eggs/total lay opportunities) and all-eggs per *used* lay opportunity (total eggs/total *used* lay opportunities) was computed. In a similar manner the mean number of viable eggs per lay opportunity and per *used* lay opportunity were calculated. Overall mean values for these were obtained for each of the control and experimental groups (Table 2).

TABLE 2. THE MEAN NUMBER OF EGGS LAID AT EACH LAY OPPORTUNITY BY PAIRS OF ZEBRAFISH EXPOSED TO γ- OR α-RADIATION. ALL-EGGS ARE THE TOTAL NUMBERS OF EGGS LAID, VIABLE EGGS ARE THOSE VIABLE AT 24H POST-LAY. LAY OPPORTUNITIES ARE ALL OPPORTUNITIES WHETHER USED OR UNUSED, USED LAY OPPORTUNITIES EXCLUDE THOSE WHERE NO EGGS WERE LAID

Group	Dose rate μGy/h	Number of pairs	Overall mean number of eggs per lay opportunity			
			Lay opportunity		*Used* lay opportunity	
			All - eggs	Viable eggs	All - eggs	Viable eggs
Control	0	11	41.1	32.4	61.8	50.0
γ-radiation						
1	300	10	39.1	29.6	55.8	46.4
2	1000	8	34.2	25.2	56.6	43.5
3	7400	12	5.8[a]	2.9[b]	41.3[c]	16.9[d]
α-radiation						
1	9.6	12	37.5	30.1	63.9	48.9
2	19	11	38.4	32.2	64.2	53.5
3	84	13	39.3	33.0	69.5	58.7
4	214	10	31.6	26.9	64.6	55.0

[a b c d] significantly different from all other groups ($P< 0.05$)

The only group to show significant effects is the γ-radiation group 3 which received 7400μGy/h. The number of all-eggs and viable eggs per lay opportunity were greatly reduced, mainly due to the cessation of egg laying of all pairs by lay opportunity 20. Even when egg laying did take place (*used* opportunites) the numbers of eggs and of viable eggs were reduced. The reduction in viable eggs was greater suggesting that the radiation was causing a reduction in all-eggs and in the proportion of them which were viable.

Hatching of eggs which were viable at 24 hours post-lay was not significantly affected by radiation in any group. The percentage hatching was 98±8.5% in controls and varied from 86.0±19.4 in γ-radiation Group 3 to 99.4±2.3% in α-radiation group 4.

4. DISCUSSION

While γ-irradiation at a dose rate of 7400μGy/h caused complete cessation of egglaying in zebrafish, lower dose rates (300-1000μGy/h) had no significant effect on egg production. α-irradiation at dose rates of 9.6–214μGy/h also had no significant effects on this endpoint. From this data it is only possible to estimate an upper limit of the relative biological effect of α-radiation compared with γ-radiation (RBE_α = dose rate of α-radiation/dose rate of γ-radiation, for production of the same effect). Thus $RBE_\alpha < 35$ (ie < 7400 / 214) for the endpoint of egg production. In Table 1 it may be seen that in the highest α-dose rate Group 4, ovary activity concentrations of ^{210}Po were considerably greater than those for whole body but they also varied greatly between fish (121-336Bq/g, Table 3). The range of dose rates to the ovary estimated from these activity concentrations was 376μGy/h - 1045μGy/h and RBE_αs derived by comparison with the γ-irradiated Group 3 fish (dose rate 7400μGy/h) were $RBE_\alpha < 20$ to $RBE_\alpha < 7.1$. Thus the $RBE_\alpha < 35$ seems likely to be a conservative upper limit. While no other data are available for fish, RBE_α as high as 50 - 90 were obtained in mice exposed to low estimated dose rates of alpha radiation from intraperitoneally injected ^{210}Po and using primary oocyte survival as an endpoint [4]. Elsewhere an RBE_α of 20 has been used for environmental biota, based on the radiation weighting factor of 20 recommended by the ICRP for use in radiation protection of humans [5, 6].

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Radiation Effects Database (FRED) – its purpose within the FASSET project

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Abstract. The FASSET EC funded project has created a generic framework for assessing the impact of radioactive contamination on non-human biota. Effects analysis forms an integral part of the framework and this paper describes the structured method used for collating information from the available literature on the biological effects of radiation. This paper illustrates the function of the FASSET radiation effects database (FRED) using a worked example as a demonstration.

1. INTRODUCTION

The FASSET (Framework for ASSessment of Environmental impacT) EC funded project has created a generic framework for assessing the impact of radioactive contamination on non-human biota. Effects analysis forms an integral part of the framework, and takes into account available data on the biological effects of radiation. In order to derive conclusions to describe dose-effects relationships, scientific information has been collated, taking into account:

- acute and high dose rate exposures;
- chronic, low-level exposures extending over a significant fraction of the life time of the organism; and,
- endpoints such as morbidity, mortality, fertility, fecundity and mutation rate.

Over 200 thousand publications under the generic search of “radiation effects” in the last 50 years were found (using Nuclear Science Abstracts (1948 - 1976), and Energy, Science and Technology (1977 - present)). This total will, of course, include non-biological endpoints such as dosimeter response and radiochemistry.

2. EXTRACTION OF INFORMATION

A number of selection criteria were used during data collation to render the exercise manageable within the constraints of the project, *inter alia*:

- concentrate on the most relevant papers, using prior experience and an informed judgement derived from an understanding of the requirements of the FASSET project;
- collect data published since 1945 due to problems in accessing the earlier literature and possible problems in interpreting the radiation exposures in the context of modern dose quantities and units;
- ignore data derived from studies of, or for application to, human radiobiology;
- note any apparently relevant references that cannot be accessed;
- divide organisms into 16 wildlife groups (plants, aquatic plants, lichen and mosses, fungi, bacteria, soil fauna, insects, birds, mammals, amphibians, reptiles, fish, crustaceans, molluscs, aquatic invertebrates and zooplankton);
- group experimental data under four umbrella effects (morbidity, mortality, reproduction capacity and mutation); and,
- split radiation exposure regime as either, acute, transitory or chronic.

Other information were also recorded, where possible, on the type and source of radiation, the dose rate and total dose, the actual biological endpoints recorded in the study, and, an indication of whether the data could be used to determine an RBE value.

The need to be open and transparent in searching the literature and gathering information for entry onto the database was also recognised. So, it was also decided that:

— data would be retained for discussion if information did not fit obviously into one of the four umbrella categories; and

— units of the original publication would be included in the database but also converted into 'standardised' units for the total dose (Gy) and the dose rate (μGy h^{-1}) for ease of interpretation.

The primary starting points for assembling the relevant literature were major reviews, such as UNSCEAR [1] and IAEA[2, 3], together with the original publications listed in these reviews. The use of a structured approach highlighted important aspects during the database design and subsequent data inputs.

It was necessary to have a clear view on what data are needed, and restrict the information gathered to that, which is relevant to the purpose of FASSET. Essentially this means to be able to provide a basis for relating the estimated dose rate to an organism in a contaminated environment to its possible consequential effects.

There was a need for guidance on data entry to ensure consistency in the content of the database. An operating manual, embedded within the database, details the information that needs to be reported.

It was necessary to apply a measure of quality assurance (QA) to the task of inputting information onto the database. Before data entry occurred, two QA exercises were undertaken to ensure that all participants were operating to the same basic requirements. The main QA exercise consisted of each participant reading ten randomly selected, but relevant, papers and then entering the data into the database. Again, this exercise identified points that needed to be addressed in the operating manual. Once data entry was completed, an overall QA exercise was performed to highlight any erroneous data.

TABLE 1. NUMBER OF REFERENCES WITHIN THE FASSET RADIATION EFFECTS DATABASE

Wildlife Group	Number of references				
	Morbidity	Mortality	Reproduction	Mutation	Totals
Amphibians	11	7	1	6	25
Aquatic invertebrates	5	5	8	11	29
Aquatic plants	13	11	2	3	29
Bacteria	6	1	0	0	7
Birds	11	20	26	1	58
Crustaceans	5	11	12	0	28
Fish	49	30	80	30	189
Fungi	2	0	0	0	2
Insects	12	17	9	8	46
Mammals	33	49	66	14	162
Molluscs	7	8	6	0	21
Mosses/lichens	5	0	0	0	5
Plants	101	20	112	112	345
Reptiles	5	5	2	3	15
Soil fauna	5	17	3	4	29
Zooplankton	6	1	1	0	8
Totals	276	202	328	192	998*

* Real total is 1047 references. Some papers are reviews, and do not belong to a single wildlife group.

3. RESULTS

The completed Microsoft© Access database contains over 1000 references. Table 1 summarises where data are available, as categorised by wildlife groups and umbrella endpoints.

The completed database allows users to search data and produce the query reports and directs users to relevant literature. There are three search options, as defined by the FASSET project:

— a sequential search on wildlife group, umbrella endpoint and dose or dose rate (with options for viewing and reporting the data after each search);

— a manual search of the database (which allows the user to search the database on any field contained within it but at the moment it is only possible to view the search results from within the database itself; and,

— a search for references that may be used to generate RBE information.

The search results can be exported to Microsoft© Excel for further interpretation and assessment. Reports can also be created for listing references, and saved as delimited text files, which can be opened in Microsoft© Word 97 or via the Microsoft© Windows Notepad. The reference list can also be viewed from within Microsoft© Access.

The database has been developed as a basis for the FASSET report [4] that aims to provide a justified level of biological hierarchy to be protected, a discussion on radiation effects including dose/response relationships, a discussion of uncertainties and proposed threshold dose rates. A series of tables have been compiled to provide information on each of the 16 wildlife groups, for acute and chronic data. Table 2 gives an example of chronic data summarised for soil fauna.

The experimental results given in Table 2 demonstrate that there is limited data available on the effects of chronic irradiation on soil fauna using two of the four FASSET umbrella endpoints, mutation and mortality as extracted from the information currently held within the FRED database. Those data that are available appear to come from the same or similar data sets (e.g. references [6, 5, 7 and 8, 9], respectively) and focus on the exposure of soil fauna such as earthworms exposed to ^{226}Ra in the soil. Earthworms particularly utilise the soil as a direct source of nutrition and thus are particularly susceptible to internal exposure by α-radiation. The majority of the experimental data are also found at dose rates of >10,000 Gy h^{-1} or at background rates.

4. SUMMARY

In summary FASSET has delivered a database in Microsoft Access® (versions 97 or 2000). The "FASSET Radiation Effects Database (FRED)" is freely available for download from www.fasset.org. The database contains over 1000 references and provides access to all the raw data extracted from the original publications. In addition to providing the information needed for the requirements of FASSET, the database has also identified gaps in knowledge, which may guide further research and can direct users to relevant literature on the effects of ionising radiation on wildlife.

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TABLE 2 SUMMARY DATA FOR CHRONIC DATA ON MOLLUSCS, DERIVED FROM THE FASSET RADIATION EFFECT DATABASE (GREY AREAS SHOW GAPS)

Soil fauna	MORBIDITY			MORTALITY			REPRODUCTIVE CAPACITY			MUTATION		
Dose-rate (μGy h^{-1})	Species	Radiation type	Data (paper)	Species	Radiation type	Data (paper)	Species	Radiation type	Data (paper)	Species	Radiation type	Data (paper)
Control-background	Mesofauna	Alpha	8(1)	Octolasium lacterum *Dendrobaena octaedae* *Eisenia nordenskioldi* Mesofauna	Alpha Alpha Alpha Beta[1]	1(1) 1(1) 1(1) 4(1)				*Tityus bahiensis Dero obtusa, Nais pseudobtusa, Nais pardalis*	Gamma Mixed	5(1) 6(1)
< 99.9				Soil community	Mixed	12(1)				*Tityus bahiensis Dero obtusa, Nais pseudobtusa, Nais pardalis*	Gamma Mixed	8(1) 2(1)
100-199.9												
200-499.9												
500-999.9												
1,000-1,999.9				Mesofauna Mesofauna	Mixed Beta[1]	1(1) 2(1)						
2,000-4,999												
5,000-9,999				Mesofauna	Beta[1]	2(1)						
> 10,000				*Octolasium lacterum* *Dendrobaena octaedae* *Eisenia nordenskioldi* Various fauna	Alpha Alpha Alpha Alpha	1(1) 1(1) 1(1) 5(1)						
Data relevant to RBE determination:												
Dbase paper IDs [reference]	752 [10] 759 [11]			208 [12] 639 [5] 561 [6] 754 [7] 762 [13] 763 [9] 757 [8] 759 [11] 752 [10]			549 [14]			549 [14] 749 [15]		
Observations "rejected" (paper ID)	(38)752,8(759)			12(754),3(759),2(757),1(763), 38(752),6(639),8(765),6(561)			None			4(549),7(749)		
Last paper published in:	1992			1992			None			1976		

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3.2. ENVIRONMENTAL TRANSFERS AND UPTAKES

Uptake of radiocaesium in soil-to-plant system in the tropical environment
A comparison between experiments and prediction

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Abstract. Soil-to-plant transfer factor (TF) of radiocaesium has been predicted using the Absalom model basing on soil properties such as pH, organic matter, exchangeable K^+ and clay contents valid for the tropical environments. Due to insufficient data of soil properties, the average values of pH, organic matter, exchangeable K^+ and clay contents have been taken as the input model parameters within the ranges given for Asia. The predicted TF values have been compared with the measured values obtained from pot and field experiments as a part of the validation of the used model. The measured TF values in the pot experiment were obtained mainly for leafy vegetables, rice, root and grass while the measured TF values from the field experiment were obtained entirely for plants grown in a contaminated land containing radiocaesium in the Atomic Energy Research Establishment (AERE) campus, Dhaka. The predicted TF values have been found to be comparable to some extent with the measured values. The sensitivity of the main soil properties on the uptake of ^{137}Cs in the soil-plant system has also been described in the present work. This data set might be useful while calculating radiological dose in the tropical environments.

1. INTRODUCTION

The environment is a great treasure for the biota. Any contamination of the environment ensues corresponding threat to the very existence of its flora and fauna. Now-a-days environment is being stressed by various newly evolved technological and social agents. One such to mention is the application of ionizing radiation in agriculture, medicine, industry and research and development, because exposure to ionizing radiation has got its deleterious effects on health. Plant system is essential for the sustenance of man and animal. Since life is very much dependent on plant system, contaminants and toxicity present in the plant system is a threat to life. Following their release in the environment, radionuclides may deposit on ground and consequently migrate into various environmental media. Among deposited radionuclides, radiocaesium (^{137}Cs) is a dominant fission product which has high relative mobility in the soil-plant system, long term bioavailability, high radiotoxicity and is long-lived. The plant uptake of ^{137}Cs from soil, commonly expressed as soil-to-plant transfer factor (TF) is widely used while calculating radiological dose. Generally TFs of ^{137}Cs vary often by more than four orders of magnitude depending upon soil type, pH, solid/liquid distribution coefficient, exchangeable K^+ and organic matter content [1]. The low solid-liquid distribution coefficient (K_D) due to the low clay content and high NH_4^+ concentration in the soil solution enhances TF value of ^{137}Cs. Moreover, the usual low K^+ level in the solid phase and in the soil solution and high organic matter content also enhance root uptake. Absalom *et al.* [2] presented a model which predicts the radiocaesium soil-plant transfer factor on the basis of easily measured soil characteristics (clay content, organic carbon content, exchangeable potassium and pH). This model has been tested in Europe through successful prediction of the fate of Chernobyl [3] and weapons [4] fallout ^{137}Cs. A comparable model is required to predict the impact of deposited ^{137}Cs based on the regional parameters derived for wet-dry tropical environments. In order to apply the Absalom model and-/or to modify the model, regional databases for model validation need to be developed for the tropical environments of Asia. This effort has been limited due to lack of appropriate data. Nevertheless, limited data of some Asian countries are available in the literature [5] which could be used to predict the TF values of ^{137}Cs in the soil-plant system.

There are many studies regarding the TF values of ^{137}Cs obtained both for tropical and temperate environments [6-10]. A pot experiment was performed for leafy vegetables, rice, root and grass while a field experiment was carried out entirely for leafy plants grown in a contaminated land containing radiocaesium in the Atomic Energy Research Establishment (AERE) campus, Dhaka [6, 7]. In the present study, the TF values of ^{137}Cs have been predicted using the Absalom model [2] in the soil-plant system basing on soil properties such as pH, organic matter content, exchangeable K^+ and clay content. Then the predicted TF values have been compared with the measured values obtained from both for pot and field experiments which might be representative for the tropical environments in Asia. Consequently, a validation of the used model has also been achieved for the tropical environments especially valid for the selected region in Bangladesh.

2. MATERIALS AND METHODS

2.1. Model description

Radiocaesium bioavailability is strongly influenced by soil properties such as organic matter content, K^+ status and clay content. Absalom *et al.* [1] presented a semi-mechanistic model, which predicts activity concentrations of ^{137}Cs in plants for stipulated soil-Cs contact times. The model utilized as input parameters/soil characteristics, which are routinely measured or can be readily estimated (% clay, exchangeable K^+, initial soil radiocaesium content). The model has been developed using data describing plant uptake of ^{137}Cs from a range of mineral soils. A revised ^{137}Cs uptake model which accounts for the effect of organic matter on ^{137}Cs sorption by soil and uptake by plants was developed by Absalom *et al.* [2]. This model can be applied to mineral and organic soils simultaneously to provide a more generally applicable simulation of ^{137}Cs dynamics.

The model of Absalom assumed that Cs sorption occurred exclusively on the soil clay fraction. The distribution of sorbed and solution ^{137}Cs was described by a labile ^{137}Cs distribution coefficient (K_{dl}, dm^3/kg) which was estimated as a function of soil clay and exchangeable K^+ contents. Plant uptake of ^{137}Cs was described by a concentration factor (CF, Bq/kg plant/Bq dm^{-3} soil solution) which was related to solution K^+ concentration (m_k, moles dm^{-3}). The calculation procedures in the present work are based on the methodologies as described in the Absalom model [2].

2.2. Data sources in the tropical environments

Soil properties (pH, organic matter content, exchangeable K^+ and clay content) are needed as the input model parameters in order to use Absalom model for the tropical environments of Asia. A comprehensive data set containing sufficient soil properties and the corresponding TF of ^{137}Cs from Japanese soils to root and leaf of radish has been reported elsewhere [8]. Using the organic carbon from this report, organic matter (OM) content was calculated as OM= Organic carbon × 1.724 [11]. Accordingly, the percentage of clay content was calculated using the formula as CEC=0.6 × (% clay) + 2.0 × (% OM) [12]. This data set can be used exclusively in the model to investigate the uptake of deposited ^{137}Cs from soil to various kinds of plants grown in the tropical environments. The TF values of ^{137}Cs from Bangladeshi soil to the main foodstuffs in the country such as leafy vegetables, ladyfinger, radish, potato, potato plant, rice, rice plant, grass, ginger, ginger plant, turmeric and turmeric plant were investigated by pot experiment described elsewhere [6]. This area mainly contains silty-clay type of soil. In this work, an average value of the corresponding soil property (such as pH, OM content, clay content and CEC) has been reported without exchangeable K^+. However, Frissel [9] reported the exchangeable K^+ for the same soil which could be used together with the other soil properties for estimation of limit valid for Asia. Limited soil properties such as pH, exchangeable K^+ and CEC can be obtained from the literature found elsewhere [10] which could be used as the partial data for the Chinese soils.

The average values of pH, OM content, exchangeable K^+ and percentage of clay content as mentioned in above literatures given for Asian regions have been selected as the input model parameters in the Absalom model. These values are found to lie within a wide range as described elsewhere [5].

3. RESULTS AND DISCUSSION

Calculated and measured TF values of ^{137}Cs from soils to vegetables and crops (mainly rice) are shown in Figures 1–4. Measured values are plotted in a convenient way because of insufficient data of the soil properties. Due to insufficient data, the ranges of pH, OM content, exchangeable K^+ and clay content were taken from the literature as 5.27-7.75, 0.008-0.074 g/g, 0.097-1.957 $cmol_c$/kg and 0.114–0.465 g/g, respectively, which lie within the limits valid for many countries in Asia [5]. There are four variables intended to incorporate as the input model parameters (soil properties). Always one parameter was variable whereas three were taken fixed averages (Figures 1–4) for calculations. This approach might provide understanding the sensitivity of the used soil properties as input model parameters on the transfer factor of ^{137}Cs in the soil-plant system. The average values of pH, OM content, exchangeable K^+ and clay content were taken as 6.30, 0.037 g/g, 0.542 $cmol_c$/kg and 0.244 g/g, respectively, valid for the tropical environments. The same empirical parameters as in the original model have been used in the calculations.

It can be seen from Figures 1, 3 and 4 that the calculated TF values decrease with increasing pH, exchangeable K^+ and clay contents. However, calculated TF values increase with increasing OM content shown in Figure 2. Within the limits of the soil properties in the tropical environments, exchangeable K^+ has been observed to be the most sensitive parameter. The sensitivities of the other parameters have been investigated using comparatively lower K^+>clay>OM>pH sequentially in the model. The measured TF values both from pot and field experiments lie within 1.8×10^{-2} to 6.61×10^{-2} which are comparable with the published data shown in Table 1. The TF values of ^{137}Cs obtained from the pot experiment have been found to be scattered compared to the TF values obtained from the field experiment. Since the same type of soil was used for both experiments, plant species would be the main reason for significant variation in the TF values obtained in the pot experiment. The predicted values carried out in the present work have been found to be comparable at a certain region for the corresponding pH, OM, exchangeable K^+ and clay contents shown in Figures 1–4. The values of pH, OM, exchangeable K^+ and clay contents in the comparable regions are observed to be nearly same as the respective value (pH =5.8, OM=0.0078 g/g, K^+ =0.51 $cmol_c$/kg and clay= 0.465 g/g mentioned elsewhere [6, 7]) in soil used in the present work.

It is observed that the predicted TF values for ^{137}Cs are comparable with the measured values especially for the leafy parts of a plant. However, the calculated values are found to be overestimated compared to the measured values obtained for rice. This indicates that a general conversion factor for each part of a plant and-/or for a variety of different plants for a specific region is suggested if the model is to be applied to tropical environments. Additionally the variation between predicted and measured TF values might arise due to the empirical parameters used in the original model. It is, therefore, the empirical parameters used could be re-evaluated for the tropical environments of Asia to predict more reliable result. Independent soil properties rather than the input model parameters such as pH, clay content, exchangeable K^+ and organic soil need to be measured experimentally in order to determine the empirical parameters for the soils in Bangladesh.

TABLE 1. TRANSFER FACTORS OF ^{137}Cs PUBLISHED ELSEWHERE [6, 7, 10]

Organization	Part of plants	TF value
IAEA	Edible parts of crops	3.0×10^{-2}
NCRP	Edible parts of crops	1.5×10^{-5}-0.29
NRPB	Edible plants	3.8×10^{-2}-2.9×10^{-1}
US DOE	Vegetable, fruit, grain	1.1×10^{-2}
US NRC	Edible parts of crops	1.0×10^{-2}
CEC	Grain	6.0×10^{-3}
Taiwan RMC*	Seed	0.03-0.188
Taiwan RWMC	Rice	4.0×10^{-2}-6.0×10^{-1}
Bangladesh (Pot experiment)	Vegetable, rice	1.8×10^{-2}-6.61×10^{-2}
Bangladesh (Field experiment)	Leafy plant	2.5×10^{-2}-4.20×10^{-2}

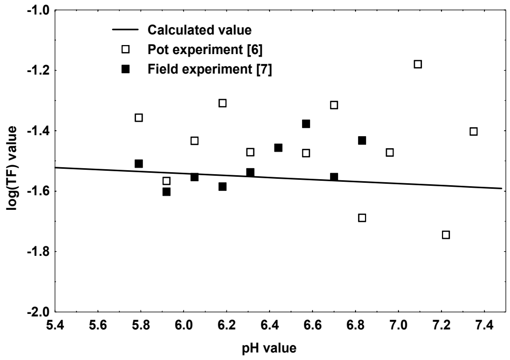

FIG. 1. Calculated and measured TF values for ^{137}Cs. Calculations were performed for variable pH values with fixed OM, K+ and clay contents valid for tropical environments.

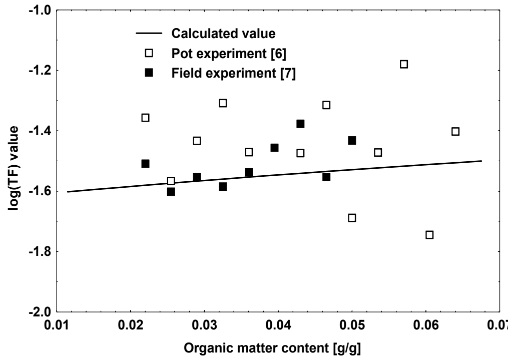

FIG. 2. Calculated and measured TF values for ^{137}Cs. Calculations were performed for variable OM contents with fixed pH, K+ and clay contents valid for tropical environments.

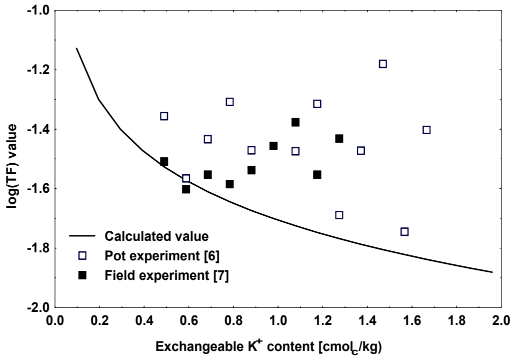

FIG. 3. Calculated and measured TF values for ^{137}Cs. Calculations were performed for variable exchangeable K^+ contents with fixed pH, OM and clay contents valid for tropical environments.

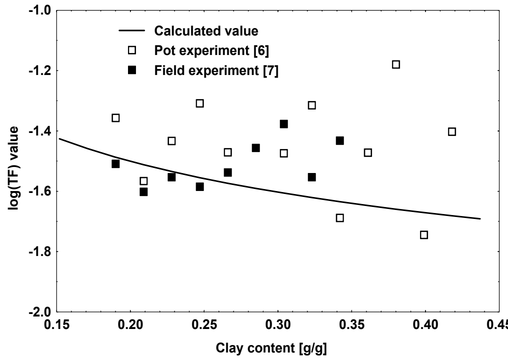

FIG. 4. Calculated and measured TF values for ^{137}Cs. Calculations were performed for variable clay contents with fixed pH, OM and K^+ contents valid for tropical environments.

4. CONCLUSIONS

In this work, soil-to-plant transfer factor (TF) of radiocaesium has been predicted using the Absalom model basing on soil properties in the tropical environments. Within the limits of the soil properties in the tropical environments, exchangeable K^+ has been observed to be the most sensitive parameter on the uptake of ^{137}Cs in the soil-plant system. The sensitivities of the other parameters were investigated using comparatively lower as K^+>clay>OM>pH sequentially in the model.The predicted TF values have been found comparable to some extent with the measured values obtained from pot and field experiments. However, the calculated values have been found to be overestimated compared to the measured values obtained for rice. Independent soil properties rather than the input model parameters such as pH, clay content, exchangeable K^+ and organic matter contents need to be measured experimentally in order to validate the used model by re-evaluating the empirical parameters for the tropical environments of Asia. Additionally, the empirical parameters need to be re-evaluated for specific parts of a plant and-/or for a variety of different plants. Alternatively, a general conversion factor for each part of a plant and-/or for a variety of different plants for a specific region is suggested if the model is to be applied to tropical environments. This data set might be useful while calculating radiological dose in the tropical environments.

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Recovery of Chernobyl-affected soils in the Republic of Belarus: *Tendencies and trends*

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Abstract. The paper presents the analysis of the developed and suggested measures and methods of decreasing soil contamination in the Republic of Belarus, and prevention of long-lived ^{137}Cs and ^{90}Sr radionuclides migration in food chains. It was found that the application of preventive measures based on a principle on decreasing ^{137}Cs and ^{90}Sr mobility in soil due, to their fixation in soil-absorbing complex, or so-called rehabilitation, is the more effective, ecologically safe and, it seems, the most perspective countermeasure in Belarus at present. The possibility of applying freshwater lake sediments, industrial waste for rehabilitation of the contaminated soils and prevention of accumulation of ^{137}Cs and ^{90}Sr radionuclides in agricultural production in Belarus is discussed.

1. INTRODUCTION

As a result of the Chernobyl NPP accident more than 70% of radioactive substances released into the atmosphere (iodine, caesium, barium, strontium, plutonium, etc.) fell out in the territory of the Republic of Belarus. Due to that fact the long-time contamination of 46,45 thousand km^2 (23%) of the territory of Belarus occurred [1]. At present the greatest risk for natural ecosystems of Belarus, and also for health of people who are still living in the contaminated territories is an internal exposure by long-lived radionuclides, ^{137}Cs and ^{90}Sr mainly.

Under the assessment of the Chernobyl NPP accident consequences on ecosystems of Belarus should be taken into account that now in the Republic more than 1.35 million ha of lands contaminated with ^{137}Cs with the density of 37–1480 kBq/m^2, from which 478 thousand ha are contaminated with ^{90}Sr with the density of 1–11 kBq/m^2 are used for agricultural production.

The main determining factor of ^{137}Cs and ^{90}Sr migration in a soil profile and in food chains is the relation of the readily soluble and the fixed forms of these radionuclides. The long-term investigations of physical- and -chemical condition of ^{137}Cs and ^{90}Sr in soils of Belarus show that the availability of ^{37}Cs for plants in soil reduces in the course of time, due to transformation of its part into a non-exchangeable absorbed state, but the mobility of ^{90}Sr remains high and has the tendency to increase. This process is aggravated with the fact that soils of the territory of Belarus are predominantly of low fertility, poor with humus, and acidic soddy podzolic and peat bog soils, with low sorption ability as to ^{137}Cs and ^{90}Sr. At present the share of mobile forms of ^{137}Cs in soddy podzolic soils of Belarus is about 10%, of ^{90}Sr up to 70%, in peat bog soils – 15 and 50%, relatively [1].

Thus, under minimization of the Chernobyl NPP accident consequences in Belarus the main task is the development of comprehensive approaches for preventing of radionuclides migration in biological chains.

2. DECONTAMINATION AND REMEDIATION METHODS

The measures and methods developed and suggested in the Republic of Belarus at different times to decrease soil pollution and to prevent of long-lived radionuclide migration of ^{137}Cs and ^{90}Sr mainly, in food chains, may be divided into cardinal measures, including removal of radionuclides out from soil or soil decontamination and preventive measures, including conditioning for decreasing of radionuclides migration in food chains or protective measures.

The cardinal measures on soil decontamination are possible by two expedients including mechanical, chemical, biological methods, as well as their combinations:

(i) Through mechanical removal of the upper layer of soil contaminated with radionuclides by a complete excavation of ground or deep ploughing with a turnover of a layer;

(ii) Through removal of radionuclides out from soil by their chemical leaching or with application of biotechnology.

The preventive measures or protective measures are based on two main principles:

(i) Conditioning for preventing uptake of radionuclides by plants or naturally protective measures;

(ii) Conditioning for fixation of radionuclides in soil or rehabilitation. The naturally protective measures have the aim of ensuring a competition on sorption places on plants rooted surface between radionuclides ^{137}Cs and ^{90}Sr and their chemical analogues that are the key macronutrients of plants.

2.1. Decontamination

The mechanical removal of the contaminated upper soil layer by excavation or deep ploughing with a turnover of the layer was widely applied at the starting period after the Chernobyl NPP accident in Belarus. However, when using this method the reconstruction of the humus layer is required by addition of a great amount of organic and mineral fertilizers into soil or complete replacement of the removed contaminated ground by pure fertile soil [1–4].

The chemical leaching of radionuclides out from soil can be used fixing surface contamination directly after fallout of radionuclides, by drawing special emulsions with the following removal of the hardened layer together with radioactive contaminants. The chemical flushing of soils can be also be applied for removal of radionuclides, excluding the cases when contamination occurred with a wide spectrum of radionuclides. The greatest effect at application of mechanical and chemical methods of soil decontamination will be obtained under the higher levels of contamination.

It is necessary to mark the application of both mechanical, and chemical methods of removal of radionuclides out from soil is effective only in case of local contamination and needs additional measures on recovery of soil fertility.

The biological methods of soil decontamination are phytoremediation, which provide maximum removal of radionuclides out from soil by plants. Phytoremediation includes the selection of agrotechnical methods and special types of agricultural crops. This method can be applied in large territories, but practically it is used at low levels of contamination only.

It is necessary to note that the application of mechanical, chemical and biological methods of soil decontamination has one general imperfection, such as the problem of utilization and storing of radionuclides removed out from soil. This problem is especially urgent in case of mechanical and biological methods, when it is necessary to process a considerable volume of the contaminated materials (soil, biomass of plants) by a special expedient. On the whole, decontamination is a laborious and expensive procedure, requiring a comprehensive approach. It is difficult to be technically implemented in the Republic of Belarus.

2.2. Rehabilitation and protective measures

Carrying out the protective measures and rehabilitation of the contaminated territories by conditioning for fixation of radionuclides in soil absorbing complex and to reduce their availability to plants are the more appropriate in the Republic of Belarus. At present, in agriculture of Belarus for obtaining of normatively pure agricultural production the protective and rehabilitation measures are being used in complex. With this purpose, the optimization of physical and chemical, and agrochemical properties of agricultural soils is carried out by addition of the advanced application of mineral fertilizers, liming and also addition of organic fertilizers. Cattle manure is applied as an organic fertilizer mainly [3, 4].

It is considered, that depending on climate and soil conditions these countermeasures allow to decrease contamination of plant-growing production to a considerable extent. The positive effect is achieved both at the expense of decreasing of the availability of ^{137}Cs and ^{90}Sr to plants, and as a result of magnification of productivity of agricultural cultures so-called "dilution effect" [4].

However, the lastest scientific data show the insufficient efficiency of the above-mentioned countermeasures and indicate that some of them are not ecologically safe [5–8]. The ecological effect of the systematic application of the advanced doses of mineral fertilizers is of particular concern. The application of this measure with a positive effect also results in a lot of negative consequences [9].

The use of the principle on decreasing of ^{137}Cs and ^{90}Sr mobility in soil due to their fixation in soil absorbing complex or natural rehabilitation is the more effective, ecologically safe and, it seems, the most appropriate measure for preventing radionuclides migration in food chains. The main task is the selection of amendments.

As the practice shows, the application of cattle manure as the amendment is an effective measure to decrease contamination with radionuclides of agricultural production in Belarus. However, the application of cattle manure and fertilizers on its basis in Belarus deals with some difficulties. Firstly, there is a lack of necessary quantity of cattle manure [10]. Secondly, it is necessary to apply clean cattle manure from the uncontaminated regions. But Belarus has no industrial processing of cattle manure, and the transportation and applying it in a virgin condition is an expensive and technically difficult procedure.

Besides, it is necessary to take into account that the application of cattle manure is effective as to ^{90}Sr mainly. But for reducing the mobility of ^{137}Cs it is necessary to apply mineral substances.

Thus, for rehabilitation of soils, contaminated with radionuclides the development of new effective, economically and technologically practical amendments on the basis of both organic and mineral raw material is very actual in Belarus. The amendment should be ecologically safe and maximum similar to natural soil humus.

2.3. Remediation by using organic-and-mineral soil conditioner

More appropriate organic-and-mineral amendments in agriculture of Belarus are freshwater lakes sediments called as “sapropel”, as well as industrial waste of hydrolyzed and potassium production like the hydrolyzed lignin and clay-salt slimes. The basic advantages of sapropels as possible amendment to soil for fixation of radionuclides are a high rate of dispersion, a great content of organic substance (up to 70%) and the content of valuable nutrient substances. The major agrochemical properties of sapropels are determined with the availability of the mobile forms of basic nutritious elements. The mobile forms of calcium are 20–40%, phosphorus 15–20%, magnesium 25–75% from their blanket content. Sapropels are contained up to 30 microelements. Under the content of such deficient microelements for Belarus, as copper, cobalt, molybdenum and zinc these fertilizers are the most appropriate for their addition to soil [11–12].

It is pointed out that the efficiency of sapropels is also determined with the quality of organic substances, especially of humic acids and nitrogen. The maximum quantity of nitrogen reaches 4.6% per dry substance, and its basic quantity enters the composition of aminoacids. Sapropels have acidity close to neutral. The sapropels of a carbonate type are a unique natural material for liming acid soils [11].

Especially perspective is the use of sapropel amendments on sandy soddy podzolic soils where are the highest migration coefficients of radionuclides from soil to plants [13, 14]. Belarus has more than 1759.1 million m^3 sapropel resources: organic, carbonate, silica and the mixed types (Table 1).

The analysis of efficiency of application of different types of sapropels as amendments shows that they sufficiently decrease radionuclides migration out from soil to plants. According to the assessment from different investigations, the application of sapropel amendments on light as to mechanical content soils will allow to reduce radionuclides migration out from soil to plants to up to 5 times, here, the effect is observed during for a long period of time [7, 8, 15].

The investigations carried out in the Ukraine and Belarus show high effectiveness of sapropel amendments application on soddy podzolic soils contaminated with radionuclides [13, 16]. The investigations carried out at the Leuven Catholic University (Belgium) have shown the influence of sapropels of different types from some Byelorussian lakes on the decreasing of the content of ^{90}Sr mobile forms in soil [7].

TABLE 1. RESOURCES AND TYPOLOGICAL BREAKDOWN OF SAPROPEL SEDIMENTS IN BELARUS LAKES

Per type structure of sapropel reserves, mill. m^3 , % mill. m^3, %							
Region	Explored reserves (mill. m^3)	Organic sapropel (mill. m^3)	Silica sapropel (mill. m^3)	Carbonate sapropel (mill. m^3)	Mixed sapropel (mill. m^3)	% of total explored reserves (mill.m^3)	Expected reserves (mill.m^3)
Brest	92.8	472 50.9	26.1 28.1	8.9 9.6	10.6 11.4	5.2	46.0
Vitebsk	1256.3	220.8 14.4	880.3 69.6	86.5 6.8	77.7 6.2	71.9	617.6
Gomel	87.3	13.0 14.9	67.3 77.1	3.1 3.5	3.9 4.5	4.9	1.5
Grodno	69.7	6.1 9.0	18.8 27.7	27.2 40.0	15.8 23.3	3.9	13.3

The hydrolyzed lignin is a by-product of the wood processing industry which is of a great considerable interest from the point of view of the content of organic substance and sorption ability [17, 18]. The content of nitrogen in quantity of 0.17–0.19%, phosphorus 0.24%, and potassium 0.21% in lignin is of the greatest interest. From the total amount of the bound nitrogen about 25% is ammonia sulphate, and 75% is chemically bound with lignin. Therefore, at addition of lignin into soil the nitrogen is not washed out, and is assimilated by plants gradually in accordance with decomposition by microorganisms to low molecular combinations [18, 19].

The hydrolyzed lignin as the component of organic fertilizers is being studied for a long time. In numerous scientific works it is shown the participation of lignin in humus formation, influence of its complex-forming properties on a mode of power supply of plants and activation of their vital activity is exhibited. The effect of lignin on soil microflora has been considered; its positive role as organic fertilizer and soil conditioner has been marked [18, 20, 21]. Many researchers when estimating fertilizers from the hydrolyzed lignin compare the efficiency of its application with peat. So, long-term examinations are exhibited, that lignoorganic fertilizers are similar to compost, provided on the basis of peat [21].

The clay-salt slimes are the waste of potassium production industry and could be the perspective mineral additives to different types of amendments. The annual formation of their clay-salt slimes is about 1.6–1.8 million tons per year. By material composition the clay-salt slimes waste are composite formations, which principal components are carbonate and magnesium carbonates, zinc sulfates of calcium, aluminosilicates, sodium and potassium chlorides. From carbonates inclusions predominates the dolomite, zinc sulfate of calcium are submitted by anhydride; aluminosilicates like clay minerals presented mainly of hydromicaceous composition [22, 23].

The clay-salt slimes have a series of the important particular properties, such as hydrophilicity, high dispersion, ability to bloating and ion exchange. The pointed out slimes contain 60–70% of particles with the size less than 0,05 mm, about 30% of them are the particles less than 0,001 mm. The particles larger than 0,01 mm are in the quantity of 35–40% and are submitted in the basic easily water-soluble halite and sylvite. The clay-salt slimes are characterized with high specific surface (40–45 m^2/g), as well as with considerable defective of crystalline structure, what increases their sorption ability. The recently executed examinations testify to expediency of usage of clay-salt slimes as amendments and structure-forming additives on low fertility sandy soils, extracted peat bogs, and also for rehabilitation of farmlands, contaminated with radionuclides in the Republic of Belarus [22].

3. CONCLUSIONS

The analysis of up-to-date existing methods and expedients on recovery of the Chernobyl-affected soils displays, that the most expedient and ecologically safe counter-measure in Belarus now is the rehabilitation of the territories, contaminated with radionuclides by decreasing of mobility of radionuclides in soils. Such rehabilitation is carried out by means of addition of organic and/or different organic-and-mineral amendments into soil.

Here, a primary task is the development of high-effective amendments economically expedient and ecologically safe. The more appropriate is the development of organic-and-mineral amendments on the basis of such local natural raw material as sapropels and such waste of industrial production as the hydrolyzed lignin and clay-salt slimes.

The basic advantages of amendments on the basis of sapropels, the hydrolyzed lignin and clay-salt slimes are as follows:

(i) Controlled physical and chemical, and agrochemical properties;

(ii) High sorption ability;

(iii) Conformity with the modern ecological requirements and standards (under their nature they will be approximate as much as possible to fertile layer of natural soils);

(iv) Sterility (does not contain pathogenic microflora, weeds, parasitic insects and parasites);

(v) Low cost of the products which will be produced on the basis of local natural raw material and waste of commercial production).

The application of the data on organomineral amendments for rehabilitation of the territories contaminated with radionuclides, will have the following advantages:

(i) Does not cause infringement in biogeochemical balance in the region (accumulation, washing-out, migration of anyone or a group of chemical substances and compounds);

(ii) Easily applied within the framework of conventional agriculture; technically and technologically is easily implemented;

(iii) Ecologically safe and does not require the constant ecological monitoring;

(iv) Conformity with the world standards oriented to high technology, ecologically pure production and promotes the development of organic agriculture in the contaminated regions.

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Radioecological monitoring of water systems of various regions of the Republic of Belarus affected by the accident at Chernobyl NPP

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Abstract. The work is aimed at studying radionuclide contamination of water system components in order to evaluate the consequences of the Chernobyl disaster for water resources of the Republic of Belarus. As a result of the studies there have been elicited peculiarities in the dynamics of the radioecological situation and roles of separate components in the overall balance of radioactive contamination of water systems in the Eastern-Belarusian province. The data on the radionuclide content in water and water vegetation allow an assessment of the danger of these components getting into human body directly, or indirectly through food chains. The established relationships in radionuclide distribution among components of the ecosystems studied can be applied to other systems of lentic and lotic types (lakes, big ponds, rivers) used for public needs.

1. INTRODUCTION

The work is aimed at studying radionuclide contamination of water system components in order to evaluate the consequences of the Chernobyl disaster for water resources of the Republic of Belarus and to use the obtained results while developing protective measures. The objects of the study are water systems of a slightly lotic and lotic type, namely, the Malinovka reservoir and the Senna river, Chericov district, Mogilev region as well as components of those systems – water, suspensions, bottom sediments, dominant water vegetation species.

As a result of the studies there have been elicited peculiarities in the dynamics of the radioecological situation and roles of separate components in the overall balance of radioactive contamination of water systems in the Eastern-Belarusian province. The data on the radionuclide content in water and water vegetation allow assessing a danger of these components getting into human body directly, or indirectly through food chains. The established relationships in radionuclide distribution among components of the ecosystems studied can be applied to other systems of lentic and lotic types (lakes, big ponds, rivers) used for public needs.

2. MATERIALS AND RESEARCH METHODS

Standard β-radiometry methods with the application of semiconductor γ-spectrometry, dosimetry, radiochemical analysis with final β- and γ-spectrometry were applied in the research. The following devices were used for measurements: β-radiometer RKB-4-2eM; β-radiometer KRVP-3AB; multi-channel semiconductor γ-spectrometer with Ge(Li) detector DGDK-100V-3 and amplitude analyzer AFORA-LP-4900 V; radiometer-dosimeters DRG-01T.

3. RESULTS AND THEIR DISCUSSION

It should be noted that the Cs-137 contamination density of the Senna river headwaters territory at the village of Chudyany is over 40 Ci/sq.km. (up to 140 Ci/sq.km), the relief is a valley outwash of the Sozh glaciation. The average maximum annual flow of the Senna river at the Chudyany village is 5.99 m. m^3, the average annual water discharge is 4.8 m^3/s. The average annual drift (sediment) flow is 0.19 thousand ton, the average annual drift discharge is 0.006 kg/s. The village of Chudyany on the Senna river is located 5 km away from the Malinovka reservoir. The Malinovka reservoir has the following hydrological features: water-surface area at the top water level is 26.6 ha and 16.1 ha. at the dead-storage level. The pondage at the top water level is 373 thousand m^3. The maximum width is 0.36 km, length – 1.8 km. The average depth is 2.0 m. The reservoir is localized on a site of the Senna river valley in its middle current situated 400 m eastward the outskirts of the village of Malinovka. The reservoir was put into operation in 1982.

According to their physical and hydrological indices and geographical peculiarities, the reservoir at the village of Malinovka and the Senna river are typical water systems of the Eastern-Belarusian province, and they were chosen as objects of long-term radioecological monitoring because of high radioactive contamination of landscape components. There have been data on the distribution of Cs-137 and Sr-90 in phytocoenoses obtained. Judging by biomass, the reservoir ecosystem is dominated by the following macrophits: Elodea canadensis Rich., Typha latifolia L., algae of Cladophora, Oedogonium families; Oenanthe aquatica L., Gliceria aquatica Waheb., Alisma plantago aqual, Equisetum fluviatile L. are represented in smaller quantities. Over the years of research the highest values of Cs-137 specific activities were observed in 1991 – from 2,110 Bq/kg in Typha latifolia L. to 377,000 Bq/kg in Elodea canadensis Rich. It has been established that in different vegetation seasons Cs-137 activity values amount to 1,920 to 377,000 Bq/kg, Sr-90 activity values - 29.5 to 721 Bq/kg in Elodea canadensis Rich. ; in Typha latifolia L. – from 387 to 15,113 Bq/kg for Cs-137 and 32-500 Bq/kg for Sr-90; in samples of filamentous algae – from 932 to 70,300 Bq/kg for Cs-137 and 22-214 Bq/kg for Sr-90. The average values of specific activities for 10 years of research have varied in the range of 1,100 – 132,000 Bq/kg for Cs-137 and 67 - 355 Bq/kg for Sr-90 with no stable decrease in average activity values with time, which is probably related to the fact that the reservoir is lotic – in different years and seasons different quantities of radionuclides may be contributed to by the Senna river. The obtained values of accumulation coefficients (AC) for Cs-137 in hydroflora are in the range of 6,790 – 284,900, while AC values for Sr-90 are by 1-2 degrees lower that those for Cs-137 and amount to 321-965. There is no stable decrease in AC for Cs-137 and Sr-90 with years; the observed values of AC for Cs-137 are always higher than those of AC for Sr-90 (AC for Cs-137/AC for Sr-90 > 1). It has been found out that seasonal changes in the development of plants may influence the distribution of radionuclides in various parts of water plants as follows: in Typha latifolia L. the Cs-137 maximum content in vegetation parts is observed in June, and in the root system – in May, i.e. in the beginning of vegetation; in the root system the Sr-90 maximum content is observed in May, and the Sr-90 maximum content in vegetation mass is in the end of vegetation, in Octovber. In Elodea canadensis Rich. that has the longest vegetation period, the Cs-137 content gradually increases from May to July, and then – to October; the Sr-90 content increases from May to July and then decreases. It has been established that that there is a proportional relationship between the Cs-137 and Sr-90 content and that of their microanalogues – the higher concentrations of macroanalogues are to be found in a plant, the more radionuclides are contained.

The studies have shown that if the concentration of Na^+ and K^+ cations (Cs-137 analogues) in water increases from June-July to October, then the Cs-137 content in Elodea canadensis Rich. decreases; and if the concentration of Ca^{2+} and Mg^{2+} cations (Sr-90 analogues) in water increases from June-July to October, then the Sr-90 content also decreases. The values of specific activities of detritus are in the range of 1,970 – 114,700 Bq/kg, and those of Elodea canadensis Rich. – 1,860 and 10,700 Bq/kg. It has been revealed that there is more Cs-137 in detritus, than in a living plant by 1.2 times on the average, and there is almost always more Sr-90 in living vegetation compared to detritus. The area of water vegetation did not exceed 1,756 sq. m or 0.66% of the water-surface area; the relatively small area of water vegetation is, in the first instance, due to the fact that the Malinovka reservoir is a young water system aged about 20 years and lotic. The maximum vegetation area is characteristic of Typha latifolia L. being from 319.2 to 1088.72 sq. m and Elodea canadensis Rich. being from 239.4 to 590.52 sq. m as well as low algae being up to 234.5 sq. m. There have been calculated the stocks of radionuclides in the phytocoenotic component of the reservoir ecosystem that are $112 \times 10^3 – 8038 \times 10^3$ Bq for Cs-137 and 22×10^3 - 213×10^3 for Sr-90. The stocks of Cs-137 in phytomass exceed those of Sr-90 by a factor of $10\text{-}10^2$ on the average. It has been established that the contribution of plant hydrobionts to the total radioactivity of major components of the reservoir ecosystem does not exceed 0.02% for Cs-137 and 0.01% for Sr-90 by virtue of the insignificant age of the reservoir and the small area of water vegetation and, consequently, low biomasses of biocoenosis-making species. The average specific activities of Sr-90 for water vegetation in the reservoir are greater than those of the river itself by 2.25 times, whilst the indices for Cs-137 are almost the same, which can be explained by a higher content of Sr-90 in the reservoir ecosystem, especially, in the bottom sediments compared to that of Cs-137 as well as by the fact that forms of strontium forms with time become more mobile, whilst caesium gets inerter.

The statistical analysis of study results was made with STATGRAPHICS Plus for Windows software. It has been established that statistically significant relationships in the correlation analyses of general population data including results for all the years and sampling seasons are observed in the Malinovka reservoir only for Cs-137 in the following comparison groups: ‘suspensions – bottom sediments’, ‘suspensions – water vegetation’, ‘bottom sediments – water vegetation’. The obtained correlation coefficients (rhos) are 0.58, 0.69 and 0.73 (for the three comparison groups respectively) show an average and close degree of statistical relationships between characteristics in the comparison groups and, alongside with the obtained determination coefficients (0.34, 0.48 and 0.53 respectively), are statistically reliable ($p < 0{,}05$, i.e. $P>95\%$).

There was a statistical analysis made for different groups of the reservoir components only in vegetation periods. A study of the results shows that correlation relationships in the comparison groups, where water vegetation is a part, are observed significantly more often compared to statistical analysis data for the general population as a whole, not taking into account the seasonal factor. In particular, close correlation relationships have been established in the following comparison groups: ‘water Cs-137 – water vegetation Cs-137', 'water Sr-90 – water vegetation Sr-90', 'suspensions Sr-90 – water vegetation Sr-90', 'bottom sediments Cs-137 – water vegetation Cs-137’, ‘soil Cs-137 – water vegetation Cs-137' with correlation coefficients r = 0.7; 0.53; 0.47; 0.99; 0.51 respectively.

As an regression analysis showed, trend lines for all the components of the reservoir and river averaging the dynamics of the radioecological situation are a second-degree parabolas y= a+bx+cx2.

It has been shown that the Malinovka reservoir is a young water ecosystem, where plant communities are essentially in the making only. The major contribution to the total radioactivity is made by bottom sediments – over 90% for Cs-137 and 57-99% for Sr-90. It is likely that with time, as the area of water vegetation and the total biomass of water vegetation increase, the contribution of this component to the total radioactivity will be more weighty, provided that hydrological parameters, technical characteristics and radioecological indices of the reservoir do not change.

Concerning the Senna river at the village of Chudyany, the radioecological situation here can change more often, for it is lotic, and is to a higher degree dependent on climatic factors, mainly on the frequency and quality of precipitation defining the ingress of radionuclides from headwaters areas.

Quantitative and temporal assessment of ^{137}Cs and ^{90}Sr biofixation by organic wastes in agro-ecosystems

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Abstract. Each year considerable quantities of residue from different crops grown on contaminated lands are tilled back into the soil and partly used as forage. Radionuclides in the crop residues are made available to succeeding plants by release thought decomposition. Relative phytoextraction of radionuclides by crops to soil content, quantitative and temporal rate of ^{137}Cs and ^{90}Sr released from plant residues and annual fixation of radionuclides by plant residues tilled into the soil for succeeding crops in rotation were studied. Different crops have shown considerable variation in their ability to uptake the radionuclides from soil. The relative accumulation of ^{137}Cs by the crop residues was lower 0.07% to the total radionuclides content in soil. The maximum quantity of ^{90}Sr extracted from soil can reach up to 4% and it is valuable in respect of the technologies of phytodecontamination. Agricultural crops accumulating the high amounts of ^{90}Sr were rapeseed, clover and Sakhalin buckwheat. The size of ^{90}Sr immobilization by above-mentioned crops can be comparable with 'self-decay' of radionuclides per year. The release of radionuclides incorporated in the straw is very slow and radionuclides of plant organic wastes tilled in the soil are unavailable to succeeding crops at least during 2 years.

1. INTRODUCTION

Agricultural production in Belarus is conducted on 1.3 million hectares of land contaminated by ^{137}Cs with deposition 37-1480 $kBq.m^{-2}$. Some part of this land, 0.46 million hectares, is simultaneously contaminated with ^{90}Sr as well (6 -111 $kBq.m^{-2}$). In some cases, the choice of a crop for a rotation depends upon the benefits of getting of food-grade products with minimum concentration of radionuclides on each separate field. In other cases, for example, cultivation of rapeseed (*Brassica napus f.*) and Sakhalin buckwheat (*Poligonum sachalinense F. Schmilt*) on the contaminated lands is frequently considered as one of the ways of obtaining alternative biodiesel and biogas sources with simultaneous soil decontamination [1, 2].

Each year considerable quantities of residue from crops grown on contaminated lands are tilled back into the soil and partly used as forage. About 2.5 million ton of crop residue made on contaminated area can be regarded as a organic wastes. The elements in crop residues are made available to plants by release thought decomposition. The rate at which the elements, including radionuclides, become available is an important factor in determine the danger of the organic wastes as a source of radionuclides to succeeding crops. The rate of straw decomposition and release of elements varies, depending on the size of residues, the part of the plant and tissue involved such as cellulose, hemicellulose, lignin, etc [3, 4].

The purpose of the investigations was to estimate the magnitude of radionuclides fixation by biomass of crops and the rate of release of radionuclides from plant residues after their tillage in soil for succeeding crops. Three factors were studied: (a) relative phytoextraction of radionuclides by crops to soil content, (b) quantitative and temporal rate of ^{137}Cs and ^{90}Sr release from plant residues, (c) annual fixation of radionuclides by plant residues tilled into the soil for succeeding crops in rotation.

2. MATERIALS AND METHODS

2.1. Experiment I (phytoextraction)

Quantitative assessment of ^{137}Cs and ^{90}Sr biofixation by crops was conducted on the database of multiyear results received from trial plots. The field experiments were conducted on the sod-podzolic loamy sand soil or Podzoluvisol of FAO classification. Agrochemical parameters of trial plots widely varied: pH (1N KCl) 4.7 - 7.3, extractable P_2O_5 - 25 -1008 ppm (0.2N HCl extraction) and K_2O - 120 -

480 ppm (0.2N HCl extraction), exchangeable Ca – 320 - 2100 ppm(1N KCl extraction) and Mg - 40 - 205 ppm (1N KCl extraction). Density of soil contamination varied for ^{137}Cs 17-2000 $kBq.m^{-2}$ and for ^{90}Sr 3 - 77 $kBq.m^{-2}$. Relative phytoextraction of radionuclides by crops to soil content was calculated by multiplying the radionuclide concentration of plant by a crop yield from a unit of area divided by total contents of radionuclide of a unit of area and after that converted in percent. Table 2 shows a set of crops evaluated.

2.2. Experiment II (phytoavailability)

In the pot experiment the sod-podzolic loamy sand soil collected from uncontaminated area was treated by contaminated straw of rapeseed (*Brassica napus f.*), Sakhalin buckwheat (*Poligonum sachalinense F. Schmilt*) and rye (Secale cereale) collected from the area of Chernobyl fallouts. The straw belonged to plants with final maturity. The straw was cut in equal lengths of 3-4 centimeters. The crop residues were added to the soil at 0.1 kg per pot or 6 kg of soil. A control was included in the treatments to take into account the global fallout.

Agrochemical parameters of soil of the pot experiment were: pH (1N KCl) 5.9, extractable P_2O_5 – 820 ppm (0.2N HCl extraction) and K_2O - 430 ppm (0.2N HCl extraction), exchangeable Ca – 1300 ppm(1N KCl extraction) and Mg - 140 ppm (1N KCl extraction).

Each treatment was replicated 4 times. The soil was brought up to a moisture content of about 60% of the field water holding capacity. In order to approach the experiment to the climatic conditions of agriculture of the contaminated zone the crop residues were incubated in the soil of plastic pots in the field conditions. Terms of the incubation of the contaminated straw into the soil were 180 days during winter season. In the first year, lupine was sown on the treated soil and green rye in the second year. Treatments are presented in the Table 2.

Before sowing of the second crop the activity of soil enzymes was analyzed. Parameters of enzymatic activity are expressed as a quantity of modified substratum for the defined time: oxidation-reduction (dehydrogenase) - mg of TPF for 24 hours per 100 g of soil, invertase - mg of glucose for 4 hours on 100 g per soil, urease - mg $N\text{-}NH_4^+$ for 4 hours per 100 g of soil. Potential cellulose-decomposing capacity was determined by weight method at the decomposition of filter paper in the soil for 60 days [5].

2.3. Measurements

The ^{90}Sr content in soil and plants was determined with use of the liquid scintillation counter (CANBERRA Tri-Carb 2750 LL). Determination of ^{137}Cs was done by Gamma-spectrometer with HP GC4019 detector (CANBERRA) [6]. The laboratory measurements have been performed at the Laboratory for Radioecology of the Belorussian Research Institute for Soil Science and Agrochemistry (BRISSA).

3. RESULTS AND DISCUSSION

3.1. Phytoextraction of radionuclides

Accumulation of ^{137}Cs and ^{90}Sr by crops with edible and secondary parts is given in Table 1. Different crops have shown considerable variation in their ability to uptake the radionuclides from soil [7]. Relative radionuclide uptake by agricultural crops depended on agrochemical soil parameters, specificity of kinds and varieties, crop yield and fertilization. Accumulation of ^{137}Cs by crops was insignificant. Cereals which annually cultivate on 50-70% of plough soils accumulate no more 0.02% of ^{137}Cs and other crops no more 0.07% to the total radionuclides content in soil. Accumulation of ^{90}Sr by crops was significantly higher than could be explained by less strong fixation of ^{90}Sr with soil complex than ^{137}Cs. The maximum quantity of ^{90}Sr is accumulated by rapeseed 3.85%, by clover 3.60 and by Sakhalin buckwheat 1.95% to the total ^{90}Sr content in the soil.

The size of ^{90}Sr immobilization by rapeseed straw, clover and Sakhalin buckwheat is comparable with ‘self-clean-up’ of soil as a result of annual decay of radionuclides and is valuable in the respect of the technologies of phytodecontamination.

TABLE 1. PHYTOEXTRACTION OF RADIONUCLIDES BY GROSS OUTPUT OF CROPS ON PODZOLUVISOL (% TO TOTAL RADIONUCLIDE CONTENT IN SOIL)

Crop	^{137}Cs accumulation			^{90}Sr accumulation		
	Edible part*	Secondary part**	Total	Edible part	Secondary part	Total
			Food crops			
Cereals	0.0004-0.002	0.003-0.02	0.003-0.02	0.02-0.06	0.20-0.31	0.22-0.37
Potato	0.006-0.009	0.01-0.02	0.016-0.029	0.04-0.06	0.09-0.14	0.13-0.20
			Forage crops			
Annual legume	–	0.02-0.03	0.02-0.03	–	0.43-1.2	0.43-1.2
Clover	–	0.03-0.04	0.03 0.04	–	2.1-3.6	2.1-3.6
Perennial grass	–	0.02-0.07	0.02-0.07	–	0.27-0.49	0.27-0.49
Corn	–	0.008-0.009	0.008-0.009	–	0.16-0.30	0.16-0.30
			Industrial crops			
Rapeseed	0.003-0.008	0.004-0.01	0.007-0.02	0.10-0.32	1.09-3.53	1.19-3.85
S. buckwheat	–	0.006-0.02	0.006-0.02	–	0.56-1.95	0.56-1.95

* grain, seed, tubes
** straw, green mass, hay, leafy tops

TABLE 2. INFLUENCE OF KIND OF CROP RESIDUE ADDED IN SOIL ON THE PERCENTAGE OF THE RADIONUCLIDES IN LUPIne (2001) and green rye (2002) PLANTS DERIVED FROM THE CROP RESIDUE

Crop residue added	Activity of radionuclides added by straw, Bq per pot		Yield of dry matter, g per pot		^{137}Cs in crops derived from crop residues, %		^{90}Sr in crops derived from crop residues, %	
	^{137}Cs	^{90}Sr	Lupine	Rye	Lupine	Rye	Lupine	Rye
None	–	–	15.2	33.4	–	–	–	–
Rye straw	10.2	15.0	19.4	36.9	2.0	6.2	1.5	3.0
S. buckwheat	13.8	36.2	18.7	40.9	0.9	4.9	0.9	2.0
Rapeseed straw	11.4	15.0	16.0	36.6	0.9	4.0	0.8	2.6
LSD 0.05	–	–	2.34	3.21	–	–	–	–

TABLE 3. ACTIVITY OF SOIL ENZYMES DEPENDING ON KIND OF CROP RESIDUE ADDED IN SOIL

Crop residue added	Dehydrogenase		Invertase	Urease	Cellulose-decomposing capacity, (%)
	Wet soil	Dry soil			
None	7.1	9.7	11	6.3	35
Rye straw	7.6	10.5	27	18.3	61
Sakhalin buckwheat	4.9	6.8	18	13.0	51
Rapeseed straw	6.2	8.3	22	15.3	56
LSD 0.05	2.1	2.4	3.2	3	5.9

3.2. Phytoavailability of radionuclides

In the first year of the experiment green mass of lupine accumulated lower then 2% of ^{137}Cs and 1.5% of ^{90}Sr contented in contaminated straw tilled. In the second year radionuclides uptake by green mass of rye was 4-6% for ^{137}Cs and 2-3% for ^{90}Sr (Table 2). Higher transfers of radionuclides from rye straw can be explained in smaller diameter of a stalk and accordingly its closer contact with soil. Experiments show that release of radionuclides incorporated in the straw is very slow and radionuclides of organic plant wastes are unavailable for green plants at least during 2 years.

As a whole all samples are characterized by the lowered level of the enzyme activity of soil. Enzymatic activity of untreated soil was in several times lower in comparison with soil treated by crop residues. Cellulose-decomposing capacity of treated pots was in 1.5-1.7 time higher than soil of the control variant (Table 3).

4. CONCLUSIONS

These preliminary studies were carried out to estimate the immobilization of radionuclides by organic wastes in agriculture. Different crops have shown considerable variation in their ability to uptake the radionuclides from soil. The relative accumulation of ^{137}Cs by the crop residues was insignificant and lower than 0.07% to the total radionuclides content in soil. The maximum quantities of ^{90}Sr extracted from soil can reach up to 4% and it is valuable in respect of the technologies of phytodecontamination. Agricultural crops accumulating the high amounts of ^{90}Sr were rapeseed, clover and Sakhalin buckwheat. The size of ^{90}Sr immobilization by above-mentioned crops can be comparable with «self-decay» of radionuclides per year. The release of radionuclides incorporated in the straw is very slow and radionuclides of plant organic wastes tilled in the soil are unavailable to succeeding crops at least during 2 years.

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Effect of soil parameters on uranium availability to ryegrass

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Abstract. When wishing to assess the impact of radioactive contamination on biota or on an ecosystem, knowledge on the physico-chemical conditions governing the radionuclide availability and speciation in the exposure medium and hence its bioavailability and incorporation is indispensable. The present study explores the dominant soil factors (18 soils collected under pasture) ruling uranium mobility and availability to ryegrass and intents to define and assess the extent of the effect. Soil pH and iron-oxi-hydroxides explained for 60% the uranium concentration found in the soil solution (which varied with factor 100). Plant U-concentration was mostly affected by the concentration of U in the soil solution, pH and total inorganic carbon content. Observed U-uptake was highest when pH was below 5.3 or around 7 or higher. Uranium speciation was found important in explaining the U-uptake observed: apparently, uranyl, $UO_2CO_3^{-2}$ and $(UO_2)_2CO_3(OH)_3^-$ were the U-species being preferentially transported.

1. INTRODUCTION

Enhanced levels of naturally occurring radionuclides (NOR) may be associated with abandoned waste dumps and the surroundings of industries involved in the extraction or processing of raw materials containing NORs. The most prominent examples in Europe are the residues of uranium mining and milling, the sludge heaps of the phosphate processing industry and the ashes from power production from coal. These contaminated environments may result in a considerable exposure to the public and the environment.

When wishing to assess the impact of radioactive contamination on biota or on an ecosystem, knowledge on the physico-chemical conditions governing the radionuclide availability and speciation in the exposure medium and hence its bioavailability and incorporation is indispensable in order to deduce a potential relation between environmental concentrations, the fraction bioaccumulated (and the location of the contamination in the target organism or compartment and this in function of time) and the observed effect.

Uranium behaviour is similar to that of other heavy metals and its physiological toxicity, other than damage from ionising radiation, mimics that of lead. Uranium is chemically toxic to kidneys and insoluble U-compounds are carcinogenic [1]. The complex, pH-dependent speciation of U in soils makes the study of U uptake by plants difficult. U is present in soil primarily (80-90%) in the +VI oxidation state as the uranyl (UO_2^{+2}) cation [2, 3]. Under acidic conditions, UO_2^{+2} is the predominant U species in the soil. Hydroxide complexes form under near-neutral conditions, while carbonate complexes predominate under alkaline conditions. Generally, negligible amounts of UO_2^{+2} are present in the soluble and exchangeable forms, due to the high solid-liquid distribution coefficient of uranium.

Although extensive work has been done on the U solubility in soils, there is comparatively little information regarding the uptake and translocation of U by plants as affected by soil properties. Most work has focused primarily on either the U content of native plants growing in U contaminated environments [3] or U uptake by field and garden crops of importance to animals and humans [4].Ebbs et al. [2] showed that free UO_2^{+2} is the U species most readily taken up and translocated by plants.

The present study explores the dominant soil factors ruling uranium mobility and availability to plants and tries to define and assess the extent of the effect.

2. MATERIALS AND METHODS

Eighteen soil types were collected under pasture. The soils were selected such that they covered a wide range for those parameters hypotised as being potentially important in determining U-availability (pH, clay content, Fe and Al oxide and hydroxide content, $CaCO_3$, organic carbon). Soils were analyzed for texture, total organic and inorganic carbon, CEC, total Fe and P, density, field capacity. Soils (3 kg for each soil type) were brought to field capacity and contaminated with 450 Bq/kg ^{238}U(VI) (added as uranylnitratehexahydrate), fertilized (NK) and incubated for 4 weeks. Before the start of the experiment, soils were analyzed for pH, exchangeable cations, Olson P. Soil solution was analyzed for cation content, phosphate, sulphate, chloride, nitrate, uranium. Uranium was also analyzed in the supernatants following extraction with ammoniumacetate (pH 5 and 7) and ammoniumoxalate. These latter extracts should give information about which fraction uranium was primarily bound. After 4 weeks' incubation, the 3-kg soil batches were distributed over 3 pots and ryegrass was grown on top. After 5 weeks' growth, ryegrass was harvested; the ryegrass was dried and analyzed for biomass and uranium content in the shoots. Statistical analysis was performed with the software *Statistica* (General Linear Models).

3. RESULTS AND DISCUSSION

Table 1 presents mean values of some selected soil characteristics and the uranium content in shoots.

Statistical analysis showed that there were no single soil parameters significantly explaining the uranium concentration in the soil solution, nor the uranium concentration in the plants.

Preliminary testing showed that soil pH and the oxalate extractable iron explained for 60% the uranium concentration found in the soil solution, as expressed by the following equation:

$$U\text{-}SoilSol=54.01(\pm 16.51) + 7.26(\pm 2.92)\ pH - 0.0006\ (\pm 0.0002)\ Fe\text{-}Oxal,\ R^2=0.60$$

Plant U concentration was mostly affected by the concentration of uranium in the soil solution, the soil pH and the Total Inorganic Carbon (TIC) content in the soil solution.

$$U\text{-}Plant=-0.45(\pm 0.21) + 0.06(\pm 0.04)\ pH + 0.025(\pm 0.013)\ TIC+0.011\ (\pm 0.002)\ U\text{-}SoilSol,\ R^2=0.71$$

TABLE 1. SOME SOIL CHARACTERISTICS AND U CONCENTRATION IN SOIL SOLUTION AND RYEGRASS SHOOTS

	Soil class	0-2 µm %	O.M. %	pH (H_2O)	CEC meq/100g	Fe (Oxal-extr) µg/g	HPO_4^{2-} mg/l	TIC mg C/L	U in soil sol. µg/l	U in plant µg/g
A	loamy clay	18.02	2.65	5.76	8.7	2922	0.98	0.77	7.61	0.182
B	light clay	24.56	11.62	6.27	43.8	20367	1.15	1.32	1.74	0.058
C	sandy clay	30.03	15.37	6.56	51.7	13090	0.88	1.85	1.81	0.012
D	sandy clay	28.76	3.20	5.10	5.1	2176	0.86	0.53	20.57	0.143
E	loamy sand	24.86	3.95	6.65	7.8	1143	2.45	0.75	6.64	0.056
F	sandy clay	24.91	7.11	7.64	32.2	1286	17.67	4.42	2.89	0.230
G	light clay	19.9	9.75	6.92	42.3	3161	9.04	4.81	1.97	0.032
H	light clay	27.33	10.34	7.61	48.2	5142	1.48	6.49	11.33	0.310
I	sandy clay	20.05	4.03	8.03	24.0	887	1.17	9.46	231.61	1.223
J	clayey sand	16.32	4.34	6.36	8.9	896	0.65	0.63	11.47	0.039
K	loamy clay	26.72	6.00	7.01	20.7	2761	0.90	5.41	19.39	0.393
L	clay	27.82	7.42	5.88	24.5	3527	1.18	1.53	7.35	0.014
M	heavy sand loam	15.66	14.65	6.77	65.8	13231	1.36	2.79	1.91	0.013
N	clayey sand	11.7	5.61	5.38	8.5	1869	9.68	0.68	17.24	0.024
O	clayey sand	14.79	4.70	5.64	9.7	2358	0.90	0.80	10.87	0.020
P	clayey sand	10.06	4.35	6.14	8.1	1469	0.78	0.80	11.96	0.021
Q	light clay	21.36	7.51	6.24	17.0	6258	1.12	1.97	3.00	0.012
R	clayey sand	15.63	4.29	6.20		1998	0.80	1.00	8.78	0.056

The next step was to assess the uranium speciation in the soil solution with a Geochemical Speciation Model and to assess if and how speciation may influence the translocation of uranium to the shoots.

Figure 1 shows that for the soils were increased transfer was observed (soils a, d, f, h, i, k) again certainly not the soil solution concentration but neither a rough-speciation repartitioning could explain the transfer factors observed.

Apart from the uranyl-species, carbonate species, occurring at pHs around 7 or higher, are reported to be rather bioavailable [2]. It can be deduced from Figure 2 that, apparently, uranyl, and just some U-carbonate species $UO_2CO_3^{-2}$ and $(UO_2)_2CO_3(OH)_3^-$ were the U-species being preferentially transported.

Relating specific U-species concentrations with the bioaccumulated fractions results in clearly better correlations than when simply taking the soil solution concentration (Figure 3).

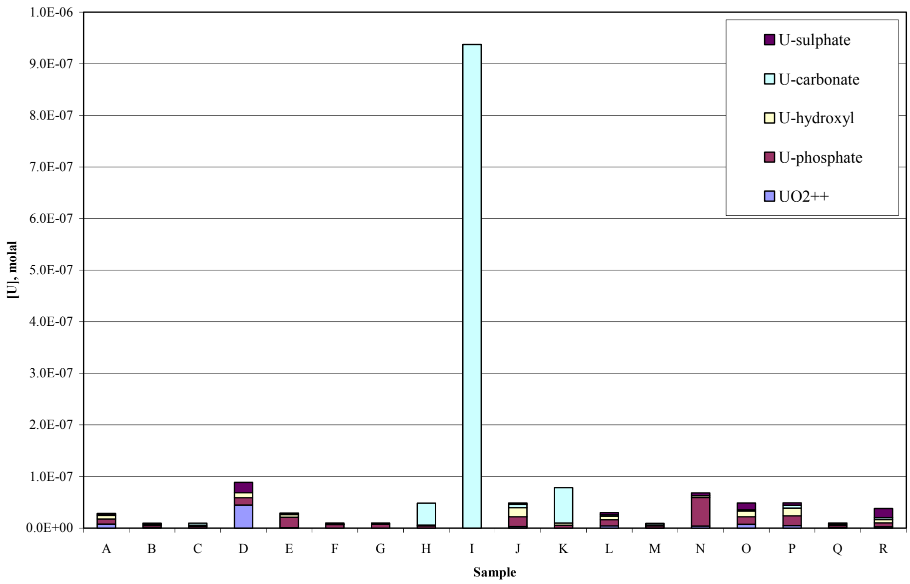


FIG. 1. Uranium speciation for the different soil samples.

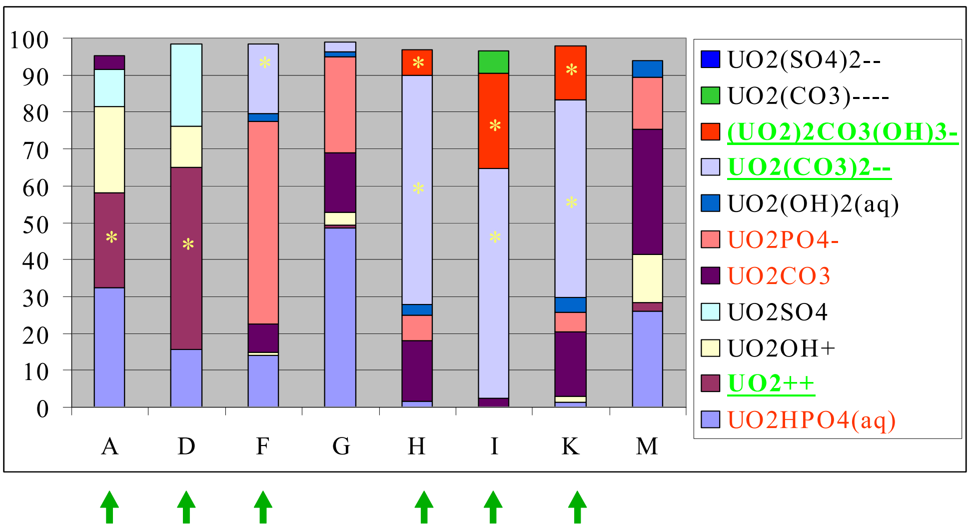
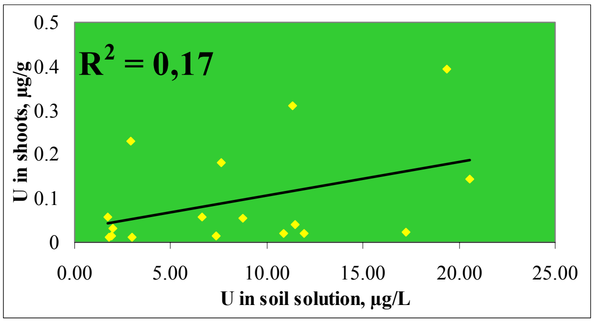


FIG. 2. Detailed U-speciation of the soil samples where higher soil-to-ryegrass shoot transfers were observed. Samples with arrows have high transfers; U-species preferentially bioaccumulated are underlined.



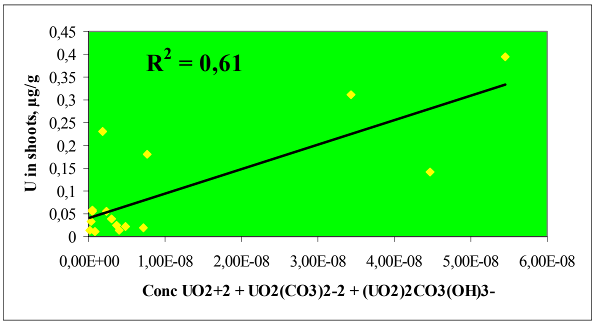


FIG. 3. Relation between the uranium concentration in the soil solution (left) or the concentrtaion of specific uranium species (right) and the uranium concentration in the shoots.

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A study of iodine aerosol deposition on some crops, vegetables and grass

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Abstract. In order to improve and verify the knowledge of radioiodine transfer in Asian biosphere system, a closed space was established to study gaseous iodine deposition in a simulated terrestrial food exposure pathway using ^{125}I as a radiotracer. The pot culture test was carried out to study airborne ^{125}I deposition on corn, soy, navy bean, cabbage, shallot and vanilla grass, and results show that (1) No plants absorb ^{125}I initiatively; (2) ^{125}I aerosol deposition on leaves could be transferred to other tissues; (3) Corn and navy bean have the biggest translocation factor. The ^{125}I soil-to-crops uptake test shows that ^{125}I deposited in soil could transfer to plants via root uptake, and millet and broomcorn have the biggest transfer factor.

1. INTRODUCTION

^{131}I ($T_{1/2}$ 8.04d) and ^{129}I ($T_{1/2}$ 1.57×10^7a), as the most concerned radioactive iodine isotope, are fission products that can be released in large quantities in the event of an accident involving nuclear power plants. Because of their abundance and biological mobility, they are critical components in nuclear fission wastes, and are the two best studied artificially produced radionuclides in our environment.

A major radioecological task in the future is a reconstruction of the radioiodine deposition to make an assessment of the individual doses to these population groups [1]. Iodine could enter human body through different pathway, in which at most start in airborne pathway. In Asia, crop and vegetable are the most important for Asian food ingredient. Therefore, air-vegetable-human is the critical pathway in Asia for iodine exposure. At the moment, not too much data is available relating to the behaviour of radioiodine in Asia. In order to improve and verify the knowledge of radioiodine transfer in Asian biosphere system, the present work aims to define its behaviour in a simulated terrestrial food exposure pathway of gaseous iodine using ^{125}I ($T_{1/2}$ 60.1 d).

2. MATERIALS AND METHODS

2.1. Study site and gaseous ^{125}I generation

A field site 22.5m^2 was established. The space closed by plastic approximately 30 m^3 was set up for gaseous ^{125}I deposition tests:

$$2Na^{125}I + MnO_2 + 3H_2SO_4 = 2NaHSO_4 + MnSO_4 + 2H_2O + {}^{125}I_2\uparrow$$

^{125}I gas was generated by above chemical reaction from a 250 mL vessel containing 407 MBq Na^{125}I. The vessel was set on a field corner in the space, and a mini-fan was used to simulate natural wind. ^{125}I gas diffused in the space immediately with the reaction.

2.2. Gaseous ^{125}I depostion test on crops, vegetables and grass

The pot culture was carried out to study gaseous ^{125}I deposition and translocation in plants. Corn, soy, navy bean, cabbage, shallot and vanilla grass were selected for the test. ^{125}I gas contamination happened when all plants grown up to seeding stage. The pots were moved into the space and placed at a distance of 5m of the source point. ^{125}I deposition lasted 1 hour. Then all pots were moved out from the site. Afterward, the crops, vegetables and grass were grown in the pot under artificial sprinkling irrigation. Samples were collected at the 15th day after contamination. Each sampling has three samples.

2.3. ^{125}I soil-to-crops uptake test

In order to study ^{125}I soil-to-crops uptake, 0.5 cm thick culture soil was spread on ground in the space, in which the gaseous ^{125}I deposited on soil for 1 hour. Then the soil was collected and moved out from the site and mixed evenly. The soil with ^{125}I specific activity of 43 Bq/kg (d.w., dry weight, same below) was prepared and filled into pot. Afterwards, pot culture test was carried out in the most representative plants as corn, wheat, rice, millet, broomcorn, cole and soy under the normal fertilization and irrigation. The leaf sample was collected at the 30th day after seedling. Each sampling has three samples.

2.4. Instrument and measurement

All samples were weighted by Sartius electronic balance (0.1mg sensitive) and measured immediately after sampling using FT613 micro process γ counter which has a counting efficiency of 75% for ^{125}I. The counting errors were less than 10% in 2σ.

3. RESULTS AND DISCUSSIONS

3.1. ^{125}I aerosol deposition test on crops, vegetables and grass

In Asia, the basic food for Chinese people are rice, wheat and vegetable. The principal exposure pathway is via these food to human. Concerning the airborne exposure, especially for iodine aerosol falls on the aerial part of plants, some Asian basic food as leafy crop and vegetable represent the important role in the pathway. Subsequently, some crop and vegetable as corn, soy, navy bean, cabbage, shallot and vanilla grass are selected to study gaseous iodine deposition.

3.1.1. Corn

^{125}I activity in different corn leaves after 1 hour deposition is 0.65 ± 0.12Bq/kg (f.w., fresh weight, same below) in the first leaf, 0.63 ± 0.25 Bq/kg in the second leaf, 0.40 ± 0.09 Bq/kg in the third leaf, 0.22 ± 0.21 Bq/kg in the fourth leaf, which show that deposition activity in different leaves differs no significantly with the same order of magnitude values. Generally, the activity deposited on flourishing leaves that have large interception area and a small inclination to ground is more pronounced than other leaves. Because of growth dilution, specific activity of ^{125}I in leaves decreases with growth period after deposition. ^{125}I deposited on the leaves can transfer and re-distribute in other corn tissues because ^{125}I was measured in these new leaves and stem.

3.1.2. Navy bean

The measurement results show that specific activity of ^{125}I in navy bean leaves is 1.7 ± 0.9Bq/kg, and 0.05 ~ 2.3Bq/kg in beanpod. ^{125}I, furthermore, can transfer into new leaves and stems, with at most accumulated in the stem tip. The result also shows that ^{125}I can translocate into fruit during growth afterward, and its specific activity is 0.002 ~ 0.03 Bq/kg.

3.1.3. Soy

Specific activity of ^{125}I in soy leaf is 2.4 ~ 2.5 Bq/kg. Afterward, ^{125}I on the leaf can transfer into new leaves and bean. The content of ^{125}I in bean is lower 2 ~ 3 order of magnitude than leaves.

3.1.4. Cabbage

Result shows that specific activity of ^{125}I in cabbage is 0.3 ± 0.1Bq/kg. ^{125}I can transfer to all plant with specific activity of 0.001 ~ 0.1Bq/kg after 15 days growth.

3.1.5. Shallot

Specific activity of ^{125}I in shallot leaves is 0.5±0.3 Bq/kg. Since leaves wrap the shallot stem, few ^{125}I contaminate the stem. The measurement show that ^{125}I can transfer to all new leaves at most accumulated at leaf tip, and further into the stem with the specific activity of 0.02 ~ 0.2 Bq/Kg in 15 days growth.

3.1.6. Vanilla grass

Specific activity of ^{125}I in the grass is 0.14 ~ 0.34Bq/kg. ^{125}I can tranfer to new growth leaves after a period of 15 days with specific activity of 0.001~0.035Bq/kg, in which ^{125}I in leafstalk less than leafage.

3.1.7. ^{125}I Translocation in the plants

Foliar absorption of radionuclides is believed to consist of two phases. The first one is nonmetabolic cuticula penetration which is generally considered to be the major route of entry and metabolic mechanisms which account for element accumulation against a concentration gradient. The second process is responsible for transporting ions across the plasma membrane and into the cell protoplast. The results obtained show that gaseous ^{125}I contamination to plants in the present test is a natural process of ^{125}I dry deposition, and no plants absorb ^{125}I initiatively. It should be noted that the impact of resuspension is not considered here because of the test was carried on a closed small space where the impact could be no pronounced.

Since all tests in this study keep the same ^{125}I diffusion and deposition pattern as described before, the results obtained are under the same condition and can be comparable. In order to compare ^{125}I translocation ability in different plants, the translocation factor (TLF) is used and expressed here as:

$$\mathrm{TLF} = \frac{\text{Specific activity of } {}^{125}\text{I translocated in other plant tissues } (\text{Bq/kg, f.w.})}{\text{Specific activity of } {}^{125}\text{I deposited on the original leaves } (\text{Bq/kg, f.w.})}$$

TABLE 1. ^{125}I TRANSLOCATION FACTOR (TFL) IN PLANT AFTER A GROWTH PERIOD OF 15 DAYS

Plants	Leaf	Fruit
Corn	0.088~0.318	–
Navy bean	0.007~0.248	0.001~0.018
Soy	0.002~0.006	0.0006~0.001
Cabbage	0.003~0.333	–
Shallot	0.001~0.0035	0.04~0.4 (stem)
Grass	0.004~0.146	–

3.2. ^{125}I soil-to-crops uptake test

There are totally 8 local representative crops and vegetables selected in the test. The leaf sample was collected for measurement. The results obtained in this test verified again that iodine is a mobile element. Iodine could uptake into plant from soil and distributed in all tissues. The soil-to-plant transfer factor (TF) was calculated to quantify the transfer of ^{125}I from soil to crop leaf, which can be defined as:

$$\mathrm{TF} = \frac{{}^{125}\text{I concentration in plant leaves } (\text{Bq/kg, f.w.})}{{}^{125}\text{I concentration in soil } (\text{Bq/kg, f.w.})}$$

TABLE 2. ^{125}I SPECIFIC ACTIVITY (SA, BQ/KG) IN DIFFERENT LEAVES AND THE CORRRESPONDING TF

Plants		1st leaf	2nd leaf	3rd leaf	4th leaf
Corn	SA	0.016	0.006	0.004	0.002
	TF	0.00037	0.00014	0.00009	0.00005
Wheat	SA	0.014	0.006	0.004	0.0005
	TF	0.00033	0.00014	0.00009	0.00001
Rice	SA	0.011	0.001	0	–
	TF	0.00026	0.00002		
Millet	SA	0.099	0.033	0.008	0
	TF	0.0023	0.00077	0.00019	
Broomcorn	SA	0.083	0.023	0.006	0
	TF	0.0019	0.00054	0.00014	
Pea	SA	0.005	0.002	0.003	0.002
	TF	0.00011	0.00005	0.00007	0.00005
Soy	SA	0.006	0.002	0	–
	TF	0.00014	0.00005		
Cole	SA	0.001	0.0001	0	–
	TF	0.00002	0.000002		

4. CONCLUSIONS

Simple pot experiments in a closed space have been done to simulate the deposition, transfer and absorption of airborne ^{125}I in some crop, vegetable and grass when the exposure occurs during the growth of the plants. The following conclusions can be drawn:

(1) ^{125}I generated in the present test may be defined as iodine aerosol. The test shows that the ^{125}I deposition decreases with increasing of release time.

(2) ^{125}I aerosol deposition on the plants is a natural dry deposition process. No plants absorb ^{125}I initiatively.

(3) ^{125}I activity deposited on different leaves of plant during the first hour of release shows no significant difference.

(4) For airborne ^{125}I exposure pathway, the major absorption of ^{125}I in plant is through the foliar deposition and absorption.

(5) ^{125}I aerosol deposition on leaves could be transferred to other tissues. Translocation factor (TFL) defined as the ratio of activity concentration in new growthing tissues to the leaf ^{125}I deposition initially. Corn and navy bean has the biggest TFL in the present test.

(6) ^{125}I deposited on soil could transfer to plants via root uptake. Transfer factor (TF) of ^{125}I in millet and broomcorn are significantly higher than other crops in the present test.

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Modelling integrated transfer of radionuclides to foodstuffs in the Faroe Islands

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Abstract. The Faroese terrestrial environment has received radioactive debris from the nuclear weapons tests in the 1950s and 1960s and from the Chernobyl accident 26 April 1986. The paper presents integrated transfer coefficients of ^{137}Cs and ^{90}Sr from wet deposition to selected foodstuffs. In this context, the integrated transfer coefficient is defined as time-integrated radionuclide concentration in a sample from a unit ground deposition, as e.g. (Bq/l)y per (kBq/m^2). Estimates are also given for the effective ecological halflife of the radionuclide concentrations in the selected foodstuffs. The work is based on the well-known UNSCEAR model, which relates the concentration of a radionuclide in a sample from a given year to the deposition rate of the nuclide from precipitation in the given year and in the year before, and to the accumulated deposition from previous years.

1. INTRODUCTION

The Faroe Islands are located in the North Atlantic Ocean between 61^{o}20'N and 62^{o}24'N and 6^{o}15'W and 7^{o}41'W. The total land area is 1399 km^2. The climate changes gradually from cool temperate oceanic climate at the coast to arctic climate in the mountains, of which the highest is 882 m above sea level (asl). The yearly average air temperature is around 6-7^{o}C in the zone from sea level to 100m asl, with average winter and summer air temperatures around 3-4^{o}C and 9-10^{o}C respectively (Refs [4] and [6]). The annual precipitation varies across the country, from around 820 mm to around 3260 mm.

Radioactivity has been measured in selected samples from the Faroe Islands since the beginning of the 1960s in a co-operation between Risø National Laboratory in Roskilde, Denmark, and various Faroese institutions. The University of the Faroe Islands has been the Faroese partner since the beginning of the 1990s.

The paper presents results from modelling long-term variation of radioactivity in cow milk, lamb meat and drinking water in the Faroe Islands. Model estimates are given for the effective ecological halflives of radionuclides in the foodstuffs and of integrated transfer coefficients, defined as time-integrated radionuclide concentration in an environmental sample from a unit ground deposition (Refs [1] and [8]).

2. MATERIAL AND METHODS

2.1. Sampling and data

The available data for the study are summarized in Table 1. The data have been obtained from annual reports from the Risø National Laboratory in Roskilde, Denmark, (e.g. Ref. [1]) and from personal communication with Dr. Sven P. Nielsen at Risø.

TABLE 1. AVAILABLE DATA FOR THE STUDY

Samples	Fallout rates (3 localities)	Cow milk (3 localities)	Lamb meat	Drinking water
Isotopes	^{137}Cs	^{137}Cs	^{137}Cs	^{90}Sr
Time period	1962-96	1962-96	1962-93	1964-93

The measured fallout rate varies across the country, showing maximum values in 1963 and a pronounced ^{137}Cs signal from the Chernobyl accident in 1986. Precipitation and cow milk have been sampled at three different locations: Klaksvík in the north of the country, the capital Tórshavn in the central part and Tvøroyri in the south. The annual mean precipitation for the 30-year period 1961-90 was 1284mm in Tórshavn and 2334mm in Klaksvík. There are no meteorological measurements from Tvøroyri. The Faroese drinking water is obtained from surface water, and the samples have been collected as tap water in Tórshavn. Lamb meat derive from countrywide samplings. Faroese lamb meat is mostly used for local specialties, while meat for other purposes is imported mainly from Iceland and New Zealand.

2.2. Analyses

The radioactivity in a given sample has been related to the fallout rates in regression models based on the following equation (Refs [1] and [8]):

$$C_i = b_1 \cdot d_i + b_2 \cdot d_{i-1} + b_3 \cdot \sum_{k=1}^{k=\infty} d_{i-k} \cdot e^{-\lambda \cdot k} \tag{1}$$

where C_i is the concentration of a given radionuclide in a sample from year (i), and d_i and d_{i-1} are the deposition rates (kBq/m^2) in the years (i) and (i-1), respectively. Index i=1 represents the year 1950, taken as the first year with radioactive fallout. The sum on the right hand side represents the accumulated deposition from preceding years assuming an effective radiological halflife of T years, corresponding to a decay factor $\lambda = \ln(2)/T$. The estimated ecological halflife of a radionuclide in a given foodstuff is the value of T that gives the best fit between model and observations. It is presumed that all fallout is deposited by the start of a given year. This will underestimate the accumulated deposition, but it is immaterial for this study of long-lived radionuclides.

The observed deposition rates at the respective locations have been used for modelling ^{137}Cs concentration in cow milk and lamb meat, and for modelling ^{90}Sr concentration in drinking water from Tórshavn. An average deposition rate for the Faroe Islands has been calculated from precipitation rates and activities in precipitation at Klaksvík, Tórshavn and Tvøroyri. This average is used for modelling ^{137}Cs concentrations in lamb meat, as the lamb meat has been sampled widely in the country.

The model-estimated integrated transfer coefficient, ITC, is calculated as follows (Refs. [1] and [3]):

$$\text{ITC} = b_1 + b_2 + b_3 \cdot e^{-\lambda}/(1-e^{-\lambda}) \tag{2}$$

where the parameters are as given in equation (1). The integrated transfer coefficient estimates the transfer coefficient of a radionuclide from fallout to the considered foodstuff.

3. RESULTS AND DISCUSSION

The results are presented in Table 2 and Figures 1–3. The model is found to represent the observations fairly well with high values for the square regression coefficient, R^2, in all cases except for lamb meat. The relative low R^2 in the case of lamb meat may partly be explained by the fact that the meat samples have been collected countrywide from a few animals each year, as it is well-known that the ^{137}Cs concentration in lamb meat varies significantly between animals even from a single pasture (Ref. [5]).

The model has been run in two steps in the case of cow milk: with all available data taken into account in the regression and with only data before 1986 taken into account. The R^2 is found to be slightly higher when only pre-Chernobyl observations are taken into account in the regression analyses.

The effective radioecological halflife of ^{137}Cs is estimated to 3-5 years for cow milk, 5.5 years for lamb meat and 5.5 years for ^{90}Sr in drinking water from Tórshavn (Table 2).

TABLE 2. COEFFICIENTS IN THE MODEL, WITH P-VALUES FROM T-TEST OF THE MODEL COEFFICIENTS GIVEN IN BRACKETS. R^2 IS THE SQUARE REGRESSION COEFFICIENT. ITC REPRESENTS THE MODEL-CALCULATED INTEGRATED TRANSFER COEFFICIENT, AND T IS THE ESTIMATED EFFECTIVE ECOLOGICAL HALF-LIFE IN YEARS

	b_1	b_2	b_3	T	R^2	ITC
^{90}Sr in drink. water; Tórshavn 1964-93	0.0205 (0.486)	0.0397 (0.018)	0.0114 (<0.001)	5.5	0.908	0.145 (Bq/l)y per kBq/m^2
^{137}Cs in lamb meat; Faroes 1962-93	0.1487 (<0.001)	0.0138 (0.594)	0.0168 (0.003)	5.5	0.675	288.0 (Bq/kg ww)y per kBq/m^2
^{137}Cs in milk from Klaksvík 1962-96	0.0075 (<0.001)	0.0086 (<0.001)	0.0040 (<0.001)	3.0	0.903	31.3 (Bq/l)y per kBq/m^2
^{137}Cs in milk from Tórshavn 1962-96	0.0093 (<0.001)	0.0171 (<0.001)	0.0041 (<0.001)	3.0	0.954	42.0 (Bq/l)y per kBq/m^2
^{137}Cs in milk from Tvøroyri 1964-93	0.0112 (<0.001)	0.0186 (<0.001)	0.0055 (<0.001)	4.5	0.982	62.6 (Bq/l)y per kBq/m^2
^{137}Cs in milk from Klaksvík 1962-85	0.0080 (<0.001)	0.0111 (<0.001)	0.0030 (<0.001)	4.0	0.968	35.2 (Bq/l)y per kBq/m^2
^{137}Cs in milk from Tórshavn 1962-85	0.0094 (<0.001)	0.0189 (<0.001)	0.0036 (<0.001)	3.5	0.977	44.7 (Bq/l)y per kBq/m^2
^{137}Cs in milk from Tvøroyri 1964-85	0.0073 (0.062)	0.0222 (<0.001)	0.0050 (<0.001)	5.0	0.986	63.3 (Bq/l)y per kBq/m^2

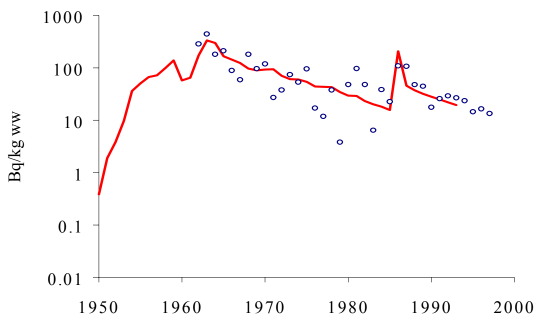

FIG. 1. Observations (ₒ) and model results (—) of ^{137}Cs in Faroese lamb meat. R^2=0.675

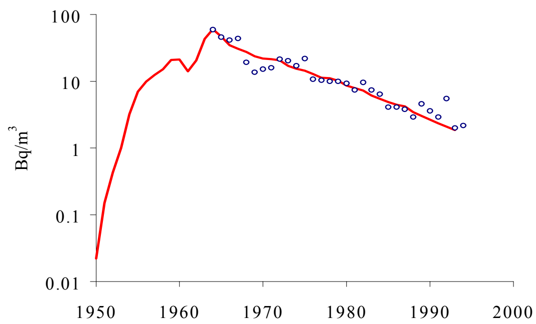

FIG. 2. Observations (ₒ) and model results (—) of ^{90}Sr in drinking water from Tórshavn R^2=0.908

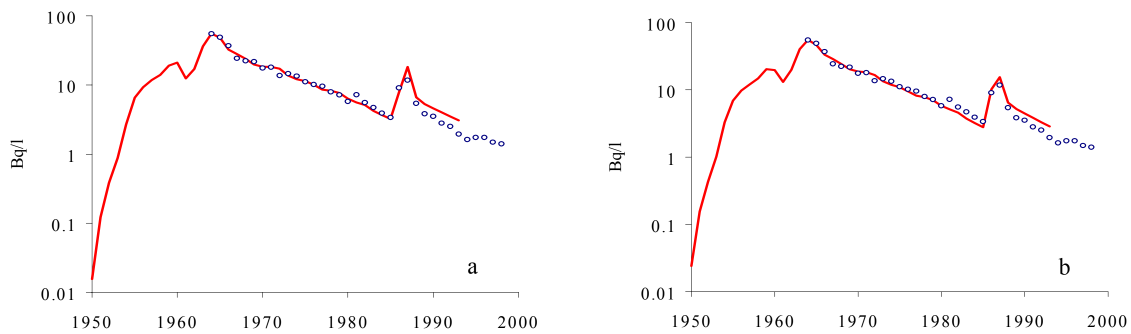

FIG. 3. Observations (ₒ) and model results (—) of ^{137}Cs in cow milk from Tvøroyri. Regression: (a) 1964-93; R^2=0.982. (b) 1964-85; R^2=0.986.

The results for cow milk express a geographical variation of the integrated transfer coefficient, with the highest value in the south (Tvøroyri). The reason for the geographical variation may partly derive from differences in the soil characteristics at the localities. This could, however, not be tested in the present study because of lack of data.

The calculated integrated transfer coefficients are high compared to other countries (Refs. [1], [7] and [8]). Ref. [8] reports integrated transfer coefficients for milk in different countries around the world with the unit $pCi(gK)^{-1}y$ per $mCi\ km^{-2}$. The highest value of 27.51 is reported for the Faroe Islands followed by 15.48 for Norway. The lowest value of 3.23 was reported for Denmark.

4. CONCLUSIONS

The long-term variation of radioactivity in the selected foodstuffs is reproduced fairly well by the UNSCEAR model. The integrated transfer coefficients in the Faroe Island are found to be high compared to other countries, indicating relatively high individual doses from ingestion of the foodstuffs in the Faroe Islands. The integrated transfer coefficients provide a comprehensive assessment of transfer over the long term of ^{137}Cs and ^{90}Sr from deposition to foodstuffs. The effective radioecological halflives of ^{137}Cs and ^{90}Sr in the foodstuffs are estimated to 3.0-5.5 years.

For a further study, more updated data sets should be available, and other sample types should be included, e.g. other foodstuffs, soil and grass. The present study shows that further modelling along the same line is worthwhile.

ACKNOWLEDGEMENTS

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Development of a concentration factor database for radionuclides in the aquatic environment: *Applications and implications for FASSET*

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Abstract. The uptake of radionuclides by aquatic biota is most generally described by the concentration factor CF, which is the ratio of the activity concentration in the organism to activity concentration in the surrounding water. A database on concentration factors for radionuclides selected for consideration in FASSET project in the freshwater environment was developed. CFs illustrate well differences between organisms, but have also shortcomings. Their application is limited in situations where isotopic equilibrium is strongly distorted.

1. INTRODUCTION

To estimate radiation doses to plants or animals activity concentrations in the habitats, data on transfer of radionuclides and uptake by biota are needed. The commonest approach to represent the uptake of radionuclides by aquatic biota is by means of a constant coefficient (the concentration factor, CF) relating the activity in the biological organism to activity concentration in the surrounding water. By their nature, CF's are a means of generalising data for comparison with other nuclides and other organisms, and such generalisation has inevitable shortcomings. CF's are usually calculated by comparison of biota activities, which are time integrated, with an average of several instantaneous water results (or even a single result). Despite the shortcomings described below, if treated with caution, CF's can be used effectively to illustrate trends and differences between species. For this reason, the CF approach is recommended in FASSET as a necessary approximation applicable to most cases but one that needs to be applied with reservations in environments in which isotopic equilibrium is heavily distorted.

2. DATABASE DESCRIPTION AND EXAMPLES

In the FASSET project a number of candidate reference organisms for various environments were selected. For freshwater environment they were: phytoplankton, zooplankton, vascular plants, gastropoda, bivalve molluscs, crustacean, insect larvae, benthic fish, pelagic fish, amphibian, birds and mammals. Radionuclides selected for consideration in FASSET are: H, C, K, Cl, Ni, Sr, Nb, Tc, Ru, I, Cs, Po, Pb, Ra, Th, U, Pu, Am, Np and Cm.

A database on concentration factors for the above mentioned organism types and radionuclides repressenting transfer from water to plants and animals in the freshwater environment was developed. This database was developed in ACCESS format, which is easily converted into EXCEL. Open literature was collated for the concentration factor data. The database includes now about 700 data on concentration factors between organisms and water and about 50 data on distribution coefficients between sediment and water. Besides data on concentration factors of radionuclides between water and whole organisms, factors to various tissues or organs of organisms were gathered as well. To estimate radiation exposure for the whole life cycle of an organism all the life stages should be considered. The youngest life stages are usually the most sensitive for the effects of radiation.

Examples of concentration factors taken from the data base are given in Table 1. If concentration factors for an organism for a radionuclide were found in several published sources, average values were calculated and the variation range was entered into the table. In some cases only variation of concentration factors was given. Variation by a factor of 10^4 in concentration factors of U for various species of macrophytes was detected, while variation between stem and roots of some species was less than a factor of two. The highest concentration factors of those nuclides given in the table were for ^{137}Cs for a benthic fish *(Lota lota)*, about 17000 (Table 1).

TABLE 1. CONCENTRATION FACTORS OF ^{137}Cs, ^{239,240}Pu, ^{90}Sr AND U FOR VARIOUS FRESHWATER ORGANISMS – EXAMPLE FROM THE DATABASE

^{137}Cs	CF (Bq/kg fresh organism/Bq/kg water)	Comments	Reference(s)
Birds	3000		[1]
Fish (*Perca fluviatilis)*	10180	muscle	[2,3,4]
	4890	bones	[4]
Fish *(Lota lota)*	12200 (7110-17350)	muscle	[5]
Insect (stonefly)	326	dry weight	[4]
Macrophytes	130-1500		[6]
Macrophytes	1000	emergent vascular plant	[1]
Mollusc	100	shell	[1]
	1000	soft tissues	[1]
Plankton	3450 (1480-5870)	dry weight	[4]
^{239,240}Pu			
Phytoplankton	8000		[6]
Zooplankton	120-650		[6]
Macrophytes	1000 (230-9000)		[6]
Fish *(Perca fluviatilis)*	17 (10-24)		[7]
^{90}Sr			
Macrophytes	200	whole plant	[1]
Mollusc	300	soft tissues	[1]
Phytoplankton	37		[6]
Zooplankton	61		[6]
U			
Macrophytes	450 (4 - 1000)		[6, 8]
Macrophytes, *(Potamogeton sp.)*	6510 (14 - 13000)		[9]
Macrophytes, *(Nyphaea violacea)*	0,1		[10]
Macrophytes, *(Myriophyllum sp.)*	9520 (33 - 19000)		[9]
Macrophytes, *(Typha sp)*	3510 (14 - 17000)	root	[9]
Macrophytes, *(Typha sp.)*	2510 (10 - 5000)	stem	[9]
Phytoplankton	88 - 156		[6]
Zooplankton	16 - 80		[6]

3. CONSIDERATIONS IN APPLYING THE CONCENTRATION FACTOR APPROACH

Substantial variability was observed in the field determinations of CFs for some biota species as quoted by different sources. It is believed that this field variability is due to the large range of water activity caused by some environmental inputs, e.g. the pulsed nature of ^{99}Tc discharges from Sellafield to sea. Organisms generally respond slowly to a change in ambient water concentration. This explains the variability in CF's reported in the literature, and why uptake of radionuclides by marine biota in the field tends to diverge from what is reported in laboratory studies [11, 12, 13]. These discrepancies are due to the non-equilibrium of the field system and are typical of environments combining non-continuous, pulsed discharges with a rapid flushing time. In addition, in the natural environment CFs are affected by a number of processes. These include temperature, chemical and physical speciation and colloidal association, interfering substances, differential uptake in the fed or starved condition, size, age, seasonality, variations in habitats/diet and the timing of processes like moulting and sexual activity in biota.

The use of concentration factors with regards to assumption of equilibrium can be very limited when investigating short-term environmental releases of radioactivity. Jackson and Vives i Batlle [14] made the observation that concentrations of ^{99}Tc in seawater exhibit a rapid response phase, whilst *Fucus vesiculosus* and winkles (*Littorina littorea*) do not. This indicates clearly the limitations of using an equilibrium CF approach when environmental discharges are of a pulsed nature.

A classic example of the above limitations is the uptake of ^{99}Tc by lobsters in coastal environments: Shortly after a short-term pulsed release of Tc activity, concentration of ^{99}Tc in lobsters sampled along the dispersion path may increase slightly. Lobsters have a biological half-life of elimination ($T_{b1/2}$) for ^{99}Tc of 60 days or more (66 days for uptake from food and 200-300 days for uptake from seawater - [15]). But ^{99}Tc is soluble in seawater and may clear quickly from the area where lobsters live, aided by tides and currents, so concentration in seawater is likely to decrease sharply within hours-days. If a lobster has been sampled within a few days after the pulse, it may appear to have a high CF because it still retains ^{99}Tc but the surrounding water does not. The apparent anomaly is just conceptual, in trying to capture a snapshot picture of the concentration capacity of the lobster for ^{99}Tc by using an equilibrium CF. It has been demonstrated that, in fact, ^{99}Tc levels in lobsters routinely sampled near Sellafield year after year are quite adequately represented by a simple model based on actual ^{99}Tc uptake and depuration transfer rates as measured in laboratory experiments, taking these factors into account [16]. Hence, some of the perceived uncertainties in bioaccumulation processes are minimised once the relevant hydrodynamic and biokinetic processes are duly considered.

An improved representation of the uptake of radionuclides by an organism in a dynamic environment is therefore a model in which the organism undergoes simultaneous radionuclide sorption and desorption from and to the surrounding medium, mediated by simple first-order kinetics with rates of uptake and excretion from water. Such rates can be expressed in terms of the equilibrium CF and the $T_{b1/2}$ following uptake of radionuclides from water and food. This kind of approach can bring more realism to assessments carried out on short (sub-annual) timescales. Olsen and Vives i Batlle [16] and Vives i Batlle *et al.* [17] have developed biokinetic models to study the uptake of ^{99}Tc in winkles (*Littorina littorea*), calibrating the model with data from laboratory experiments to reproduce monitored ^{99}Tc concentrations in winkles from the Cumbrian coast. It is believed that, generally, there is enough biokinetic data to attempt a basic biokinetic model for the most radiologically significant FASSET reference organisms and radionuclides (e.g. Tc, Pu, Am, Cs and I), capable of bringing more realism to assessments carried out on short (sub-annual) time scales.

Despite the above shortcomings, if treated with caution, CF's can be used effectively to illustrate trends and differences between species. For example, similarities have been found between the distribution of ^{99}Tc in lobster and *Nephrops norvegicus*, whilst a significant difference exists between these two species and crab (*Cancer pagurus* - [12]). For this reason, the CF approach is recommended in FASSET as a necessary approximation applicable to most cases but one that needs to be applied with reservations in environments in which isotopic equilibrium is heavily distorted CF. The database developed for this project is seen as a significant contribution to assessments within the FASSET framework which will provide sufficient approximation for most ecosystems of interest.

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Vegetable-to-soil Concentration Ratio (CR) for ^{226}Ra in a highly radioactive region

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Abstract. Radium-226 is one of the main natural radionuclides, which is present in the environmental samples such as soil, water, food, etc. In order to assess the transfers and uptakes of ^{226}Ra, the vegetable-to-soil concentration ratio (CR) has been studied in a highly radioactive region of Ramsar. This area is of special interest in the world, due to the presence of several hot springs with high ^{226}Ra concentrations. Concentrations of ^{226}Ra were measured in different types of vegetables and substrate samples, using gamma spectrometry. The results show that the average vegetable-to-soil CR value in leafy vegetables (1.5×10^{-2}) is more than root vegetables (4×10^{-3}). In fact, uptake of ^{226}Ra via roots of vegetables is directed toward the leaves. In addition, the CR values measured in edible vegetables in a highly radioactive region of Ramsar are similar to those values in low background radiation areas.

1. INTRODUCTION

Survey on environmental transfers and uptakes of radionuclides play an important role for the establishment of scientific basis of environmental radiation assessments. The vegetable-to-soil concentration ratio (CR), which is defined as concentration of a radionuclide in the vegetable to its concentration in the substrate (subsoil), has been used to predict the transport of radionuclides and other elements of interest through the food chain [1]. Many factors such as temperature, type of soil and plant, environmental conditions, as well as chemical and physical characteristics of radionuclides in the environment can affect the concentration ratio [2].

Radium-226 is one of the most important alpha emitting radionuclides in the ^{238}U decay series with a half-life of 1620 years. Due to metabolic similarities of ^{226}Ra to calcium, it is called a “bone seeker element”. ^{226}Ra decay products, present to a varying extent in soil, rock, sediment, water, etc. Transfer and accumulation of radium from one environmental compartment to another, for example through root uptake in vegetables, depend on different environmental parameters such as the availability of the nuclides in soil, and also on the rate of loss from the internal structure of leaves after translocation [3, 4].

In the present study, investigations have been carried out on several areas in “Ramsar”, a northern coastal city of Iran, which has been the subject of concern as a highly radioactive region for the past 40 years [5]. The origin of elevated radioactivity in this area is mainly due to presence of natural radionuclides, especially ^{226}Ra in hot springs flowing into the region, leading to the creation of hot spots with occasional high values of up to 100 μGy/h [6]. The geographical location of Ramsar is illustrated in Figure 1.

To assess the environmental transfers and uptakes, vegetable-to-soil concentration ratio for ^{226}Ra was studied in “Talesh Mahalleh”, which is a highly radioactive district in Ramsar. Different areas with low background radiation in that region were also considered for further investigations in this study.

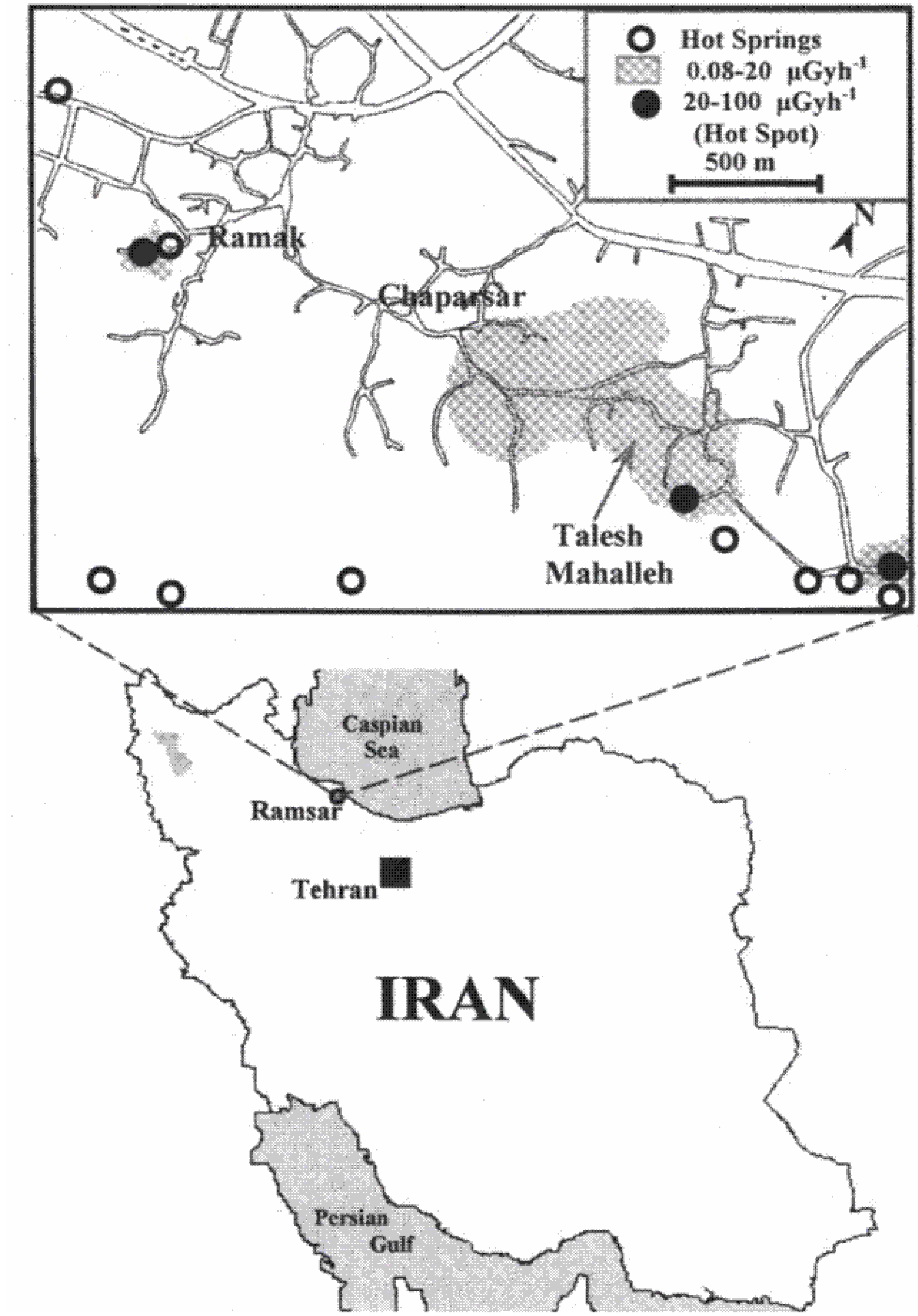

FIG. 1. Location of Ramsar.

2. MATERIALS AND METHODS

In order to determine the concentration ratio for ^{226}Ra in Talesh Mahalleh, different types of leafy and root edible vegetables were cultivated in low background radiation and in an area with high radioactivity levels. After harvesting and washing the vegetables, the fresh weighted samples were dried at 105 ºC, and subsequently ground and sieved. Substrate samples were also dried at 105 ºC after grinding and sieving. Each of soil and vegetable samples were kept after preparation, in a sealed calibrated geometry, for at least three weeks in order to reach secular equilibrium between ^{226}Ra and ^{214}Bi. The concentration of ^{226}Ra was measured by the 609 keV photopeak of ^{214}Bi using a Canberra high purity germanium (HPGe) gamma spectrometer with a relative efficiency of 40% (Ghiassi-Nejad, et al., 2001). ^{226}Ra concentrations in four samples of water used for irrigation were also analysed by the emanation method with a minimum detection limit (MDL) of less than 2 mBq/l [7].

The CR values were derived, using the following equation:

$$CR = \frac{\text{Concentration of } ^{226}\text{Ra in dried edible vegetables} \left(\text{Bq kg}^{-1}\right)}{\text{Concentration of } ^{226}\text{Ra in dried soil (substrate) in Bq kg}^{-1}}$$

3. RESULTS AND DISCUSSION

Vegetable-to-soil concentration ratio for ^{226}Ra in different types of edible vegetables studied in low and high background radiation areas for leafy and root vegetables are illustrated in Table 1 and Table 2, respectively.

The results show that in both high and low background radiation areas, the average vegetable-to-soil CR is about 1.5×10^{-2} for leafy vegetables. The CR value for root vegetables in these regions are found to be about 4×10^{-3}. On the other hand, the maximum uptake of ^{226}Ra occurs mainly in leafy vegetables and the minimum in root vegetables. These results also reconfirm the assumption that the concentration of ^{226}Ra in vegetables increase linearly with increasing ^{226}Ra concentration in soil in not very high radioactive regions. The non-linearity assumption for CR values in very high background radiation areas seems to require further specific experiments [6].

It should be noted that the ^{226}Ra concentration in samples of irrigation water was negligible compared to the soil concentrations and does not affect on the CR values.
In general, the vegetable-to-soil concentration ratio (CR) for leafy and root vegetables in highly radioactive and low background regions are very close to each other and their mean values are in the same order of 10^{-2} to 10^{-3}, respectively.

TABLE 1. VEGETABLE-TO-SOIL CONCENTRATION RATIO FOR ^{226}Ra IN LEAFY VEGETABLES

Type of Vegetable	Concentration ratio (High Background Radiation)	Concentration ratio (Low Background Radiation)
Basil	1.79×10^{-2}	1.47×10^{-2}
Beet leaf	1.98×10^{-2}	2.1×10^{-2}
Coriander	5.3×10^{-3}	1.3×10^{-2}
Cress	1.55×10^{-2}	1.71×10^{-2}
Leek	1.12×10^{-2}	1.01×10^{-2}
Parsley	3.07×10^{-2}	2.4×10^{-2}
Spearmint	7.5×10^{-3}	1.0×10^{-2}
Spinach	1.03×10^{-2}	1.2×10^{-2}
*Mean ± SD**	$1.5\times10^{-2} \pm 0.8\times10^{-2}$	$1.5\times10^{-2} \pm 0.5\times10^{-2}$

*Standard Deviation.

TABLE 2. VEGETABLE-TO-SOIL CONCENTRATION RATIO FOR ^{226}Ra IN ROOT VEGETABLES

Type of Vegetable	Concentration ratio (High Background Radiation)	Concentration ratio (Low Background Radiation)
Carrot	1.09×10^{-2}	7.1×10^{-3}
Garlic	5.0×10^{-3}	6.5×10^{-3}
Onion	1.1×10^{-3}	2.0×10^{-3}
Potato	1.0×10^{-3}	1.63×10^{-3}
Radish	4.9×10^{-3}	4.1×10^{-3}
Mean ± SD	$4.6\times10^{-3} \pm 3.6\times10^{-3}$	$4.3\times10^{-3} \pm 2.2\times10^{-3}$

*Standard Deviation.

4. CONCLUSION

According to the results of this study we can conclude that:

(1) There is no significant difference between the CR values of edible vegetables in a highly radioactive region and the corresponding values in the low background radiation areas.

(2) The maximum and minimum uptake of ^{226}Ra occurs mainly in leafy and root vegetables, respectively.

(3) Uptake of ^{226}Ra via roots of vegetables is directed toward the leaves.

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Bioaccumulation of technetium by bacterial community in water covering a rice paddy field soil

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Abstract. Bioaccumulation of technetium (Tc) by bacterial community in water body covering a rice paddy field soil, is studied to gain insight on the behaviour of Tc in agricultural ecosystem. The radioactive tracer, pertechnetate ($^{95m}TcO_4^-$), was added to aqueous solution collected from a waterlogged paddy field soil. The solution was incubated statically at 25˚C under dark conditions. Up to 80% of the total ^{95m}Tc was removed from the solution for 4 days. However, little ^{95m}Tc was removed from both a cell-free filtrate solution and a filtrate mixed with autoclaved cell particles at day 4. The maximum removal of the ^{95m}Tc, about 85%, was achieved by addition of nutrients to the solution. When activities of bacteria inhibited by addition of antibiotics, only 23% of the ^{95m}Tc was removed. Addition of cycloheximide, however, did not affect on the removal of the ^{95m}Tc. These results indicate that the bacterial community living in water body accumulates Tc. In order to determine the effects of oxygen level on Tc removal, bacterial communities were incubated under aerobic and anaerobic conditions. It is reasonable that the bacterial community cultured under anaerobic conditions removed 80% of Tc from the solution, however, interestingly, the bacterial community grown under aerobic conditions also showed Tc removal of 30%. The result implies that aerobic bacterial communities have a potential for removing Tc from aqueous solutions.

1. INTRODUCTION

Technetium-99 (^{99}Tc) has been created by human activities involving fissionable materials and released to the environment from nuclear facilities [1]. Levels of discharge of ^{99}Tc into the environment will be increased because of its long half-life of 2.1×10^5 years. In the presence of oxygen, most stable chemical form of Tc is the pertechnetate (TcO_4^-) [2], and this form is highly soluble and readily available for plants. These factors have raised questions concerning the environmental behaviour of ^{99}Tc including its transfer from soil to plants and from plants to humans.

The behaviour of Tc in rice paddy fields could be influenced by microorganisms. Tagami and Uchida [3] reported that TcO_4^- is removed from solution covering water logged paddy field soils. The authors suggested that the Tc removal was caused by indirect microbial effect: microbial activity brings their habitat into extremely reductive conditions and then physicochemical modifications of Tc in the local environment leading to a reduction of TcO_4^- inducing the Tc removal from solution. As well as indirect effects, microbial direct effects on Tc removal must be considered. It is well known that anaerobic bacteria *Desulfovibrio* sp. and facultative aerobic bacteria *Escherichia coli* can directly accumulate Tc in their biomass [4]. Their capability of Tc accumulation exercises under anaerobic conditions. The depth of water covering paddy field soils is generally about 10 cm in Japan, and thus anaerobic conditions in water would rarely found. Removal of Tc by aerobic microbial community must be also examined.

The objects of this research were 1) to investigate time course of Tc removal from aqueous solution by microorganisms in water body of a waterlogged paddy field soil, 2) to determine the Tc removing organisms and removal mechanisms, and 3) to examine a potential for Tc removal by aerobic and anaerobic microbial communities.

2. MATERIALS AND METHODS

2.1. Preparation of water covering a paddy field soil

Gray lowland soil was collected from a rice paddy field in Koriyama City, Japan. The soil was air-dried and passed through a sieve (2 mm mesh). Five grams of the soil were flooded with 7 ml of 0.3% glucose water in a 50-ml centrifugation tube and then the waterlogged soil was incubated at 25˚C for 7 days. After the incubation, only the water body covering the soil was collected and used following tracer experiments. All experiments were carried out tripricate.

2.2. Time course experiment of Tc removal

The radioactive tracer, pertechnetate ($^{95m}TcO_4^-$), was added to the solution collected from the waterlogged soil. The solution was incubated statically for 42 days at 25˚C under dark conditions. Samples were drawn out from the solution and filtered through a 0.2 µm-pore-size cellulose acetate membrane filter at day 0, 1, 2, 3, 4, and 42. The ^{95m}Tc in each sample measured before and after the filtration with a NaI (Tl) scintillation counter.

2.3. Tc removing organisms and their removal mechanisms

Using the solution collected, the following 5 sub-samples were prepared: (1) a cell-free filtrate of the solution by passing through a 0.2 µm-pore-size cellulose acetate membrane filter; (2) a filtrate mixed with an autoclaved cell suspension; (3) a solution supplemented with nutrients (1 mg of tryptone, 0.5 mg of yeast extract, and 1 mg of NaCl per ml in the solution); (4) a solution added nutrients and antibiotics (ampicillin, streptomycin, and tetracycline, each 25 µg/ml solution); and (5) a solution with nutrients and cycloheximide (50 µg/ml solution). Percentages of Tc removal from these 5 sub-samples were compared to clarify factors of the removal. After 4 days of incubation with the ^{95m}Tc in each sample, the ^{95m}Tc in the samples was measured as described above.

2.4. Aerobic and anaerobic culture

Microorganisms in the solution were cultured under aerobic and anaerobic conditions. The aerobic culture was incubated at 25˚C for 4 days and shaken at 200 rpm on a shaker. The anaerobic-static culture (1.48 ml) was incubated in a sealed 1.5-ml plastic screwcap microtube at 25˚C for 4 days. The tube was sealed with parafilm and stored in anaerobic jar (Aneropack-Kenki, Mitsubishi Gas Chemical, Tokyo, Japan). The removal percentages of the ^{95m}Tc in the cultures were examined by the same method described above.

3. RESULTS AND DISCUSSION

3.1. Time course of Tc removal

The ^{95m}Tc added was removed from the solution over time (Figure 1). The maximum rate of ^{95m}Tc removal occurred between 0 and 2 day after the beginning of incubation. After 4 days of incubation, approximately 80% of the ^{95m}Tc added was removed from solution. This removal of the ^{95m}Tc could be directly caused by bioaccumulation of microbial community because it takes a long time to remove the ^{95m}Tc. If reduction of Tc by reducing conditions or biosorption were mechanisms of the removal, the ^{95m}Tc would be removed more rapidly. After 42 days of incubation, the ^{95m}Tc removal from the solution was the same as the maximum level (Figure 1). It was therefore suggested that once accumulated, Tc did not easily released into solution. The strong fixing of Tc by bioaccumulation may be one of the mechanisms for discharge of Tc on soil surface [5]. The long term fixation also means that the potential of Tc removal can be determined by 4 days of incubation with Tc. Thus 4 days incubation time was applied for subsequent studies.

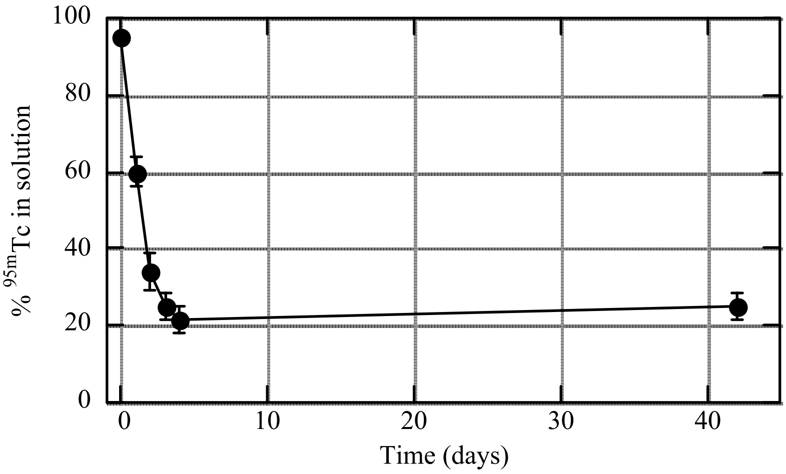

FIG. 1. Removal of ^{95m}Tc from solution as a function of time.

3.2. Bioaccumulation of Tc by bacteria

To confirm the bioaccumulation, percentages of ^{95m}Tc removal from sub-samples were compared (Table 1). For control, that is, the untreated water covering the paddy field soil, 74.5% of ^{95m}Tc was removed from the culture solution, but the percentage remarkably decreased when the microorganisms were removed by filtration from the control. Since the filtrate did not contain > 0.2 µm-size particles, this lack of suspended particles on which Tc adsorbed might cause little ^{95m}Tc removal from the solution of the filtrate sample. Then, cells in the control were autoclaved and suspended in the filtrate. Significant removal of Tc, however, did not observed by that treatment. Based on the Eh value (data not shown), solutions of the control and the filtrate had reductive conditions. These results therefore show that the ^{95m}Tc removal was not merely the result of reducing conditions in the solutions. When nutrients were supplemented to the control, the ^{95m}Tc removal percentage increased about 10% by comparison with that of the control. The difference was statically significant (Student's *t*-test, $P < 0.01$). Because nutrients promote microbial activity, microbial direct effects on Tc removal were indicated.

Soil microorganisms dominantly consist of bacteria and fungi. Using antibiotics, it was examined which microorganisms, bacteria or fungi, contributed the ^{95m}Tc removal. Antibiotics inhibiting bacterial activity caused the decrease of ^{95m}Tc removal, while no significant decrease of the removal was observed by the addition of cycloheximide (Table 1). Therefore we conclude that the removal of the ^{95m}Tc from the water covering paddy field soils is caused by bacterial accumulation.

TABLE 1. REMOVAL PERCENTAGE OF THE Tc FROM SOLUTION OF THE CONTROL AND 5 SUB-SAMPLES

sample	% of Tc removal (mean ± sd, n = 3)		
control (untreated)	74.5	±	4.4
Filtered	3.6	±	0.5
filtrate with autoclaved cell particles *	2.9		
culture with nutrients	85.7	±	2.1
Nutrients culture with bacterial abtibiotics	23.4	±	0.8
Nutrients culture with cycloheximide	87.8	±	1.7

* no replicate data

3.3. Removal of Tc under aerobic and anaerobic conditions

In this series of experiments, microorganisms were cultured statically. The static culture does not keep strict aerobic conditions even though sufficient headspace of air exists. For example, the bottom of cultures may develop low oxygen levels. Thus a shaking culture was carried out to examine the Tc removal under aerobic conditions. Interestingly, 30% of Tc was removed from the solution under aerobic conditions (Table 2). To our knowledge, there are no reports for the Tc removal under aerobic conditions. Therefore, isolation and identification of the Tc removing aerobic bacteria and the removal mechanisms under aerobic conditions must be very important subjects in future researches.

In anaerobic culture, approximately 83% of ^{95m}Tc added was removed from the solution (Table 2). The percentage was slightly higher than that of static culture, but the difference was not statically significant (Student's *t*-test, $P > 0.05$). The high removal ratio of ^{95m}Tc under the anaerobic condition is reasonable because many kinds of Tc reducing bacteria exist [4] and because there is a similar report on Tc removal by soil bacteria under anaerobic conditions [6]. The high removal ratio of ^{95m}Tc from the static culture, however, does not account for the developing anaerobic conditions at the bottom of the static culture because the top of the culture is aerobic at least. Both aerobic and anaerobic bacterial communities might cause the removal of ^{95m}Tc in the static culture.

TABLE 2. REMOVAL PERCENTAGES OF Tc UNDER AEROBIC AND ANAERBIC CONDITION

sample	% of Tc remocal (mean ± sd, n = 3)		
static culture	76.1	±	4.2
aerobic	30.5	±	3.0
anaerobic	82.8	±	0.6

4. CONCLUSION

Previously, removal of Tc from water covering paddy field soils had not been evaluated. In this study, it is indicated that Tc is removed from the water body by bacterial bioaccumulation and that the accumulated Tc retains in the bacterial biomass for long time. The possibility of Tc bioaccumulation under aerobic conditions is also suggested.

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Multi-element analyses of environmental samples for radioecology and ecotoxicology

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Abstract. About 60 major and trace elements as well as radiocaesium in environmental samples were determined in order to obtain basic information for environmental radiation protection and ecotoxicology. Levels and distribution in plants, mushrooms and soils collected in forest ecosystems were discussed. Data of radiocaesium and related stable elements indicated that the high ^{137}Cs, Cs and Rb concentrations and low Ca and Sr concentrations, in comparison with plants, could characterize mushroom composition. Relationships between radiocaesium and stable Cs in mushrooms and plants were discussed. The results indicated that the ratio between radiocaesium and stable Cs is useful for judging the equilibrium of deposited radiocaesium in different components comprising a forest ecosystem. The stable Cs analyses might be also useful to predict the long-term radiocaesium contamination of mushrooms and plants. Transfers of the elements from media to earthworms, and distribution of the elements in an aquatic microcosm were also discussed.

1. INTRODUCTION

Increasing discussion about the environmental radiation protection realizes that more information is required on the transfer and accumulation of radionuclides in the biological compartments of the ecosystems. In order to compare the effect of radiation with another toxic materials, it is also important to obtain data sets for toxic materials, which can be discussed together with radionuclides.

Research efforts concerning the migration of elements in natural and semi natural ecosystems have concentrated mainly on the major nutrient elements (e.g. K, Ca, Mg, P) and some heavy metals (e.g. Cu, Zn, Pb, Cd). Therefore, the data for other trace elements are still limited. The behaviour of radionuclides have also been studied from radiation protection viewpoint. However, information is focused on some limited nuclides such as ^{137}Cs and ^{90}Sr, and the fate of long-lived radionuclides in the systems is still difficult to predict. Determination of stable elements in ecosystems is one of the most powerful ways to obtain the *in-situ* transfer information, which might be useful for predicting the behaviour of long-lived radionuclides and toxic elements in the systems. However, available data set for this purpose is still limited, since radionuclides and stable elements have been studied separately by the different institutes, in most cases.

Recent development of analytical technique such as inductively coupled plasma-mass spectrometry (ICP-MS) provides a possibility for accurate and precise determination of trace elements in a variety of materials including environmental samples. Due to its low detection limits, analytical speed, and multi-element capability, ICP-MS has been applied to more than 50 elements in environmental samples [e.g. 1, 2].

In this study, the stable major and trace elements with wide range of concentrations in plants, mushrooms and soils collected in forest ecosystems were determined by ICP-MS, inductively coupled plasma-atomic emission spectrometry (ICP-AES) or X-ray fluorescence spectrometry (XRF), and the data of stable elements were summarized together with those of radiocaesium. Equilibrium of radiocaesium with stable caesium within the biological cycles of contaminated forests was discussed. Concentrations of the elements in earthworms and their growth media were also estimated in order to obtain basic information for the possible use of earthworms for the indicator of soil contamination and its effect on the ecosystems. Aquatic microcosms used for the estimation of ecological effects of radiation and other environmental stresses were also subjected to analyses in order to understand the behaviour of the elements in the system.

2. ANALYTICAL METHODS FOR ENVIRONMENTAL SAMPLES

2.1. Sample pre-treatment

Plants, mushrooms, earthworms and earthworm growth media were freeze-dried or oven-dried, and pulverized with a cooking blender. Soil samples were air-dried or oven-dried, sieved (1 mm) and ground to powders. Aquatic microcosms were filtered, and both filtrates and residues were proceeded to the analyses of stable elements.

2.2. Determination of ^{137}Cs

Each dried sample was placed in a plastic bottle and concentrations of ^{137}Cs and ^{134}Cs were determined by counting with a Ge-detector. The approximate detection limit for ^{137}Cs by the counting system for a 30 g dry sample was 3 Bq kg^{-1} [3].

2.3. Determination of stable elements

Samples were digested in Teflon pressure decomposition vessels with acids (HNO_3, HF and $HClO_4$) by using microwave oven. After digestion, the samples were evaporated to dryness. Then, the residues were dissolved in 1 - 2% HNO_3 to yield the sample solutions. The solutions were analyzed for major and trace elements with wide range of concentrations by the combination of several analytical methods. Trace elements (Cs, Sr, Zn, Cu, Cd, La, Ce, Th, U, etc.) were measured mainly by quadrupole ICP-MS. Internal standards such as Rh, In and Bi were used to compensate for changes in analytical signals during the operation. Major elements (K, P, Mg, Na, Ca, etc.) were determined by ICP-AES. Standard reference materials were used to validate the analytical procedure. Details for the analyses using ICP-MS have been described in elsewhere [2, 4]. Wavelength dispersive XRF was also used for some major elements analyses in soil samples. Powdered soils were treated as fused glass beads with lithium tetraborate and analyzed. The high resolution ICP-MS (HR-ICP-MS) was used for the determination of U and Pu isotopes [5, 6]. About 60 elements were determined totally, although the accuracy and precision were different depend on the elements and their concentrations.

3. TRACE ELEMENTS AND RADIOCAESIUM IN SOIL-PLANT SYSTEMS

Concentrations of some trace elements such as Pb and Cd in Japanese forest soils tended to be the highest in the surface soil layer [2], indicating the importance of atmospheric deposition on the total contents in the soils of these elements. The activity concentration of ^{137}Cs in soil was the highest in the surface organic rich layers and decreased with soil depth [7], even in Japanese forests, where the most of ^{137}Cs was deposited mainly during the 1960s. Contrastively, the concentrations of stable Cs in soils were almost constant with depth [2].

In comparison with the elemental composition of plants, the mushroom composition could be characterized by the high ^{137}Cs, Cs and Rb concentrations and low Ca and Sr concentrations [4]. The Cs concentrations of mushrooms were one or two orders higher than those of other plants growing in the same forest. Accumulations of Cs and Rb in mushrooms were also observed from cultivation experiments in flasks using radiotracers [8]. These results are consistent with the high concentrations of radiocaesium in mushrooms reported in many countries.

In order to estimate the accumulation of each element by plants and mushrooms, tentative transfer factors (TFs) were estimated from the ratio of "concentration in plants and mushrooms on dry weight bases" to "concentration in the surface soils on dry weight bases" [2]. In case of a Japanese pine forest, the TFs of Co, Ba, La, Ce, Pb, Th and U were very low for most plants and mushrooms. On the other hand, those of Cu, Zn, Rb and Cd were relatively high. The low TFs of lanthanide elements, Th and U suggested transuranium elements would have low TFs. Ferns tended to accumulate the lanthanide elements.

4. EQUILIBRIUM OF RADIOCAESIUM WITH STABLE CS IN FORESTS

Long-term prediction of the behaviour of radiocaesium in contaminated forest ecosystems is the matter of concern for the radiation protection of human and ecosystem itself. As the chemical behaviour of radiocaesium is expected to be almost identical to that of stable Cs, analyses of stable elements should be useful to understand the long-term behaviour of radiocaesium and its equilibrium distribution. We have determined the concentrations of stable Cs and related alkali and alkaline earth elements in mushrooms, plants and soils collected in forests with different contamination levels in Japan, Germany, Finland, Italy, Ireland and Belarus [9-11]. Data of stable elements were discussed together with those of radiocaesium.

Relationships between ^{137}Cs and stable Cs for mushrooms collected from 5 different forests in Finland, Germany and Japan were studied. A good correlation between ^{137}Cs and stable Cs was observed for each site independently, although several different species of mushrooms are included. This finding suggests that mushrooms take up ^{137}Cs together with stable Cs. A good correlation between ^{137}Cs and stable Cs was also observed in samples collected from different parts of trees in Irish and Italian forests. The ^{137}Cs/stable Cs ratios were fairly constant for samples collected at the same site. The results for different sites, however, showed different degrees of variability. The ^{137}Cs/stable Cs ratio might be a useful criterion for judging the equilibrium of deposited ^{137}Cs to stable Cs in a forest ecosystem. Standard deviation of the ^{137}Cs/stable Cs ratio was the lowest in Japanese forests (Tokai-mura: 27%), in which most ^{137}Cs originated from the global fallout, and the highest in Hochstadt, Germany (48%). This finding suggests that ^{137}Cs deposited during the 1960s has already attained a dynamic equilibrium within the soil-mushroom system.

We are currently studying samples collected in 1998 in 4 different forests with different contamination levels in Belarus. This study is expected to yield comprehensive information of radiocaesium and stable Cs and their interrelation in the whole forest ecosystem. Although several different species of plants and mushrooms are included, a good correlation between ^{137}Cs and stable Cs was observed for each sampling site independently. The ^{137}Cs/stable Cs ratio in plants, mosses, mushrooms and soil organic layers observed for each site were about the same. This finding suggests that ^{137}Cs deposited on the forest ecosystems from the Chernobyl accident was almost mixed with stable Cs within the biological cycle in the system in 1998. Fungal and microbiological activity might play an important role on the mixing and the long-term retention of radiocaesium [12].

5. EARTHWORMS

Earthworms play important roles on ecosystems, and might be a good indicator of soil contamination and its effect on the ecosystem. Elemental composition of earthworm gives useful information on background level and possible accumulation of the metals. In addition, change of the composition itself might be a possible indicator of the effect. There are several works for some specific heavy metals such as Cd, Zn, Pb and Cu about concentration, transfer, chemical form and controlling factors [e.g. 13, 14]. However, the information for other elements is limited. In order to obtain the basic information on the elemental composition and metal accumulation, the earthworms (*Eisenia foetida*) fed in the laboratory were analyzed for stable elements. Before the analyses, the worms were kept in water to remove the materials inside their guts. The growth media, litter and cotton linter, were also analyzed.

TFs were estimated from the ratio of "concentration in earthworms on dry weight bases" to "concentration in the media on dry weight bases". Relatively high transfer was observed for Na, Mg, P, K, Ca, Co, Ni, Cu, Zn, Rb, Mo and Cd. The TFs for Al, Sc, Ti, Y, Nb and lanthanide elements were low. The TFs for Sr and Cs were intermediate.

6. AQUATIC MICROCOSM

Microcosms are experimental ecosystems constructed in the laboratory to have some of the physical, chemical and biological functions of natural ecosystems. We have successfully used the microcosm consisting of three species of microorganisms, *Euglena gracilis* Z, *Tetrahymena thermophila* B and *Escherichia coli* DH5α, for the estimation of ecological effects of radiation and other environmental stresses [e.g. 15, 16]. Multi-element analyses of the microcosm provide the basic information of the behaviour of both essential and trace elements in the system. Elements were added to the systems, and the solid phase (microorganisms) and solution phase (medium) were separated by the filtration after 2 weeks incubation. Both phases were analyzed by using ICP-MS and ICP-AES. Preliminary results indicate that the percentage of the elements found in the solid phase were quite different among the elements. Relatively high percentages in the solid phase were observed for e.g. Cr and lanthanide elements. Cs and Sr tended to stay in the solution phase.

7. CONCLUSIONS

Multi-element analyses can provide information on the distribution of many major and trace elements including toxic metals in ecosystems. Such studies must be useful to identify the target organisms where the element can be accumulated. Analyses of stable elements are also useful to understand the long-term behaviour of the respective radionuclides and its equilibrium distribution in the radio-contaminated systems.

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Tritium and ^{14}C in the environment around Wolsong nuclear power plant

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Abstract. To understand the environmental dispersion trend of ^{3}H and ^{14}C discharged from the Wolsung nuclear power plant, the present levels of ^{3}H and ^{14}C in environmental samples in the vicinity of the Wolsong site were studied. During 1992 through 2002, the annual means of HTO and ^{14}C concentrations in air collected at three sampling stations around Wolsong nuclear power plant were in the range of 1.13 ~ 29.2 Bq m^{-3} and 0.33 ~ 0.40 Bq g-C^{-1} respectively, decreasing with distance from the site. The annual mean of TFWT concentrations in pine needle was in the range of 104 ~ 790 Bq L^{-1} at N-1 and 62.8 ~ 298 Bq L^{-1} at S-2, while the annual mean of ^{14}C concentration was in the range of 0.335 ~ 0.897 g-C^{-1} at N-1 and 0.254 ~ 0.332 Bq g-C^{-1} at S-2, showing elevated value than a 0.25 Bq g-C^{-1} of background level.

1. INTRODUCTION

Since the Test Ban Treaty came into effect in 1963, the levels of ^{3}H and ^{14}C have gradually declined to a background level. Recently, both nuclides production by nuclear power reactors have increased and will in time become the dominant source. The annual production of ^{3}H would be about 218 PBq by ternary fission and 370 PBq from deuterium activation in heavy water reactors [1]. Most of the ^{14}C produced in Canadian deuterium uranium (CANDU) pressurized heavy-water reactors (PHWR) has resulted from the large inventory of ^{17}O in the heavy-water moderator, and about 8.7 PBq of ^{14}C were projected to be released through 1990 from the nuclear power industry with 40 percent from the operation of heavy water reactors [2]. As described above, the release rates of ^{3}H and ^{14}C are relatively higher in heavy water reactors than in pressurized water reactors. Since 1983, Wolsung Nuclear Power Plant (Wolsong NPP) Unit 1, which is a CANDU-type reactor (heavy water reactor), has released small amounts of triated water during routine operations. Now, in addition to the Wolsong NPP Unit 1, Unit 2, 3 and 4 are in the commercial operation. However, sufficient field data on ^{3}H and ^{14}C in the environment around Wolsung site has yet to be accumulated. For this reason, it is necessary to obtain ecological data which would enable the estimation of public exposure to ^{3}H and ^{14}C released from Wolsong NPPs. In a previous paper [3], ^{3}H and ^{14}C levels in agricultural products in the vicinity of the Wolsong site were evaluated. In this work ^{3}H and ^{14}C concentrations in air, pine needle and precipitation in the vicinity of the Wolsung site have been determined to estimate the environmental dispersion trends of both nuclides discharged from Wolsong nuclear power plant, and their correlation with gaseous discharge rates of ^{3}H and ^{14}C were discussed.

2. MATERIAL AND METHODS

2.1. Site description and sampling

The Wolsong site is located in Naa-ri, Yangnam-myon, Kyongsangbuk-do in the southeast of the Korean Peninsula bordered by the East Sea (Sea of Japan). Wolsung NPP Unit 1, began its operation in 1983, is a CANDU-type reactor with a total gross capacity of 679 Mwe. Unit 2, 3 and 4 are the same type of reactors as Wolsong NPP 1, and their operation began in 1997, 1998 and 1999, respectively, with 700 Mwe of total gross capacity for unit.

As shown in Figure 1, air, pine needle and precipitation were monthly collected at N-1, N-2 and S-2 sampling stations located within a 2km radius from Wolsong NPP 1 stack during 1992 through 2002. Air sample was collected by an air sampler which is composed of a series of molecular sieve† adsorption columns and two traps filled with 12% NaOH solution. Pine needle was sealed tightly in thick polyethylene bag and brought back to the laboratory as quickly as possible. Precipitation was collected by a 50cm diameter of rain sampler made of stainless steel.

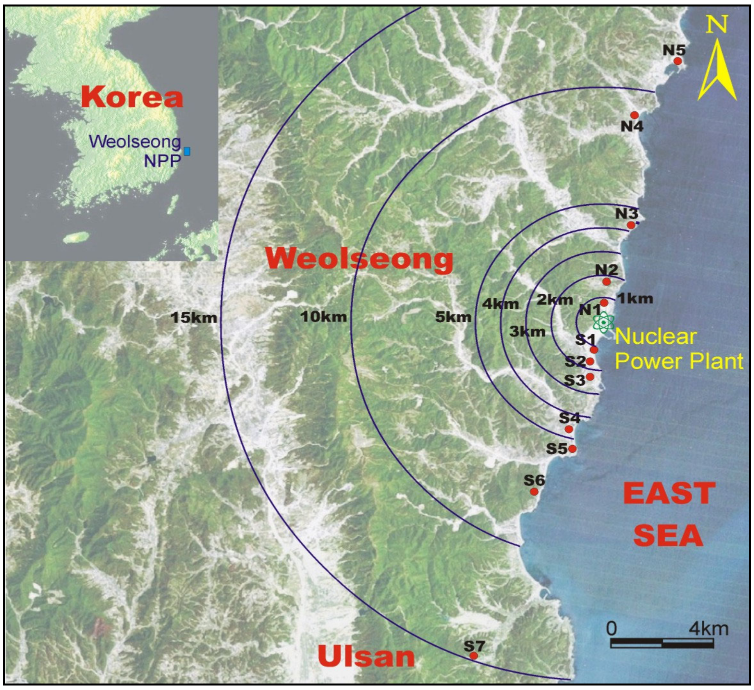

FIG. 1. Sampling locations around Wolsong nuclear power plant.

2.3. Preparation and measurement of counting samples for ^{3}H and ^{14}C analysis

Atmospheric water vapor (i.e. HTO) was recovered from the molecular sieve column (45cm × 5cm i.d.) in the laboratory by heating at 450°C for 2h. 12% NaOH solution in two of the $^{14}CO_2$ traps were transfered to a 3 L beaker containing a stir-bar and precipitated as $CaCO_3$ with $CaCl_2$. Tissue free water tritium (TFWT) of pine needle was collected by lyophilization. The freeze-dried sample was transferred to a combustion cup and burned in the combustion bomb‖ . After burning the sample, the combustion bomb was cooled with ice. After cooling, $^{14}CO_2$ remaining in the combustion vessel was passed through two bubblers containing 500 ml of ammonium hydroxide solution free carbonate for $^{14}CO_2$ absorption. The sample water remaining in the vessel was used for the determination of organically bound tritium (OBT) concentration. The ^{3}H and ^{14}C in the counting sample were measured for 500 min (50 min × 10 repeats) by liquid scintillation counter (LSC) ‡ .

3. RESULTS AND DISCUSSION

3.1. ^{3}H and ^{14}C concentrations in air and pine needle

As shown in Table 2, the annual means of atmospheric HTO concentrations at sampling stations N-1, N-2 and S-2 during 1992 through 2002 are in the range of 3.39 – 29.2, 1.95 – 4.66 and 1.13 – 2.21 Bq m^{-3}, respectively.

The long-term atmospheric dilution factors of ^{3}H at sampling stations N-1, N-2 and S-2 are in the range of 1.29 ~ 7.15 × 10^{-7} s m^{-3}. During 1999 to 2002, the annual mean of ^{14}C concentrations at sampling stations N-1, N-2 and S-2 is 0.40, 0.36 and 0.33 Bq g-C^{-1}, respectively. The long-term atmospheric dilution factors of ^{14}C at sampling stations N-1, N-2 and S-2 are in the range of 1.67 ~ 1.91 × 10^{-5} s g-C^{-1}.

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‡ Pure Chemical Co. Ltd, Molecular sieve 5Å.

§ Wallac and EGandG Co., Model 1220 Quantulus, P.O. Box 10, FIN-20101 Turku, Finland.

‖ Model 1121 Oxygen Combustion Bomb, Parr Instrument Co., 211 Fifty-Third Street, Moline, IL 61265.

TABLE 1. ANNUAL MEANS AND LONG-TERM ATMOSPHERIC DILUTION FACTORS OF HTO IN THE ENVIRONMENT AROUND WOLSONG SITE

Year	Emission rate of tritiated water vapor[a] (A) ($MBq\ s^{-1}$)	Average atmospheric HTO concentration (B) ($Bq\ m^{-3}$)			Atmospheric dilution factor (B)/(A) ($s\ m^{-3}$) ($x10^{-7}$)		
		N-1	N-2	S-2	N-1	N-2	S-2
1992	12.3	3.80	–	1.31	3.09	–	1.07
1993	11.7	-[b]	–	–	–	–	–
1994	15.3	29.2	–	1.84	19.1	–	1.20
1995	14.0	22.0	–	1.56	15.7	–	1.11
1996	9.8	–	–	–	–	–	–
1997	19.8	8.68	4.66	1.89	4.39	2.36	0.96
1998	12.9	7.33	2.76	2.21	5.67	2.13	1.71
1999	8.9	5.44	1.95	1.17	6.13	2.20	1.32
2000	6.5	8.07	3.23	1.33	12.4	4.99	2.06
2001	7.3	3.39	2.22	1.53	4.62	3.02	2.08
2002	10.4	7.79	3.19	1.42	7.51	3.07	1.37
Ave.	11.7	10.6	3.0	1.6	7.15	2.96	1.29

[a]This data was cited from the annual report of Korea Hydro and Nuclear Power Co., Ltd.
[b]The samples were not obtained.

Pine needle may be very useful indicator to monitor the variation of ^{3}H and ^{14}C levels in the environment around Wolsong site, because of their availability regardless of places and seasons in Korea and a capability of simultaneous collection of present and old pine needle samples. During 1992 to 2002, the annual mean of TFWT concentrations in pine needle was in the range of 104 ~ 790 Bq L^{-1} at N-1 and 62.8 ~ 298 Bq L^{-1} at S-2, while the annual mean of ^{14}C concentration was in the range of 0.335 ~ 0.897 g-C^{-1} at N-1 and 0.254 ~ 0.332 Bq g-C^{-1} at S-2, showing elevated value than a 0.25 Bq g-C^{-1} of background level.

3.2. ^{3}H concentrations in precipitation

During the period of 1992 – 2002, the annual means of ^{3}H concentrations in precipitation in sampling stations N-1 and S-2 were in the range of 67.1 ~ 253 and 69.7 ~ 235 Bq L^{-1}, respectively. At sampling station S-2, the annual means of ^{3}H concentrations in precipitation were correlated with the annual accumulated gaseous tritium discharge rate as shown in Figure 2.

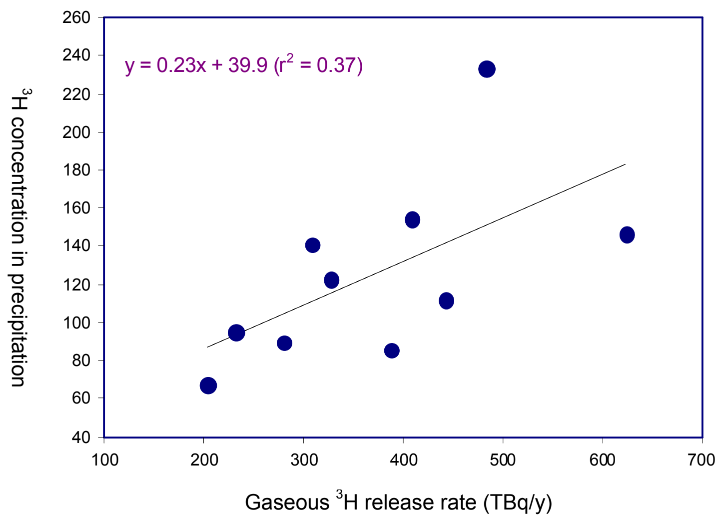

FIG. 2. Correlation between annual mean of ^{3}H concentration in precipitation and gaseous ^{3}H discharge rate.

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Comparison of the measured and calculated results of ^{137}Cs and ^{90}Sr concentration change in the Baltic Sea after Chernobyl Power Plant accident

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Abstract. The concentration data of artificial radionuclides in the Baltic Sea were averaged for the every year in the period of time 1986-2001. The averaged data consist of the measured results in the Baltic Sea proper, in the southeastern area and in the coastal waters. A slow decrease of the average concentration of ^{137}Cs in 1991-1995 and swift decrease in 1996 is defined. The measured and calculated average concentration of ^{137}Cs values were in a good agreement in 1997-1998 and in 2000. The discrepancy of mentioned results in another period of time is possible because of an extra radioactive contamination penetrating from the Bothnian Sea and the local effects. The discrepancy of measured and calculated results of average concentration of ^{90}Sr in the waters of the Baltic Sea is found out. The theoretic results show the tendency of the self-cleaning process of this radionuclide, however the measured data don't indicate this tendency. The measured results are higher of the theoretic one in 1991-2000. The ratio of $^{137}Cs/^{90}Sr$ concentration changed in 1986-2000: the maximum number is 6.5 and minimum is 2.8. The slow self-cleaning of the Baltic Sea waters is possible because of the penetration an extra amount of artificial radionuclides.

1. INTRODUCTION

Concentration of radionuclides ^{137}Cs and ^{90}Sr instability in the Baltic Sea proper, in the southeastern area of the Baltic Sea (SEABS) and in the coastal waters was investigated over the period 1985-2001 following the Chernobyl Power Plant (ChPP) accident [1-4]. The average concentration of ^{137}Cs in the surface waters increased one order more after ChPP accident in comparison with the formed radioactive background (12 Bq/m^3) after the nuclear bomb test [1]. The heterogeneous field of ^{137}Cs concentration was formed in the surface water because of irregular fallout of mentioned radionuclide onto the sea surface. This heterogeneity was resolved very slowly, and only by 1989-1990 the concentration distribution could be described as homogenous with an average value about 150 Bq/m^3 [2, 5]. The ^{90}Sr concentration values were not changed considerably. Before and after the ChPP accident almost the same average concentration of this radionuclide was observed in the Baltic Sea waters [1, 4].

The mathematical models of the self-cleaning process of artificial radionuclides are presented by some authors [6, 7].

The comparison of measured and calculated results of ^{137}Cs concentration in the Baltic Sea since 1989 is optimum, when the concentration values were levelled in the surface waters. The analogues comparison for ^{90}Sr is possible since an earlier period of time, because the after-effect in the Baltic Sea after the ChPP accident was negligible.

The obtained results of ^{137}Cs average concentration in the Baltic Sea waters were in a good agreement with models data (50-60 Bq/m^3) in 2000. The measured average concentration of ^{90}Sr was two times higher (16-19 Bq/m^3) than the calculated result (8 Bq/m^3) [7].

Nowadays the high deviation values of measured concentration of ^{137}Cs from the average data and constantly increased concentration of ^{90}Sr in the Baltic Sea are problematic in the explanation by the natural effects.

2. METHODS

The radiochemical precipitation method of radionuclide concentration and separation was used in the present investigation. The precipitation of ^{137}Cs and ^{90}Sr were carried out from the same water sample of 40-50 *l* and after this procedure the radionuclides were separated [3]. The caesium yield was defined gravimetrically in $Cs_3Sb_2I_9$ form. ^{137}Cs activity was registered by gamma-spectrometers with scintillation and semiconductor detectors.

After purification from additional beta-emitters, ^{90}Sr was measured by ^{90}Y-decay using a beta-radiometer. The strontium yield was determined by atomic absorption spectroscopy method and yttrium yield was measured gravimetrically. Measurement results for ^{137}Cs were obtained with an error of 10 per cent, ^{90}Sr – 15 per cent. The water sampling was carried out during the vessel "Vėjas" cruises.

3. RESULTS AND CONCLUSIONS

A control over the radioactive state of the Baltic Sea waters after the ChPP accident was carried out in the Baltic Sea proper (1986-1989); in the southeastern area (1986-1992, 1995-2001) and in the coastal waters (1986-2001). (Figures 1 and 2).

For the most part the measurement results of ^{137}Cs concentration were heterogeneous. The more stable concentration values of ^{90}Sr radionuclide were observed. The example of radioactive situation in the southeastern area of the Baltic Sea is illustrated in Table 1.

In Table 1 the heterogeneous concentration of radionuclides is shown. The correlation between concentration of radionuclides and hydrometeorological data is not fixed. A heterogeneity of concentration values of radionuclides is possible in a certain degree because of a hydrometeorological situation with an effect of the Curonian Lagoon waters.

It is naturally that the sum of separate measurement results influences upon the averaged data and the tendency of self-cleaning process investigation of the sea. Heterogeneous average values of the radionuclides concentration are observed during the different seasons. (Table 2).

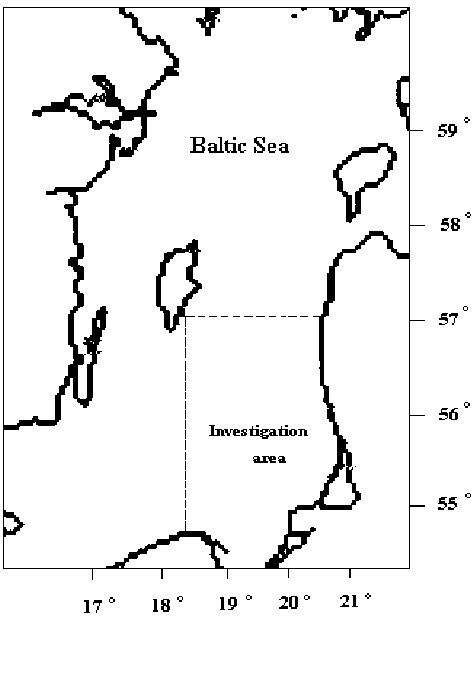

FIG. 1. The investigation area of the radioactive contamination in the Baltic Sea proper and in the southeastern area in 1986-2001.

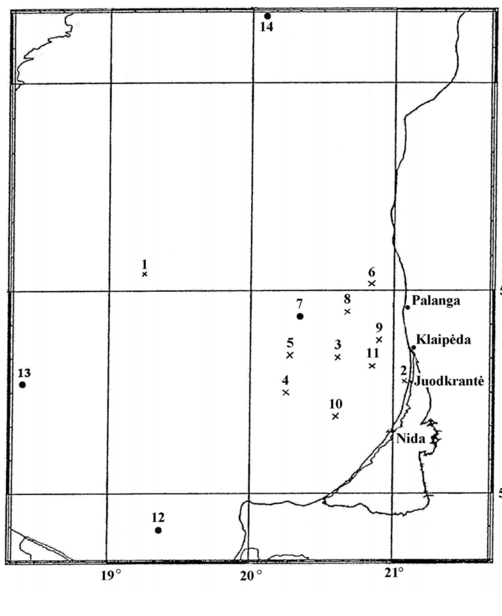

FIG. 2. Oceanographic stations in the southeastern area of the Baltic Sea:
• – international, ✕ – national

TABLE 1. DISTRIBUTION OF ^{137}Cs AND ^{90}Sr CONCENTRATION AND SOME HYDROMETEOROLOGICAL DATA IN THE SURFACE WATERS OF THE SOUTHEASTERN AREA OF THE BALTIC SEA IN FEBRUARY, 1998

Station (Figure 2)	Data	^{137}Cs Bq/m^3	^{90}Sr Bq/m^3	Salinity, %	Tempera-ture, ^{o}C	Wind	
						Velocity, m/s	Direction
1	15.02	-	18	7.25	3.57	7.2	200^{o}
2	14.02	-	18	6.81	2.26	8.6	220^{o}
3	14.02	75	21	7.26	3.92	5.9	270^{o}
4	14.02	89	18	7.14	3.56	11.7	240^{o}
5	13.02	78	17	7.00	3.27	2.9	280^{o}
6	13.02	46	10	6.81	1.99	7.3	270^{o}
7	13.02	73	17	7.02	2.59	4.4	280^{o}
8	13.02	85	18	6.97	2.45	9.1	350^{o}
9	13.02	81	17	6.58	2.57	9.1	360^{o}

TABLE 2. MONTHLY AVERAGE, MAXIMUM AND MINIMUM CONCENTRATION OF ^{137}Cs AND ^{90}Sr (Bq/m^3) DURING THE SAME PERIOD OF TIME IN THE SOUTHEASTERN AREA OF THE BALTIC SEA IN 1996-2000

Date		1996			1997			1998			1999				2000		
		05	08	10	02	05	10	02	08	11	02	05	07	11	02	05	07
^{137}Cs	aver	66	77	75	74	67	50	89	70	51	50	78	120	50	58	48	54
	max	83	84	92	83	79	70	185	83	70	69	130	212	63	64	52	61
	min	65	59	61	65	61	32	46	59	35	44	55	76	36	55	44	51
^{90}Sr	aver	17	26	18	16	19	16	17	17	17	16	18	18	17	21	19	20
	max	27	64	23	22	28	20	21	20	20	23	22	20	21	24	19	22
	min	15	15	14	13	15	12	10	14	14	10	15	13	15	18	19	15

In some cases a little variation of average seasonal concentration of the radionuclides is observed. In other cases the abnormally high concentration of ^{137}Cs is fixed, for instance, in February, 1998 and in May and July, 1999 (Table 2).

To define the general tendency of the self-cleaning process of the Baltic Sea, the concentrations of radionuclides in the Baltic Sea were averaged by every year: for ^{137}Cs – 1986 –2001 and ^{90}Sr – 1985 – 2000. Here the known experimental results of concentration of these radionuclides in the surface waters of the Baltic Sea were used [1-5, 8-11]. The obtained experimental data are given in Figures 3 and 4 (curves 1).

The theoretical curves are illustrated in the same figures too (curves 2). In the both cases a discrepancy of measured and calculated results is observed (Figures 3 and 4). Theoretical and experimental data of ^{137}Cs concentration were in a good agreement only in 1996-1998, 2000 (Figure 3).

A convergence of the same results for ^{90}Sr is observed during a period of time 1985-2000 (Figure 4). Here a self-cleaning process is presented by the calculated data, however experimental results don't show this process.

The ratio of average concentration ^{137}Cs/^{90}Sr was changed in the range of 2.8-6.5 in 1986-2000. The decrease of this ratio depends on faster decrease of ^{137}Cs concentration in the sea water.

The convergence of calculated and measured results of ^{137}Cs concentration in 1991-1995 (Figure 3) can be explained by an extra amount of this radionuclide penetrating from the Bothnian Sea. This fact is not taken into consideration in the theoretic models. Different results between experiment and theory in 1999 and 2001 (Figure 3) is possible because of the local effects.

The convergence of measured and calculated average concentration of ^{90}Sr has place probably as the result of appearance an extra amount of this radionuclide in the Baltic Sea.

An assumption of additional pollution may be a possible cause of the slow self-cleaning of the Baltic Sea of artificial radionuclides.

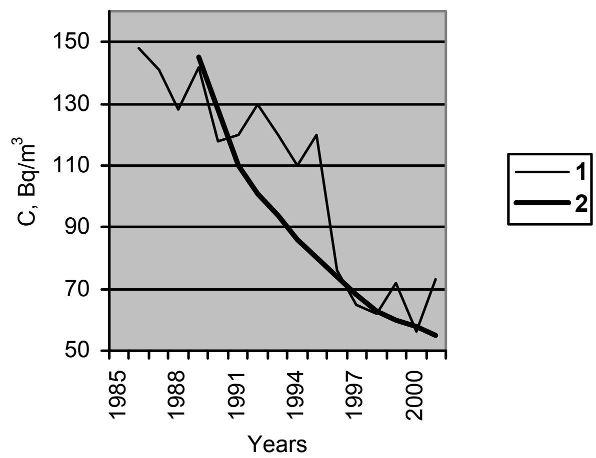

FIG.3. Annual average concentration of ^{137}Cs (Bq/m3) in the surface waters of the Baltic Sea (1986-2001): 1 – measured, 2 – calculated.

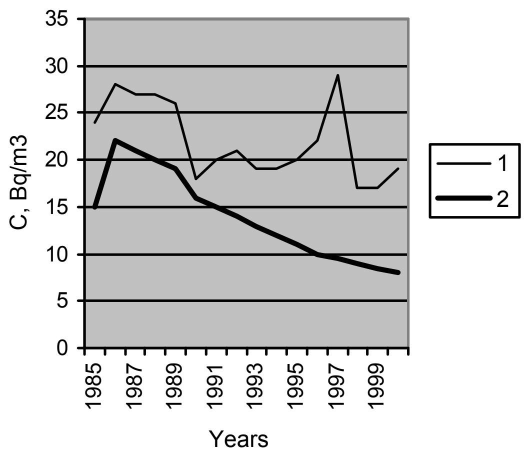

FIG. 4. Annual average concentration of ^{90}Sr (Bq/m3) in the surface waters of the Baltic Sea (1986-2001): 1 – measured, 2 – calculated.

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Kinetic modelling of radionuclide transfer in northern European marine food chains

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Abstract. A biokinetic modelling approach with allometric considerations has been used to estimate the trophic transfer of ^{137}Cs and ^{239}Pu in a marine foodchain consisting of 4 trophic levels including phytoplankton, zooplankton, polar cod, and with harp seal as a top predator. The time-dependent transfer of radionuclides within the foodchain is defined by uptake rates, assimilation efficiencies and egestion/excretion rates and can be described by a system of linear 1st order differential equations. The Matlab based software ECOLEGO was used to solve the equation system numerically, and the activity concentration for animals in each trophic level was found as a function of time. For the simulation concerning ^{137}Cs, the preliminary results suggest that equilibrium is not attained for higher trophic levels, i.e. polar cod and harp seal, before time > 2000 days. Biomagnification appears to occur for the lower trophic levels but is not occurring at the highest trophic level, i.e. seal. For ^{239}Pu, transfer to successively higher trophic levels is low – there is a fall of several orders of magnitude between primary producers, represented by phytoplankton, and polar cod representing trophic level 3-4. However, the model predicts that this decreasing trend in activity concentrations along the food-chain is reversed for the highest trophic level, represented by seal. The simulated results for seal display equilibrium activity concentrations about 2 orders of magnitude higher than those observed for polar cod (one of its prey species). However equilibrium (165 years) is not reached during the life span of a seal. The equilibrium ^{137}Cs CFs are approximately 50, 130 and 70 ml/g for zooplankton, polar cod and seal respectively. The predicted equilibrium ^{239}Pu CFs for zooplankton and polar cod are $2.5 \cdot 10^3$ and 25 ml/g. For seal, following a 1 year equilibration period, a CF of approximately 75 ml/g is predicted.

1. INTRODUCTION

In environmental assessment models the trophic transfer of radionuclides is often described with equilibrium based distribution coefficients or concentration factors. The concentration factor (CF) approach is open to criticism because:

- it provides no information concerning the types of processes/mechanisms in operation during biological uptake,
- the relationship between a radionuclide water concentration and the radionuclide concentration within (the organs or whole body of) a high trophic-level organism, deriving most of its contaminant load from ingested food, may not be a simple, linear one and
- the assumption that the system is under equilibrium, a requirement for CFs to be truly applicable, is often invalid.

Biokinetic models do not suffer from the aforementioned criticisms, and allow a more realistic description concerning the dynamic response of an ecological system to be made, at the expense of a more extensive parameterisation exercise. However, where data are lacking, allometric relationships may provide surrogate values. The allometric approach is based on the observation that metabolic parameters, including basal metabolic rates, ingestion rates, biological half times etc., are proportional to the size of an organism.

2. MODEL STRUCTURE

A food-chain model has been developed to consider the transfer of selected radionuclides (^{137}Cs and ^{239,240}Pu) to "reference" organisms in a pelagic foodchain – specifically harp seal, polar cod and zooplankton. The structure of the foodchain is based on information in the open literature [1] and is represented in Figure 1.

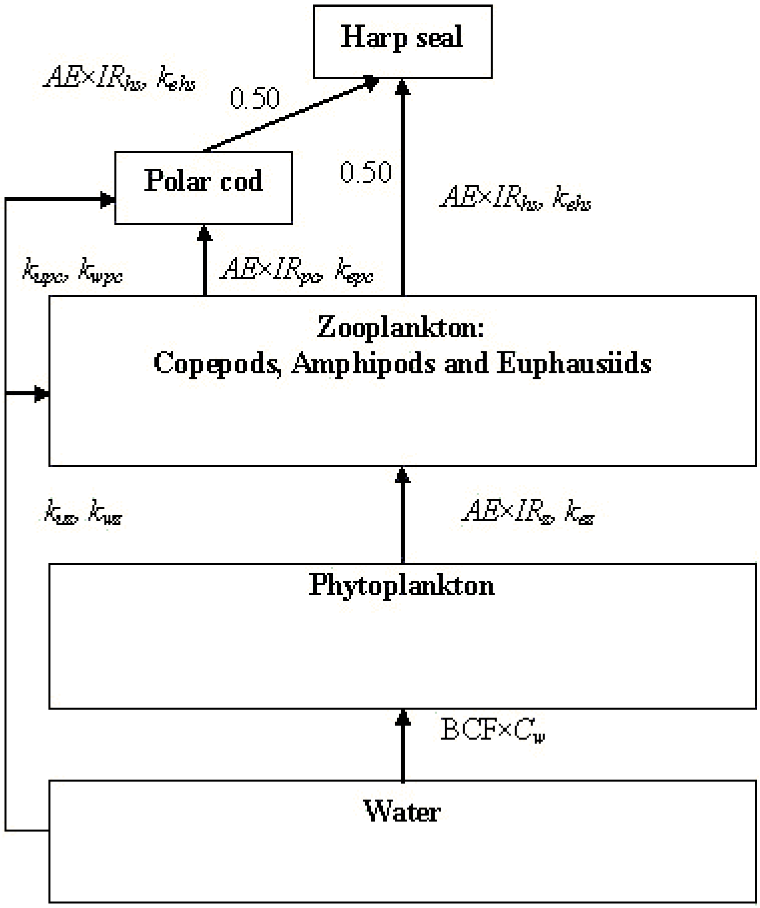

FIG. 1. Foodchain model for harp seal in the Barents Sea, simplified from [1].

The model, based on earlier work ([2], [3], [4]) considers uptake via food and water for aquatic organisms, while the excretion/elimination rate is considered to be independent of the uptake route, and the assimilation efficiency is considered to be independent of food type. A further simplification is that the phytoplankton and the zooplankton (trophic levels 1 and 2) are considered as homogeneous groups described by specified parameter values rather than ranges. We also make the simplifying assumption that the growth rate for all organisms is zero. The time-dependent transfer of radionuclides within the foodchain can be described by simple first order differential equations.

3. PARAMETERISATION OF MODEL

For phytoplankton, equilibrium with water concentrations is assumed and the generically based CFs of 20 and $1 \cdot 10^5$ for Cs and Pu respectively reported in [5] have been used.

Ingestion rates (normalized to the wet weight of the organism), water uptake rates, k_u (d^{-1}), excretion rates k_e (d^{-1}) and assimilation efficiencies, AE (dimensionless) for zooplankton and fish used in the simulations have been tabulated by [2] for different trophic levels. The following allometric relationship for adult seals provided by [6] has been used:

$$\text{IR (kg/day)} = 0.079\text{Wt}^{0.71} \qquad (1)$$

For a seal weighing 160 kg, a weight-normalized IR of 0.018 d^{-1} is found, while AEs for both ^{137}Cs and ^{239}Pu have been set to the same value as that representative of lower levels in the food-chain. The direct uptake from the water column is assumed to be zero.

Allometric relationships may be used to estimate ^{137}Cs and Pu k_es for seal. The following equation has been applied by the USDoE [7] based on earlier considerations [8].

$$\lambda_i = \frac{\ln 2}{3.5 W^{0.24}} \tag{2}$$

$$\lambda_i = \frac{\ln 2}{0.8 W^{0.81}} \tag{3}$$

where:

λ_i = biological decay constant (d^{-1});
W = mass of animal (g, w.w.)

This yields excretion rates of 0.0112 d^{-1} and $5 \cdot 10^{-5}$ d^{-1} for ^{137}Cs and Pu respectively for seal.

4. IMPLEMENTATION OF MODEL

The compartmental model described above, including concomitant parameter values defined thereafter, has been incorporated into and solved numerically using the ecosystem modelling tool ECOLEGO. The water concentration has been set to unit concentration (of ^{137}Cs and ^{239}Pu) through the whole simulation. Radioactive decay is accounted for during the simulation.

The simulation results using the biokinetic allometric model are shown in Figure 2 for ^{239}Pu and ^{137}Cs. For the simulation concerning ^{137}Cs, the preliminary results suggest that equilibrium is not attained for higher trophic levels, i.e. polar cod and harp seal, before time > 2000 days. This has obvious implications in relation to the interpretation of field data if the activity concentrations in water are changing rapidly with time. Biomagnification appears to occur for the lower trophic levels but is not occurring at the highest trophic level, i.e. seal. It should be noted, however, that the uncertainty associated with the excretion rate (k_e) of ^{137}Cs for seal is large and that this parameter has a significant effect on the equilibrium CF. The equilibrium ^{137}Cs CFs are approximately 50, 130 and 70 ml/g for zooplankton, polar cod and seal respectively. These values appear sensible. They compare to IAEA [5] recommended values of 30 for zooplankton and 100 for generic fish.

Several points of interest arise from the simulation for ^{239}Pu (Figure 2). Transfer to successively higher trophic levels is low – there is a fall of several orders of magnitude between primary producers, represented by phytoplankton, and polar cod representing trophic level 3-4. It should be noted, however, that the model predicts that this decreasing trend in activity concentrations along the food-chain is reversed for the highest trophic level, represented by seal. The simulated results for seal display activity concentrations about 2 orders of magnitude higher than those observed for polar cod (one of its prey species) once the system has equilibrated. This prediction is strongly influenced by the fact that the other component of the seal's diet, zooplankton, has a much higher activity concentration associated with it. Equilibrium is attained very slowly for "seal" (reflecting, in part, the very low, allometrically-derived excretion rate). In this case, equilibrium is only truly obtained after 165 years of simulation, i.e. not attained during the animals life time. The model predictions compare quite favourably with the recommended values reported by the IAEA. The equilibrium CFs of $2.5 \cdot 10^3$ and 25 ml/g predicted from model runs for zooplankton and polar cod compare with IAEA recommended values of $1 \cdot 10^3$ and 40 for zooplankton and generic fish respectively. For seal, following a 1 year equilibration period, a concentration ratio of approximately 75 ml/g is predicted. The appropriateness of applying a Pu CF value to a high level predator, like seal, is clearly open to question.

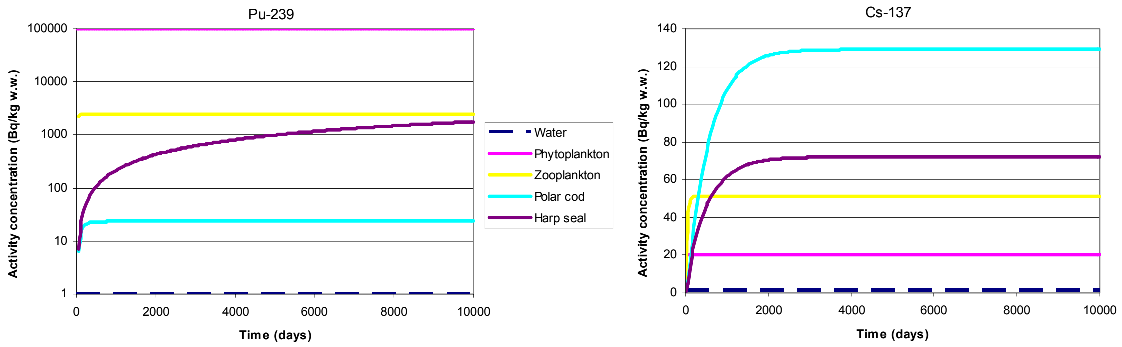

FIG. 2. Activity concentrations (w.w.) of ^{239}Pu and ^{137}Cs for selected marine organisms derived from biokinetic allometric modelling. The simulation was run for a unit activity concentration in water.

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Experiments on radionuclide soil-plant transfer

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Abstract. Some experimental studies were performed in our institute to assess site specific soil-plant transfer factors.A full characterization of an experimental site was done both from pedo-chemical and radiological point of view. Afterwards, a certain number of culture plants were grown on this site and the evolution of their radionuclide burden was then recorded. Using some soil amendments one performed a parallel experiment and the radionuclide root uptake was evaluated and recorded. Therefore, transfer parameters were calculated and some conclusions were drawn concerning the influence of site specific conditions on the root uptake of radionuclides.

1. INTRODUCTION

Due to the great sensitivity of radioactivity measurements a very small amount of radionuclides could be identified and measured in different environmental compartments. This fact gives the possibility to detect and study many complex phenomena. Ecology took advantage from this kind of studies and its progress in the last few years was greater than the whole progress from the previous century.

In order to understand the radionuclides transfer in agricultural lands and semi-natural systems it is necessary to know the mechanisms that determine their behaviour because this kind of systems are both complex and various.

2. EXPERIMENT DESCRIPTION

A radioecological study was initiated in 1998, within the Institute for Nuclear Research, to assess the site-specific transfer factors of radionuclides from soil to plants. One identified a contaminated area within the limits of the waste-water treatment station. A proper delimitation and radiological characterization of the selected site was performed followed by preparative works. Radioactive contamination was identified as being spread sludge from the settling stage of the spent industrial water treatment.

The lot was homogenized and soil samples were collected for radionuclide content determination. The main radionuclides identified were: ^{137}Cs, ^{60}Co, ^{234m}Pa, ^{235}U, ^{134}Cs and ^{54}Mn. The lot was divided into several parcels considering the contamination pattern obtained from soil samples measurement.

A suggestive image of radioactive contamination is given in Figure 1 where ^{137}Cs activity distribution is plotted. A similar pattern was obtained for ^{235}U activity, also, showing that radioactive contamination originated from the same source, namely the sludge (see Figure 2).

Over the selected parcels one cultivated pea, carrots, potatoes and maize. Vegetation samples were collected in different plant growth stages and soil samples were collected at the maturity of cultivated plants.

Laboratory analyses were performed to determine the main physiochemichal parameters of the soil, and one established that the soil was of alluvial type with a slight alkaline reaction and enhanced clay content.

The soil samples were collected over a depth down to 20 cm, to assure their representativeness for the root developing soil layer. The soil samples were prepared by free drying, milling, sieving and oven drying to a constant mass; the vegetation samples were prepared by free drying, oven drying to constant mass and volume reduction by turning into ash. Each sample was brought in a standard measurement geometry and then counted using a high resolution spectrometer.

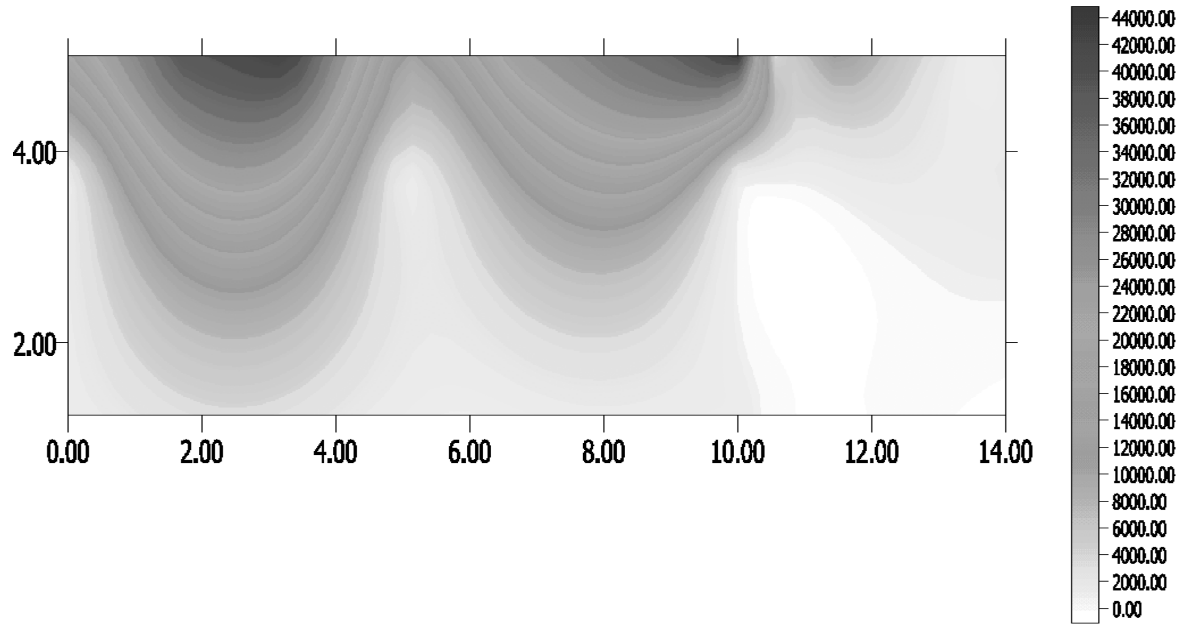

FIG. 1. Concentration distribution map for ^{137}Cs.

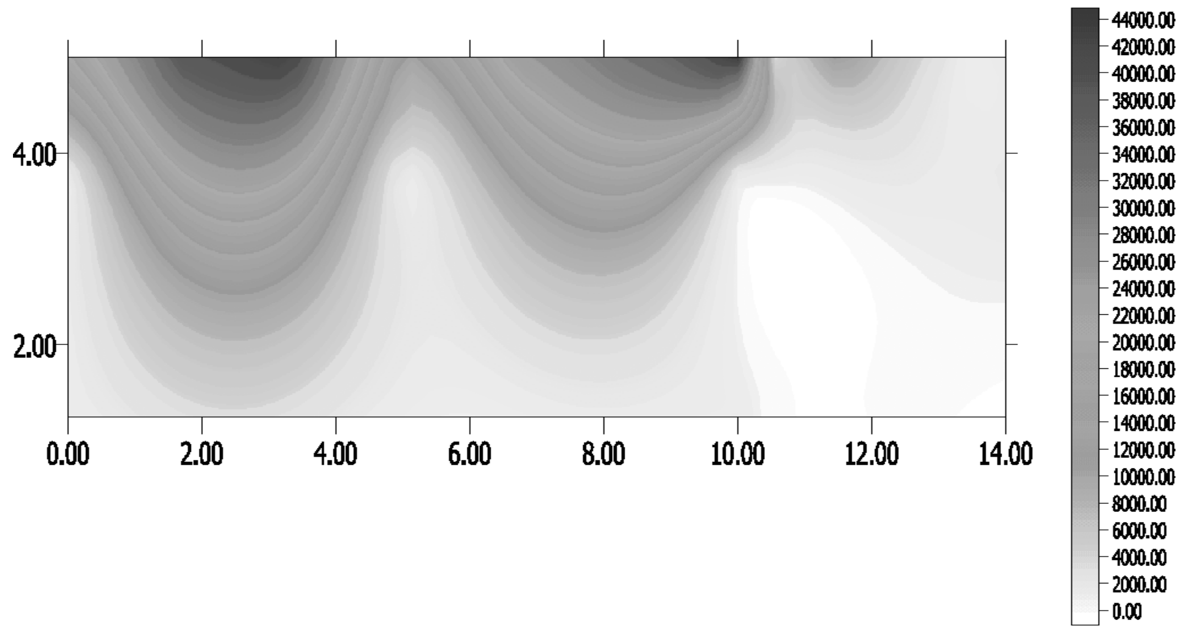

FIG. 2. Concentration distribution map for ^{235}U.

3. RESULTS

Transfer factor was calculated for each radionuclide using the formula below:

$$TF = \frac{\text{radionuclide activity per unit mass of dry vegetation (Bq/kg)}}{\text{radionuclide activity per unit mass of dry soil (Bq/kg)}}$$

For the pea samples the transfer factors were calculated and are given in Table 1.

TABLE 1. TRANSFER FACTORS OBTAINED FOR PEA SAMPLES

Radionuclide	Transfer factor		
	Aerial part during growing period	Aerial part at maturity	Seeds
^{228}Th	0.11	0.04	0.0478
^{54}Mn	0.05	0.05	0.0186
^{60}Co	0.01	0.01	0.0008
^{134}Cs	0.04	0.03	0.0152
^{137}Cs	0.04	0.03	0.0164
^{234m}Pa	0.01	0.01	–
^{235}U	0.01	0.01	0.0005

TABLE 2. TRANSFER FACTORS OBTAINED FOR POTATOES AND CARROTS SAMPLES

Radionuclide	Transfer factor		
	Potatoes (tuber)	Carrots (leaves)	Carrots (root)
^{228}Th	–	0.078	0.051
^{54}Mn	0.0030	0.103	–
^{60}Co	0.0006	0.059	0.017
^{134}Cs	0.0052	0.097	0.003
^{137}Cs	0.0057	0.089	0.047
^{234m}Pa	0.0006	0.059	0.014
^{235}U	0.0005	0.058	0.022

TABLE 3. TRANSFER FACTORS OBTAINED FOR MAIZE SAMPLES

Radionuclide	Transfer factor	
	Maize seeds	Maize plant
^{228}Th	0.014	0.303
^{60}Co	0.0002	0.086
^{137}Cs	0.025	0.110
^{234m}Pa	0.014	0.122
^{235}U	0.0003	0.118

Potatoes and carrots showed a different behaviour in radionuclide uptake the transfer factor being one order of magnitude lower for potatoes comparing with the carrot for all radionuclides.

The transfer factor calculated for the maize samples revealed that maize seed accumulate a lesser amount of radionuclides than the rest of the plant.

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Does size matter?

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Abstract. A challenge in the estimation of doses to biota is the large number of potential radionuclide-organism transfer parameters which are required. To address this allometric (weight dependent, trans-species) relationships have been proposed. Here, these relationships are considered and used to derive a hypothesis for trans-species concentration ratios describing the radionuclide transfer from diet to animals.

1. INTRODUCTION

An important step in assessments of the potential impact of ionising radiations on biota is the estimation of radionuclide transfer to organisms. This means that transfer parameters are required for a large number of biota-radionuclide combinations. Recently, a number of workers have suggested the use of allometric (weight dependent) relationships to provide qualitative estimates of transfer to animals over orders of magnitude of body mass [1-3]. Here, these proposed relationships are discussed and used to suggest a simpler approach.

2. ALLOMETRY

Size influences rates of many biological structures and of a variety of processes from cellular metabolism to population dynamics [4]. The dependence of a biological variable Y on a body mass M is typically characterised by an allometric scaling law of the form:

$$Y = aM^b \tag{1}$$

where a and b are constants.

Many biological phenomena scale as quarter powers of mass [4]. For example: metabolic rates scale as $M^{0.75}$; rates of cellular metabolism and maximal population growth rate as $M^{-0.25}$; life-span and embryonic growth and development as $M^{0.25}$; cross sectional areas of mammalian aortas and tree trunks as $M^{0.75}$. Discussions for this phenomena can be found within the literature [4, 5].

2.1. Allometry in radiological assessment

A number of authors have proposed allometric relationships describing the (long-component of) biological half-life ($T_{0.5bio}$) for a range of radionuclides derived from data including a number of phyla [1,6-11]. For many of the radionuclides considered (including Cs, Sr, Co, U and organically bound ^3H), the scaling constant is close to 0.25, as observed for many biological parameters. This is perhaps to be expected as biological processes control the uptake and turnover of all elements, including radionuclides, by animals. Of the radionuclides considered to date, exceptions which do not scale to *circa* 0.25 include the actinide elements [7].

Table 1 compares collated aggregated transfer values (Tag ($m^2\ kg^{-1}$); the ratio of the whole body fresh weight activity concentration ($Bq\ kg^{-1}$) and radionuclide concentration in soil ($Bq\ m^{-2}$)) for Cs to those derived using an allometric $T_{0.5bio}$ approach [12]. The predicted values generally compare well to the available measurements demonstrating the potential of the approach for use within the modelling of the exposure of wild animals to ionising radiation.

TABLE 1. A COMPARISON OF TAG VALUES FOR CS TO DIFFERENT SPECIES WITH COLLATED DATA (FROM [12])

	Reindeer ($m^2\ kg^{-1}$)	Vole ($m^2\ kg^{-1}$)	Fox ($m^2\ kg^{-1}$)	Willow ptarmigan ($m^2\ kg^{-1}$)
Allometric prediction	1.0×10^{-1}	2.4×10^{-2}	9.9×10^{-2}	2.7×10^{-2}
Collated data mean (range)	1.3×10^{-1} (8.8×10^{-4}-5.8×10^{-1})	4.5×10^{-2} (2.2×10^{-2}-5.7×10^{-2})	8.3×10^{-3} (1.3×10^{-3}-2.2×10^{-2})	1.0×10^{-2} (2.4×10^{-4}-4.1×10^{-2})

Other authors have proposed allometric relationships to estimate equilibrium transfer coefficients (F_f)1 for Cs and I [3, 13]. Unsurprisingly, given the relationship between F_f and $T_{0.5bio}$, these have scaling constants close to 0.75. On the basis of these results, Sheppard [2] hypothesised that a similar scaling factor would be appropriate for all radionuclides.

3. A CONSTANT CONCENTRATION RATIO?

The allometric relationship derived by MacDonald [3] for the Cs F_f for birds and mammals is:

$$F_f = 10.2M^{-0.777} \tag{2}$$

and the definition of F_f (see footnote 1) can be represented as:

$$F_f = \frac{[WB_{Cs}]}{[Diet_{Cs}] \times DMI} \tag{3}$$

where:

$[WB_{Cs}]$ is the animal whole body activity concentration of Cs (Bq kg^{-1} fresh weight);
$[Diet_{Cs}]$ is the Cs activity concentration of the diet (Bq kg^{-1} dry matter);
DMI is the daily dietary dry matter intake (kg d^{-1} dry matter).

Many authors have proposed allometric relationships for the DMI of animals, in the work of MacDonald [3] the following was adopted (for mammals):

$$DMI = 0.0687M^{0.822} \tag{4}$$

Equation (3) can be rewritten incorporating Equations (2) and (4) as:

$$10.2M^{-0.777} = \frac{[WB_{Cs}]}{[Diet_{Cs}] \times 0.0687M^{0.822}} \tag{5}$$

and can be further rearranged to:

$$\frac{[WB_{Cs}]}{[Diet_{Cs}]} = 10.2M^{-0.777} \times 0.0687M^{0.822} \tag{6}$$

1 The equilibrium transfer coefficient (F_f; d kg^{-1}) is the ratio of the activity concentration (Bq kg^{-1} fresh weight) in an animal or tissue to the daily intake of radionuclide (Bq d^{-1}).

TABLE 2. A COMPARISON OF CS CR VALUES FOR DIFFERENT ANIMAL SPECIES

	CR	Reference
Red Deer	0.44±0.04	[14]
Rabbit	0.27±0.02	[14]
Blue Hare	0.48±0.03	[14]
Black Grouse	0.63±0.22	[14]
Red Grouse	0.75±0.12	[14]
Sheep	0.6-0.7	[15]

At this stage, we can propose the hypothesis that the scaling constants on the two allometric relationships are not numerically significantly different, and therefore:

$$\frac{[WB_{Cs}]}{[Diet_{Cs}]} = 10.2 \times 0.0687 = 0.70 \tag{7}$$

where $\frac{[WB_{Cs}]}{[Diet_{Cs}]}$ is the concentration ratio (CR) between whole body and dietary Cs activity concentrations which is hypothesised to be a constant value (of 0.7) for bird and mammal species. To test this hypothesis, Table 2 presents reported CR values for a number of species. Whilst this cannot represent a validation of Equation (7) it does demonstrate that the variation in the CR values for a range of different herbivorous species is not large.

An alternative derivation of the hypothesis for a constant CR value can also be derived from the allometric relationship of $T_{0.5bio}$ starting with the equation:

$$\frac{d[WB_{Cs}]}{dt} = \frac{f_1 \times [Diet_{Cs}] \times DMI}{M} \times \frac{0.693}{T_{0.5bio}} \times [WB_{Cs}] \tag{8}$$

where:

t is time (d);
f_1 is the fractional gastrointestinal absorption which will here be assumed to be 1 for Cs [6].

At equilibrium, and using allometric relationships for DMI (as above) and Cs $T_{0.5bio}$ (from [12]), Equation 8 can be rearranged to:

$$\frac{0.693 \times [WB_{Cs}]}{13.22M^{0.237}} = \frac{[Diet_{Cs}] \times 0.0687M^{0.822}}{M} \tag{9}$$

and from Equation (9) an expression describing CR can be obtained:

$$\frac{[WB_{Cs}]}{[Diet_{Cs}]} = \frac{0.0687M^{0.822} \times 13.22M^{0.237}}{M} \tag{10}$$

Since the product of $M^{0.822}$ and $M^{0.237}$ is unlikely to be significantly different to M^1, then from Equation (10) a Cs CR value of 0.9 can be derived.

4. DISCUSSION

Equations (7) and (10) both suggest a constant CR value describing transfer of Cs from the diet to tissues of different animal species. The limited data reviewed within Table 2 goes some way to support this hypothesis. Obviously, the values of CR derived for Cs here are open to discussion. For instance, many allometric expressions for describing DMI can be found in the literature and f_1 may be lower in ruminant than monogastric animals [6]. Similar algebraic derivations could be made for other radionuclides for which allometric relationships of $T_{0.5bio}$ or F_f have been shown to scale to *circa* 0.75 and 0.25 respectively. Adopting a hypothesis similar to that of Sheppard [2] leads to the suggestion of trans-species CR values for all radionuclides. However, the scaling constants derived for $T_{0.5bio}$ for some radionuclides, including the actinides (currently reported to scale to *circa* 0.8 [7]), suggest that an assumption of a trans-species CR may not be valid for all radionuclides. In the case of actinides, this may reflect the lack of any biological role or analogues with a biological role. Alternatively, it may more simply be the result of the paucity of available data for these radionuclides.

ACKNOWLEDGEMENTS

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Predicting tritium and radiocarbon in wild animals

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Abstract. We have assessed the tritium and ^{14}C concentrations in wild animals under conditions of continuous emission. For ^{14}C we have used a specific activity approach,. For tritium we have distinguished between tritiated water and organically bound tritium, adapting a metabolic model previously tested for farm animals. We have considered different European climatic zones (Arctic, Continental, Maritime and Mediterranean) assuming a constant yearly deposition (1 Bq y^{-1}) or constant atmospheric concentration (1 Bq m^{-3}). Less uncertainty is obtained assumining constant atmospheric concentrations, as tritiated water loss, required when assuming a constant deposition, within the different climatic zones is poorly defined.

1. INTRODUCTION

Biota can be exposed to ionising radiation as a consequence of the routine operation of nuclear installations or in the event of accidents at such facilities. While initially radiological protection was focused on human, today it is being extended to include other organisms. Here we describe the approach taken to assess ^{14}C and tritium concentrations in selected biota within two European projects which are developing environmental protection frameworks: FASSET [1] and EPIC [2]*. Tritium can be present in and emitted by some nuclear installations at comparatively high levels (e.g. heavy water reactors, fuel reprocessing plants, military factories or future fusion reactors). Due to the high potential release the environmental impact needs to be assessed, even though tritium has a low radiotoxicity. Carbon-14 is among the radionuclides of concern in waste management. Our assessment takes into account some of the major differences between animals concerning dietary habits and distinguishes between tritiated water (HTO) and organically bound tritium (OBT) in diet, whereas previous assessments have considered *total* ^{3}H [3]. In order to cover typical conditions across the Europe, we have considered Arctic, Continental, Maritime and Mediterranean climate.

2. BASIC APPROACH

Within the EPIC and FASSET projects radionuclide concentrations in plants and animals are being compared to the deposited concentration (Bq m^{-2}). However, tritium and ^{14}C enter the environment as reversible gases ($^{14}CO_2$, HTO) and the current practice is to assume a constant air concentration and compare concentrations in biota to this. An effective yearly deposition can be defined for a constant air concentration by using the radionuclides stored yearly in surface crops (biomass production) and, for the case of HTO, escaped to groundwater. While biomass production can be defined with acceptable reliability across European ecosystems, the HTO escape under conditions of the different climatic zones is poorly defined. Due to this we have concentrated on modelling a constant air concentration but we have also considered a constant annual deposition.

The ^{14}C concentration in herbage and animals was assessed using the specific activity approach for both a constant air concentration or a constant yearly deposition:

$$C_{p(a)} = (X/X_C) * M_{C,p(a)} \quad \text{or} \quad C_{p(a)} = (D_{C14}/B_C) * M_{C,p(a)} \tag{1}$$

* EPIC is specifically considering the European Arctic.

where:

$C_{p(a)}$ the ^{14}C concentration in fresh mass of plant or animal;
X is the ^{14}C concentration in air;
X_C is the carbon content in the air (0.18 g m^{-3});
D_{C14} the yearly deposition;
B_C the biomass production of organic carbon;
$M_{C,p(a)}$ the mass of carbon per kg fresh weight (fw) of plant or animal body.

A more elaborate formalism was used for tritium, in order to distinguish between tritiated water and organically bound tritium produced in plants during photosynthesis and consequently entering the food chain. An updated theory for wet deposition of tritiated water was used [4]. The dry and wet contribution of HTO deposition was considered for both plant HTO and OBT concentrations using recent results of the IAEA BIOMASS CRP Tritium Working Group [5]. To describe transfer to animals we used a metabolic derivation of transfer factors from HTO and OBT in diet to HTO and OBT in animal produce [6]. This has previously been successfully tested for farm animals. The fresh weight HTO and OBT concentrations in animals (C_{HTO} and C_{OBT}) are given by:

$$C_{HTO} = \nu_{bw} * C_{med} + F_{Dom} * \frac{\nu_{bw}}{WF} * (I_{f,dm}/FD) * C_{OBTf} \qquad (2)$$

$$C_{OBT} = \frac{0.25 m_{ot}}{0.111} * C_{med} + (0.75 * m_{ot}/C_{obh}) * C_{OBTf}/FD \qquad (3)$$

where:

ν_{bw} body water fraction;
m_{ot} body bound hydrogen (BH) content kg BH kg^{-1} body;
F_{Dom} digestibility coefficient of organic matter in food;
WF total water flux per day (drinking, food and metabolic) L d^{-1};
C_{obh} is the bound hydrogen concentration in food (kg BH kg^{-1} dry matter);
FD the dietary dry matter fraction;
C_{OBTf} is the OBT concentration in the fresh food;
C_{med} is the daily HTO intake from food and drinking water, divided by the water flux;
$I_{f,dm}$ is the daily dry matter intake given by the field metabolic rate and the utilisable energy density in food.

3. DATABASE

For each climatic region, data for average biomass production, growing season, relative and absolute humidity, seasonal temperature, and precipitation have been collated from European databases. For tritium, the loss of HTO to groundwater, as needed for an annual deposition scenario, is highly uncertain. Both the FASSET and EPIC projects have suggested reference organisms and species representative of these [1, 2]. Typical diets have been established and data on plants and animal compositions (carbon, water and organic hydrogen) collated.

Considerable effort was dedicated to the best assessment of water flux and food intake by wild animals. For water flux the most recent references [7] use a mass dependent allometric equation distinguishing between carnivorous, granivorous, omnivorous and herbivorouss mammals, and carnivorous, passerine and non-passerine birds. After a thorough assessment of the literature on food we have used the mass allometric equations of Nagy [8] based on experimental field metabolic rate and average diet characteristic for carnivorous, omnivorous, granivorous and herbivorous mammals and birds. We also considered a universal equation of field metabolic rate for small mammals [9] dependant on body mass, ambient temperature and site latitude. This equation shows a factor of two variability across Europe in the estimated food intake of small mammals of constant mass. When applying it to our relationships for tritium concentration in animals we have observed that the tritium concentration was over-predicted compared with the upper limit defined by the specific activity approach. This is explained by the mismatch between the empirical relationships of water flux and food intake, the former depending only on body mass and not on food intake and an excess temperature, as physiologically required if used in conjunction with the latitude dependent food intake equation. Until better relationships for water flux are obtained, we cannot investigate the predicted variability in animal concentrations across Europe more thoroughly.

4. RESULTS

For a yearly deposition of 1 Bq m^{-2} of ^{14}C, and linking biomass and deposition, the air concentration varies between 0.9 and 4.3 mBq m^{-3}; biomass production explaining predicted variability in animal concentrations between climates. Using the specific activity approach, variability of ^{14}C concentrations in a defined climate depends on the carbon contents of plants and animals. Seeds have about 2.5 more ^{14}C than composite food for herbivores. Worms have the lowest concentrations (high water content) and a lemming (high fat content) the highest. Due to low biomass production, the highest predicted ^{14}C concentrations are in the Arctic. Results are highly dependent on biomass and less dependent on the carbon content of plants and animals which are with moderate certainty. When modelled as a constant air concentration, there are no differences between climates and the variability of ^{14}C concentration across plants and animals depends solely on their carbon content.

We only report results considering constant air concentration, as commonly practiced for reversible gases, for ^{3}H (i.e. we do not attempt to model a constant deposition scenario as we do not have the required climaticly varying loss via ground water inputs). Tables 1-3 present results assuming 1 Bq y^{-1} m^{-3} ^{3}H for HTO, OBT and total tritium, for animals representative of FASSET reference organisms. Across climates the highest concentrations are predicted for the Arctic and the lowest for the Mediterranean. Considering various animals, there is a factor of two difference in total tritium concentrations but more than a factor of four for OBT, largely reflecting the influence of body composition.

For EPIC the list of reference Arctic animals differs and results are given in Table 4. Lemming and Arctic fox are predicted to have the highest concentrations.

One of our goals was to assess the importance of OBT contribution to total ^{3}H concentrations, as OBT has a slower turnover. The highest contribution of 40% is for lemming (high fat content) and the lowest for worm and red grouse. The dose coefficient for OBT ingestion (Sv Bq^{-1}) is 2-3 times higher than for HTO intake.

In conclusion we have demonstrated the potential contribution of OBT to wild animal tritium concentrations under equilibrium conditions and have shown that our metabolic approach together with a careful choice of all environmental parameters gives results close to the specific activity limit. The environmental influence on animal concentrations cannot be fully argued in the absence of correlated food and water intakes across changes in ambient temperature. The work comprises a first step to establishing a database for a further dynamic modelling.

TABLE 1. PREDICTED HTO CONCENTRATIONS (Bq kg^{-1} fw)

Climate	Worm	Mole	Rabbit	Moose	Weasel	Red fox	Red grouse	Lark egg
Medit	5.76E+01	4.23E+01	5.08E+01	5.16E+01	5.56E+01	5.19E+01	4.93E+01	4.23E+01
Cont	7.44E+01	5.59E+01	6.79E+01	6.90E+01	7.51E+01	6.97E+01	6.58E+01	5.50E+01
Marit	9.80E+01	6.96E+01	8.20E+01	8.29E+01	8.80E+01	8.30E+01	7.99E+01	7.50E+01
Arctic	8.82E+01	7.73E+01	8.54E+01	8.71E+01	9.61E+01	9.25E+01	8.26E+01	6.59E+01

TABLE 2. PREDICTED OBT CONCENTRATIONS (Bq kg^{-1} fw)

Climate	Worm	Mole	Rabbit	Moose	Weasel	Red fox	Red grouse	Lark egg
Medit	8.81E+00	2.16E+01	2.90E+01	2.48E+01	2.95E+01	2.67E+01	1.90E+01	1.13E+01
Cont	1.19E+01	2.89E+01	3.90E+01	3.35E+01	3.98E+01	3.60E+01	2.56E+01	1.51E+01
Marit	1.39E+01	3.45E+01	4.61E+01	3.95E+01	4.67E+01	4.26E+01	3.03E+01	1.92E+01
Arctic	1.88E+01	4.41E+01	4.97E+01	4.27E+01	5.10E+01	4.82E+01	3.26E+01	1.89E+01

TABLE 3. PREDICTED TOTAL ^3H CONCENTRATIONS (Bq kg^{-1} fw)

Climate	Worm	Mole	Rabbit	Moose	Weasel	Redfox	Red grouse	Lark egg
Medit	6.64E+01	6.39E+01	7.98E+01	7.64E+01	8.51E+01	7.86E+01	6.83E+01	5.36E+01
Cont	8.63E+01	8.48E+01	1.07E+02	1.02E+02	1.15E+02	1.06E+02	9.15E+01	7.00E+01
Marit	1.12E+02	1.04E+02	1.28E+02	1.22E+02	1.35E+02	1.26E+02	1.10E+02	9.42E+01
Arctic	1.07E+02	1.21E+02	1.35E+02	1.30E+02	1.47E+02	1.41E+02	1.15E+02	8.48E+01

TABLE 4. PREDICTED ^3H CONCENTRATIONS (Bq kg^{-1} fw) IN REPRESENTATIVE SPECIES OF ARCTIC REFERENCE ORGANISMS

Animal	Lemming	Vole	Reindeer	Ptarmigan	Arctic fox	Egg ptarmigan
HTO	8.05E+01	9.46E+01	7.26E+01	8.26E+01	9.67E+01	9.91E+01
OBT	6.66E+01	4.93E+01	5.20E+01	4.26E+01	5.25E+01	3.26E+01
Total	1.47E+02	1.44E+02	1.25E+02	1.25E+02	1.49E+02	1.32E+02

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A survey of the radionuclide concentrations in some characteristic bioindicators in Montenegro

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Abstract. The results presented are obtained in the scope of the Programme of systematic monitoring of radionuclide concentrations in the environment in the Republic of Montenegro in the year 2002. Gamma-spectrometry analyses of mushroom samples, as well as of two sea organisms (mussels and squids) are shown. In parallel, gamma-spectrometry analyses of fall-out samples (Podgorica) and of seawater (collected on the locations near Bar and Herceg-Novi) are given, too. At the other side, two types of meat samples, from the food control programme are analyzed: beef and lamb. All results given are in form of the mean value and the range of a number of single samples of the particular type.

1. INTRODUCTION

Systematic monitoring of the radionuclides in the environment in the Republic of Montenegro has been in place for some twenty years, with the Center for Eco-toxicological Research, Podgorica, being engaged from 1998. The monitoring programme includes: radionuclide concentrations in the air, fall-outs, lake and sea waters, soil, drink waters, food, forage, building construction materials, etc., as well as indoor (schools, dwellings, offices, public places, etc.) and outdoor radiation exposition measurements and control.

As to bio-indicator organisms, the following ones are routinely and regularly analyzed by gamma-spectrometry methods: mushrooms, sea mussels and squids. In this work we show the results for the year 2002. The results for the fall-out samples and seawater are given for the same period for the comparison and interpretation. Results for the beef and lamb samples from the animals grown in our region are also shown.

Mushrooms are well known bio-indicators of the radionuclides in the environment. In Montenegro, one particular type is especially interesting, as it is collected in large quantities and exported, mostly to EU countries (however, not so frequent in local diet), namely cep (*Boletus Edulis*). The cep appearance and collection in the summer 2002 was by far the largest ever known, some individual examples weighing up to 4–5 kg. The samples analyzed were obtained from the commercial collection/buy-out network.

Sea water is sampled at two coastal locations, Bar and Herceg-Novi. Every sample is a cumulative 30 liter lot, collected during one month period (i.e. 30 times 1 liter). Fall-out samples are collected also during one month periods in Podgorica.

Mussels and squids, as indicator organisms, are sampled once in six months at the same spots where sea water is taken for analyses.

Beef and lamb are analyzed also once per six months, the samples being taken directly from the farmers, or from the butchers (taking care about the origin).

2. MEASUREMENT METHOD

All gamma-spectra are recorded by a low background gamma-spectrometry system with 40% efficiency HPGe detector. Energy calibration is performed with a set of sources containing ^{152}Eu, ^{137}Cs, ^{133}Ba, ^{88}Y, ^{60}Co and ^{57}Co, while efficiency is calibrated by means of a voluminous radionuclide mixture ^{139}Ce, ^{57}Co, ^{60}Co, ^{137}Cs, ^{203}Hg, ^{113}Sn, ^{85}Sr and ^{88}Y (supplier: Czech Metrological Institute, Prague). Data processing is effectuated by commercial soft wares Gamma Vision 32, Nuclide Navigator and Angle [1].

Mushrooms samples are prepared from the fresh fungi: cleaned, washed, homogenized and set into 1 liter Marinelli beakers and then counted for 10 000 to 25 000 s. For mussels, edible part is separated from the shell, homogenized, and counted for 30 000 to 170 000 s. Squids and meat samples are processed in the same way.

Cumulative sea water samples are evaporated from 30 liter to 1 liter volumes and counted in Marinelli beakers for 30 000 to 90 000 s. The same is done with the fall-out ones. The results obtained are normalized to the collection date for those isotopes where such decay periods are relevant.

3. RESULTS AND DISCUSSION

The results are given in the following Tables 1-6, with:

N - number of single samples
R - range of the results obtained
A - mean value
M - median

TABLE 1. MUSHROOMS

N	Isotope	R (Bq/kg)	A (Bq/kg)	M (Bq/kg)
6	^{40}K	27.4–96.2	63.8	70.7
6	^{137}Cs	2.35–17.3	10.2	10.3
6	^{226}Ra	0.14–1.85	0.99	1.19
6	^{232}Th	0.07–1.94	0.51	0.28

TABLE 2. SQUIDS

N	Isotope	R (Bq/kg)	A (Bq/kg)	M (Bq/kg)
3	^{40}K	62.6–88.4	87.3	79.4
3	^{137}Cs	0.04–0.19	0.09	0.11
3	^{226}Ra	0.11–0.97	0.36	0.48
3	^{232}Th	0.02–0.43	0.10	0.18

TABLE 3. MUSSELS

N	Isotope	R (Bq/kg)	A (Bq/kg)	M (Bq/kg)
3	^{40}K	40.5–46.8	44.6	43.9
3	^{137}Cs	0.08–0.12	0.1	0.10
3	^{226}Ra	0.05–0.36	0.15	0.19
3	^{232}Th	0.03–0.1	0.09	0.07

TABLE 4. PRECIPITATION

N	Isotope	R (mBq/l)	A (mBq/l)	M (mBq/l)
12	^{40}K	5.17–180	61.0	32.3
12	^{137}Cs	0.20–13.1	2.08	1.17
12	^{226}Ra	1.25–29.2	5.76	3.50
12	^{232}Th	0.18–9.73	2.01	1.44

TABLE 5. SEA WATER

N	Isotope	R (mBq/l)	A (mBq/l)	M (mBq/l)
24	^{40}K	6930–12600	9980	10300
24	^{137}Cs	1.67–5.27	3.33	3.40
24	^{226}Ra	1.61–19.8	6.69	5.87
24	^{232}Th	1.09–7.75	3.72	3.46

TABLE 6. BEEF AND LAMB MEAT

N	Isotope	R (Bq/kg)	A (Bq/kg)	M (Bq/kg)
12	^{40}K	77 - 115	100	99.5
12	^{137}Cs	0.13 – 3.68	1.07	0.80
12	^{226}Ra	0.10 – 026	0.19	0.18
12	^{232}Th	0.02 – 0.23	0.08	0.07

In all the samples analyzed ^{40}K, ^{226}Ra, ^{232}Th and ^{137}Cs radionuclides are found. Besides, ^{7}Be is found in mussels. ^{7}Be concentration results are as on the sampling day. For the sea water ^{7}Be activity was below detection limit, which is (0.04–0.14) Bq/kg, depending on the counting geometry and counting time. ^{40}K, ^{226}Ra and ^{232}Th are natural isotopes, while ^{137}Cs is mainly from the Chernobyl accident. ^{7}Be has cosmogenic origin, from the interaction of cosmic radiation with oxygen and nitrogen in the atmosphere (about 75% ^{7}Be being produced in the stratosphere and 25% in the upper troposphere). It falls to lower atmosphere, and further reaches ground waters and seas. ^{7}Be is being used as a tracer when studying air masses flows and displacements. Its concentrations in the air layers above the ground depend on winds and fall-outs. In Belgrade, for instance these concentrations vary in (0.8–10) mBq/m^3 range, with similar values being found elsewhere in Europe, too [2, 3].

4. CONCLUSIONS

Radionuclide concentrations in mushrooms are relatively high. As to human diet, where they are in most cases represented with a tiny portion, this is not contributing significantly to the total radiation dose by intake. However, it would be interesting to investigate how this influences the animals for which mushrooms represent a significant portion of the diet (e.g. bugs, worms, snails and higher animals further in the food chain).

As radionuclide indicators, mushrooms are significant organisms, the results therefore being more important in particular samples than as average values.

In all mussel samples ^{7}Be is found in relatively high concentrations (9.9–42.1) Bq/kg. Since ^{7}Be concentrations are below detection limit (0.14 Bq/l) in sea water, and in (0.02–2.65) Bq/l range in fall-out, mussels obviously do accumulate this cosmogenic radionuclide in its particular way.

As to the other radionuclides and other bio-indicators, the concentrations found are comparable or even below those in beef and lamb meat samples.

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Investigation of vertical migration of ^{54}Mn, ^{58}Co, ^{63}Ni, and ^{55}Fe in the soil-water-plants system

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Abstract. The work studied the vertical migration of radionuclides ^{54}Mn, ^{55}Fe, ^{58}Co, and ^{63}Ni in coriander, fennel, and onion, which are widely used in food of Tajikistan's population. The results showed that these radionuclides accumulate only in these plants' roots. It is connected with formation of complex forms of isotopes of ^{54}Mn, ^{55}Fe, ^{58}Co, and ^{63}Ni with organic part of soil. A size of such a complex considerably increases a capillary of the plant, which makes impossible its proliferation into stalks and other parts of a plant by the way of self-diffusion.

1. INTRODUCTION

The problem of radioisotope penetration into plants, their distribution to parts of plants, return to soil as a result of pulling-down and dying of plants and their parts has important theoretical and practical significance. A considerable amount of material has been accumulated about conduct of long-lived radioisotopes, such as ^{60}Co, ^{90}Sr, ^{137}Cs and etc. in the system soil-water-plants. This allows scientific substantiation of the methods of decreasing accumulation of these radionuclides in different grain and legume crops [1].

In scientific literature there is practically no data about the migration of ^{54}Mn, ^{58}Co, ^{55}Fe, and ^{63}Ni in such dry crops as coriander, fennel, and onion and the distribution in them from the nutrient medium, i.e. on different parts of a plant.

The purpose of this research is an investigation of the vertical migration of radionuclides ^{54}Mn, ^{55}Fe, ^{58}Co, and ^{63}Ni in the following plants: coriander, fennel, and onion, which are widely used in food among Tajikistan's population.

2. EXPERIMENT

We added solutions containing the radionuclides ^{54}Mn, ^{55}Fe, ^{58}Co, and ^{63}Ni into metallic tins having 10 cm diameter and 250 g soil capacity. The average concentration of solution was $C_A = 5.4 \times 10^5$ Bq/l. We then planted seeds of coriander, fennel, and onion in these tins. Thus, we had:

(1) with radioactive soil - four coriander tins, four fennel tins, and four onion tins;

(2) without radioactive soil – one coriander tin, one fennel tin, and one onion tin as control.

The test lasted two months. After ageing, the soil and plants were taken from tins; roots were carefully separated from soil and then washed. It should be mentioned that onion seeds did not grow in radioactive soil, but grew successfully in control tins.

The roots, leaves, and pedicels of coriander, fennel, and onion were separated from each other, and were placed in separate glass cones with a capacity of 50 ml each. Then 30 ml of spirit was added in each cone and was left for 48 hours. Spirit extract was tested for radioactivity using fluid dish on radiometer PSO-5.2. The results of analysis are shown in the Table 1.

TABLE 1. THE PERCENTAGE OF RADIONUCLIDES IN CORIANDER AND FENNEL PLANTS, FOLLOWING APPLICATION OF SPIRIT

Radioactive isotopes in spirit extract	Coriander			Fennel		
	Roots	Pedicels	Leaves	Roots	Pedicels	Leaves
^{54}Mn	0.25%	–	–	0.35%	–	–
^{55}Fe	0.28%	–	–	0.27%	–	–
^{58}Co	0.30%	–	–	0.28%	–	–
^{63}Ni	0.29%	–	–	0.30%	–	–

The results of the experiment show that radioactive isotopes of ^{54}Mn, ^{55}Fe, ^{58}Co, and ^{63}Ni are not accumulated in stems and other parts (except roots) of experimental plants of coriander and fennel. It is connected with formation of complex form of radioactive isotopes of ^{54}Mn, ^{55}Fe, ^{58}Co, and ^{63}Ni with the organic part of soil. The size of this complex form is larger than that of the plant's capillary and this fact results in non-distribution of these isotopes among different parts of a plant by the way of self-diffusion.

Using the reagent Na-EDTA and an ammonia solution to separate fractions, we determined by the method of desorption the presence of ^{54}Mn, ^{55}Fe, ^{58}Co, and ^{63}Ni radionuclides, which were added to soil. The results are presented in Table 2.

We carried out an experiment to investigate the sorption of ^{54}Mn, ^{55}Fe, ^{58}Co, and ^{63}Ni in soil depending on the pH of the initial radioactive solution at constant concentration of the sorbing component.

As may be seen from Figure 1 (curve 1-4), sorption of ^{54}Mn, ^{55}Fe, ^{58}Co, and ^{63}Ni at $C_A = 5.4 \times 10^5$ Bq/l, at low values (of pH=2) these radioactive elements are in the form of $Me(H_2O)$. With increase of pH the hydrolytic forms $Me(OH)^+$, $Me(OH)$, $Me_2(OH)_2^{2+}$ and $Me(OH)^o$, (Me = Mn, Fe, Co, and Ni) are present. Obviously, making hydrolytic forms promote sorption of these radioisotopes by soil.

The desorption of radionuclides ^{54}Mn, ^{55}Fe, ethylene diamine tetraacetate and Na (Na-EDTA) was investigated using a Na-EDTA complex with a concentration of 0.1-1 mole/liter. We took 5 g of soil from a depth of 5 cm, where radioactive isotopes ^{54}Mn, ^{55}Fe, ^{58}Co, and ^{63}Ni were present at different concentrations of Na-EDTA (0.1-1 mole/liter). In the case of ^{58}Co and ^{63}Ni a solution of NH_4OH as desorbing reagent.

As demonstrated in Figure 2 (curve 1-4), the results of the experiment show that, with increase of Na-EDTA concentration, there is an increase in desorption of radioactive Mn and Fe, which practically reaches 100% at $C_{Na\text{-}EDTA}$ = 1 mole/liter. For ^{58}Co and ^{63}Ni solution of four moles of NH_4OH at values of pH=10.5 practically bind (93%) $Co(NH_3)_4$ and $Ni(NH_3)_4$ and leach from soil to solution.

Thus, Na-EDTA is a good desorbent for Mn and Fe, and ammonia solution is a good desorbent for ^{58}Co and ^{63}Ni.

TABLE 2. THE PERCENTAGE OF RADIONUCLIDES IN CORIANDER AND FENNEL PLANTS FOLLOWING APPLICATION OF Na-EDTA AND AMMONIA SOLUTIONS

Reagents	Radionuclide	Coriander			Fennel		
		Roots	Pedicels	Leaves	Roots	Pedicels	Leaves
Na-EDTA	^{54}Mn	99.75%	100%	100%	99.65%	100%	100%
	^{55}Fe	99.72%	100%	100%	99.73%	100%	100%
Ammoniacal solution at pH=10.5	^{58}Co	99.70%	100%	100%	99.72%	100%	100%
	^{63}Ni	99.71%	100%	100%	99.70%	100%	100%

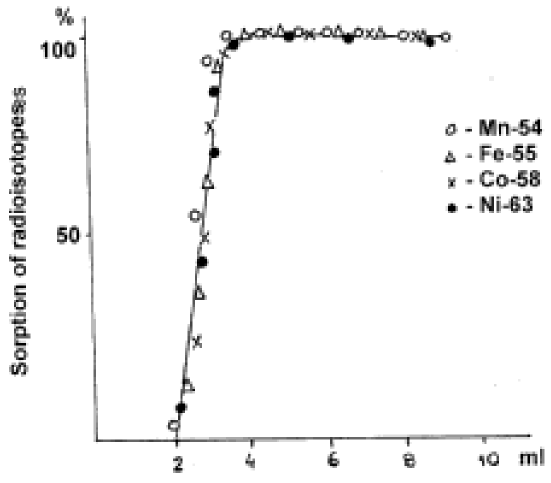

FIG. 1. Dependence of radioisotopes' sorption by soil from pH value.

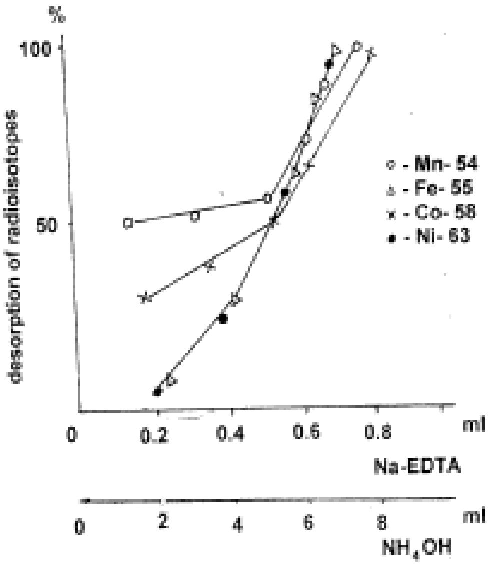

FIG. 2. Desorption of Mn-54 and Fe-55 in Na-EDTA and desorption of Co-58 and Ni-63 in NH_4OH.

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Parameters of radiation situation on the territory of the Red Forest site in the Chernobyl exclusion zone as impact factors for wild non-human species

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Abstract. Detailed description of parameters of radiation situation on the territory of the Red Forest site in the Chernobyl exclusion zone is given. Results of measurements of soil contamination by ^{90}Sr, ^{134,137}Cs, ^{154,155}Eu, ^{241}Am and ^{238,239,240}Pu are provided. Some parameters of a spatial dynamic many–nuclides source of radiation exposure formation for wild animals are calculated. Typical profiles of radionuclides distribution in soil are demonstrated.

1. INTRODUCTION

The dynamics of the formation of dose burdens for wild animals in the area of near traces of fallout from the ChNPP emergency emission is characterized by a series of specific features caused by an inhomogeneous spatial distribution of emissions in this area [4], variety of physical and chemical forms of radioactive fallout and various dynamics of their transformation in soil [5], compound radionuclide compositions represented by fission products, induced activity radionuclides and actinides [3]. In its turn, the heterogeneity of the soil cover is the reason for essentially various intensity of radionuclide distributions in the soil profile [6]. Since 1994 the comprehensive surveys of small mammals has been performed on the territory of the Red Forest. The surveys obviously demonstrated the need for detailed consideration for radiation situation parameters to obtain correct assessment results.

The paper presents assessments for some parameters of a spatial dynamic of a multi-nuclide source of radiation exposure for wild animals in the area adjacent to the ChNPP on the territory of so called Red Forest, where the above mentioned factors are maximised.

2. OBJECTS AND METHODS

The experimental polygon is located in the area of the terrace above the flood plain of the right bank of the river Pripyat with underdeveloped mesorelief. Before the accident the territory was occupied by pine forests aged 35-40 years. The soil cover is represented by mineral automorphic soils of light mechanical structure, in some areas – by hydromorphic mineral and organogenic soils. The indicated territory has been exposed to maximum radiological contamination; during first months after the accident the gamma exposure dose rate was $n \cdot 10$ R·hour^{-1}. As a consequence of this situation the greater part of pine plantations perished.

Sampling procedures were performed in 2001 at three sites in 435 points. The sampling points at each site were placed along 12 radially disposed rays 100m in length. The selection of a radial sampling procedure was stipulated by a model used to assess the density of small mammals population (Anderson et al., 1983). The soil samples in 349 points were selected using a layer-by-layer method at a depth of 10 cm to assess the territory contamination density. In 86 points the soil samples were taken at a depth of 30 cm to assess the vertical distribution of radionuclides in the soil profile and the contamination density of the territory.

The specific activity of radionuclides in soils was determined with the help of the following methods: ^{90}Sr – Beta spectroscopy method [1], $^{134,137}Cs$, $^{154,155}Eu$, ^{241}Am – Gamma spectroscopy method, alpha emitting isotopes $^{238,239,240}Pu$ - by L_x-spectroscopy method [2] as well as using radiochemical methods. The contamination density of the territory was calculated the relation of a sum of radionuclide activity margin in a 0-30 cm layer (kBq) to a core sample sectional area (m^2) or in a 0-10 cm layer, on the assumption that no less than 90% of activity margin are still located in this horizon. The parameters of radionuclide vertical transfer in the soil profile were calculated with the help of a convectively diffusion transfer model.

3. RESULTS AND THEIR DISCUSSION

The research results have demonstrated discontinuity in the spatial distribution of radioactive fallout in the area of experimental sector. Assessments of the contamination density distribution of different radionuclides in the area of experimental sector are given in Table 1.

Average values of contamination density of polygon area sectors vary within the limits of ($MBq \cdot m^{-2}$): ^{90}Sr - 43.4–82.6, ^{137}Cs - 74.3–166, ^{241}Am - 1.0–2.5, $^{238,239,240}Pu$ – approximately 1.4–2.8, at the same time, contamination density values of ^{137}Cs in some “spots” are more by a factor of 2.8. The trace direction of radioactive fallout can be observed from northeast to southwest through the first and the second sites, whereas the third site (the clearest one) is left to the northwest from the trace. The log normal distribution law can characterize contamination density distribution in space of the experimental territory.

The analysis of calculated values for isotopic ratios of $^{90}Sr/^{154}Eu$, $^{239,240}Pu/^{154}Eu$, $^{241}Am/^{154}Eu$, $^{239,240}Pu/^{137}Cs$, $^{241}Am/^{137}Cs$, $^{137}Cs/^{90}Sr$ and their comparison with values calculated for fallouts of emission fuel component in the near area of the ChNPP exclusion zone [3] has shown that different sectors of the polygon area are characterized by fuel emission factors or a superposition of fuel and condensate emission factors.

The analysis of vertical distribution through the soil profile has shown a significant impact of physical and chemical emission forms, specific features of the soil cover, radionuclide behaviour as isotopes of specific chemical elements, fossorial activity of animals etc. Both exponential forms of radionuclide distribution profiles in soil and distributions with “gaussiana” maximum were indicated, depending on the water regime of soils.

The range of transfer parameters (diffusion coefficient is D and convective transfer speed is V_k) of radionuclides in the soil profile of the experimental polygon is given in Table 2. Assessments of isotope relation values for each radionuclide and ^{154}Eu in different horizons of the soil profile and the values of radionuclide vertical transfer parameters, calculated with the help of convectively diffusive transfer model, demonstrate that ^{90}Sr leached from the fuel particles matrix is characterized by maximum migration ability.

According to 2001 status, the most stock of radionuclides is located in soil top horizons. Within the sectors where radionuclides are distributed in soil profile according to exponential mode, more than 90% of ^{90}Sr margin was located in top soil layer of 10 cm, margin of ^{137}Cs and $^{238,239,240}Pu$ was 93% and 97% correspondingly.

Figure 1 presents typical profiles of radionuclide distribution in soil. Based on the analysis of radionuclides distribution profiles in soil within different sectors of experimental polygon, as well as of $^{90}Sr/^{154}Eu$, $^{239,240}Pu/^{154}Eu$, $^{241}Am/^{154}Eu$, $^{239,240}Pu/^{137}Cs$, $^{241}Am/^{137}Cs$ and $^{137}Cs/^{90}Sr$ isotopic ratio values and taking into account the fact that ^{154}Eu can be a fuel particles marker, a conclusion can be made that differences in character and intensity of the considered radionuclides redistribution in soil profile depend on a number of reasons.

TABLE 1. GENERAL DESCRIPTION OF VALUE DISTRIBUTION FOR DENSITY OF SOIL CONTAMINATION AT 3 SITES OF THE RED FOREST, MBq·m^{-2}

		^{90}Sr	^{134}Cs	^{137}Cs	^{154}Eu	^{155}Eu	^{241}Am	^{238,239,240}Pu
Site 1	Mean	82.6	0.59	143.3	0.93	0.49	2.20	2.75
	Geometric mean	62.1	0.45	107.9	0.63	0.34	1.49	1.65
	Median	70.2	0.50	118.6	0.76	0.40	1.79	1.59
	Standard deviation	62.5	0.44	111.4	0.81	0.42	1.92	4.83
	Variation coefficient, %	75.7	74.5	77.7	87.5	87.2	87.5	175.4
Site 2	Mean	67.6	0.74	166.0	1.08	0.56	2.46	2.37
	Geometric mean	56.5	0.6	139.3	0.9	0.5	1.9	1.6
	Median	55.3	0.65	151.8	0.99	0.51	2.21	1.46
	Standard deviation	43.8	0.42	94.2	0.67	0.35	1.52	2.45
	Variation coefficient, %	64.8	56.9	56.7	62.1	61.5	61.6	103.1
Site 3	Mean	43.4	0.35	74.3	0.45	0.25	1.01	1.39
	Geometric mean	31.1	0.27	53.6	0.31	0.18	0.70	0.96
	Median	34.0	0.29	61.4	0.37	0.20	0.85	1.09
	Standard deviation	33.5	0.26	59.4	0.38	0.20	0.84	1.34
	Variation coefficient, %	77.1	76.2	80.0	83.4	79.8	83.2	96.5

TABLE 2. PARAMETERS OF VERTICAL RADIONUCLIDES TRANSPORT IN EXPERIMENTAL POLYGON SOILS PROFILE

	D, cm^2·s^{-1}	V_k, cm·s^{-1}	D, cm^2·s^{-1}	V_k, cm·s^{-1}
	Areas with normal hydro regimes of soil		Areas with intensive drained regimes of soil	
^{137}Cs	2.0E-9 – 7.7E-8	9.0E-16 – 4.1E-13	1.4E-9 – 5.8E-9	5.4E-9 – 1.4E-8
^{90}Sr	1.7E-9 – 4.2E-7	4.1E-11 – 3.2E-11	2.3E-9 – 1.8E-8	5.5E-9 – 1.7E-8
^{154}Eu	9.7E-10 – 6.5E-8	6.6E-12 – 2.8E-9	1.9E-10 – 1.0E-8	4.1E-9 – 1.0E-8
^{241}Am	1.2E-9 – 6.6E-8	3.7E-12 – 1.3E-9	1.1E-9 – 1.0E-8	3.3E-9 – 1.10E-8
^{239,240}Pu	1.3E-9 – 1.7E-8	5.7E-14 – 1.5E-12	1.3E-9 – 8.7E-9	2.7E-9 – 1.20E-8

For automorphic mineral soils with standard water regime the following sequence of radionuclides can be accepted in accordance with their migration mobility: ^{90}Sr > ^{137}Cs > ^{241}Am > ^{154}Eu ~ ^{239,240}Pu. In hydromorphic organogenic soils ^{137}Cs migration mobility is comparable to ^{90}Sr migration mobility or exceeds it. In case fuel particle destruction is of low level and radionuclides have transferred from their matrix to soil composition, similar transmission intensity is observed for all radionuclides. Clearly, in case of soil mechanical mixing, for instance, as a result of animals' activity, practically similar radionuclide distribution profiles are observed (Figure 1(d)).

Thus, heterogeneity of the spatial distribution of fallout within the territory, plurality of the radioactive fallout physical and chemical forms and different dynamics of their transport in soil, complex composition of fallout radionuclide, soil cover heteroginity, wild animals digging activities predetermine various intensity of radionuclides redistribution in soil profile.

Differences in main dose-forming radionuclide distribution in soil profile, as well as differences in organogenic and mineral soils density have significant influence on both gamma fields exposure dose rate (EDR) buildup and its dynamics within various sectors of experimental polygon. Heterogeneity of radioactive fields spatial distribution within the polygon together with other above-mentioned reasons predetermine significant differences in EDR values within various sectors of experimental polygon. In 2001, exposure dose rate values of gamma emission within the polygon area varied between 3 and 24 mR·hour^{-1}.

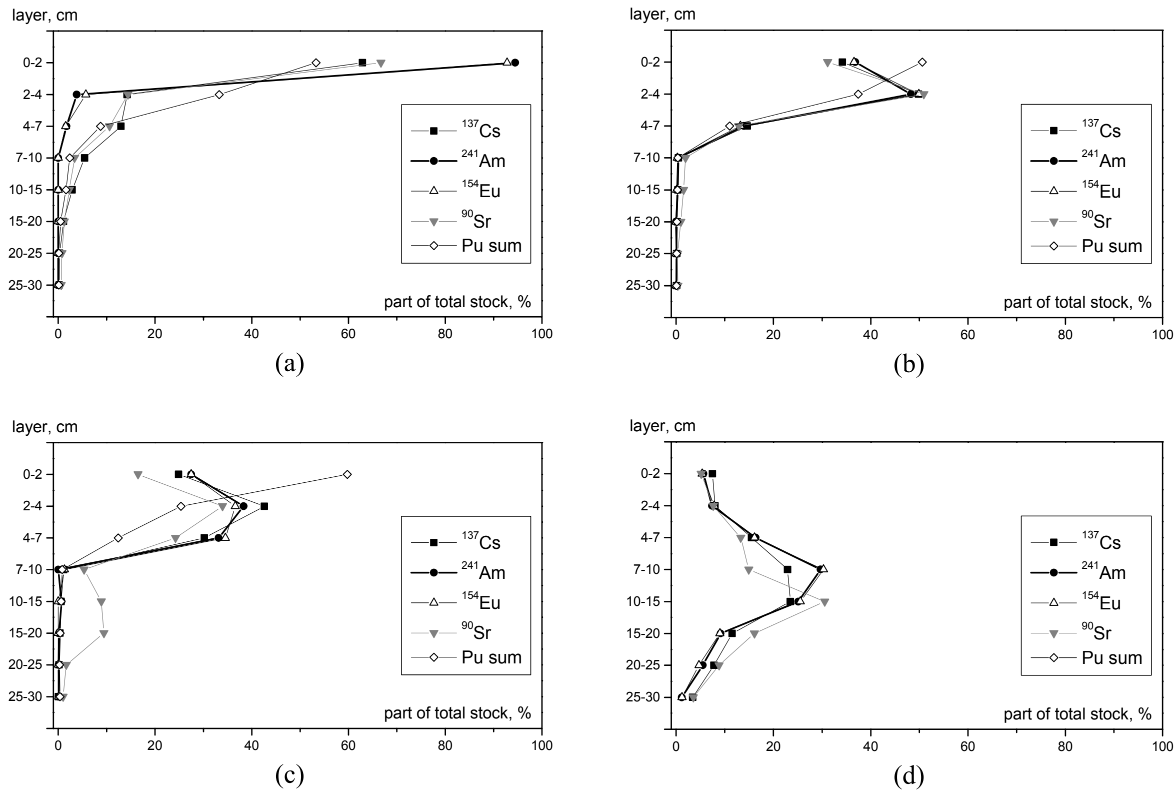

FIG. 1. Typical profiles of radionuclides distribution in soil profile of experimental polygon various sectors: (a) exponential radionuclides distribution in automorphic mineral soil; (b) radionuclides distribution in automorphic mineral soil, are mostly transported as a part of fuel particle; (c) radionuclides distribution in mineral soil with intense washing regime; (d) radionuclides distribution in soil as a result of animals' digging activities.

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The transfer of Cs-137 and Sr-90 to wild animals within the Chernobyl exclusion zone

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Abstract. More than 700 ^{137}Cs and ^{90}Sr activity concentrations for the tissues of 8 species of mammals and 13 species of birds sampled from within the Chernobyl exclusion zone between 1988 and 2000 have been collated. These data present a useful contribution to the current development of environmental impact assessment approaches with regard to ionising radiation. Summarised transfer values are presented and discussed.

1. INTRODUCTION

The removal of humans from the 30 km exclusion zone surrounding the Chernobyl nuclear power plant has led to increases in wild animal populations [1]. Many measurements of radionuclide activity concentrations in the tissues of wild animals have subsequently been made. Whilst transfer data for aquatic biota have been relatively well reported and analysed within the international literature [e.g. 2-5], those for terrestrial animals have not. Here we summarise a database of observations made by the Ukrainian authors of ^{137}Cs and ^{90}Sr transfer to mammalian and bird species.

2. DATABASE

More than 700 measurements made throughout the exclusion zone between 1988 and 2000 were collated; these include 8 species of predominantly large mammals and 13 species of birds. Data are available for a range of tissues (muscle, liver, spleen, lung, bone, foetus, castoreum gland) for both nuclides although bone predominates for ^{90}Sr. Wild boar (*Sus scrofa*) and roe deer (*Capreolus capreolus*) contribute a large proportion of the data, whilst relatively few measurements are available for each bird species. At the time of sampling, five soil samples were obtained in the vicinity of where the animal was shot and were used to provide estimates of ^{90}Sr and ^{137}Cs deposition.

Tables 1 and 2 summarise the data for bird and mammalian species respectively. For brevity, only muscle and bone data are presented for ^{137}Cs and ^{90}Sr respectively. Data are summarised as aggregated transfer values (Tag; the fresh weight (fw) activity concentration in animal tissue (Bq kg^{-1}) relative to the deposited activity (Bq m^{-2})). Over the observation period, there was no long-term temporal trend in the transfer of ^{90}Sr or ^{137}Cs to the sampled species.

3. DISCUSSION

There is currently much development of environmental impact assessment approaches with regard to ionising radiations. The availability of much of the data required to parameterise the transfer components of these frameworks is often scarce and the data described here have usefully contributed to the derivation of soil-biota transfer parameters [6, 7].

Whilst a complete analysis of all of the available data is not possible within the confines of this paper a number of observations can be made.

TABLE 1. SUMMARY OF TRANSFER DATA AVAILABLE FOR BIRD SPECIES SAMPLED WITHIN THE CHERNOBYL EXCLUSION ZONE (TAG VALUES AND SAMPLE NUMBERS REFER TO MUSCLE FOR Cs AND BONE FOR Sr ONLY)

Latin name	English name	Radionuclide	Tag (m^2 kg^{-1} fw) Mean	Range	n
Hirundo rustica	Barn swallow	Cs	5.78×10^{-4}		1
		Sr	1.78×10^{-2}		1
Sylvia nisoria	Barred warbler	Cs	2.76×10^{-3}		1
		Sr	4.82×10^{-3}		1
Larus ridibundus	Black-headed gull	Cs	6.73×10^{-5}		1
Gavia sp.	Diver	Cs	9.73×10^{-6}		1
		Sr	8.72×10^{-4}		1
Turdus pilaris	Fieldfare	Cs	3.69×10^{-4}	$(2.07\text{-}6.87)\times10^{-4}$	3
		Sr	2.30×10^{-1}	$(0.11\text{-}5.58)\times10^{-1}$	3
Accipiter gentilis	Goshawk	Cs	6.62×10^{-4}		1
Circus sp.	Harrier	Cs	7.73×10^{-4}		1
		Sr	2.07×10^{-2}		1
Anas Platyrhynchos	Mallard	Cs	6.78×10^{-4}	$(0.21\text{-}1.15)\times10^{-3}$	2
		Sr	2.53×10^{-2}		1
Alectoris spp.	Partridge	Cs	5.83×10^{-2}	$(0.90\text{-}116)\times10^{-3}$	2
Lanius collurio	Red-backed shrike	Cs	3.50×10^{-3}	$(3.26\text{-}3.71)\times10^{-3}$	2
		Sr	4.60×10^{-2}	$(1.44\text{-}7.72)\times10^{-2}$	2
Sturnus vulgaris	Starling	Cs	1.24×10^{-3}		1
		Sr	1.00×10^{-1}		1
Anas crecca	Teal	Cs	4.92×10^{-4}		1
		Sr	1.36×10^{-2}		1
Haliaeetus albicilla	White tailed eagle	Cs	$5.92\text{x}10^{-5}$		1
		Sr	4.38×10^{-4}		1

TABLE 2. SUMMARY OF TRANSFER DATA AVAILABLE FOR MAMMALIAN SPECIES SAMPLED WITHIN THE CHERNOBYL EXCLUSION ZONE (TAG VALUES AND SAMPLE NUMBERS REFER TO MUSCLE FOR Cs AND BONE FOR Sr ONLY)

Latin name	English name	Radionuclide	Tag (m^2 kg^{-1} fw) Mean	Range	n
Castor fiber	Beaver	Cs	7.53×10^{-3}		1
Vulpes vulpes	Fox	Cs	8.30×10^{-3}	$(0.13\text{-}2.16)\times10^{-2}$	5
Lepus spp.	Hare	Cs	1.29×10^{-3}	$(1.23\text{-}1.36)\times10^{-3}$	2
		Sr	2.91×10^{-3}		1
Alces alces	Moose	Cs	4.46×10^{-3}	$(0.41\text{-}10.8)\times10^{-3}$	3
Cervus elaphus	Red deer	Cs	5.12×10^{-3}		1
Capreolus capreolus	Roe deer	Cs	3.44×10^{-2}	$(0.03\text{-}97.5)\times10^{-2}$	66
		Sr	2.05×10^{-1}	$(1.13\text{-}7.68)\times10^{-1}$	52
Sus scrofa	Wild boar	Cs	1.34×10^{-2}	$(0.03\text{-}10.3)\times10^{-1}$	75
		Sr	1.56×10^{-1}	$(0.19\text{-}66.9)\times10^{-2}$	59
Canis lupus	Wolf	Cs	5.27×10^{-2}	$(0.19\text{-}1.65)\times10^{-1}$	8
		Sr	8.98×10^{-2}	$(0.15\text{-}2.31)\times10^{-1}$	8

The data for birds presented in Table 1 are limited, few individuals being sampled within each species. However, radionuclide transfer data are only available for a limited number of bird species within the literature and this additional data is consequently of value. Assessments of the impact of ionising radiation conducted in EC member states may need to especially consider birds as a consequence of the large number of species covered within the Birds Directive [8].

Whilst no long-term temporal trends in the transfer of ^{90}Sr or ^{137}Cs were observed, seasonal variations were observed for ^{137}Cs transfer to both wild boar and roe deer; no seasonal trends were observed for ^{90}Sr. For wild boar, an order of magnitude reduction in transfer was observed from February/March to July/August compared to a four-fold increase in the transfer to roe deer from spring to autumn. For roe deer this is in agreement with seasonal trends observed in other countries as a consequence of changes in dietary composition [9].

Some results for ^{137}Cs in wild boar foetuses are available: the activity concentration in foetuses were 21±3% (mean±SE) of that in the muscle of the mother.

For those muscle samples in which ^{90}Sr activity concentrations were determined, these were typically in the range 1-10% of those measured in the bone of the same animals.

Both wild boar and roe deer data demonstrate no trend in ^{90}Sr transfer to bone with animal age (as may be expected given the long biological half-life of ^{90}Sr [6]).

Caesium-137 Tag values for wolves are higher than for their likely prey species in agreement with other observations of a concentration of radiocaesium from prey to carnivorous species [10]; Tag values for ^{90}Sr to bone for wolves are similar to those observed for prey species.

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3.3. DOSE ASSESSMENT

Estimation of radiation doses from ^{137}Cs to perch in a Finnish lake

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Abstract. The biota dose calculation code NHDC (Non-Human Dose Calculation) was developed at STUK - Radiation and Nuclear Safety Authority. Perch (*Perca fluviatilis*) from southern parts of Lake Päijänne, contaminated by ^{137}Cs from the Chernobyl deposition, was the object of this study. Dose rates from ^{137}Cs to perch were estimated and annual doses of a hypothetic perch were calculated with NHDC using the measured ^{137}Cs activity concentrations in perch in 1986-2001.

1. INTRODUCTION

A dose calculation code (NHDC) for estimation of radiation to non-human biota was developed. The code was applied to estimate the annual dose of a perch (*Perca fluviatilis*) from southern part of Lake Päijänne. Lake Päijänne is a large (1000 km^2) oligotrophic lake located in the middle part of Finland. The deposition of ^{137}Cs in the southernmost basin (53 km^2) of the lake was about 64 000 Bq/m^2 in 1986. Perches from this basin were analysed annually for ^{137}Cs and the results [1, 2] were used for this study. The hypothetic perch was born in the year 1985 and lived 16 years to 2001 with a hypothetic annual grow.

2. DESCRIPTION OF MODELS

2.1. Description of NHDC

The NHDC (Non-Human Dose Calculation) is a dose calculation code which simulates physical interactions using the Monte Carlo method and point energy distributions. The code solves both internal and external doses with mono-energetic photons or electrons and with a few simple geometries. The shape of the target can be a sphere, a cylinder or an ellipsoid. The element contents and densities of the target and environment can be selected. The target can have a shell which can also be a source. The environment can be divided into multiple planes.

2.1.1. Photon simulation

First, if a particle is too far from an object or its energy is too low, the Russian roulette should be used. In the Russian roulette a particle is killed by 75% probability and if it stays alive its effects are multiplied by four. Thereafter, a photon route is divided by object cuts. In the first part of the route all cross-sections are defined and also the possible photoelectric event is determined with a random number distribution. If the photoelectric event occurs an electron with the same energy and direction is created. The energy is divided between a photon and an electron by the Klein-Nishina distribution [3] in the Compton event. Directions and energies of a photon and an electron complies with the physical laws. When the pair production occurs, a positron annihilates immediately and two 0.511 MeV electrons moving opposite directions are created and the remaining energy is absorbed. The whole photon route is repeated and the absorbed dose to the target is recorded from all events that occurred inside the object. Concerning the whole dose, numerous photons are simulated.

2.1.2. Electron simulation

First, if the energy of an electron is too low, it is absorbed. Second, the stopping power is defined [4], and if an electron is too far from the object, it is absorbed. In the Møller scattering [5], a secondary electron is created and the energy is divided to these electrons. After this, the secondary electron is simulated and only then the first electron. The direction of the electron with a multiple scattering within simulation step is defined by the Moliér theory [6, 7]. In each step, the absorbed energy is defined by the stopping power. The absorbed energy is a sum of all energies absorbed by the stopping power in each step and each created electron. To estimate the whole dose, numerous simulations must be made.

2.2. Model of perch used in the simulations

In the simulations the perch was modelled as an ellipsoid. The perch had the same shape during all its lifetime. The proportional diameters of ellipsoids used were 1:0,24:0,184. These size ratios were measured from one perch sample. To determine the annual dose, the activity concentration and the size of the perch were kept the same through the year. The activity concentrations of ^{137}Cs were measured from the perches from the southern part of Lake Päijänne in 1986-2001. In the simulations all the activity was assumed to be homogenously distributed in the perch [8].

3. RESULTS

Both the internal and external doses to a perch from ^{137}Cs with a different size were simulated. The electron energy used was the average energy of the beta spectrum of ^{137}Cs, 188.4 keV, and the photon energy was 662 keV with 85% intensity.

Internal and external dose rates per unit pGy/h per Bq/kg (Bq/kg in perch or in water) are presented in Table 1. The internal dose rates have been divided into dose rates caused by betas and photons. Most of the internal dose, 74 - 95%, is caused by the beta radiation of ^{137}Cs (Table 1).

Both internal annual and lifetime doses of hypothetic perches, born in 1985 and grown at hypothetic annual rate, are presented in Table 2 and in Figure 1. The external annual dose is very small because the activity concentration of ^{137}Cs of the water in Lake Päijänne is low, 0.35-2.6 Bq/kg, which causes 1-7 µGy of the annual dose. That is a small proportion compared to that of internal dose to a perch (Table 2). The highest annual dose with a highest measured activity concentration (5300 Bq/kg, in 1988) was 6.2 mGy/a per 250g-sized perch.

4. DISCUSSION

The internal radiation doses to a perch from ^{137}Cs in the southern part of Lake Päijänne has been rather high, up to few mGy/a. The external doses from ^{137}Cs are small compared to the internal, only few µGy/a. The internal annual doses of a perch have first risen after the Chernobyl deposition, gained its maximum at about two years after the accident, and then reduced again (Figure 1).

TABLE 1. THE INTERNAL AND EXTERNAL DOSE RATES (pGy/h per Bq/kg) OF A PERCH FROM ^{137}Cs

Weight (g)	Size of ellipsoid (cm)	$\dot{E}(\gamma)_{int.}$ (pGy)	$\dot{E}(\gamma)_{ext.}$ (pGy)	$\dot{E}(\beta)_{int.}$ (pGy)
5	5.93 : 1.48 : 1.09	6.425	301.65	107.05
10	7.47 : 1,87 : 1.37	8.210	300.39	107.14
15	8.55 : 2.14 : 1.57	9.346	303.24	107.40
20	9.41 : 2.35 : 1.73	10.579	304.97	107.77
25	10.14 : 2.53 : 1.86	11.163	299.29	107.68
30	10.77 : 2.69 : 1.98	11.909	303.61	107.74
40	11.86 : 2.96 : 2.18	13.532	306.68	107.91
50	12.77 : 3.19 : 2.35	14.310	300.81	107.91
75	14.62 : 3.66 : 2.69	16.647	296.66	108.00
100	16.09 : 4.02 : 2.96	18.204	296.63	107.83
125	17.33 : 4.34 : 3.19	19.373	293.61	108.04
150	18.42 : 4.61 : 3.39	21.028	295.39	108.11
200	20.27 : 5.07 : 3.73	22.910	291.11	108.21
250	21.84 : 5.46 : 4.02	24.759	291.69	108.18
300	23.21 : 5.80 : 4.27	26.252	288.03	108.24
400	25.54 : 6.39 : 4.70	28.718	288.74	108.25
500	27.52 : 6.88 : 5.06	30.535	287.31	108.35
750	31.50 : 7.87 : 5.80	34.981	290.26	108.38
1000	34.67 : 8.67 : 6.39	38.291	278.75	108.43

TABLE 2. THE INTERNAL LIFETIME AND ANNUAL DOSES OF A HYPOTHETIC PERCH FROM ^{137}Cs IN THE SOUTHERN PART OF LAKE PÄIJÄNNE IN 1986-2001

Year (g)	Size Bq/kg	^{137}Cs mGy/a	Annual dose mGy	Cumulative dose
1986	5	1950	1.937	1.937
1987	10	2801	2.827	4.764
1988	25	2739	2.859	7.623
1989	40	2135	2.273	9.896
1990	50	1460	1.565	11.461
1991	75	1001	1.092	12.553
1992	100	776	0.859	13.412
1993	125	679	0.758	14.170
1994	150	514	0.580	14.750
1995	200	735	0.843	15.593
1996	250	533	0.620	16.213
1997	300	478	0.562	16.775
1998	400	346	0.415	17.190
1999	500	276	0.335	17.525
2000	750	265	0.331	17.856
2001	1000	242	0.310	18.166

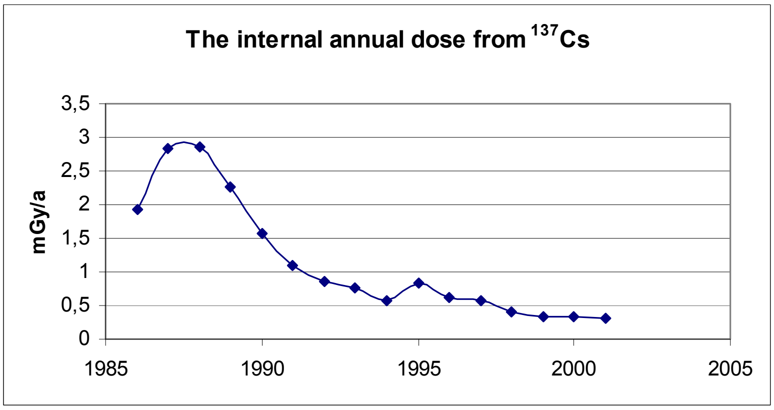

FIG. 1. *The internal annual doses to a hypothetic perch in Lake Päijänne due to* ^{137}Cs.

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Concentration factor method:
A tool for the evaluation of radiation effect on the environment

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Abstract. The increased risk to other species may be resulted from higher doses due to elevated concentration of radionuclides in environmental media or biota and/or different sensitivity to ionizing radiation. The methodology of the dose estimation for human beings is adequate basically for other species, too. The activity concentration of radionuclides can be estimated by the concentration factor method commonly used for terrestrial and aquatic food-chain models. The radiosensitivity of different species is characterized by the lethal doses, however there is less data on the risk of stochastic effects on non-human species.

1. INTRODUCTION

It is obvious that radioactive material in the environment may cause higher doses to living organisms other than human. The relative risk of species is determined by two main factors:

— dose to the individuals of the given species;

— the sensitivity to ionizing radiation.

It should be noted, that the concept of risk has also different meaning for human and non-human species. The dose limitation system focuses on individuals of human beings, as the individuals have also great value and importance. For other species the population itself is recognized as the risked target.

A possible methodology of dose estimation to biota is based on two main steps:

— estimation of steady state concentrations in the different environmental media;

— estimation of dose to biota.

2. THE ESTIMATION OF RADIONUCLIDE CONCENTRATION IN ENVIRONMENTAL MEDIA

2.1. The concentration factor (CF) method

The concentration factor (CF) method is widely used for the characterization of environmental migration of radionuclides. The models applied for the calculation are based on the methodology and parameters of IAEA publications [1,3].

The concentration of radionuclides was estimated in environmental media in direct connection with biota and in biota itself. We have focused on the pathways of aquatic and terrestrial food-chain models leading to human exposure only, as the models and parameters used are limited for them. Consequently, the following media and species were considered:

(a) Aquatic food-chain:

— water;
— sediment;
— fish;
— irrigated soil;
— plants growing on irrigated fields;
— animals fed by plants growing on irrigated fields;
— human.

(b) Terrestrial food-chain:

— plants;
— animals;
— eggs;
— human.

As the current radiation protection system is based on the dose limitation of human the radionuclide concentrations in environmental media were compared to the concentrations resulted in human. The relative concentrations in the above elements are summarized in Tables 1 and 2. For demonstration the following radionuclides were selected: ^{90}Sr, ^{131}I and ^{137}Cs.

The highest estimated concentrations were found for radionuclides ^{131}I and ^{137}Cs in fish and for ^{90}Sr and ^{131}I in sediment. These values indicate the possibility of doses to some species much higher than for human. In several elements the concentration of some radionuclides may be lower by a factor of 5–20 comparing to the human values.

For the terrestrial pathways the radionuclide concentrations relative to the human values are lower usually. Significantly higher values were only estimated for ^{131}I (except for animals). It is favourable that in animals the relative concentrations are lower than in human.

2.2. The estimation of absorbed dose

There is no sufficient data to confirm the use of tissue weighting factors, equivalent or effective dose concept for non-human species. Therefore the absorbed dose estimation is the only adequate tool to compare the risk of human and non-human beings.

The basic equation of the absorbed dose-rate in target organ T [4]:

$$\dot{D}_T(t) = k \cdot A(t) \cdot \sum_i y_i \cdot E_i \cdot SAF_i \qquad (1)$$

where:

k is the number of joules in 1 MeV (J/MeV);
A(t) is the activity of the radionuclide in source organ S (Bq);
y_i is the yield of energy line i;
E_i is the energy of line i (MeV); and
SAF_i is the specific absorbed fraction (1/kg).

The SAF values can be considered as 1 for beta and alphas. For gamma-radiation the SAF values are tabulated for different sources and target organs and gamma-energies [5]. The SAF values depend also on mass of target (Figure 1).

TABLE 1. RELATIVE RADIONUCLIDE CONCENTRATIONS OF THE ELEMENTS OF AQUATIC PATHWAYS

Radionuclide	Water	Fish	Sediment	Soil	Plants	Animal
Sr-90	0.19	12	190	0.88	0.90	0.06
I-131	2.3	900	230	0.54	8.8	6.5
Cs-137	0.04	75	40	0.17	0.14	0.05

TABLE 2. RELATIVE RADIONUCLIDE CONCENTRATIONS OF THE ELEMENTS OF TERRESTRIAL PATHWAYS

Radionuclide	Soil	Plant	Animal	Egg
Sr-90	0.56	6.0	0.41	0.09
I-131	6.8	480	0.48	18
Cs-137	0.45	1.3	0.62	0.04

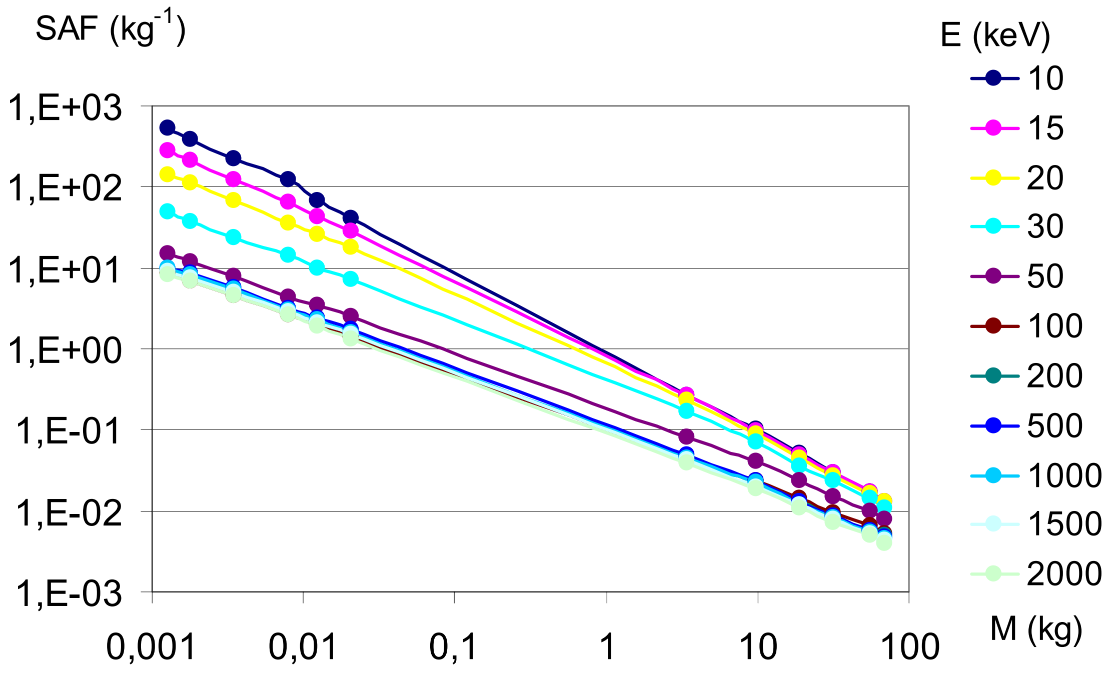

FIG. 1. Dependence of SAF values on mass of target organs and photon energy [5].

The relative concentrations in Tables 1 and 2 and SAF values together result the absorbed dose relative to human body. If we consider ^{137}Cs in fish of 1 kg, the absorbed dose will be approx. 85 times higher than the dose in human of 70 kg. This value differs from the result of Ref. [6], because of two main reasons: the bioaccumulation factor used by us is 5 times higher and the fish consumption rate is 50 times lower than in Ref. [6]. It should be noted that the above estimation refers to the absorbed dose of fish from internal exposure only and Table VI of Ref. [6] indicates a dominant contribution of the external exposure from the contaminated sediment. Using the same dose rate coefficient for the exposure from sediment the total dose will be about 100 times higher than human.

The radiosensitivity of living organisms varies in a very wide range. Considering the lethal effects mammals are the most sensitive, followed by birds, fishes, reptilia and insects. The range of radiosensitivity of plants is similar to the animals. The less sensitive living organisms to acute radiation are mosses, lichens, algae and microorganisms, i.e. viruses and bacteria. However, it should be noted that embryos and young species are more radiosensitive than aged ones, usually. Alevins can be mentioned as a typical example for this effect.

The UNSCEAR states that even for the most sensitive animal species, mammals, “there is little indication that dose rates of 400 μGy h^{-1} to the most exposed individuals would seriously affect mortality in the population” and “for dose rates up to an order of magnitude less, the same statement could be made with respect to reproductive effects” [7].

If we accept the above statement, the annual absorbed dose up to a few hundreds mGy will be no harmful to fishes. The estimation above yielded somewhat lower dose, but the “safety margin” is not high. Consequently, we have to pay particular attention to the radiation protection of living organisms other than human beings in the future.

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Modelling approach for environmental impact assessment from radioactive contamination of marine environment

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Abstract. One approach for modelling the consequences for the marine environment from radioactive contamination is promoted. The method is based on the modified approach for box modelling, which includes dispersion of radionuclides during time (non-instantaneous mixing in oceanic space). The model can provide information concerning the dispersion of radionuclides in water, sediment and biota phases of the marine environment. The assessment of internal and external doses to the marine organisms is based on assumptions about uniform distributions of radionuclides in organisms as well as surrounding water and sediment boxes.

1. INTRODUCTION

Modelling approach for environmental impact assessment can be considered as an essential component of system of radiological protection with special regards to the nonhuman components [1].

The dose assessment modelling methodology for flora and fauna has to cover whole processes such as dispersion of radionuclides in oceanic space, transfer of radioactivity between sea water and sediments, uptake of radionuclides by biota and, finally, dose calculations.

The modelling approach for environmental impact assessment, which will be used in this paper, is based on the modified approach for box modelling [2]. Modelling includes dispersion of radionuclides during time (non-instantaneous mixing in oceanic space). This approach was created in order to provide a better and more realistic/physical description of the radionuclide dispersion comparing to traditional box modelling.

2. MODEL DESCRIPTION

The equations of transfer of radionuclides between the boxes are of the form:

$$\frac{dA_i}{dt} = \sum_{j=1}^{n} k_{ji}A_j - \sum_{j=1}^{n} k_{ij}A_i\gamma(t \geq T_j) - k_iA_i + Q_i,\ t \geq T_i$$

$$A_i = 0, \quad t < T_i$$

where $k_{ii}=0$ for all i, A_i and A_j are activities at time t in boxes i and j; k_{ij} and k_{ji} are rates of transfer between boxes i and j; k_i is an effective rate of transfer of activity from box i taking into account loss of material from the compartment without transfer to another, for example radioactive decay; Q_i is a source of input into box i ; n is the number of boxes in the system; T_i is the time of availability for box i (the first times when box i is open for dispersion of radionuclides) and γ is an unit function:

$$\gamma(t \geq T_i) = \begin{cases} 1,\ t \geq T_i \\ 0,\ t < T_i \end{cases}.$$

The times of availability T_i

$$T_i = \min_{\mu_m(v_0,v_i)\in M_i} \sum_{j,k} w_{jk}$$

are calculated as a minimized sum of the weights for all paths μ_m (v_0,...,v_i) from the initial box (v_0) with discharge of radionuclides to the box i on the oriented graph $G=(V, E)$ with a set V of nodes v_j correspondent to boxes and a set E of arcs e_{jk} correspondent to the transfer possibility between the boxes j and k. Every arc e_{jk} has a weight w_{jk} which is defined as the time required before the transfer of radionuclides from box j to box k can begin (without any way through other boxes). Weight, w_{jk}, is considered as a discrete function F of the water fluxes f_{jk}, f_{kj} between boxes j and k, geographical information g_{jk} and expert evaluation X_{jk}. M_i is a set of feasible paths from the initial box (v_0) to the box i (v_i).

The contamination of marine organisms is calculated from the radionuclide concentrations in filtered sea water in the different water regions using concentration factors.

The model can provide information concerning the dispersion of radionuclides in water, sediment and biota phases of marine environment using site-specific data. Calculation of doses to biota in the marine environment is considered for the selected set of reference organisms [1] and based on the assumption that radionuclides are distributed uniformly in organisms and the surrounding environment [3].

Internal absorbed dose calculations to biota are based on an equilibrium absorbed dose constant model that accounts for the total energy emitted by individual components of electromagnetic and/or particulate radiations after the decay of a specific radionuclide [4]. Assuming that a radionuclide is uniformly distributed throughout the body of an organism, the absorbed dose rate to the biota for the internal radiation can be given by :

$$\dot{D}(t) = f \sum_j A_j(t) \sum_i k_i E_i^{(j)} y_i^{(j)} \phi_i^{(j)} ,$$

where $\dot{D}(t)$ is a dose rate (Gy y^{-1}); A_j is activity concentration for radionuclide j; $E_i^{(j)}$ is energy emitted by component i of α, β, γ radiation for radionuclide j; $y_i^{(j)}$ is a fractional abundance or yield of radiation particles/photons of energy $E_i^{(j)}$; $\phi_i^{(j)}$ is an absorbed fraction for energy $E_i^{(j)}$; k_j is a biological factor (relative biological effectiveness) for component i of (α, β, γ) radiation; $f = 5.04 \cdot 10^{-6}$ is a factor to account for conversions of MeV to Joules (1 MeV = 1.602×10^{-13} J) and seconds to years (1 year = 3.145×10^{7} seconds). All radionuclide data have been derived from ICRP Publication 38 [5].

It is necessary to note that evaluation of the dose conversion factors for internal and external radiation for wide set of reference organisms is under development in the FASSET (Framework for assessment of environmental impact) project of EC 5th Framework Programme [6] and can be easily adopted by the present methodology.

3. MODEL APPLICATIONS

Applications of the present box model in the context of an environmental impact assessment are illustrated in Figures 1–3. All simulations have been made for a 1 TBq discharges of radionuclides into Obskaya Guba (the Ob Estuary of the Kara Sea). Dynamic concentrations of ^{137}Cs and ^{239}Pu are shown for cod, crab (muscles) and Greenland seal for the Obskaya Guba and the Barents Sea in Figures 1 and 2. Calculations relating to the uptake by and transfer to biota are based on generic concentrations factors derived specifically for Arctic marine reference organisms.

The Influence of innovative components of the present box modelling approach is illustrated in Figure 3. The simulation corresponds to the dispersion of 1 TBq of ^{241}Am discharged into Obskaya Guba and accounts for ice transport of radionuclides from the Kara Sea to the Greenland Sea through the Central Arctic Basin. The dynamics of ^{241}Am concentrations in lobsters living in the Greenland Sea is calculated with a generic Arctic concentration factor. Figure 3 clearly demonstrates that ice transport of radionuclides can be a significant factor for some scenarios and radionuclides. It has been shown that the influence of ice transport increases with increasing K_d values for radionuclides [7-8].

Doses to cod from ^{137}Cs are shown in Figure 3 with regards to the concentration plots in Figure 1. The absorbed fractions for characteristic energies of ^{137m}Ba were calculated using a preliminary version of modelling tools [29], which can be easily used to calculate ϕ for a wide set of geometries and energies. Preliminary intercomparison with results (for uniform distributions of photon-emitters) derived from Monte-Carlo calculations [17] provided satisfactory agreement.

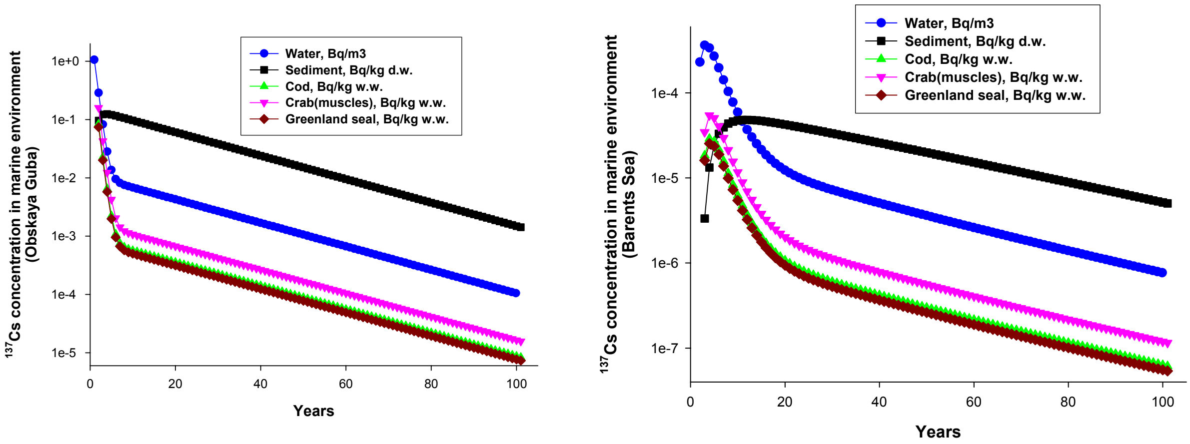

FIG. 1. ^{137}Cs dynamic concentration in marine environment.

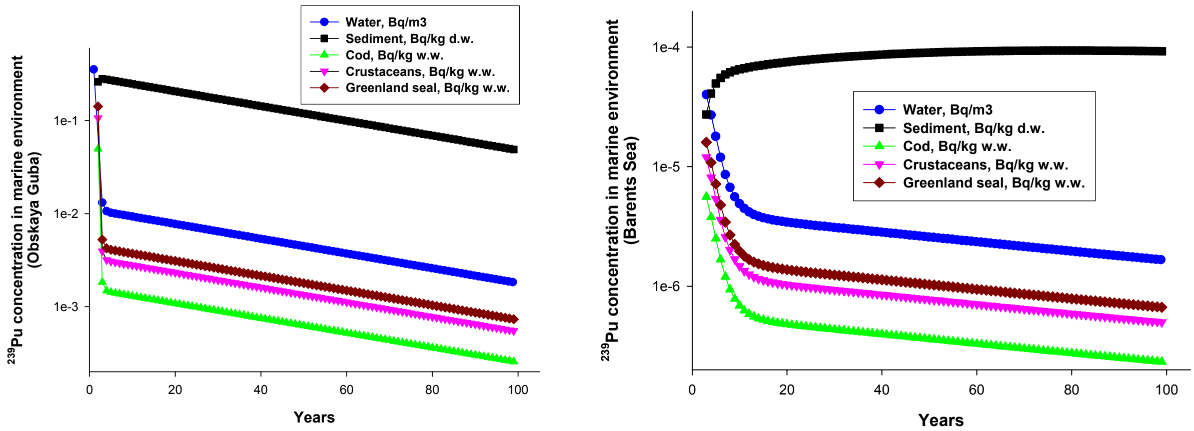

FIG. 2. ^{239}Pu dynamic concentration in marine environment.

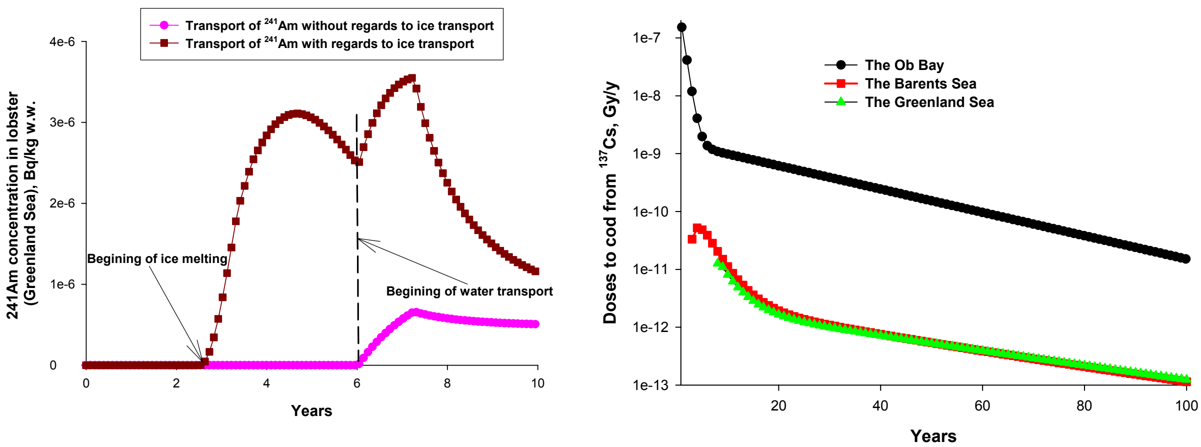

FIG. 3. Innovative elements of the model.

ACKNOWLEDGEMENTS

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Doses to selected seawater organisms from ^{137}Cs, ^{226}Ra and ^{239,240}Pu

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Abstract. The dose assessment for aquatic animals was performed, based on the methodology elaborated by U.S. Department of Energy. Baltic Sea fish (cod, sprat, herring, plaice) and crustaceans (Sanduria entomon and Mytilus edulis) were taken in to consideration and the annual doses from ^{137}Cs, ^{226}Ra and ^{239}Pu to organisms were calculated at average concentrations of these radionuclides observed in water, concentration of radionuclides determined in bottom sediments from two sub-areas (Gdansk Basin and Bornholm Basin) and average concentrations of radionuclides in animal tissues. The total maximal annual doses for seawater organisms do not exceed a one percent of recommended dose limits of 3.6 Gy y^{-1}. The default animal-water bioacumulation ratios that are provided by screening metodology can be used with caution when they are aplied to dose evaluation for biota.

Determination of radionuclides in Baltic Sea environment in the frame of MORS [1] gives opportunity to evaluate doses for aquatic organisms from the Southern Baltic Sea. The environmental data from the last few years concerning ^{137}Cs (bomb-tests-fallout and Chernobyl origin), ^{226}Ra (natural radionuclide) and ^{239,240}Pu (bomb-tests-fallout) concentrations in biota, bottom sediment and water are presented in Table 1.

Concentrations of radionuclides in sea water taken to dose calculation belong to own determinations (^{226}Ra), published data by IMGW (^{137}Cs) [4] and FMHG (^{239,240}Pu) [5].

The doses assessment of ^{137}Cs, ^{226}Ra and ^{239,240}Pu for aquatic animals was performed, based on the screening methodology elaborated by U.S. Department of Energy [2,3]. The following pathways of exposure for marine organisms were considered i.e. external doses from the concentrations of radionuclides determined in water and sediments, and internal doses from radionuclides concentrations observed on animal tissues. Calculations were performed using conservative assumptions about external dose conversion factors for simplified geometry and uniform distribution of radionuclide in animal tissues. The dose conversion factors used in the assessment are presented in Table 2. The calculated doses were compared to the relevant recommended biota dose limits. These limits have been proposed to avoiding impairment of reproductive capability. The absorbed dose to aquatic animals should not exceed 10 mGy d^{-1} (3.6 Gy y^{-1}) from exposure to radiation or radioactive material releases into the aquatic environment [3].

Baltic Sea fish (cod, sprat, herring, plaice) and crustaceans (sanduria entomon and mytilus edulis) were taken in to consideration and the annual doses from ^{137}Cs , ^{226}Ra and ^{239}Pu to organisms were calculated at average concentrations of these radionuclides observed in water, concentration of radionuclides determined in bottom sediments from two sub-areas (Gdansk Basin and Bornholm Basin) and average concentrations of radionuclides in animal tissues. Doses for ^{239}Pu are overestimated of about 20% because the concentrations of ^{239,240}Pu were used in the calculations. (The ratio of ^{240}Pu to ^{239}Pu in global fallout is about 0.18).

The results of doses evaluation are summarized in Tables 3–5 for fish, saduria entomon and mytilus edulis respectively.

The total maximal annual doses for seawater organisms do not exceed a one percent of recommended dose limits however, the dominate contribution to the total dose depends on analysed radionuclide. For ^{137}Cs a maximum contribution to the total dose gives external dose from bottom sediment (about 0.5 mGy y^{-1}). Only about 1% of the total ^{137}Cs dose is derived from internal dose however animal-water ratio obtained from measurements is much lower (in a range 30-300 L kg^{-1}) comparing with recommended by DOE Standard value (22000 L kg^{-1}). For ^{226}Ra internal doses for fish and mytilus are similar (0.2 mGy y^{-1}, 0.6 mGy y^{-1} respectively) and they are comparable with external doses from sediment (0.3 mGy y^{-1}) whereas internal dose for sanduria is about 10 times higher (7 mGy y^{-1}). A measured animal- water ratios (20 - 500 L kg^{-1}) differs remarkably from default DOE Standard value (3200 L kg^{-1}). For plutonium ^{239}Pu the maximum contribution to the total dose gives internal dose and external dose from sediment contributes less then 1%. The contribution of external dose from water is negligible for all analysed radionuclides. A measured animal-water ratios for Mytilus edulis (2600 L kg^{-1}) is much higer then default DOE Standard value (1000 L kg^{-1}) and it sugests careful veryfication parameters used in the screening metodology.

TABLE 1. THE ENVIRONMENTAL DATA FROM THE LAST YEARS CONCERNING ^{137}Cs, ^{226}Ra AND ^{239,240}Pu CONCENTRATIONS IN BIOTA, BOTTOM SEDIMENT AND WATER. RANGES OF MINIMUM AND MAXIMUM OBSERVED VALUES ARE SHOWN IN PARENTHESIS

Radionuclide	Bottom sediments layer: 0-10 cm period 1997-2001 Bq kg^{-1} dry	Baltic Sea surface water period 2000, 2001. Bq L^{-1}	Baltic Sea fish: cod, sprat, herring, plaice period 1998-2001. Bq kg^{-1} fresh mass	Baltic Sea bottom animals, period 1998-2001 Bq kg^{-1} fresh mass	
				Sanduria entomon	Mytilus edulis
^{137}Cs	1.2×10^{2} (5.8×10^{1}–2.2×10^{2})	5.8×10^{-2} (1.7×10^{-2}–7.4×10^{-2})	9.0 (6.6–1.2×10^{1})	4.0 (2.2– 6.3)	1.2 (5.8×10^{-1}–2.1)
^{226}Ra	3.6×10^{1} (2.6×10^{1}–4.9×10^{1})	3.0×10^{-3} (2.0×10^{-3}–3.7×10^{-3})	5.0×10^{-2} (2.9×10^{-2}–7.1×10^{-2})	1.6 (1.1–2.5)	1.7×10^{-1} (1.4×10^{-1}–2.1×10^{-1})
^{239,240}Pu	2.8 (8.7×10^{-1}–5.5)	3.6×10^{-6} (1.6×10^{-6}–5.6×10^{-6})		9.5×10^{-3} (5.7×10^{-3}–1.9×10^{-2})	4.0×10^{-3} (2.5×10^{-3}–5.9×10^{-3})

TABLE 2. DOSE CONVERSION FACTORS APPLIED IN DOSE CALCULATION ACCORDING TO DOE-STD-1153-2002

Radionuclide	Dose convertion factors			
	External dose		Internal dose	
	Sediment $\frac{mGy y^{-1}}{Bq kg^{-1} dry}$	Water $\frac{mGy y^{-1}}{Bq L^{-1}}$	Internal $\frac{mGy y^{-1}}{Bq kg^{-1} \text{fresh mass}}$	Animal-water ratio $\frac{L}{\text{kg fresh mass}}$
^{137}Cs+^{137m}Ba	2.0×10^{-3}	2.0×10^{-3}	4.3×10^{-3}	2.2×10^{4}
^{226}Ra +D* (RBEα=20)	6.8×10^{-3}	6.8×10^{-3}	3.0	3.2×10^{3}
^{239}Pu (RBEα=20)	1.4×10^{-3}	1.4×10^{-4}	5.3×10^{-1}	1.0×10^{3}

TABLE 3. ANNUAL DOSES FOR BALTIC SEA FISH (COD, SPRAT, HERRING, PLAICE). RANGES OF MINIMUM AND MAXIMUM CALCULATED VALUES ARE SHOWN IN PARENTHESIS 2002

Radionuclide	Sediment external dose mGy year^{-1}	Water external dose mGy year^{-1}	Inernal dose mGy year^{-1}	Organism to Water L kg^{-1} fresh mass	Total dose mGy year^{-1}
^{137}Cs	2.4×10^{-1} *(1.2×10^{-1}–4.5×10^{-1})*	1.2×10^{-4} *(3.4×10^{-5}–1.5×10^{-4})*	3.9×10^{-2} *(2.8×10^{-2}–5.2×10^{-2})*	155 *(160–390)* *22000*∗	2.8×10^{-1} *(1.4×10^{-1}–5.0×10^{-1})*
^{226}Ra	2.4×10^{-1} (1.7×10^{-1}–3.3×10^{-1})	2.0×10^{-5} (1.4×10^{-5}–2.5×10^{-5})	1.5×10^{-1} (8.7×10^{-2}–2.1×10^{-1})	17 (14–19) *3200*∗	3.9×10^{-1} (2.6×10^{-1}–5.4×10^{-1})
Sum	4.8×10^{-1} (2.9×10^{-1}–7.8×10^{-1})	1.4×10^{-4} (4.8×10^{-5}–1.7×10^{-4})	1.9×10^{-1} (1.2×10^{-1}–2.7×10^{-1})		6.7×10^{-1} (4.0×10^{-1}–1.0)

∗ Value adapted by DOE Standard.

TABLE 4. ANNUAL DOSES FOR IN BALTIC SEA BOTTOM ANIMALS (SADURIA ENTOMON). RANGES OF MINIMUM AND MAXIMUM CALCULATED VALUES ARE SHOWN IN PARENTHESIS

Radionuclide	Sediment external dose mGy year^{-1}	Water external dose mGy year^{-1}	Inernal dose mGy year^{-1}	Organism to water L kg^{-1} fresh mass	Total dose mGy year^{-1}
^{137}Cs	2.4×10^{-1} (1.2×10^{-1}–4.5×10^{-1})	1.2×10^{-4} (3.4×10^{-5}–1.5×10^{-4})	1.7×10^{-2} (9.6×10^{-3}–2.7×10^{-2})	69 (85–132) 22000∗	2.6×10^{-1} (1.3×10^{-1}–4.8×10^{-1})
^{226}Ra	2.4×10^{-1} (1.7×10^{-1}–3.3×10^{-1})	2.0×10^{-5} (1.4×10^{-5}–2.5×10^{-5})	4.9 (3.2–7.4)	540 (520–660) 3200∗	5.1 (3.4–7.7)
^{239}Pu	3.9×10^{-5} (1.2×10^{-5}–7.7×10^{-5})	5.0×10^{-11} (2.2×10^{-11}–7.8×10^{-11})	4.9 (3.2–7.4)	2600 (1600–5200) 1000∗	5.0×10^{-3} (3.0×10^{-3}–1.0×10^{-2})
Sum	4.8×10^{-1} (2.9×10^{-1}–7.8×10^{-1})	1.4×10^{-4} (4.8×10^{-5}–1.7×10^{-4})	4.9 (3.2–7.4)		5.4 (3.5–8.2)

∗ Value adapted by DOE Standard.

TABLE 5. ANNUAL DOSES FOR IN BALTIC SEA BOTTOM ANIMALS *(MYTILUS EDULIS)*. RANGES OF MINIMUM AND MAXIMUM CALCULATED VALUES ARE SHOWN IN PARENTHESIS

Radionuclide	Sediment external dose [mGy year^{-1}]	Water external dose [mGy year^{-1}]	Inernal dose [mGy year^{-1}]	Organism to water [L kg^{-1} fresh mass]	Total dose [mGy year^{-1}]
^{137}Cs	2.4×10^{-1} (1.2×10^{-1}–4.5×10^{-1})	1.2×10^{-4} (3.4×10^{-5}–1.5×10^{-4})	5.1×10^{-3} (2.5×10^{-3}–9.2×10^{-3})	30 (20÷35) *22000**	2.4×10^{-1} (1.2×10^{-1}–4.6×10^{-1})
^{226}Ra	2.4×10^{-1} (1.7×10^{-1}–3.3×10^{-1})	2.0×10^{-5} (1.4×10^{-5}–2.5×10^{-5})	5.2×10^{-1} (4.2×10^{-1}– 6.3×10^{-1})	60 (56÷70) *3200**	7.7×10^{-1} (5.9×10^{-1}–9.6×10^{-1})
^{239}Pu	3.9×10^{-5} (1.2×10^{-5}–7.7×10^{-5})	5.0×10^{-11} (2.2×10^{-11}–7.8×10^{-11})	2.1×10^{-3} (1.3×10^{-3}–3.1×10^{-3})	1110 (690÷1640) *1000**	2.1×10^{-3} (1.3×10^{-3}–3.2×10^{-3})
Sum	4.8×10^{-1} (2.9×10^{-1}–7.8×10^{-1})	1.4×10^{-4} (4.8×10^{-5}–1.7×10^{-4})	5.3×10^{-1} (4.2×10^{-1}–6.4×10^{-1})		1.0 (7.1×10^{-1}–1.4)

∗ Value adapted by DOE Standard.

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Assessment of background radiation exposure to Arctic freshwater fish

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Abstract. Internal and external background radiation dose rates to Arctic freshwater fish are estimated. Internal dose rate to Arctic freshwater fish is about 0.3 mGy/year. The following radionuclides are the main contributors to this dose: ^{40}K (91%), ^{210}Po (6%), ^{232}Th (1%) and ^{238}U (2%). The external exposure from bottom sediments varies considerably for different fish species. The exposure to bottom-dwelling fish from sediments is much higher (by a factor of 2000-5000) than that from water. The main radionuclides responsible for the natural exposure from bottom sediments are ^{40}K (27%), ^{232}Th series (40%), and ^{238}U series (33%). The total background exposure to Arctic freshwater fish, including cosmic rays, external and internal exposure from natural radionuclides, is about 0.5-0.8 mGy/year.

1. INTRODUCTION

The subject of this paper is the assessment of internal and external background radiation exposure to freshwater fish in the Arctic region. The following species of fish may be considered as the reference species in the Northern freshwater ecosystems:

— Fish-planktophage: Shallow-water cisco *(Coregonus albula)* in all freshwater ecosystems except High Arctic lakes; peled *(Coregonus peled)* in High Arctic lakes.

— Fish-benthophage: Cisco *(Coregonus lavaretus)* in the freshwater ecosystems located in the basin of Barents, White and Kara Seas, including High Arctic lakes; bream *(Abramis brama)* in the freshwater ecosystems located in North-West Russia, Karelia, Onega Lake and its basin.

— Predatory fish: Pike *(Esox lucius)* and burbot *(Lota lota)* in all freshwater ecosystems, including High Arctic lakes.

2. MATERIALS AND METHODS

Assessments of exposure to freshwater fish from natural radionuclides were made using the methods described by [1-3]. The data of the specific activity of natural radionuclides in water, bottom sediments and fish were used for estimation background radiation exposure to reference species in Northern freshwater ecosystems.

2.1. Natural radionuclides in North freshwater ecosystems

The data on the concentrations for radionuclides of natural origin in Northern freshwater ecosystems were compiled and analyzed. The database includes the data on the activity concentrations of ^{3}H, ^{40}K, ^{210}Pb, ^{210}Po, ^{238}U in water; ^{40}K, ^{226}Ra, ^{228}h, ^{232}Th, ^{238}U in bottom sediments; and ^{40}K, ^{210}Pb, ^{210}Po in freshwater fish [4-8]. The activity concentrations of ^{40}K in water were in the range 0.012-0.089 Bq/L; ^{210}Pb, 0.0001-0.0017 Bq/L; ^{210}Po, 0.0002-0.008 Bq/L; ^{238}U, 0.0005-0.005 Bq/L; and ^{3}H, 2.7-7.9 Bq/L. The activity of ^{40}K in bottom sediments were in the range 220-540 Bq/kg; ^{226}Ra 22-68 Bq/kg; ^{228}Th 23-48 Bq/kg. The activity concentrations of ^{40}K in the fish were in range 55-190 Bq/kg; ^{210}Pb 0.0037-0.0074 Bq/kg; ^{210}Po 0.1-1.1 Bq/kg. Assessment of concentration factors was used to estimate the activity concentrations of ^{232}Th and ^{238}U in fish [9-10].

2.2. Internal dose assessment

If the dimension of an organism sufficiently exceeds the path length of alpha and beta particles, the dose rates to an organism are approximately equal to the dose rates within the infinite volume of an absorbing material uniformly distributed with alpha and beta emitter (D_{α}^{∞} and D_{β}^{∞}, in Gy/day) [1-3]:

$$D_{\alpha}^{\infty} = 1.38 \cdot 10^{-8} \cdot E_{\alpha} \cdot y; \quad (1)$$

$$D_{\beta}^{\infty} = 1.38 \cdot 10^{-8} \cdot E_{\beta} \cdot y; \quad (2)$$

where E_{α}, E_{β} are the average energies of alpha and beta particles per decay of the particular radionuclide, MeV, and *y* is the activity concentration of the radionuclide in the organism, Bq/kg w.w.

The average dose rate to organism from incorporated gamma emitters is calculated by the following equation:

$$D_{\gamma}^{\text{int}} = 8.64 \cdot 10^{4} \cdot y \cdot \rho \cdot \Gamma_{\delta} \cdot g_{ave}; \quad (3)$$

$$g_{ave} = \frac{1}{V} \int_{V} g_{P} dV \; ; \; g_{P} = \int_{V} \frac{\exp(-\mu_{eff} \cdot r)}{r^{2}} dV \; ; \quad (4)$$

where D_{γ}^{int} is the internal dose rate, Gy/day; Γ_{δ} is the kerma radiation constant of radionuclide, $Gy \cdot m^{2} \cdot s^{-1} \cdot Bq^{-1}$; *y* is the activity concentration of the radionuclide in the organism, Bq/kg w.w.; ρ is the density of the biological material, kg/m^{3}; g_{ave} is the average geometric factor of organism, m; V is the body volume of organism, m^{3}; μ_{eff} is the effective attenuation factor of the biological tissue, m^{-1}. Standard Γ_{δ} values are tabulated for a point source in the air, for biological tissues and water $\Gamma_{\delta} \approx 1.09 \cdot \Gamma_{\delta}^{air}$. The value of the geometric factor g_{ave}, can be calculated analytically for simple symmetrical figures, such as sphere, plate, cylinder, truncated cone, etc. [2-3].

2.3. External dose assessment

External gamma-radiation dose rate $D_{\gamma,ext}^{W}$ to aquatic organisms from a gamma emitter of average energy E_{γ} uniformly distributed in the water is calculated as:

$$D_{\gamma}^{ext} = D_{\gamma}^{\infty} - D_{\gamma}^{\text{int}}; \quad (5)$$

where $D_{\gamma}^{\infty} = 1.38 \cdot 10^{-8} \cdot E_{\gamma} \cdot C_{wat}$, and D_{γ}^{int} can be calculated using the equations (3)-(4).

The external exposure from the bottom sediments is calculated by representing of them as a layer of finite thickness *h*, only exposure from gamma emitters is taken into account. The dose rate at the surface of bottom sediments is calculated with the following formula [3]:

$$D_{\gamma,ext}^{sed}(h) = 0.5 \cdot D_{\gamma}^{sed}(\infty) \cdot (1 - E_{2}(\mu_{eff}^{sed} \cdot h)) \cdot \tau_{sed}; \quad (6)$$

where $D_{\gamma,ext}^{sed}(h)$ is the dose rate at the surface of bottom sediments; *h* is the thickness of the layer of sediments, cm; τ_{sed} is the portion of time, which the organism spends at the bottom sediments; $D_{\gamma}^{sed}(\infty)$ is the dose rate in a sediment layer of infinite thickness with a uniformly distributed gamma-emitter; μ_{eff}^{sed} is taken for the material of sediments; $E_2(x)$ is the tabulated integral exponential function, or so called King's function:

$$E_{2}(x) = x \int_{x}^{\infty} \frac{\exp(-y)}{y^{2}} dy = \int_{0}^{1} \exp(-x/u) du \; . \quad (7)$$

3. RESULTS AND CONCLUSIONS

The results of the dose calculations are presented in Table 1. The following sources of natural radiation were taken into account: external exposure from water and bottom sediments and internal exposure from incorporated natural radionuclides in fish. The irradiation doses to fish are closely related to their ecology. Exposure from bottom sediments is an important factor in formation of doses to bottom-dwelling species. External dose rates from bottom sediments were evaluated as follows:

fish-benthophage - cisco 0.27 mGy/year; predatory fish – pike 0.12 mGy/year. This source of radiation is insignificant for fish-planktophage - shallow-water cisco.

External exposure to fish from water is about $6x10^{-8}$ Gy/year. The exposure to bottom-dwelling fish from sediments is much higher (by a factor of 2000-5000) than those from water. The main contributors responsible for the natural exposure from bottom sediments are ^{40}K (27%), ^{232}Th series (40%), and ^{238}U series (33%). Internal dose rate to Arctic freshwater fish is about 0.3 mGy/year. The main contributors to the internal dose are ^{40}K (91%), ^{210}Po(6%), ^{232}Th (1%) and ^{238}U (2%).

For practical purposes, it is important to apply the radiation weighting factors to estimate the equivalent doses to biota from a mixture of natural radionuclides. As a conservative default, it is proposed to apply a quality factor of 20 to the absorbed dose from α-particles and to quantify the biologically equivalent dose in Sv units [11]. In this case equivalent dose to Arctic freshwater fish from internal natural sources is about 0.77 mSv/year (Table 1). Contributions of natural radionuclides to equivalent dose are as follows: ^{40}K (33%), ^{210}Po(42%), ^{232}Th (9%) and ^{238}U (16%).

Total absorbed dose (internal and external) from natural radionuclides is as follows: shallow-water cisco 0.27 mGy/year; cisco 0.55 mGy/year; pike 0.4 mGy/year. Contributions of natural radionuclides to total absorbed doses are as follows: fish-planktophage - shallow-water cisco ^{40}K (90.3%), ^{210}Po (6%), ^{232}Th (1.4%), ^{238}U (2.2%) and ^{3}H 0.04%; fish-benthophage - cisco, ^{40}K (59%), ^{210}Po (3%), ^{232}Th (20%) and ^{238}U (18%); predatory fish – pike ^{40}K (72%), ^{210}Po (4%), ^{232}Th (13%) and ^{238}U (11%).

Annual doses from natural radiation sources, which are estimated to freshwater fish in Arctic (Table 1) and other regions [12], are listed in Table 2. The total background radiation exposure to Arctic freshwater fish (including cosmic rays, external dose from sediments and water, internal dose) is 0.5-0.8 mGy/year. According to estimates, freshwater fish in Arctic generally receives exposure from natural sources of the same order of magnitude as freshwater fish in other regions [12]. The external doses from sediments vary considerably for different species of freshwater fish and various aquatic ecosystems as a result of differing habits and content of natural radionuclides in local bottom sediments.

TABLE 1. ASSESSMENTS OF EXPOSURE FROM NATURAL RADIONUCLIDES TO ARCTIC FRESHWATER FISH, 10^{-6} Gy/year

Reference species	Radio-nuclide	External dose from water	External dose from sediments	Internal dose	Sum
Shallow-water cisco	H-3	0	0	0.11	0.11
	K-40	0.038	0	242	242.038
	Pb-210	0	0	0.013	0.013
	Po-210	0	0	16.2 (324)	16.2 (324)
	Th-232	0.0036	0	3.7 (74)	3.704 (74)
	U-238	0.022	0	6 (120)	6.022 (120)
	Sum	*0.064*	*0*	*268 (760)*	*268 (760)*
Cisco	H-3	0	0	0.11	0.11
	K-40	0.036	71.4	253	324.436
	Pb-210	0	0	0.013	0.013
	Po-210	0	0	16.2 (324)	16.2 (324)
	Th-232	0.0033	108.6	3.6 (72)	112.203 (180.6)
	U-238	0.021	89.3	6 (120)	95.321 (209.3)
	Sum	*0.06*	*269.3*	*278.9 (769)*	*548 (1038)*
Pike	H-3	0	0	0.11	0.11
	K-40	0.036	31.4	253	284.436
	Pb-210	0	0	0.013	0.013
	Po-210	0	0	16.2 (324)	16.2 (324)
	Th-232	0.0036	48.6	3.6 (72)	52.204 (120.6)
	U-238	0.021	39.3	6 (120)	45.321 (159.3)
	Sum	*0.061*	*119.3*	*278.9 (769)*	*398 (888)*

Note. Assessments of exposure with RBE=20 for for α- radiation are given in brackets.

TABLE 2. ESTIMATES OF ANNUAL DOSES (mGy/year) RECEIVED BY FRESHWATER FISH FROM NATURAL SOURCES OF RADIATION IN THE ARCTIC AND OTHER REGIONS

Source of radiation	Arctic	Other regions [12]
Cosmic	0.24	0.19-0.24
Water	0.00006	0.00004-0.06
Sediment	0-0.27	0-3.2
Internal	0.28	0.32-0.42
Sum of natural sources	0.5-0.8	0.5-3.8

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Environmental impact assessment approach in an agricultural ecosystem

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Abstract. The overall aim of the FASSET project is to develop a framework for the assessment of the impact on non-human biota of a radioactive contamination the environment, with a focus on European ecosystems. Here we present the approaches used to assess the radioactive contamination impact in an agricultural ecosystem.

Modelling approaches used are: (i) the interaction matrix development for a generic agricultural ecosystem. (ii) From the matrix processes screening analysis the conceptual model for this ecosystem was developed. (iii) Mathematical models developed to calculate activity concentration in vegetation and animals. In addition, reference organisms approach has been used. Representative species for each reference organism have been selected and their physical and ecological characteristics defined from a wide parametric review. Assessment results of concentration in the selected species have been obtained for 34 radionuclides.

1. INTRODUCTION

An agricultural ecosystem results from the human influence on a natural system to adjust it for theirs own needs. To develop agricultural systems, the human being causes alterations such as: changes in the composition of the ecosystem species, genetic improvement of the species, soil disturbances by adding fertilisers and other elements to improve their agricultural potential, alterations in the soil moisture with artificial irrigation, develop a greenhouse's agriculture and, intensive and extensive livestock.

The diversity of the farming and cropping systems is enormous, each presenting very distinct characteristics which clearly show their adaptation to the environment in which they are found. It is possible to divide them into six mayor groups: Pastoralism, mixed rain-fed farming, annual rain-fed cropping, irrigated agriculture, multiple cropping, covered cropping. In this paper we discuss the CIEMAT approaches to consider an agriculture ecosystem with the aim of the protection of this particular environment from the radiological assessment perspective in the context of FASSET Project [3].

2. MODELLING APPROACHES

First, we developed an interaction matrix, including transfer processes with different significance. These processes have been screened and from this analysis, features and processes are considered or not within the conceptual and mathematical models.

To begin with, we have represented the Matrix of the migration of radionuclides in an agricultural ecosystem (Figure 1). The Leading diagonal elements (LDEs) correspond to the various components of the system that have been identified as being relevant conceptual model objects in the representation of the contaminant migration within the ecosystem. The Off-diagonal elements (ODEs) are interactions between LDEs (transfer processes between components). In order to identify the transfer processes the matrix should be read clockwise.

Furthermore, from the screening analysis, the conceptual model developed for an agricultural ecosystem is shown in Figure 2. This includes four compartments representing environmental media (atmosphere, soil, water and sediment), two compartments representing concentrations in biota (Crop concentration and animal concentration) and two biota final receptors receiving doses (Crop total dose and Animal total dose).

Source term	Release of gaseous effluent	Release of liquid effluent					
	Atmosph.	Dry and wet deposition		Rainfall Deposit	Rainfall Deposit. Intercept. Ext.Irrad.	Inhalat. Ext.Irrad.	
		River Water	Sediment.	Irrigat.	Irrigat.	Ingestion	Outflow
		Resuspen.	River sediment				Burial
	Resuspen.			Agricul. soil	Resuspen. Root uptake	Ingestion Ext.Irrad.	Erosion Percola.
	Transpirat.			Weathe.	Crops	Ingestion	Harvest.
	Exhalation			Excret.		Domest. animals	Slaught.
							Sink

FIG. 1. Matrix representation of the migration of radionuclides in an agricultural ecosystem.

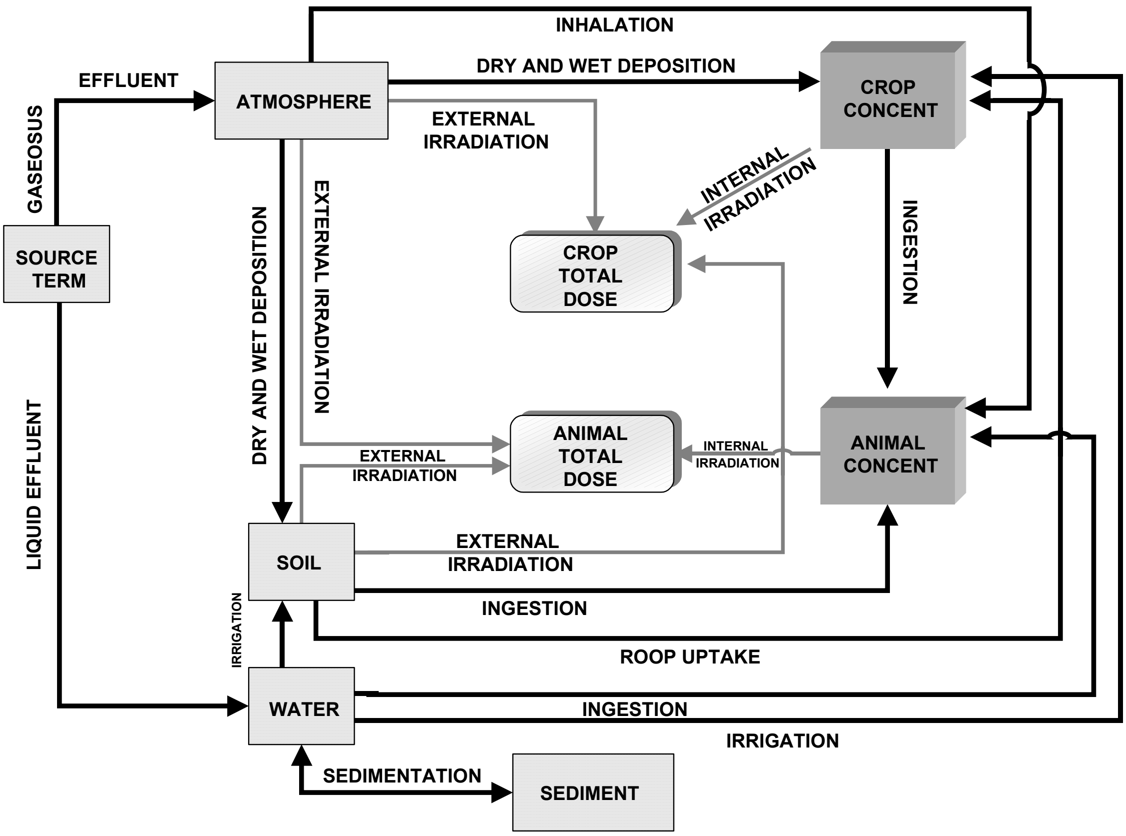

FIG. 2 Conceptual model developed for an agricultural ecosystem.

Finally, we have developed mathematical models based in reference [1], to obtain activity concentrations in the selected compartments.

Source term consist in the stablished concentrations in air, water and soil. To calculate activity concentration in vegetation we used a common mathematical expression for every vegetable type. The parameters specificity determine the radionuclide concentration of different crop types (root vegetables, fruit vegetables, leafy vegetables, cereals and fruits). The general equation is:

$$C_{i,veg,hum} = \left[\frac{\dot{d}_i \ \alpha_1 (1-\exp(-\lambda_{ief}^{v} \ t_e))}{\lambda_{ief}^{v}} + \frac{\dot{d}_i \ (1-\exp(-\lambda_{ief}^{s} \ t_b)) \ F_{1vi}}{\rho \ \lambda_{ief}^{s}} \right] \exp(-\lambda_i \ t_h)$$

Symbol	Unit	Description
$Ci,_{w}$	Bq m^{-3}	Radionuclide water concentration
d_i	Bq m^{-2} d^{-1}	Atmospheric deposition rate
F_{1vi}	---	Soil to plant transfer factor
I_w	m^3 m^{-2} d^{-1}	Irrigation rate
t_b	d	Duration of the discharge of radioactive material
t_e	d	Time period that crops are exposed during the growing season
t_h	d	Delay time between harvest and food consumption
α_1	m^2 kg^{-1}	Interception factor
λ_i	s^{-1}	Rate constant for radioactive decay
λ_{ief}^{s}	s^{-1}	Effective rate constant for reduction of the activity concentration in the root zone of soils
λ_{ief}^{v}	s^{-1}	Effective rate constant for reduction of the activity concentration from crops
ρ	kg m^{-2}	Surface density for the effective root zone in soils

And, to calculate activity concentration in animals we have used the following equation:

$$C_{i,anim} = \left[\sum (F_{i,veg,anim} \ C_{i,veg,anim} \ M_{veg,anim}) + F_{i,water,anim} \ C_{i,w} \ V_{w,anim} \right] \exp(-\lambda_i \ t_{anim})$$

Symbol	Unit	Description
$C_{i,anim}$	Bq kg^{-1}	Radionuclide concentration in animal
$C_{i,w}$	Bq m^{-3}	Radionuclide concentration in water
d_i	Bq m^{-2} d^{-1}	Atmospheric deposition rate
$F_{i,water,anim}$	d kg^{-1}	Fraction of the animal's water intake of a radionuclide that appear in meat at equilibrium
$F_{i,veg,,anim}$	d kg^{-1}	Fraction of the animal's daily intake of a radionuclide that appear in meat at equilibrium
$M_{veg,anim}$	kg d^{-1}	Amount of feed consumed by the animal per day
$V_{w,anim}$		Amount of water consumed by the animal per day
t_{anim}	d	Average time between slaughter and consumption
λ_i	s^{-1}	Rate constant for radioactive decay

2.1. Reference organisms approach

Following the International Union of Radiecology [2] and Strand and Larson [3], which have proposed a working definition of reference organism within the context of the radiological protection of the environment as *"a series of imaginary entities that provide a basis for the estimation of radiation dose rate to a range of organisms which are typical, or representative, of a contaminated environment"*. We have identified a set of candidate reference organisms, which have been suggested primarily on radioecological criteria. Based upon the knowledge of the distribution of radionuclides within the environment, a simplified compartmentalisation of the ecosystems has been used: soil, herbaceous layer and canopy for terrestrial ecosystems. Representative species for each reference organism has been selected and their physical and ecological characteristics defined (Tables 1 and 2).

TABLE 1. ECOLOGICAL INFORMATION FOR REFERENCE ORGANISMS – FAUNA

FLORA	Latin/common English name	Habitat	Distribution	Average life expectancy
Soil associated plants	*Solanum tuberosum* potatoe	Agricultural, in soil (-20 to 40 cm)	generalised	3-6 months, 1-2 harvest per year, anual crop
	Daucus carota carrot	Agricultural, in soil (-20 to 20 cm)	generalised	1-2 months, continued in the year anual crop
	Allium cepa/onion	Agricultural, in soil (-10 to 50 cm)	generalised	1-2 months, continued in the year anual crop
Herbaceus layer	*Lactuca sativa* lettuce	Agricultural/green house´s agriculture, soil surface	generalised	1-2 months, continued in the year in temperate zones, anual crop
	Lycopersicum esculentum tomato	Agricultural/green house´s agriculture, 5-130 cm	generalised	3-4 months, continued in the year in temperate zones, anual crop
	Triticum sativum wheat	Agricultural, 60 cm	dry and temperated zones	4-6 months, 1 harvest per year, anual crop
Shrubs	*Vitis vinifera* grapevine	Agricultural, 5-100 cm	temperated zones	1 harvest per year, ligneous crop
Tree	*Citrus sinensis* orange	Agricultural, canopy layer	temperated zones, no freeze	1 harvest per year
	Pyrus malus apple	Agricultural, canopy layer	no high temperatures	1 harvest per year
	Olea europaea olive	Agricultural, canopy layer	temperated zones (-10, 35 °C)	1 harvest per year

TABLE 2. ECOLOGICAL INFORMATION FOR REFERENCE ORGANISMS – FLORA

FAUNA.- Herbivorous mammals			
Latin/common English name	**Habitat dinamic**	**Average life expectancy**	**Feeding habits (kg/d, L/d)**
Box taurus cow	12 h Indoor, 12 h outdoor	14-16 y. Slaughtering age: <10 months 10-18 months, 18-36 months	meat cow: water 40, fodder 8, pasture 30 milk cow: water 60, fooder 10, pasture 30
Ovix sp Sheep	12 h Indoor, 12 h outdoor	8-10 y. Slaughtering age: <1.5 months 1.5-3 months, 3-12 months	water 6, pasture 8
Sus sp pig	Indoor, outdoor	10-12 y. Slaughtering age: <1 months 6-8 months	water 5, grain 3, root 2, milk 3

2.2. Assessment results

Two different release scenarios have been accomplished. First one, considering a continuous atmospheric deposit of 1 Bq m^{-2} and the second one, considering an homogeneous soil concentration of 1 Bq kg^{-1}. Calculations were made for 34 radionuclides selected in FASSET project [3]. The concentration in crops and farm animals has been calculated using the CROM code [4] develop by CIEMAT following the IAEA methodology [1]. Results, as an example, for ^{129}I and ^{241}Am in form of look-up tables for the reference organisms present in the agricultural ecosystem, are presented below.

Reference Organism		Bq/kg fresh per Bq/kg soil	Bq/kg pe Bq/m^2 y	Confidence	Comments
Radionuclide: ^{129}I					
Soil associated plants	Roots	2.822×10^{-2}	7.337×10^{0}	High	
Herbaceous layer	Leafy veget.	2.733×10^{-2}	7.105×10^{0}	High	
	Fruit vegt.	2.822×10^{-2}	7.337×10^{0}	High	
	Cereals	2.842×10^{-2}	7.388×10^{0}	High	
Shrubs	Shrubs	2.842×10^{-2}	7.389×10^{0}	High	
Trees	Fruit trees	2.842×10^{-2}	7.388×10^{0}	High	
Herbivorous mammals	Cow	1.365×10^{-1}	3.549×10^{1}	High	
	Sheep	1.149×10^{-2}	2.987×10^{0}	High	
	Pork	1.645×10^{-2}	4.278×10^{0}	High	
Radionuclide: ^{241}Am					
Soil associated plants	Roots	2.282×10^{-2}	5.933×10^{0}	Medium	
Herbaceous layer	Leafy veget.	2.193×10^{-2}	5.701×10^{0}	Medium	
	Fruit vegt.	2.282×10^{-2}	5.933×10^{0}	Medium	
	Cereals	2.302×10^{-2}	5.985×10^{0}	Medium	
Shrubs	Shrubs				No soil-plant T_f
Trees	Fruit trees	2.312×10^{-2}	6.012×10^{0}	Medium	
Herbivorous mammals	Cow	2.408×10^{-4}	6.262×10^{-2}	Medium	
	Sheep	5.769×10^{-5}	1.500×10^{-2}	Medium	
	Pork	2.237×10^{-2}	5.815×10^{0}	Medium	

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Ecolego – A toolbox for radioecological risk assessment

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Abstract. Ecolego is a Matlab toolbox for modelling dynamic systems and to perform risk assessments using model simulations. The toolbox supports linear and non-linear systems, modelled in continuous time, sampled time, or a hybrid of the two. The Graphical User Interface of Ecolego facilitates the transition from a conceptual model, represented with an interaction matrix, to a mathematical model implemented in Simulink. Ecolego includes routines for probabilistic simulations, correlation between parameters, sensitivity analyses, uncertainty analyses and parameter optimisation. Even though the toolbox can be used to simulate any species and dynamic system, it has been specially designed to fulfil specific needs in the field of radioecological risk assessment. Ecolego includes a library with radionuclide data and it supports automatic consideration of radionuclide decay chains. This paper provides an overview of the functioning and features of Ecolego.

1. INTRODUCTION

Radioecological risk assessment often involves simulations with models of radionuclide transfer in the environment and with dosimetric models to estimate the exposure of man and biota to ionising radiation. There exist several commercial software packages [1, 2], which can be used for model implementation. Their capabilities are, however, limited to linear models (Ordinary Differential Equations – ODE) and cannot be fully customised, since their code is hidden to the users. Besides, they have a limited number of numerical solvers, which are not universally applicable and in some cases outdated. An alternative is to use Simulink, which is a Matlab [3] toolbox to model, simulate and analyse dynamical systems. Simulink is designed to be universal, in the sense that models are built at the “atomic level”, i.e. the user defines each elementary block and signal. This property, together with its full integration with Matlab, makes it possible to implement in Simulink practically any type of model and simulation method. A drawback, though, is that substantial experience and effort is required to implement a model and to take advantage of all available capabilities. Moreover, users not familiar with Simulink may have difficulties to understand and run models implemented by others. In an effort to overcome these negative aspects, whilst keeping the advantages, we initiated a project dedicated to development of an interface, which we call Ecolego, to facilitate the implementation of radioecological models and to perform radioecological risk assessments in a Matlab/Simulink environment.

2. STRUCTURE AND FUNCTIONING

The main components of Ecolego and their interactions are shown in Figure 1. The Graphical User Interface (GUI) allows the user, who does not need to be familiar with Matlab/Simulink, to define the conceptual model, transfer functions, initial conditions, model parameters, simulation settings, etc (see epigraph 3). The Matlab code processes this information and prepares the inputs needed by the Simulink code. The Simulink code generates the Simulink model using Simulink construction commands, for instance **add_block** adds a new block to the model, **set_param** is used to specify the block properties, **add_line** is used to connect two blocks, etc.

The simulations are controlled by the Matlab code, which receives the simulation results from the Simulink code and passes them to the GUI. Note that the more blocks and connections are added, the more complicated becomes the Simulink model. However, since the construction process is fully done by the Simulink code (about 800 lines of Matlab code were need to cover all possible situations in a generic way), the user will always deal with a simple system.

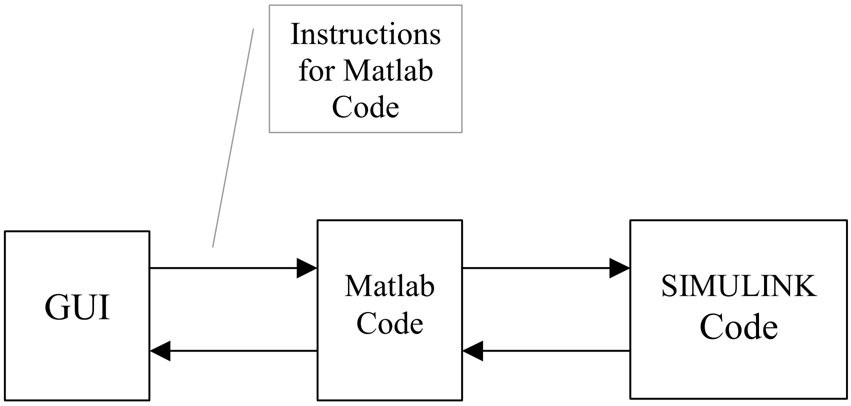

FIG. 1. Main components of Ecolego and their interactions

2.1. Advantages of the Matlab/Simulink environment

The Ecolego implementation in a Matlab environment has the following advantages:

- The whole code is open and not compiled, since Matlab is an interpret-type programming language, which allows the user to add new functionalities and introduce modifications.
- The user has full direct access to all functions, numerical solvers, toolboxes and graphical capabilities of Matlab.
- Codes written in either Matlab or other programming language can be used directly in Ecolego. This functionality increases substantially the scope of application of Ecolego well outside systems that can be represented with ODE. For instance, a geosphere transport model based on partial differential equations can be implemented as a Matlab function and fully integrated with a biosphere compartment model, a near field model, etc.
- Matlab and Simulink have been widely used by the scientific community for a long time and are constinuously upgraded. This is an advantage for the quality assurance of Ecolego and provides for a constant improvement.
- The code is platform independent and can run on Windows, Macintosh and Linux computers.

3. MODELLING WITH ECOLEGO

The GUI includes a routine for representation of the conceptual model with the help of interaction matrices [4] as illustrated in Figure 2, which shows the same model represented with a traditional flow diagram and with an interaction matrix. Each matrix element is treated as an object, which could also be a matrix, i.e. a sub-model. This facilitates using a top-down approach, which might be especially useful when modelling complex systems and to create hierarchical models.

A special routine has been implemented to handle several radionuclides in the same model. Basically, the user has only to select the radionuclides, and radionuclide decay chains, that will be considered in the model, and Ecolego will scale accordingly the signals dimensions, make corrections for decay and ingrowths, etc.

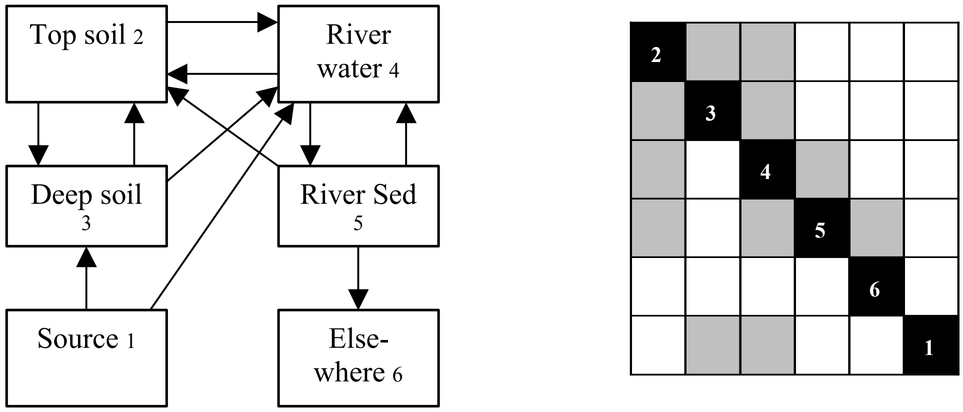

FIG.2. Conceptual representation of the model used in the comparison exercise PSACOIN Level 1b [5] with a traditional flow diagram (left) and with an interaction matrix (right). The diagonal elements in the matrix correspond to the components of the system and the off-diagonal elements to their interactions. The matrix should be read clockwise.

3.1. Towards a "LEGO" functionality

One important element in our development strategy has been to give to Ecolego what we call "LEGO" functionality. By this we mean that the user will be able to build a model in a way that resembles playing with pieces of LEGO. To achieve this we are integrating in Ecolego a database of objects. When a model is created in Ecolego, the model itself and all his elements (diagonal and off-diagonal elements in the matrix – see Figure 2) can be added to the database. The elements in the database can be afterwards combined with each other to create new models. When doing this they acquire new properties and in this way the database can be continuously enlarged. Parameter names and values, transfer functions, probability distributions, etc can also easily be added to the database. Models created with other applications can also be built-in the database.

4. PERFORMING SIMULATIONS

The Simulink model created by Ecolego can be integrated using any of the numerical solvers available in Matlab. Ecolego currently supports the basic simulation routines needed for conducting risk assessments. Probabilistic simulations can be carried out using an engine for Monte Carlo or Latin Hypercube sampling, with the option of taking into consideration binary and multiple correlations [6] between parameters. It is also possible to conduct sensitivity analyses [7] and parameter optimisation [8]. The results of the simulations can be viewed using the powerful Matlab graphic capabilities and/or directly exported to Microsoft Excel.

5. BENCHMARKING AND APPLICATIONS

Even though Ecolego uses well known and quality assured Matlab/Simulink routines, we have nevertheless conducted a series of benchmark calculations for quality control of the whole toolbox. For instance, we implemented in Ecolego the model used in the OECD/NEA intercomparison exercise PSACOIN Level 1b (shown in Figure 2) and did the specified calculations [5]. The results of the deterministic and probabilistic calculations were practically identical with the reported for all radionuclides, including those for radionuclide chains. The suit of forest models FORESTLAND [9], originally implemented in Stella [1], was set up in Ecolego and comparisons were made for several simulation cases using the same numerical solver. The results obtained with both tools were again practically identical. We have also obtained satisfactory results in comparisons against Model Maker [2] using the biosphere models described in [10] and against AMBER [11] using the near field and geosphere models described in[12].

Ecolego has been used within the EC projects FASSET [13] and BORIS [14] for implementation and development of models. Following our strategy of giving to the toolbox “LEGO” functionality, we are implementing in Ecolego “state of the art” models available in the literature for different radionuclides and ecosystems of interest. We have adopted the classification of ecosystems used in FASSET comprising forests, seminatural ecosystems, agricultural lands, wetlands, fresh water and marine ecosystems. Hitherto models for forest, seminatural and marine ecosystems have been implemented.

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The weighting of absorbed dose in environmental risk assessments

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Abstract. Practical applications of using absorbed dose for radiation protection purposes forces consideration of the biological effectiveness of different types of radiation. For human radiation protection this is taken into account by applying dimensionless radiation weighting factors. Similarly, environmental dosimetry requires weighting factors, based on experimentally obtained RBE values, relevant for biota. The expected doses and dose rates in contaminated environments are low and the dose distribution is assumed to be highly heterogeneous. The identification of relevant RBE values for biota implies the recognition of critical end-points leading to reproductive disturbances. Concurrently, a mechanistic understanding of these end-points, whether they are of stochastic (single track) or deterministic origin, has to be elucidated. In a risk assessment framework it seems necessary to postulate the criteria for selecting appropriate RBE values, and to indicate a span for weighting factors applicable to different exposure situations and ecosystems.

1. INTRODUCTION

The definition of relevant quantities, and the identification of their associated units, is of major concern for environmental risk assessments, related to ionising radiation. The basic quantity for radiation protection purposes is the absorbed dose averaged over tissue, organ or the whole body. In an environmental context, there is no alternative to the use of the absorbed dose, unit gray (Gy), for the measure of the transfer of energy from the radiation field to the biological tissue. The absorbed dose alone cannot predict the biological effect (risk) since other parameters have to be considered such as radiation quality, temporal distribution of the irradiation and level and type of biological effect.

In radiation protection, the equivalent and effective doses are formed by weighting the absorbed dose by radiation and tissue weighting factors, thus correlating the expected biological effects to the physical absorbed dose [1]. The doses are used to assess the risks of stochastic effects, and for setting dose-limits. The objective for the environmental dosimetry must be to derive risk related (biologically relevant) quantities for biota, analogous to the effective dose employed in human radiation protection practice. It is known that the exposure of the wild flora and fauna, whether from the natural background or from contaminant radionuclides, arises from radiations with a wide range of qualities and, hence, the biological effectiveness of the radiation differs. Concurrently, the dose distribution is assumed to be more heterogeneous for non-human organisms than for humans. In addition, the biological effects of concern after exposure of biota are most likely different to human radiation protection. The challenge is to provide relevant values for the radiation regimes - radiation types, dose rates and accumulated doses - in contaminated environments, and for the biological endpoints of concern. End-points that have received increased attention after exposures to genotoxic substances include: heritable changes (mutation frequency), physiological changes, teratologic effects and lethality at the individual level and, reproductive impairment, productivity reduction, and altered life-span at the population level. Unfortunately, there is a limited number of studies conducted on radiation qualities and relevant environmental end-points, which makes it difficult to draw any firm conclusions.

Nevertheless, the application of absorbed dose in environmental radiation protection requires some new weighting factors - a radiation weighting factor and probably some ecological weighting factors. We will discuss the criteria for defining weighting factors for biota dosimetry recognising the limitation of knowledge in this field.

2. INFLUENCE OF RADIATION QUALITY

The influence of radiation quality on biological systems is usually quantified in terms of the relative biological effectiveness - RBE. The RBE for a specific type of radiation, X, is defined as:

$$\text{RBE (X)} = \frac{\text{absorbed dose of reference radiation required to produce a given biological response}}{\text{absorbed dose of radiation X required to produce an equal response}}$$

The initial insult to cells, irrespective of their origin, from virtually all radiations is in the form of individual structured tracks of ionisation from charged particles [2]. The capacity of radiation to cause permanent change to cells depends on how effectively these individual radiation tracks induce severe damages in DNA. This is determined by the probability of a radiation track passing through the cell and the micro-distribution of the energy deposition. On logical grounds the observed differences in biological effectiveness of different radiation qualities must originate from differences in track structures and intrinsic radiosensitivities [3].

The linear energy transfer (LET) (keV/μm) is commonly used as a description of radiation quality, and characterises the rate of energy transfer per unit distance along a charged-particle track. X- and γ- rays are considered to be low-LET radiations, while α-particles are a high-LET radiation. This qualitative difference in energy transfer between the α-particles and the electrons of low LET radiations arises primarily from the lower velocity of an α-particle relative to an electron at a given energy (due to its greater mass); the greater charge of the α-particle is also a contributory factor. Typical LET values for different types of radiation broadly reflect their capacity to produce damage; the more effective radiations, such as α-particles, have higher LET (~100 keV per μm on average) than less damaging radiations such as X- or γ-rays (on average, about 2 and 0.3 keV per μm respectively).

The LET dependence of RBE has often been demonstrated although it is generally acknowledged that RBE is not a unique function of LET [4]. The numerical values of RBE for a given LET can vary considerably depending not only on intrinsic radiosensitivities but also on features of radiation tracks not adequately described by the LET concept. The RBE tend to be greater for mutation than for cell killing, and radioresistant cells generally show higher RBE than radiosensitive cells [5]. There are, however, many exceptions from these generalities, and this makes it difficult to predict an RBE value for a given situation with any certainty. The reference radiation, X- and γ-rays, are commonly assumed to be interchangeable with each other although X-rays have slightly higher biological effectiveness than γ-rays. For low doses applied at low dose rates it might be important to consider whether X- or γ-rays have been taken as the standard for the derivation of RBE values.

The random distribution of electrons generated by low LET radiation has ranges great enough to traverse many tens of cells. One electron track crossing a cell nucleus (diameter 8 μm) results to an energy deposition which generates 60-80 ionisations in the nucleus or a cellular dose of on average 1 mGy [6]. For α-particles (at an initial energy of 5.5 MeV), one track crossing each nucleus (at 8 μm diameter) results an average dose of 370 mGy, generating about 2.4 10^4 ionisations in the nucleus [2]. Accordingly, a mean dose of 3 mGy from α-radiation results in only about 1**%** of the cells being hit - and essentially all hit cells will have experienced the passage of just one α-particle, with extremely few cells being hit by two or more particles.

The dose response relationship of cellular end points for the α-radiation is shown as being approximately linear ($f(D) = \alpha D$), and generally that for the γ-radiation as linear-quadratic ($f(D) = \alpha D+\beta D^2$). It follows, therefore, that the RBE value depends on the dose and dose rate at which the biological effect is measured and that the RBE decreases as the dose increases.

The prime determinant of radiation effectiveness for different radiation types is the formation of ionisation clusters resulting in local multiple damages in DNA. The proportions, as well as the complexity, of this damages within a single track are higher for high LET than for low LET radiation, and by assuming that the repair will be least efficient for this complex damage, the RBE will be enhanced. The much larger number of sparsely-distributed ionisations produced by low LET radiation is assumed to play a minor role in regard to the induction of late biological effects as compared with the random production of ionisation clusters mainly at the end of the particle tracks [2]. An increase in the yield of severe clustered damage in DNA may lead to an expectation for high RBE values for high LET radiations. Conversely, the complex damage may be more lethal for the cells and tend to reduce RBE values for long-term effects in viable cells. Taken together, these factors indicate that the RBE may vary considerably.

In a contaminated environment, low doses and low dose rates are to be expected, and the exposure pattern is chronic rather than acute. Accordingly, single track events dominate which plays a significant role in the interpretation of RBE relationships. It seems highly unlikely that individual tracks should overlap, and interact, with one another at any doses and dose rates of interest in biota dosimetry.

3. BIOLOGICAL END-POINTS

The effects important for radiation protection of man are induction of cancer, hereditary effects and effects to the developing brain [1]. They are assumed to be mainly stochastic effects i.e. induced by a single particle track showing no threshold. Concomitantly, in radiation protection, an acceptable risk level is discussed. The question remains if the same philosophy is applicable for the environment and the protection of non-human organisms.

Radiation-induced mortality of an organism is due to the death of a large number of cells - for mammals usually in the bone marrow. Even if the death of a single cell is a stochastic effect, the death of many thousands of stem cells in the bone marrow will be a deterministic effect showing a sigmoid dose-response with an effective threshold. During the developmental stages the threshold is less pronounced due to increased radiosensitivity.

Induction of cancer and hereditary effects are supposed to be of minor importance for the protection of non-human organisms. Assuming that the risk for cancer induction is similar to humans, the acceptance of such a risk would probably be much greater, since effects on populations or ecosystems are generally not expected. Hereditary effects are supposed to affect non-human organism similarly as humans but a higher frequency of mutations in non-human organism is likely to be tolerated.

Impaired reproductive success, encapsulating effects on fertility and fecundity, as well as teratogenic effects, is most critical for protecting non-human organisms. These effects are induced at low dose levels and can, in some cases, be regarded as stochastic effects. The gametogenesis has been found to be extremely radiosensitive in mice. For example, the primary oocytes have shown LD_{50} values of a few mGy after both external and internal irradiation [7, 8]. Similarly, spermatogonial cells have also been found to be extremely radiosensitive [9]. Whether these radiosensitivities of gametogenesis apply to other organisms in the environment is not so well known. Teratogenic effects are difficult to study but a decline in mental capacity affecting the fitness of organisms may potentially lead to decreased survival of the individual [10].

5. TENTATIVE STRATEGIES

Clearly, the effects on reproduction are of great importance in relation to the protection of the biota. Similar to human radiation protection, any proposed dose limits or regulatory standards must be based on experimental as well as theoretical considerations. Therefore, the identification of relevant RBE values for biota implies the recognition of critical end-points leading to reproductive disturbances. Concurrently, a mechanistic understanding of these end-points, whether they are of stochastic (single track) or deterministic origin, has to be elucidated.

Assessment of the possible impact of incremental radiation exposures on biota requires values for radiation weighting factors appropriate to the absorbed dose rates likely to be experienced in a contaminated environment. In respect of the radiation effects of interest, it seems most probable to concentrate on stochastically induced end-points, since such a strategy would have an overall effect of being conservative for all groups. However it might not be necessary in an assessment framework, to pertain to absolute figures on weighting factors, but merely indicate a span for relevant values at different exposure situations and ecosystems, as illustrated in the FASSET-project [11]. The advantage could be flexibility and transparency but probably mean more tedious work for the assessors. Finally, it needs to be emphasised that radiation weighting factors should be based on radiobiological knowledge solely.

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Radionuclide accumulation and dose burden in small mammals in Chernobyl zone

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Abstract. By results of complex research in the Chernobyl zone, peculiarities of radionuclide accumulation in small mammal organisms have been studied and radiation doses have been assessed. It is shown, that radionuclide intake is defined by trophic specialization of the animals, and how that depends on season and local changes of the animals diet. A detailed description of ^{90}Sr and ^{137}Cs distribution in organism of bank vole has been presented. Radiation dose burden has been assessed for 4 small mammal species, and for conditions of the last 7 years. As a rule, the major contributor into total dose – external beta irradiation, some less – external gamma irradiation. Internal dose depends on species radionuclide accumulation and it is mainly determined by ^{90}Sr, with the exception of the bank vole, for which an internal irradiation is more important than external, due to extremely high ^{137}Cs accumulation.

1. INTRODUCTION

Small mammals, mouse rodents, and shrews, in particular, have for a long time been objects of radioecological research. Nevertheless, many aspects of their radioecology have not yet been studied, especially when we consider consequences of the Chernobyl disaster. As the wild mice are constant objects of Chernobyl research, this question is very important for correct interpretations of biological effects observed there. For example, there are no detailed data for radionuclide distribution through wild rodent organisms. There are much more data of assessment of dose burdens received by small mammals [5-7]. But those data concern conditions of the first post-accident years. Taking into account that radioecological situation has changed over last 10 years, the dose assessment should be repeated.

2. GENERAL CHARACTERISTICS OF RADIONUCLIDES ACCUMULATION

On the whole, even in 10 years after the accident ^{90}Sr activities concentration (in skeleton) and those of ^{137}Cs (in muscles) within some of the Chernobyl zone central areas reached hundreds, sometimes thousands Bq per gram of the tissue's weight [1, 3, 4]. At the same time, values vary for the same area by up to 1-2 orders of the value. During research activities, it was determined that frequency distribution of the measured values (both specific activity, and transfer factors), even within one area, while for one animal species log-normal (or seldom exponential) characteristics with more skew towards bigger values [3, 4].

According to the research results of 1995-1997 it was determined that radionuclides TF substantially depends on trophic specialization of the animals. So, forest inhabitants – euryphags – *Clethrionomys glareolus* are characterized with the most ^{137}Cs accumulation, and grasseaters (stenophags and oligophags) living in the fields – *Microtus spp.* are characterized with the minimum (Table 1). On the contrary, species that prefer plants vegetative mass, and insectivore shrews *Sorex*, accumulate radioactive strontium in a greater extent. Research performed in 1997-2000 had similar results (concerning to interspecific differences) (Table 2). Evidently, general dispersion of data is caused by peculiarities of individual diet, as well as by season and soil-vegetation conditions of the animal catching locations. As result, the TF indices, calculated on analysis results for at least 50-70 animals, are most confidence.

TABLE 1. RADIONUCLIDES TRANSFER FACTOR (LG TF) 'SOIL-ANIMAL TISSUE' FOR DIFFERENT SPECIES OF SMALL MAMMALS (THE AVERAGE FOR THE CHERNOBYL ZONE, DATA FOR 1995-1997), MEAN ± STANDARD ERROR

TF	Species	lg (TF)	n	(Bq/g)/(MBq/m^2)
^{137}Cs 'soil-muscle tissue'	*Microtus oeconomus*	-0.66±0.13	67	0.22
	Microtus arvalis (+M. rossiaemeridionalis)	-0.05±0.07	184	0.90
	Sorex araneus	0.07±0.16	40	1.19
	Sylvaemus flavicollis	0.10±0.21	20	1.26
	Apodemus agrarius	0.16±0.10	102	1.43
	Mus musculus	0.29±0.16	30	1.95
	Sylvaemus sylvaticus	0.33±0.21	33	2.11
	Clethrionomys glareolus	1.49±0.07	169	30.8
^{90}Sr 'soil-bone tissue'	*Mus musculus*	0.79±0.10	28	6.21
	Apodemus agrarius	0.93±0.06	72	8.43
	Sylvaemus flavicollis	1.02±0.17	17	10.58
	Clethrionomys glareolus	1.11±0.04	170	12.78
	Microtus oeconomus	1.12±0.06	44	13.24
	Microtus arvalis (+M. rossiaemeridionalis)	1.17±0.05	158	14.90
	Sorex araneus	1.35±0.09	32	22.23
	Sylvaemus sylvaticus	1.46±0.09	25	29.00

TABLE 2. RADIONUCLIDES TRANSFER FACTOR (TF) FOR DIFFERENT SPECIES OF SMALL MAMMALS (AVERAGE FOR THE CHERNOBYL ZONE, DATA FOR 1997-2000), MEAN ± STANDARD ERROR

TF	Species	lg (TF)	n	(Bq/g)/(MBq/m^2)
^{137}Cs 'soil-muscle tissue'	*Muscardinus avellanarius*	0.44±0.08	33	2.7
	Sorex araneus	0.62±0.13	50	4.2
	Apodemus agrarius	1.07±0.09	45	11.7
	Sylvaemus flavicollis	1.12±0.05	234	13.3
	Microtus sp.	1.14±0.12	25	13.9
	Clethrionomys glareolus	1.40±0.06	198	25.4
^{90}Sr 'soil-muscle tissue'	*Apodemus agrarius*	-0.48±0.19	30	0.33
	Microtus sp.	-0.13±0.09	12	0.75
	Sylvaemus flavicollis	0.16±0.04	91	1.44
	Sorex araneus	0.29±0.17	12	1.95
	Clethrionomys glareolus	0.43±0.05	77	2.69

3. PECULIARITIES OF RADIONUCLIDE DISTRIBUTION IN BODY TISSUES

In the study on *Microtus* voles we investigated ^{137}Cs distribution in tissues. It was determined that the largest ratio values of ^{137}Cs concentration in organs to one in muscles are in skin (2.02±0.45) and kidneys (1.57±0.29), in other tissues they are much lower. The comparison of the ^{137}Cs specific concentration in tissues with their size (mass index) has shown that the major portion of the total radionuclide concentration in the animal body is in the bone and muscle tissues aggregate (44.1 ± 2.8%) and in the skin (24.8 ± 2.5%). The total content of ^{137}Cs in gastrointestinal tract is about 20-25%, but nevertheless, it may be assumed that it is strongly dependent on the food radioactivity.

In this complex of works ^{90}Sr accumulation in bones was estimated only for yellow-necked mouse *Sylvaemus flavicollis*: TF=169.8 (Bq/g)/(MBq/m^2), lg (TF) = 2.23±0.06 (n=37).

A more detailed research was conducted on the bank vole (*Clethrionomys glareolus*), ^{90}Sr and ^{137}Cs contents in tissues were calculated. Table 3 shows that ^{137}Cs distribution corresponds with the tissue portion in the animal's body mass. Comparison of the activity concentration of each tissue with the ^{137}Cs average concentration in the body give more evident differences between the tissues. The highest radionuclide concentration is in the skin, and less in spleen and eyes.

The ^{90}Sr distribution in the vole body, as it was anticipated, is uneven – more than 80% of overall content is in the bone tissue (Table 4). In comparison with ^{90}Sr average activity in the body, after the skeleton come eyes and spleen, and their values are much higher than in other tissues and organs. This is interesting, as for instance eyes are surrounded by the bone tissue in any case (i.e., exposed to ^{90}Sr+^{90}Y beta-particles), besides, eyes are affected by the external exposure from soil. Also, just like haematogenic organs, the spleen and the bone morrow, eyes are a critical organ for small mammals. It is worth noting that these organs and tissues have highest relative concentrations and ^{137}Cs (see above).

TABLE 3. ^{137}Cs DISTRIBUTION IN *CLETHRIONOMYS GLAREOLUS* (N=13), MEAN ± STANDARD ERROR

Type of tissue	% of the total content in the body	Ratio of ^{137}Cs concentration in tissues to the average one in the body
eyes	0.23±0.03	1.20±0.18
spleen	0.33±0.04	1.22±0.11
heart	0.74±0.06	0.78±0.05
fat tissue	1.13±0.37	0.81±0.10
lungs	1.16±0.11	0.85±0.05
kidney	1.87±0.18	1.05±0.07
brain	1.97±0.24	0.75±0.02
uterus	5.10±2.10	0.95±0.15
liver	5.67±0.42	0.84±0.05
skeleton	5.76±0.48	0.70±0.05
testicles	8.37±1.48	1.09±0.06
skin	16.7 ± 1.4	1.48±0.10
gastrointestinal tract	21.3 ± 2.0	0.90±0.05
muscles	38.3 ± 1.3	1.04±0.04

TABLE 4. ^{90}Sr DISTRIBUTION IN *CLETHRIONOMYS GLAREOLUS* (N=13), MEAN ± STANDARD ERROR

Type of tissue	% of total content in the body	Ratio of ^{90}Sr concentration in tissues to the average one in the body
brain	0.36±0.09	0.16±0.03
eyes	0.23±0.04	1.30±0.23
fat tissue	0.32±0.07	0.47±0.14
gastrointestinal tract	4.20±0.81	0.19±0.03
heart	0.25±0.05	0.28±0.06
kidney	0.26±0.05	0.15±0.03
liver	0.31±0.06	0.05±0.01
lungs	0.29±0.06	0.24±0.05
muscles (values anticipated)	6.4-7.2	0.15-0.20
skeleton	83.4±5.9	10.9 ± 0.9
skin	3.85±0.56	0.36±0.06
spleen	0.23±0.04	1.02±0.24
testicles	0.42±0.10	0.07±0.02
uterus	0.50±0.14	0.19±0.09

TABLE 5. RATIO OF RADIATION BURDEN OF SMALL MAMMALS FORMED BY DIFFERENT RADIATION SOURCES UNDER EQUAL RADIOECOLOGICAL CONDITIONS (SOIL CONTAMINATION DENSITY, MBq/m^2: ^{90}Sr – 0.4, ^{137}Cs – 1.0; BETA-RADIATION – 300 COUNT/cm^2×min; GAMMA-RADIATION – 3.5 µGy/h)

	Clethrionomys glareolus	*Apodemus agrarius*	*Microtu arvalis*	*Sorex araneus*
Radionuclides specific activity, Bq/g				
^{137}Cs in soft tissues, average	30.8	1.4	0.9	1.2
^{137}Cs in bones	21.0	1.0	0.6	0.8
^{90}Sr soft tissues, average	0.6	0.4	0.7	1.1
^{90}Sr in bones	2.9	1.9	3.4	5.1
Dose formed by various sources, µGy/day				
external beta-radiation	81.6	81.6	81.6	81.6
external gamma-radiation	63.0	63.0	63.0	63.0
^{137}Cs in soft tissues, average	93.7	4.3	2.7	3.6
^{137}Cs in bones	112.4	5.2	3.3	4.3
^{90}Sr in soft tissues, average	11.9	7.9	13.9	20.7
^{90}Sr in bones	16.1	10.6	18.8	28.0

4. RADIATION BURDEN ASSESSMENT

Our research work included not only assessments of small mammals organisms radioactive contamination but also assessments of formed dose burdens [1, 2, 4]. Generally, the structure of dose burdens formed by different irradiation sources varies. Even rather small areas and one species can have different daily dose burdens (varying by a hundred or thousand times) [1]. At the same time under present conditions (years 1995-2000) small mammals inhabiting areas of so-called “Red Forest” obtain overall dose burdens up to 30 mGy/d, some of them obtain up to 100 mGy/d. Under present conditions external beta irradiation dose exceeds external gamma as before, and they are more than burdens from incorporated radionuclides (Table 5). Doses from incorporated ^{90}Sr of the majority of small mammals exceed doses formed by incorporated ^{137}Cs, with the exception of the bank vole. Among the greatest number of small mammals, the highest overall doses (from all sources) are obtained by bank voles and shrews. Therewith, incorporated ^{137}Cs is the main contributor to internal doses of the first species, and ^{90}Sr – of the latter. It is clear that these estimations ignore a set of factors determining both accumulation of radionuclides, their distribution in tissues and the intensity of external irradiation. Thus, we have shown the effects of soil agrochemical characteristics on the formation of absorbed doses [4]. For example, on account of higher ^{137}Cs transfer factors under conditions of moist peat soils the dose burdens of small mammals can be one order of magnitude greater that those under conditions of dry soddy-podzol soils with the same contamination level. It may be assumed that the seasonal change of foodstuff can also have a significant impact on absorbed doses formation.

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Doses to Black Sea fishes and mussels from the naturally occurring radionuclide ^{210}Po

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Abstract. ^{210}Po contributes significantly to radiation doses absorbed by marine biota. The measured specific activities of this radionuclide and the estimated doses delivered to various species of Black Sea fish depend on the ecological group to which the investigated species belongs, increasing from benthic, to demersal and to pelagic ones. The highest doses were calculated for the commercially valuable Black Sea mussel and pelagic fishes. Their levels are much lower than limits estimated as significant for the protection of aquatic organisms from ionizing radiation.

1. INTRODUCTION

The natural radionuclide ^{210}Po contributes significantly to ionising radiation doses absorbed by marine biota [1, 2]. The investigation of the ability of different marine organisms from the Ukrainian coast of the Black Sea to accumulate ^{210}Po was initiated in 1999 [3, 4]. The purpose of this work was to determine ^{210}Po concentrations in common Black Sea fish species and in mussels *Mytilus galloprovincialis* Lam. as well as to calculate the internal doses they received from this radionuclide.

2. MATERIAL AND METHODS

The determination of ^{210}Po in marine biota was performed according to the radiochemical procedure presented in [5]. ^{208}Po was added as a yield tracer. Polonium was spontaneously plated onto silver disks. Alpha counting of ^{208}Po and ^{210}Po was done using an EGandG ORTEC silicon surface-barrier detector and alpha-spectrometer system. The sampling places were chosen in different bays of Sevastopol and along the coast of Crimea from Cape Lukul to the Cape Sarych (Figure 1). Sixteen common species of fish were investigated (see Table 1) during 1999-2000. The mean values of the ^{210}Po specific activities for each species, in $Bq \cdot kg^{-1}$ wet weight (ww) and the standard errors (SE) of the means are given in Table 1. During the year 2000 also 308 mussel samples (soft tissues) were analysed. The number of individuals used in each determination depended on the size of the mussels and varied between 5 and 20. The internal doses received by all investigated fish species calculated using the criteria and equations given in [6, 7, 8] are presented in Table 1.

3. RESULTS AND CONCLUSIONS

^{210}Po concentrations in the investigated species of Black Sea fish were found to vary widely (see Table 1) and depend on the ecological group, decreasing from pelagic to demersal and benthic species. The absorbed dose rates decrease in the same order.The specific activities of ^{210}Po and the absorbed doses delivered by this radionuclide are highest in the Black Sea sprat and anchovy, which are the main commercial fishes in the investigated area.

^{210}Po is accumulated by marine organisms through food intake. The specific activities of ^{210}Po determined in the viscera of Black Sea sprat were 2.5-4 times higher than those determined for the whole body of this fish species. The calculated rates of the absorbed doses to its viscera ranged from 2.2 to 3.5 $mGy \cdot a^{-1}$.

In the commercial species of the Black Sea mussel *M.galloprovincialis*, ^{210}Po specific activities reached up to 60 $Bq \cdot kg^{-1}$ ww giving absorbed dose rates of up to 1.64 $mGy \cdot a^{-1}$.

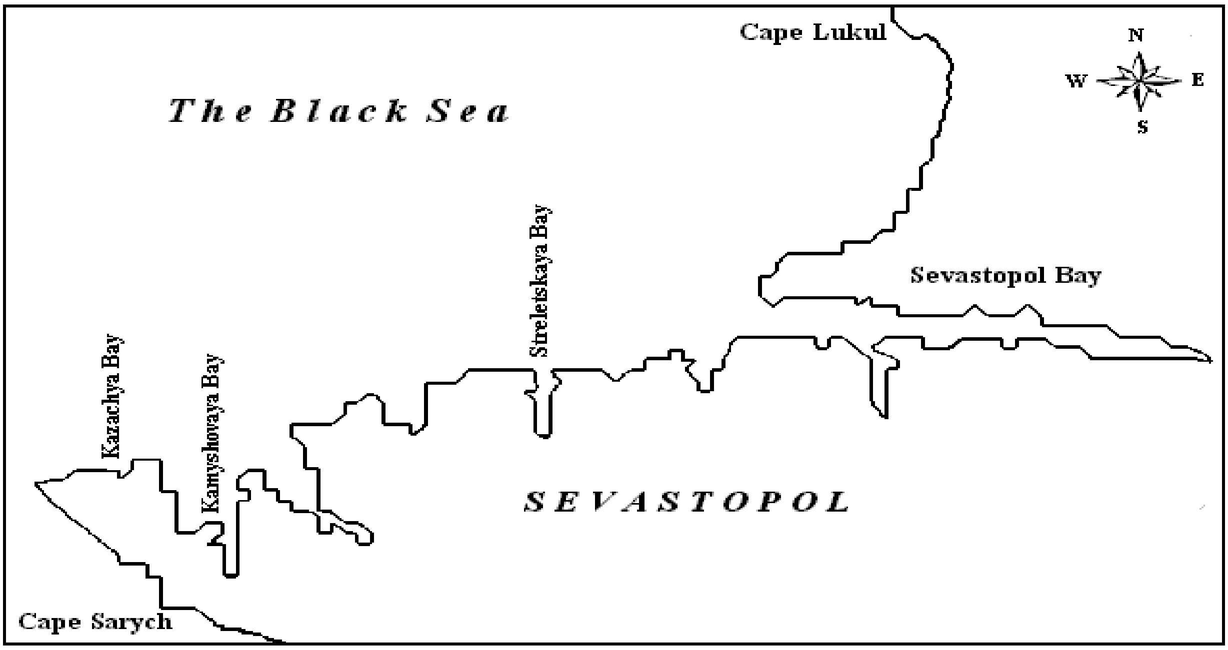

FIG. 1. The schematic map of the Sevastopol Bays area, where sampling took place.

TABLE 1. THE SPECIFIC ACTIVITIES OF ^{210}Po IN COMMON SPECIES OF THE BLACK SEA FISHES (SEVASTOPOL BAYS) AND THE ABSORBED DOSE RATES FROM THIS RADIONUCLIDE

English name of fish	Latin name of fish	^{210}Po specific activity, Bq·kg^{-1} ww	Absorbed dose rate, mGy·a^{-1}
Benthic species:			
Smallscaled scorpion fish	*Scorpaena porcus Linne*	0.98 ± 0.08	$2.7 \cdot 10^{-2}$
Common stargazer	*Uranoscorpus scaber Linne*	0.37 ± 0.04	$1.0 \cdot 10^{-2}$
Grass goby	*Zosterisessor ophiocephalus Pallas*	1.0 ± 0.1	$2.6 \cdot 10^{-2}$
Round goby	*Neogobius melanostomus Pallas*	1.7 ± 0.2	$4.7 \cdot 10^{-2}$
Black goby	*Gobius niger Linne*	2.1 ± 0.3	$5.7 \cdot 10^{-2}$
Toad goby	*Mesogobius batrachocephalus Pallas*	0.58 ± 0.07	$1.6 \cdot 10^{-2}$
Blunt-snouted mullet	*Mullus barbatus ponticus Essipov*	4.7 ± 0.5	$1.3 \cdot 10^{-1}$
Demersal fishes:			
Ocellated wrasse	*Crenilabrus ocellatus Forsäl*	2.0 ± 0.3	$5.5 \cdot 10^{-2}$
Long-striped wrasse	*Symphodus tinca Linne*	6.6 ± 0.9	$1.8 \cdot 10^{-1}$
Whiting	*Merlangius merlangus euxinus Nordmann*	10.7 ± 1.2	$2.9 \cdot 10^{-1}$
Pelagic species:			
Mediterranean sand smelt	*Atherina hepsetus Linne*	1.7 ± 0.2	$4.6 \cdot 10^{-2}$
Flat needlefish	*Belone belone euxini Günther*	7.4 ± 0.8	$2.0 \cdot 10^{-1}$
High-body pickerel	*Spicara smaris Linne*	13.6 ± 1.4	$3.7 \cdot 10^{-1}$
Black Sea scad	*Trachurus mediterraneus ponticus Aleev*	5.2 ± 0.5	$1.4 \cdot 10^{-1}$
Black Sea sprat	*Sprattus sprattus phalericus Risso*	32.0 ± 3.2	$8.7 \cdot 10^{-1}$
Black Sea anchovy	*Engraulis encrasicolus ponticus Aleksandrov*	40.7 ± 4.3	1.1

A radiation weighting factor of 20 [6, 8, 9] was applied for the calculation of equivalent doses for the investigated species of the Black Sea biota from ^{210}Po. The highest values were thus estimated for the Black Sea anchovy (around 22 mGy·a^{-1}), sprat (17.5 mGy·a^{-1}) and mussel *M.galloprovincialis* (33 mGy·a^{-1}).

The data presented above show that the internal radiation doses received by the Black Sea fishes from the natural radionuclide ^{210}Po depended on their belonging to different ecological groups, increasing from benthic, demersal to pelagic species. The highest values are determined in the Black Sea anchovy and sprat. The highest doses, calculated for the Black Sea mussel *M.galloprovincialis* and pelagic fishes, are much lower than the dose limits proposed in [10] for the protection of aquatic organisms from ionizing radiation.

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A method for calculation of dose per unit concentration values for aquatic biota within FASSET

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Abstract. A dose per unit concentration database has been generated for application to ecosystem assessments within the FASSET framework. Organisms are represented by ellipsoids of appropriate dimensions, and the proportion of radiation absorbed within the organisms is calculated using a numerical method implemented in a series of spreadsheet-based programs. Energy-dependent absorbed fraction functions were derived for calculating the total dose per unit concentration of radionuclides present in biota or their surrounding medium. All radionuclides and reference organism dimensions defined within FASSET for marine and freshwater ecosystems are included. With regard to the applicability of the method, it is assumed that differences in density between the organism and the surrounding environmental media may be ignored; for the aquatic environment this is an extremely good approximation.

1. INTRODUCTION

The FASSET programme brings to radiation protection a framework for the assessment of environmental impact of ionising radiation. This framework links together current knowledge about sources, exposure, dosimetry and environmental effects for reference organisms and ecosystems. A key component of the framework has been to provide a method for assessing radiation dose factors to aquatic organisms per unit radionuclide concentration in the surrounding media and to the organism itself. These are the factors that relate radionuclide concentrations to the resulting radiation dose rate.

To calculate dose rate per unit concentration factors (DPUC's) the assumption is made that organisms are represented as ellipsoids whose dimensions are given in Table 1. Density differences between the organism and the surrounding media are ignored, and resulting absorbed doses are averaged throughout the volume of the organism. In calculating the internal DPUC it is assumed that radionuclides are distributed uniformly through all tissues, whereas for the external DPUC it is assumed that the organism is immersed in an infinite absorbing medium with the stated concentration.

2. DESCRIPTION OF METHOD

Absorbed fractions (AF's) were derived from simple functions for energy deposition in water by photons and electrons from point isotropic sources. Photons and electrons were treated separately due to physical differences in their dose distributions around point sources and biological effects.

For γ rays, a Monte Carlo code was developed to replicate the interactions between photons and tissue, based on Berger [2]. This author provided dose distribution data for photons in terms of a point isotropic specific AF, $\Phi_E(r)$. Polynomial functions were derived from Berger's tabulated data to provide a continuous interpolation of $\Phi_E(r)$ for each of the discrete photon energies provided. From here it was possible to calculate numerically the AF for a uniformly contaminated absorbing volume.

For the calculation of AF's for β particles a more complex (but numerically more efficient) method was adopted. Berger [3] tabulated values of r_p, the radius r of a sphere within which p% of the energy is absorbed from a point β source located at the centre. These values were transformed to values of a fractional absorption from a point β emitter within a sphere of radius equal to r/r_{90} around the point source. This radial transformation makes the fractional absorption relatively independent of energy, facilitating calculation of the fraction of total energy emitted and absorbed by reference organisms [4].

TABLE 1. FASSET REFERENCE ORGANISM DIMENSIONS [1]

Organism (marine)	Length (cm)	Width (cm)	Depth (cm)	Area/Vol. (m^{-1})	Organism (freshwater)	Length (cm)	Width (cm)	Depth (cm)	Area/Vol. (m^{-1})
Bacteria	2.0E-04	5.0E-05	5.0E-05	9.7E+06	Phytoplankton	8.0E-03	7.0E-04	7.0E-04	6.8E+05
Phytoplankton	5.0E-03	5.0E-03	5.0E-03	1.2E+05	Zooplankton	2.0E-01	1.4E-01	1.6E-01	3.7E+03
Zzooplankton	6.2E-01	3.1E-01	6.1E-01	1.3E+03	Crustacean	1.0E+00	3.0E-01	1.0E-01	3.2E+03
Benthic mollusc	5.0E+00	2.5E+00	2.5E+00	2.1E+02	Insect larvae	1.5E+00	1.5E-01	1.5E-01	3.2E+03
Benthic worm	2.3E+01	1.2E+00	1.2E+00	3.9E+02	Vascular plant	1.0E+02	1.0E-01	2.0E-02	1.4E+04
Vascular plant	2.0E+01	5.0E+00	5.0E+00	9.7E+01	Gastropod	3.0E+00	1.5E+00	1.5E+00	3.4E+02
Pelagic fish	3.0E+01	6.0E+00	6.0E+00	8.0E+01	Amphibian	7.0E+00	3.0E+00	2.0E+00	2.1E+02
Marine bird	1.5E+01	1.1E+01	7.6E+00	5.9E+01	Bivalve mollusc	1.0E+01	4.5E+00	3.0E+00	1.4E+02
Macroalgae cluster	2.5E+01	2.5E+01	2.5E+00	1.1E+02	Pelagic fish	3.0E+01	6.0E+00	3.5E+00	1.1E+02
Benthic fish	4.0E+01	2.0E+01	3.0E+00	9.3E+01	Benthic fish	5.0E+01	8.0E+00	7.0E+00	6.4E+01
Ccrustacean	3.0E+01	1.0E+01	1.0E+01	4.9E+01	Mammal	3.3E+01	1.5E+01	1.5E+01	3.4E+01
Mmammal	1.8E+02	4.4E+01	4.4E+01	1.1E+01	Bird	4.0E+01	2.0E+01	1.8E+01	2.7E+01

Energy absorbed fraction functions (EAFF's) were fitted separately for photons and electrons, the key issue being to provide a reliable interpolation between calculated values, avoiding the instabilities that can occur when fitting data to polynomials. EAFF's are of the set form:

$$F_\gamma(E) = e^{-\left(\frac{E}{2\sigma}\right)^n} + a.e^{-\lambda E^m} \quad ; \quad F_\beta(< E >) = \frac{1}{1 + a < E >^n}$$

where a, m, n, λ and σ are "best fit" constants to the Monte Carlo calculated data. Photon and β AF's for the FASSET organism geometries as defined by the derived fitting constants are given in Table 2.

For α-particles the range in living tissue is very small - typically, about 50 µm. Hence, the assumption was made that internally incorporated radionuclides are distributed uniformly within the organisms of interest. Therefore, the AF for α–particles is unity for all organisms except bacteria. The AF for internally incorporated α, β and γ emitters is zero for bacteria, as their dimensions are around a few microns. Doses to micro-organisms were assumed to be equal to the absorbed dose (including the absorbed dose from α emissions) in the medium within which they are incorporated.

TABLE 2. ABSORBED FRACTION FUNCTIONS FOR β AND γ RADIATION - MARINE ECOSYSTEM

a) Coastal - estuarine ecosystem

	Phyto-plankton	**Zoo-plankton**	**Mollusc**	**Worm**	**Vascular plant**	**Pelagic fish**	**Benthic fish**	**Bird**	**Macro-algae**	**Crusta-cean**	**Mammal**
Coefficients for $AF_\beta = 1/(1+aE^n)$											
a	5.9E+02	2.1E+00	2.0E-01	4.2E-01	8.7E-02	7.4E-02	7.4E-02	4.8E-02	8.5E-02	4.2E-02	9.2E-03
n	1.9E+00	1.5E+00	1.6E+00	1.5E+00	1.4E+00	1.6E+00	1.5E+00	1.5E+00	1.5E+00	1.5E+00	1.3E+00
Coefficients for $AF_\gamma = \exp[-(E/2\sigma)^n] \times a \exp[-\lambda E^m]$											
σ	5.0E-04	9.6E-03	2.0E-02	1.5E-02	2.3E-02	2.3E-02	2.2E-02	2.6E-02	2.2E-02	2.3E-02	1.6E-02
n	5.2E-01	1.2E+00	1.7E+00	1.4E+00	1.5E+00	1.4E+00	1.3E+00	1.5E+00	1.5E+00	1.2E+00	8.2E-01
a	5.8E-05	1.4E+00	2.9E+00	2.6E-02	1.1E-01	1.4E-01	1.8E-01	2.2E-01	1.7E-01	2.7E-01	6.1E-01
λ	7.4E-02	5.7E+00	4.6E+00	3.1E-01	4.6E-01	4.6E-01	7.7E-01	7.5E-01	8.1E-01	7.5E-01	2.6E-01
m	1.6E+00	1.8E-02	3.1E-02	6.4E-01	4.7E-01	4.7E-01	2.3E-01	2.5E-01	2.2E-01	2.8E-01	8.0E-01

b) Freshwater ecosystem

	Phyto-plankton	**Zoo-plankton**	**Crusta-cean**	**Insect larvae**	**Vasc. plant**	**Gastro-pod**	**Amphi-bian**	**Bivalve mollusc**	**Pelagic fish**	**Benthic fish**	**Mam-mal**	**Bird**
Coefficients for $AF_\beta = 1/(1+aE^n)$												
a	3.8E+03	7.9E+00	4.6E+00	5.9E+00	2.5E+01	3.6E-01	1.9E-01	1.2E-01	9.1E-02	5.7E-02	2.8E-02	2.3E-02
n	1.9E+00	1.7E+00	1.5E+00	1.6E+00	1.6E+00	1.6E+00	1.6E+00	1.6E+00	1.5E+00	1.6E+00	1.5E+00	1.5E+00
Coefficients for $AF_\gamma = \exp[-(E/2\sigma)^n] \times a \exp[-\lambda E^m]$												
σ	7.8E-03	5.5E-03	6.3E-03	6.2E-03	2.4E-03	1.7E-02	2.0E-02	2.2E-02	2.2E-02	2.2E-02	2.4E-02	2.0E-02
n	3.1E+00	9.5E-01	9.8E-01	9.8E-01	7.1E-01	1.6E+00	1.7E+00	1.6E+00	1.4E+00	1.2E+00	1.1E+00	9.2E-01
a	2.8E-05	4.5E-01	2.8E-03	2.9E-01	5.1E-04	1.9E+00	9.7E-02	7.2E-02	1.0E-01	2.2E-01	3.8E-01	5.4E-01
λ	1.0E+00	5.6E+00	1.7E-01	4.9E+00	5.2E-01	4.6E+00	1.1E+00	4.0E-01	4.4E-01	7.1E-01	7.5E-01	9.0E-01
m	1.6E-01	1.5E-02	9.1E-01	2.0E-02	5.7E-01	2.7E-02	1.4E-01	4.9E-01	4.7E-01	2.9E-01	2.7E-01	2.3E-01

The full range of FASSET radionuclides (^{3}H, ^{14}C, ^{36}Cl, ^{40}K, ^{59}Ni, ^{63}Ni, $^{89,90}Sr$, ^{94}Nb, ^{99}Tc, ^{106}Ru, $^{129,131}I$, $^{134,135,137}Cs$, ^{210}Pb, ^{210}Po, ^{226}Ra, $^{227,228,230\text{-}232,234}Th$, $^{234,235,238}U$, ^{237}Np, $^{238\text{-}241}Pu$, ^{241}Am, $^{242\text{-}244}Cm$) was used for DPUC calculations. For each radionuclide and reference organism energies and yields of all α, β and γ emissions were extracted from the literature [5] and overall β and γ AF's were calculated as:

$$\overline{F_\beta} = \left(\sum_i p_i F_\beta(<E_i>)\right)_\beta \left(\sum_i p_i(<E_i>)\right)_\beta^{-1} \qquad \overline{F_\gamma} = \left(\sum_i p_i F_\gamma(E_i)\right)_\gamma \left(\sum_i p_i E_i\right)_\gamma^{-1}$$

where E_i is energy (MeV) and p_i denotes the fractional yield of individual emissions. An Excel-based VBA code was developed to sum the low β component (<10 keV) separately from the $\beta\gamma$ component, which refers to γ's and all other β particles, for all radionuclide decays. As many radionuclides have a progeny in secular equilibrium, it is necessary to calculate a combined AF for all the intervening radionuclides. An example is the merging of ^{227}Th with its progeny of ^{223}Ra, ^{219}Rn, ^{215}Po, ^{211}Pb, ^{211}Bi, ^{207}Tl and ^{211}Po, requiring the amalgamation of 37, 197 and 114 α, β and γ decay modes.

For internal exposure the DPUC values (in units of μGy h^{-1} per Bq kg^{-1}) were calculated as:

$$DPUC_{low\beta}^{int} = 5.77\times10^{-4}\times\overline{F_{low\beta}}\times\sum_i (p_i E_i)_{low\beta} \quad DPUC_{\beta\gamma}^{int} = 5.77\times10^{-4}\times\left[\overline{F_\beta}\times\sum_i (p_i E_i)_\beta + \overline{F_\gamma}\times\sum_i (p_i E_i)_\gamma\right]$$

$$DPUC_{\alpha}^{int} = 5.77\times10^{-4}\times\overline{F_\alpha}\times\sum_i (p_i E_i)_\alpha \qquad DPUC_{total}^{int} = DPUC_{low\beta}^{int} + DPUC_{\beta\gamma}^{int} + DPUC_{\alpha}^{int}$$

Where 5.77×10^{-4} is the conversion factor from MeV s^{-1} to μJ h^{-1}. For external exposure a simple formula is used for a uniformly contaminated isotropic infinite absorbing medium:

$$DPUC_{low\beta}^{ext} = 5.77\times10^{-4}\times\left(1-\overline{F_{low\beta}}\right)\times\left(\sum_i p_i E_i\right)_{low\beta} \quad DPUC_{\beta\gamma}^{ext} = 5.77\times10^{-4}\times\left[\left(1-\overline{F_\beta}\right)\times\left(\sum_i p_i E_i\right)_\beta + \left(1-\overline{F_\gamma}\right)\times\left(\sum_i p_i E_i\right)_\gamma\right]$$

$$DPUC_{\alpha}^{ext} = 5.77\times10^{-4}\times\left(1-\overline{F_\alpha}\right)\times\left(\sum_i p_i E_i\right)_\alpha \qquad DPUC_{total}^{ext} = DPUC_{low\beta}^{ext} + DPUC_{\beta\gamma}^{ext} + DPUC_{\alpha}^{ext}$$

These equations approximate the DPUC to an organism immersed in an infinite contaminated medium, neglecting density differences between organism and medium, allowing for self shielding by the organism and averaging the dose rate throughout the organism volume. No provision was made for applying weighting factors to the α or low energy β DPUC components because although the system of dosimetry for humans is well defined, such a system for wildlife has not yet been widely agreed.

Using the above procedure DPUC's for coastal ecosystems (marine) were tabulated for the following FASSET reference organisms: bacteria, phytoplankton and zooplankton, mollusc, worm, vascular plant, pelagic and benthic fish, seabird, macroalgae cluster, benthic crustacean and mammal. For the freshwater ecosystem the organisms are phytoplankton, zooplankton, crustacean, insect larvae, vascular plant, gastropod, amphibian, bivalve mollusc, pelagic and benthic fish, mammal and bird.

3. RESULTS AND DISCUSSION

Calculated EAFF's for β and γ radiation as represented by the fitting constants a, m, n, λ and σ, are given in Table 2. With 20,000 iterations the statistical uncertainty of the individual β EAFF's was kept to < ± 2.2**%** (2σ). For the γ EAFF statistical uncertainty typically ranged 3 - 20% (2σ) depending on energy, but on average it was ± 13**%**. The final product of using the above EAFF's to calculate DPUC's for different radionuclides is a comprehensive set of dose factors reported for both internal and external irradiation. The full dataset is reported separately within FASSET [1].

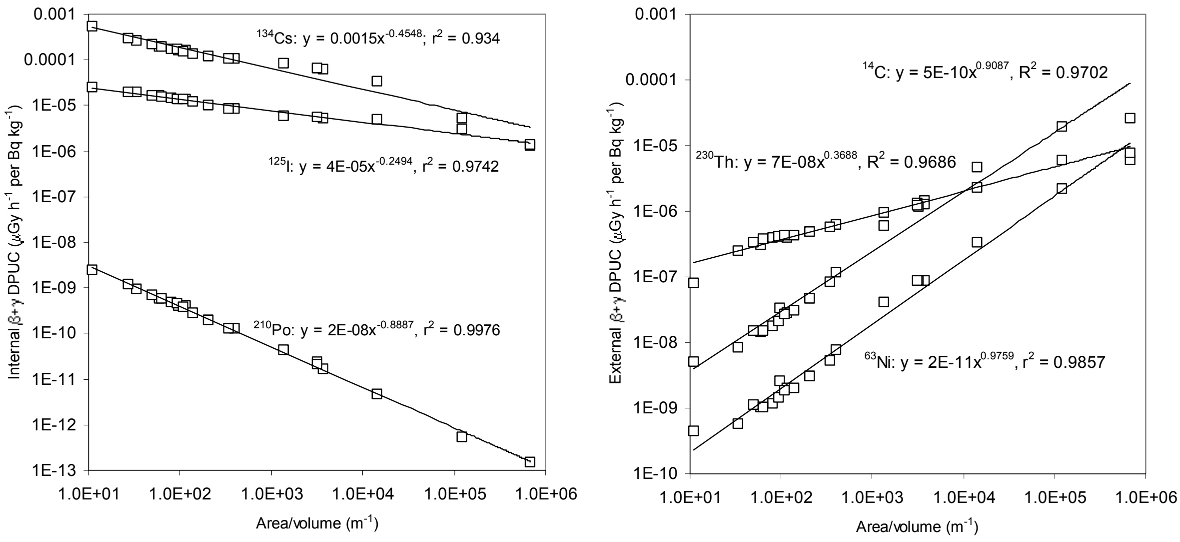

FIG. 1. Examples of DPUC (β + γ) dependency with respect to area/volume. Internal irradiation: ^{125}I, ^{137}Cs and ^{210}Po (left), and external irradiation: ^{14}C, ^{63}Ni and ^{230}Th (right).

DPUC data were represented graphically in order to investigate whether there was a regular pattern in respect of organism dimensions and indeed a possible dependence was found between DPUC and area/volume parameter. For both low β internal and external irradiation, DPUC's tend to correlate linearly with area/volume of the reference organism ellipsoid (mean r^2 for 43 radionuclides = 0.999 ± 0.001, range: 0.997 - 1.000, n = 23). For β + γ internal irradiation a power relationship tends to yield good results (mean r^2 = 0.80 ± 0.12, range: 0.50 - 0.998, n = 23), with only a handful of radionuclides (17 out of 43) yielding r^2 < 0.8. For external irradiation power functions also fit the data for most radionuclides (mean r^2 = 0.81 ± 0.17, range: 0.35 - 0.986, n = 24) with only 9 radionuclides having r^2 < 0.8. The overall trends are decreasing for internal irradiation and increasing for external irradiation. Examples of DPUC power relationships for ^{14}C, ^{63}Ni, ^{125}I, ^{137}Cs, ^{210}Po and ^{230}Th are given in Figure 1.

Inputting the area/volume ratio for an organism of dimensions different from those of the pre-set FASSET ellipsoids in the above mathematical relationships should result in a reasonable approximation for the new DPUC's for most radionuclides.

4. CONCLUSIONS

Absorbed fraction functions and DPUC's for the coastal - estuarine and freshwater ecosystems comprising all FASSET radionuclides are now available in respect to the dimensions specified for the reference organisms. With regard to the applicability of the method, the most important assumption is that differences in density between the organism and the surrounding environmental media may be ignored; for the aquatic environment this is an extremely good approximation.

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Radiation doses to aquatic organisms from natural radionuclides

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Abstract. A framework for protection of the environment is likely to require a methodology for assessing dose rates arising from naturally-occurring radionuclides. This paper addresses this issue for European aquatic environments through a process of a) data collation, mainly with respect to levels of radioactivity in water sediments and aquatic flora and fauna, (b) the use of suitable distribution coefficients, concentration factors and global data where data gaps are present and (c) the utilisation of a reference organism approach whereby a finite number of suitable geometries are selected to allow Dose per unit Concentration factors to be derived and subsequent absorbed dose calculations (weighted or unweighted) to be made. The majority of the calculated absorbed dose, for both marine and freshwater organisms arises from internally incorporated alpha emitters, with ^{210}Po and ^{226}Ra being the major contributors. Calculated doses are somewhat higher for freshwater compared to marine organisms, and the range of doses is also much greater. This reflects both the much greater variability of radionuclide concentrations in freshwater as compared to seawater, and also variability or uncertainty in concentration factor values. This work has revealed a number of severe gaps in published empirical data especially for European aquatic environments.

1. INTRODUCTION

In order to assess the potential consequences of exposures to radiation on non-human biota, arguably, two points of reference may be used. These are (a) natural background dose rates and (b) dose rates known to have specific biological effects on individual organisms [1]. With respect to the former reference point, a specific task within the EU-funded project “FASSET” [2] has been to assess doses, arising from naturally-occurring radionuclides, for selected organisms inhabiting European environments. This has initially involved the application of numerous criteria, e.g. [3], in order to select “reference” organisms - a series of entities that provide a basis for the estimation of the radiation dose rate to a range of organisms that are typical, or representative, of a contaminated environment. For the aquatic environment (freshwater and marine), the focus for this paper, the reference list includes bacteria, phytoplankton, macrophytes, macroalgae, zooplankton, mollusc, crustaceans, fish (benthic and pelagic), mammals and birds. These organism groups have allowed the construction of suitable geometries and target-source configurations thus enabling suitable Dose per unit concentration factors to be derived and applied to activity concentration data. Where relevant data are available, the assessment has focussed on doses received by organisms in European waters. However, there are substantial gaps in the data for European waters, particularly for freshwater ecosystems. Therefore, applicable non-European data have been used to fill gaps where possible in the interest of obtaining the most complete appraisal possible of the doses likely to be received by European aquatic organisms.

2. CONCENTRATIONS OF NATURALLY OCCURRING RADIONUCLIDES IN AQUATIC ENVIRIONMENTS

A literature review was conducted in order to provide information on the levels of naturally occurring radionuclides in marine sediment, waters and biota. Data are presented in terms of nominal median and 95th percentile values; the derivation of these figures from cited ranges is, to a degree, subjective but is intended to provide a realistic representation of the likely range of variability. Where only a single value is cited in the original source, the 95th percentile has been assumed to be a factor of 2 higher than the stated value. An example of the information collated – in this case for natural radionuclide concentrations in the body of marine organism is presented in Table 1. The highest concentrations are observed for potassium-40 and polonium-210.The values in Table 1 have been selected as representative of activity concentrations within the whole organism, usually taken to be 'soft tissue' for invertebrates and 'muscle' for vertebrates. For some radionuclides significantly elevated concentrations are observed in some organs. These elevations will lead to corresponding elevations of dose to these organs, but the biological and ecological significance of such elevation is at present unclear. The ranges quoted for concentrations in the broad classification of organisms reflects in some cases differences between taxa - for example, between bivalve and gastropod molluscs, or dolphins and whales.

Concentrations of natural series radionuclides in freshwater bodies are liable to be much more variable than those in the marine environment, since they are heavily influenced by the local geochemistry of the watershed. This lack of information creates some difficulty in interpreting these data in terms of mean values and ranges for European freshwaters. However use of global average data for the mean values, and the upper value cited by [4] as nominal 95th percentile values, appears reasonable in relation to the data presented. No references citing overall means and ranges for the concentrations of radionuclides in freshwater sediments have been identified. Therefore, for the purpose of making a general assessment of doses to freshwater biota, empirical distribution coefficients (K_ds) have been applied to collated water concentration data.As for water and sediments, published data on natural series radionuclides in freshwater organisms is sparse and no references citing data specific for Europe have been identified. Therefore, to make an indicative assessment of likely doses, concentration factors derived from the few published studies in other parts of the world have been used. The most useful studies have been those of the Kaveri river system in India (e.g. [5]) and the Alligator River system in Northern Territories, Australia (e.g. [6]). Some use has also been made of a Japanese compilation [7].

3. RADIATION DOSES TO AQUATIC ORGANISMS

Radiation doses are calculated using the methods described by [8]. Absorbed doses (i.e. doses which do not include a weighting factor to account for the relative biological effectiveness of α and low energy β radiations), to marine reference organisms are presented in Table 2. No doses are cited for seabirds, because of the lack of data on internally incorporated radionuclides. Doses for marine mammals include contributions only from internal potassium-40 and polonium-210, for which data are available. Doses are cited for benthic bacteria, which because of their small size are assumed to receive the same absorbed dose as the sediment which they inhabit. The likely range for total dose to each class of organism is evaluated by a statistical simulation in which concentrations in water, sediment and organisms are represented by lognormal distributions with the median and 95th percentile. The ranges cited are the 5th and 95th percentile values obtained by the simulation, based on 2,000 evaluations of the total dose. The calculated dose increases closely in proportion to any weighting factor applied for alpha radiation, since the radionuclides contributing most significantly to dose are polonium-210 and radium-226. Crustacea receive the widest range of doses, largely because of the spread of concentrations of polonium-210 between different taxa.

The range of likely doses in the freshwater environment has been assessed by a simulation method which includes variability (or uncertainty) in the concentration factors for organisms. The concentration factor data are derived from vary sparse data, so CFs have been represented in the simulation by lognormal distributions using a geometric mean derived from the data collation exercise, and the 95th percentile value a factor of 3 higher than this geometric mean. The results of the calculation, in terms of an unweighted absorbed dose, are given in Table 3. Important classes of organism for which there are insufficient data to support any estimate of dose include zooplankton, water birds, amphibians and aquatic mammals. As for marine organisms, the majority of the calculated absorbed dose arises from internally incorporated alpha emitters, with ^{210}Po and ^{226}Ra being the major contributors. The dose attributed is therefore closely proportional to the weighting factor assumed for alpha radiation. Calculated doses are somewhat higher than for marine organisms, and the range of doses is also much greater, reflecting both the much greater variability of radionuclide concentrations in freshwater as compared to seawater, and also variability or uncertainty in concentration factor values.

TABLE 1. NATURAL RADIONUCLIDE CONCENTRATIONS IN MARINE ORGANISMS

Concentration in organism, Bq kg^{-1} wet weight, as: Median (95th percentile)								
Nuclide	Phytoplankton	Zooplankton	Macroalgae	Mollusc	Crustacea	Fish pelagic/benthic	Bird	Mammal
^{40}K			300 (600)	50 (100)	50 (100)	100 (200)		
^{210}Po	2.5 (3.5)	25 (80)	2.1 (15)	37 (110)	50 (1000)	11 (500)		20 (100)[a]
^{226}Ra	2.7 (30)[a]	0.2 (0.8)[a]	0.3 (1)[a]	0.7 (1.5)[a]	0.7 (1.5)[a]	0.2 (0.5)[a]		
^{228}Ra								
^{228}Th	1 (3)[a]	0.3 (1)[a]	0.4 (2)	0.7 (2)	0.035 (0.2)			
^{230}Th	0.1 (0.3)[a]	0.06 (0.2)[a]	0.08 (0.2)	2 (80)	0.045 (0.2)	0.002 (0.01)		
^{232}Th	0.1 (0.3)[a]	0.06 (0.2)[a]	0.19 (0.8)	0.5 (1.5)	0.006 (0.012)	0.0007 (.002)		
^{238}U	0.4 (0.6)[a]	0.2 (0.5)[a]	1 (5)	1 (5)	0.14 (0.2)	0.008 (0.015)		

[1] Data from Cherry and Shannon (1974), for general oceanic waters

TABLE 2. CALCULATED UNWEIGHTED ABSORBED DOSES TO MARINE ORGANISMS

Calculated absorbed dose, microGy h^{-1}								
Nuclide	Bacteria	Phytoplankton	Zooplankton	Macroalgae	Molluscs	Crustacea	Pelagic fish	Mammals
^{40}K	6.7E-02	7.0E-03	4.9E-03	3.8E-02	2.6E-02	3.2E-02	3.2E-02	3.8E-03
^{210}Po	1.1E-02	7.6E-03	7.6E-02	6.4E-03	1.1E-01	1.5E-01	6.1E-03	6.1E-02
^{226}Ra	2.2E-01	4.6E-02	3.5E-03	5.2E-03	1.9E-02	2.0E-02	4.1E-03	2.1E-03
^{228}Ra	2.4E-06	8.2E-07	6.5E-07	6.6E-07	4.6E-07	5.2E-07	4.8E-07	3.3E-07
^{228}Th	1.7E-01	1.8E-02	5.6E-03	7.5E-03	1.7E-02	5.5E-03	3.5E-04	1.3E-03
^{230}Th	1.7E-01	2.7E-04	1.6E-04	2.2E-04	7.0E-03	2.5E-03	3.3E-05	4.9E-05
^{232}Th	8.3E-03	2.3E-04	1.4E-04	4.4E-04	1.2E-03	1.5E-05	1.7E-06	2.3E-07
^{238}U	2.3E-02	2.1E-03	1.1E-03	5.3E-03	5.9E-03	1.5E-03	5.4E-05	1.6E-05
Total	6.7E-01	8.1E-02	9.2E-02	6.3E-02	1.9E-01	2.1E-01	4.3E-02	6.8E-02
5th %	5.0E-01	3.7E-02	4.1E-02	4.6E-02	1.3E-01	7.3E-02	2.7E-02	1.9E-02
95th %	1.0E+00	5.3E-01	2.5E-01	1.4E-01	5.6E-01	2.9E+00	1.0E-01	3.1E-01

TABLE 3. CALCULATED UNWEIGHTED ABSORBED DOSE TO FRESHWATER ORGANISMS

Calculated absorbed dose, microGy h^{-1}							
Nuclide	**Bacteria**	**Phytoplankton**	**Macrophyte**	**Mollusc**	**Crustacean**	**Pelagic fish**	**Benthic fish**
^{40}K	8.1E-02	2.1E-05	1.1E-02	2.8E-02	2.8E-02	5.6E-03	1.3E-02
^{210}Po	4.9E-02	1.5E-01	9.2E-03	3.1E-01	6.1E-02	1.8E-03	6.1E-02
^{222}Rn	6.0E-03	8.3E-03	8.4E-03	8.3E-03	8.3E-03	8.4E-03	8.3E-03
^{226}Ra	2.2E-01	8.4E-02	1.7E-01	3.3E-02	9.4E-02	2.7E-02	1.1E-01
^{228}Ra	3.8E-03	1.2E-03	2.6E-03	6.8E-04	1.6E-03	4.4E-04	1.9E-03
^{230}Th	2.9E-03	7.0E-04	7.0E-04	3.8E-04	2.0E-04	1.5E-05	3.3E-05
^{232}Th	1.0E-03	2.5E-04	2.5E-04	1.3E-04	6.4E-05	5.1E-06	1.0E-05
^{238}U	1.3E-03	5.7E-03	2.3E-01	6.1E-03	9.2E-03	9.3E-04	1.6E-02
Total	**3.7E-01**	**2.5E-01**	**4.4E-01**	**3.8E-01**	**2.0E-01**	**4.4E-02**	**2.1E-01**
5th %	1.4E-01	9.6E-02	1.5E-01	1.5E-01	9.0E-02	1.7E-02	9.1E-02
95th%	5.5E+00	3.0E+00	6.1E+00	2.8E+00	2.6E+00	7.3E-01	3.4E+00

4. CONCLUSIONS – DATA GAPS AND RESEARCH REQUIREMENTS

The uptake of natural radionuclides by marine organisms has been quite extensively studied, and the range of natural doses experienced by most marine organisms can be quite easily established. There is a need for data on uptake of natural series radionuclides by seabirds, and data on European marine mammals would also be highly desirable. Because the majority of the dose is from alpha radiation, internal distribution of dose may be significant and information on localisation of dose in organs of possible importance to ecological endpoints, such as gonads, would be of great interest.

There is clearly a major need for data on the concentrations of natural series radionuclides in European freshwater organisms in order to understand the exposure of these organisms to natural radioactivity. Data on amphibians, water birds and aquatic mammals is lacking even at the global level. The predicted levels of radiation dose at the upper end of the expected distribution of radionuclide concentrations in water suggests that sites with naturally elevated concentrations may present opportunities for field research on biomarkers or radiation effects.

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The RESRAD-BIOTA Code for Application in Biota Dose Assessment*

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Abstract. The RESRAD-BIOTA code was developed through a partnership involving offices of the U.S. Department of Energy, the U.S. Environmental Protection Agency, and the U.S. Nuclear Regulatory Commission. RESRAD-BIOTA is designed to provide a full spectrum of analysis capabilities, from practical conservative screening methods to more realistic organism-specific dose assessments. The RESRAD-BIOTA code has many advanced features such as dose conversion factors for eight ellipsoid organism geometries, sensitivity analysis capability for studying parameter sensitivities, and text and graphic reports for easy interpretation of results. An improved version of the RESRAD-BIOTA code is currently available for test and evaluation.

1. INTRODUCTION

The development of the RESRAD-BIOTA code was initiated by the U.S. Department of Energy (DOE) in June 2000. The code was developed through a partnership among offices of the DOE, the U.S. Environmental Protection Agency (EPA), and the U.S. Nuclear Regulatory Commission (NRC). The DOE's graded approach for evaluating radiation doses to biota [1] was used as the starting foundation for code development. An interagency ECORAD Work Group is being used to coordinate and implement the development of the code. A beta version was released in 2002 for test and evaluation and the RESRAD-BIOTA code was presented in the Third International Symposium on the Protection of Environment from Ionising Radiation (SPEIR 3) held in Darwin, Australia in July 2002 [2]. Many new and advanced features have been added to the code after the SPEIR 3 Symposium. These advanced features and a new, redesigned, user interface are described in this paper.

2. FEATURES OF RESRAD-BIOTA

RESRAD-BIOTA is easy to install and user-friendly. The application's main window is shown in Figure 1. Three levels of biota dose evaluation are available. At Level 1, conservative assumptions are made through provision of a general screening process employing DOE's biota concentration guides (BCGs), and few inputs are required. At Level 3, fewer assumptions are made a priori but more site- or receptor-specific input data are required, and greater user flexibility is offered. Analysis can start at the general screening level and proceed to Level 3 if the assumptions in Level 1 are too conservative or if screening values are exceeded.

There is also an option for choosing which type of ecological system to assess: terrestrial or aquatic. However, the layout of the form of an aquatic assessment is almost identical to that of a terrestrial assessment. The only differences in the forms are the default organisms considered and the ability within to specify a distribution coefficient if only one of the two media concentrations is known.

After a contaminant (radionuclide) is added to the *Contaminants List Box*, its characteristics are shown in the corresponding boxes. Once selected, the characteristics of the contaminant can be modified. The dose rate limits or guidelines applied for each organism also can be modified. Once the code is run, a *results window* will pop up. Users then can view the results by clicking on the "BCG", "Dose", or

"Sensitivity" tabs. Figure 2 is an example of the BCG results screen. The results can also be shown in graphics (bar charts). Figure 3 is a bar chart of calculated doses.

The layout of the Level 2 form is the same as in Level 1. However, data entry is more extensive in Level 2 than in Level 1. For Level 3, users can enter allometric parameters for riparian or terrestrial animal organism types that contribute to the estimation of internal dose. The allometric parameters include weight, ratio of active to basal metabolic rate, fraction of energy ingested that is assimilated and oxidized, caloric value of food, fraction of soil in diet, and airborne dust loading. Figure 4 is one of the input screen for allometric parameters.

One of the new features in this version of the RESRAD-BIOTA code is an organism wizard that guides the users through the addition of user-defined, specific organisms for dose evaluation. Figure 5 is one of the screens of this new wizard.

Another new feature is the sensitivity analysis capability. Users can study the sensitivity of input parameters by selecting a parameter and assigning a factor for the code to run. Figure 6 is the window for sensitivity analysis. The program will multiply and divide the base parameter value by the factor, run the code three times, and calculate the sensitivity of the dose to the parameter value. There is no limit on the number of parameters that a user can select for sensitivity analysis. This feature is especially useful at Level 3 when data collection may be required. The sensitivity analysis results can be used to prioritize data collection efforts.

3. FUTURE PLAN

Coordination and partnerships with U.S. agencies through the interagency ECORAD Work Group will be continued; coordination and partnerships with international organizations are desired. At an April 2003 meeting of the ECORAD Work Group held at Argonne National Laboratory, a consensus-based work scope to meet the anticipated analysis needs of each agency was agreed upon by the participating agencies. The work scope provides for the inclusion of additional evaluation approaches and capabilities, such as (1) the development of BCGs for an additional 20 radionuclides, (2) additional flexibility for importing and exporting organism specification through an improved organism editor, (3) the compilation of environmental transfer factor parameter data sets for 20 additional radionuclides, (4) the development of additional dose conversion factors for eight ellipsoids selected as a reference organism for 20 radionuclides, (5) a capability that allows users to input tissue concentrations for Level 3 analysis, and (6) the development of a food diet model for Level 3. A user's manual will be prepared and a pilot training course will be conducted by the end of 2003.

4. CONCLUSION

The RESRAD-BIOTA code provides a cost-effective and flexible tool for conducting biota dose evaluations that could be applied within an international framework for protecting the environment from radiation. Once finalized, it will support a variety of environmental assessment needs, including: (1) demonstrating compliance of routine facility and site operations with available dose limits or "dose rate guidelines" for biota; (2) conducting ecological screening assessments of radiological impact at contaminated sites; (3) estimating doses to biota in an environmental impact statement, when coupled with predictive dispersion codes that model a facility's effluents prior to construction; and (4) predicting future doses to biota, when coupled with pathway codes as part of assessing the decommissioning of facilities.

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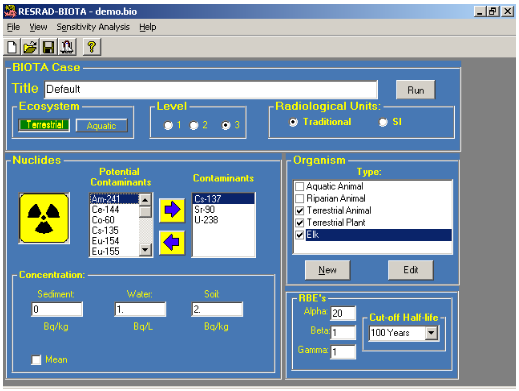

FIG. 1. RESRAD-BIOTA main window.

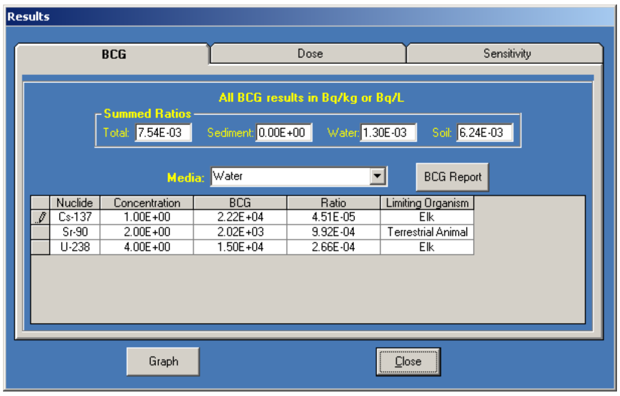

FIG. 2 BCG results.

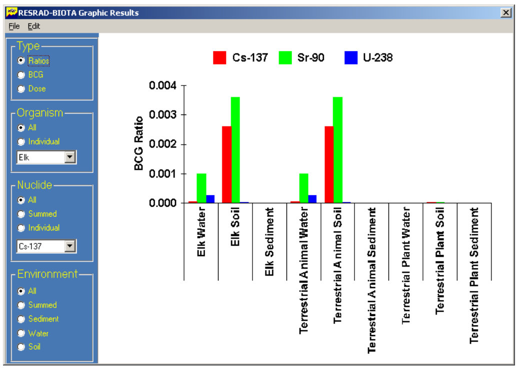

FIG. 3. Graphic results.

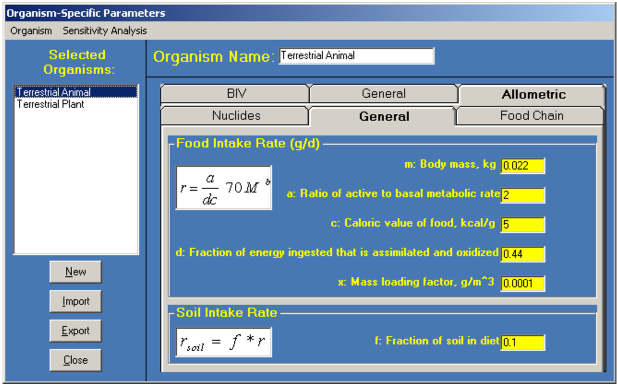

FIG. 4. Allometric parameter input screen.

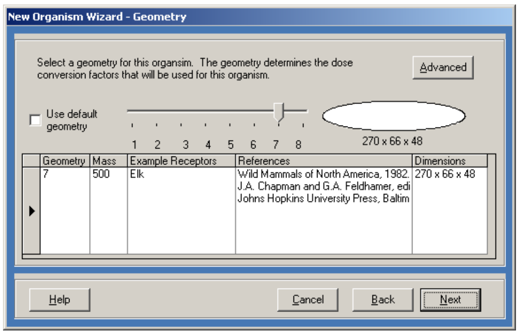

FIG. 5. New organism wizard.

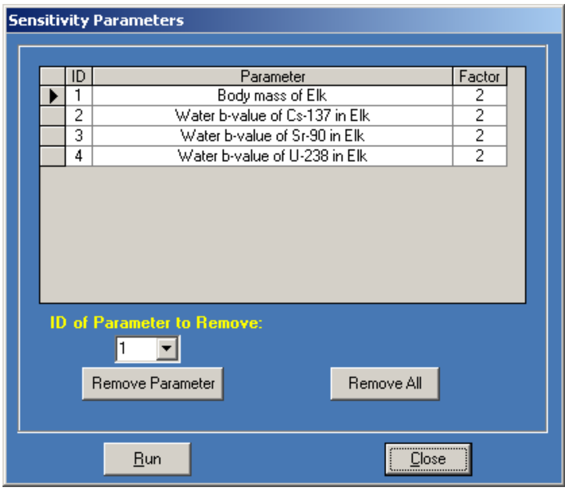

FIG. 6. Sensitivity analysis screen.

4. DEVELOPMENT OF AN INTERNATIONAL ASSESSMENT FRAMEWORK FOR INTERNATIONAL RADIATION PROTECTION

4.1. ETHICS, PRINCIPLES AND ENDPOINTS

Wildlife chronic exposure to environmentally relevant radionuclide concentrations
Experimental results are needed to compensate for the current lack of knowledge

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Abstract. The debate on the need for a system of radiological protection of the environment drives regulators to urge scientists on conceptualisation of methods demonstrating explicitly that the environment is protected against radioactive contaminants. As regards the FASSET database on radiation effects to non-human biota, one of the major difficulty in the implementation of ecological risk assessment for radioactive pollutants is the lack of data for chronic low level exposure. A general way to deal with situations for which there are no relevant data as regards the actual situation where the risk is to be estimated is to use safety factors. The higher their values, the higher the uncertainty on the risk estimate, but, on the other hand, any safety factor could be reduced as more data become available. Going back to radionuclides for which data are sparse, both concerning fate and effects in ecosystems, derivation of ecologically relevant and scientifically defensible benchmarks become a critical issue in ERA. The scope of this paper is to illustrate the relevance of the development of a greater depth of understanding of radionuclide fate and biological effects at several hierarchical levels to support quantitative risk assessments with defined and acceptable uncertainty bounds. The examination of the Fasset Radiation Effects Database content draw the conclusion that gaps on wildlife chronic internal exposure to α or β emitters are among the most critical for ERA. The following crucial issues are discussed and exemplified for uranium in this paper : (i) Radionuclide bioavailability is key to an accurate assessment of both exposure and effect : media quality criteria are needed; (ii) Chronicity of exposure is key to dose estimate and induced effects; (iii) Considering different scales for biological effects (from early to delayed, from subcellular to high organisation level) is crucial to evidence ecologically relevant indicators.

1. CONTEXT AND OVERALL SCOPE

Currently, international debates concerning the necessity of a system of environmental radiological protection drives regulatory authorities to urge scientists to develop methods demonstrating explicitly that the environment is protected against radioactive contaminants. Important concerns regarding the preservation of resources, habitats and genetic and biological diversity have evidently been raised in the context of this debate, however these issues do not point to any particular stressors besides general pollution. The recent FASSET project, funded by the European Commission, has provided a framework for environmental impact assessments for this category of pollutants, with, among others, an extensive database concerning the radiation effects on non-human biota [3]. As regards chemicals, the Ecological Risk Assessment approach constitutes the traditional method used to help demonstrate the foresight of an appropriate level of ecosystem protection [4]. ERA is a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors. According to this method, in complementarity with health risk examinations, any risk assessment to biota following exposure to radionuclides is to be associated with (1) different source-terms and environnemental release scenarios, (2) exposure pathways and potential biological effects at different organisation levels, (3) estimates of no-effects values, and finally, (4) risk calculations (based on the ratio between predicted concentrations at the exposure source and estimated no-effect concentrations). One difficulty in implementing an Ecological Risk Assessment for radioactive pollutants is the lack of data regarding chronic low-level exposures. Indeed, most studies on the effects of radionuclides on non-human organisms have focused on acute exposures to high-doses of radionuclides (mainly external γ irradiation). A general approach used to handle situations for which no data exists (or at least no relevant data concerning the current situation in which the risk is to be estimated), involves the use of assessment/safety factors, which can subsequently be used to propose a method to manage the unknown exposure levels and effects. The use of these factors has been proposed by the European Commision in its Technical Guidance Document [5], to accommodate the uncertainty associated with existing and new chemical risk assessment data and to reduce the probability of causing deleterious effects on the ecosystem. An expected ecologically meaningful acceptable level (known as the Predicted no-effect value) is derived from ecotoxicological data from the so-called "base-set" organisms (*e.g.* for freshwater organisms: primary producers *–algae*; primary consumers *–daphnids;* predators *–fishes).* The limitation of this method lies in the requirement to make allowances for the various extrapolation issues from the laboratory base-sets to the actual ecosystems (*i.e.* acute to chronic exposure conditions, single species to multiple species systems; laboratory to field). The extrapolation method involves applying to the lowest observed effect values $L(E)C_{x\%}$ from the base-set, (a highly arbitrary, conservative and protective factor ranging from 1 to 1000 as a function of the quality and relevance of existing data on effects). Obviously, this pragmatic method represents a precautionary approach to environmental management mainly applied during the screening stage. The higher the safety factors, the higher the uncertainty associated with the risk estimate, and flexibility of the method recommended for their determination is crucial as any safety factor can be reduced as more data become available. Relative to radioactive substances, for which little data is available concerning their fate and effects on ecosystems, the determination of ecologically relevant and scientifically defendable benchmarks becomes critical during the implementation of any ecological risk assessment method. Within this overall framework, the scope of this extended abstract is to propose and illustrate the importance of a research plan to (1) scientifically support the previously cited extrapolation issues, (2) develop a greater depth of understanding of the biological effects of radionuclides at several hierarchical levels, (3) fit with proper interpretation of biomonitoring data, and finally (4) support quantitative risk assessments with defined and acceptable uncertainty bounds. In other words, at the present time, "we know what we do not know" and we are aware of the extent to which these knowledge gaps will be critical during the development of a robust ERA method for radionuclides.

2. INFORMATION GAPS OF THE CHRONIC INTERNAL EXPOSURES OF WILDLIFE TO α OR β EMITTERS ARE AMONG THE MOST CRITICAL FOR ERA

2.1. Background

Based on an examination of the Fasset Radiation Effects Database [6], the crucial information gaps notably concern the chronic exposures of some taxonomic groups (regardless of the irradiation pathway), and internal contaminations by α and/or β emitters (regardless of the wildlife group). Concerning this latter, situations of chronic low level exposures are likely to cause toxic responses distinct from those observed after acute exposures to high doses, due to the bioaccumulation phenomena. Biochemical mechanisms can lead to the gradual accumulation of elements present in trace amounts in the external medium, thereby inducing highly localised depositions within tissues or cells. These highly localised radionuclide accumulations, coupling radiological and chemical toxicities, (particularly in the case of "heavy" elements such as actinides), may give rise to particular biological responses in cell groups, and possibly resulting functional or structural abnormalities. Assessment of these bioaccumulation phenomena investigated within the ENVIRHOM programme [7], is primordial with regard to internal exposure to radionuclides since, via this phenomena, both the radionuclide concentration and the biological effect of the delivered dose can be increased. Knowledge gaps in this domain constitute a strong limitation to our capability to make a reasonable risk estimate. Internal doses cannot be accurately calculated and potentially associated biological effects at any organisation level remain, for the most part, unknown.

Throughout the following paragraphs, only examples related to uranium on the element behaviour (whatever the considered isotope) are given. Experiments committed to establishing the dose (in terms of delivered energy)-effect relationship are in progress with several U isotopes such as U-233 to enhance the delivered dose and to contribute to the understanding of the interaction between chemotoxicity and radiotoxicity. In any case, the issues discussed hereafter apply to all radionuclides.

2.2. Radionuclide bioavailability is key to an accurate assessment of both exposure and effect: media quality criteria are needed

The biogeochemical behaviour of a radionuclide depends on the chemical properties of its corresponding element (redox states, ability to form complexes with inorganic and/or organic ligands…). It is presently well established that the knowledge of the distribution of the various physico-chemical forms (speciation) of an element (radioactive or not) is needed to comprehend both the mobility (transport) and biological reactivity (transfer and effect). It is commonly accepted (with, however, a few exceptions!) that the bioavailability and toxicity of dissolved metals are closely related to the metal's chemical speciation in solution. Metal uptake, nutrition and toxicity normally vary as a function of the concentration of the free-metal ion in solution (Free-Ion Model; FIM), thus metal complexation generally leads to its decreased bioavailability. The prevailing paradigm regarding a metal's uptake by aquatic organisms, *i.e.* the Free Ion Model or its derivative the Biotic Ligand Model, assumes that metals enter living cells via facilitated cation transport. Antagonism with other cations is implicit. The following example involving uranium and freshwater phytoplankton illustrates the importance of considering a radioelement's geochemical behaviour (or medium quality criteria) to better predict its bioavailability to organisms. Regardless of the radioactive isotopes considered, H^+ / UO_2^{2+} antagonism explain the large discrepancies observed under different pH exposure conditions, and the FIM has been successfully applied to describe the uranium – algae interactions in simple well-defined exposure media. Cellular uptake as a function of U concentration follows a Michaelis-Menten saturation processes with a somewhat similar half-saturation but clearly contrasting maximum uptake rates. Uranium uptake was seen to markedly increase (up to ~4X) with an increase in pH (5 → 7) in spite of the substantial decrease (55 → 0.02%) in the proportion of calculated free uranyl ion concentration in solution over this pH range [8]. Overall, these marked differences in uptake rates will lead on consequences in terms of delivered dose and, obviously, observed effects, thereby underlining the need for media quality criteria. Bioaccumulation studies performed on a bivalve (*Corbicula fluminea*) also revealed a significant pH effect on the bioaccumulation rate (x14 when pH 8.1 → 7) [9].

2.3. Chronicity of exposure is key to an accurate assessment of both dose calculation and effect: exposure conditions (concentration and duration) strongly modify the internal radionuclide distribution at various biological scales

For a given living organism, the chronicity of any exposure to a pollutant obviously leads to different biokinetics, but also to different toxicity mechanims than in the case of an acute exposure. For example, a set of experiments were conducted using an invertebrate model (the bivalve *C. fluminea*), to compare the bioaccumulation rates and tissue distributions after a short-duration exposure combined with a high contamination level, with a semi-chronic exposure to a low concentration. The marked difference in the U distribution in the organs was observed to be function of the exposure level and duration. Gills were favored in the case of high exposure levels (this organ contributes to 40% of the total internalised U quantity when bivalves were exposed to 6.3 μM during 7 days), whereas the visceral mass presented higher accumulation levels for environmentally-relevant U concentration levels (85% of the total internalised U quantity when bivalves were exposed to 0.4 μM during 42 days). These results suggest that the primary toxicity of uranium should potentially take place in the gills for acute exposure and in the digestive gland for chronic exposure. Obviously, acute to chronic extrapolations lead to large uncertainties in the exposure assessment and delivered dose to the target organs. As regards chemicals, acute-to-chronic ratios used for ERA vary over a wide range (from 1 to 20000) as a function of the species and nature of the chemical; and many are inferior to 50 [4, 5].

2.4. Considering different scales for biological effects (from early to delayed and from subcellular to high level of organisation) is crucial to evidence ecologically relevant indicators

When pollutant exposure level increases (dose and duration), organisms counteract this stress with a wide range of physiological responses in the dose-effect continuum, from exposure to resultant disease. Effects at higher hierarchical level are always preceded by early changes in biological processes, (from subtle biochemical disturbances to impaired physiological functions involving behaviour, growth, and reproduction), increased susceptibility to any additional stresses, and reduced life span. For example, laboratory experiments were performed to analyse the first valve closure response of *C. fluminea*, exposed to uranium during a 5-hours period and equipped with a non invasive method of valve recording. The minimal sensitivity threshold, expressed as the aqueous uranium concentration inducing valve closure in fifty percent of the bivalves, was evaluated at 0.05 µM of total uranium at pH 5.5 after 300 minutes of exposure [9]. For the same organism, exposure to uranium at 0.25 µM resulted in a significant decrease in ventilation rate (threefold decrease) relative to the reference condition. For unicellular algae, growth inhibition experiments (pH=5, EC_{50}=340 nM) showed that toxicity occurred at a critical concentration of 10^{-6} nmol/cell of internalised U, and then was proportional to internalised uranium by the cells. Most of these data could be applied to population dynamics and used initially to bridge the gap between the (sub)individual and ecosystem levels, simplified life-history models may be used to make ecological relevant links between test results on various effects endpoints and their implications for population dynamics (density, survival, recruitment, time for reproduction, capacity to produce offsprings).

3. CONCLUSION AND PERSPECTIVES

The existing knowledge gaps have been identified and scientists can fill these gaps through further testing and investigations whose quality and quantity leads to lower uncertainties with the magnitudes of the current uncertainties associated with these gaps. The previously cited unresolved risk assessment issues are to be investigated to propose in a rational and transparent process, scientifically defendable environmental criteria for radionuclides based on ecological relevant endpoints. Based on the discussion and illustration presented herein, the determination of the no-effect dose or dose rate is of primary importance and should be linked to the trend to integrate the behaviour of pollutants (bioavailability, bioaccumulation, biotransformation) to develop an understanding of this domain. Besides introducing media quality criteria, another challenge would be to link the effects observed at the infra or/and individual scale with the ecosystem population dynamics using on the ecological modelling approach. The set of experimental data needed will also contribute to answering questions regarding how and to what extent radionuclides and other stressors may affect different organisms and therefore the community structure, distinguishing direct (toxicity) or indirect (food-chain) effects.

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Assessment of the human impact on the abiotic environment
Indicators for a sustainable development

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Abstract. Environmental protection against the dangers arising from ionizing radiation, radioactive materials and other harmful substances is more than to avoid acute dangers or risks for humans or for non-human living organisms. To allow for a sustainable development the abiotic part of the environment must not be neglected in concepts of environmental protection. To this end, indicators are needed for the assessment of the human impact on the abiotic environment which allows to compare different human actions with respect to sustainability and to choose appropriate measures in the competition for a sustainable development. In this paper, the needs for an assessment of the abiotic environment are exemplified and some considerations for suitable indicators are presented.

1. INTRODUCTION

ICRP has presented a draft for discussion on the Protection of Non-Human Species from Ionizing Radiation [1] which shall represent a conceptual framework of radiological protection of the environment. ICRP defines the term environment as composed of biotic and abiotic components that together form a system. But, then the goal of the draft is limited to the living environment and it is stated, on the one hand: *"Abiotic components are considered in terms of their impact on biota, but the ethical issue raised by the mere presence of radionuclides in abiotic components, independent of possible effects, is beyond the scope of this report."* On the other hand, it is also stated that *"...any framework for environmental protection that is developed for radiation therefore needs to acknowledge and accommodate the principles outlined above, and needs to be compatible with other environmental protection approaches that will be in place for non-radiological emission ...".*

These two statements are in severe contradiction and the first one violates today's principles of environmental management and assessment of environmental risks. The ICRP draft is not sufficient to represent a conceptual framework of radiological protection of the environment. The limitation to the living environment is an approach that gives just an *a posteriori* justification of the earlier statements of ICRP 60 [2] that the environment is protected when man is protected. The ICRP recognizes the concepts discussed and partially already used in other fields of environmental protection such as sustainability, biological resources, and biodiversity, but the ICRP approach is not compatible with these concepts. In order to comply with a sustainable development one has to develop a concept for the protection of the environment against the dangers of ionizing radiation and radioactive substances which considers the environment as an entity and which includes aside of the living environment also the abiotic environment, i.e. the compartments atmosphere, hydrosphere, and geo-, or at least, pedosphere. Such a system of environmental protection against the dangers of ionizing radiation and radioactive substances should generally be discussed in the context of protection against other harmful substances as e.g. non-radioactive chemicals or other pollutants. This is necessary to allow comparisons with respect to the choices of policies in the context of sustainable development.

2. SUSTAINABLE DEVELOPMENT AND ENVIRONMENTAL IMPACT

At the conference of the United Nations on the environment and development in Rio de Janeiro in 1992, a framework was set for the elaboration of concepts for the establishment of a sustainable development with the Agenda 21 [3]. In chapter 40 of the Agenda 21 the development and application of measurable quantities and assessment criteria is demanded by which the national and international developments can be assessed with the goal to comply with a sustainable development. The United

Nations established a Commission on Sustainable Development which developed a set of 134 indicators for sustainability; see e.g. [4]. This set of indicators was further elaborated and assessed with respect to their applicability until the 2nd world summit in Johannesburg 2002 by 22 testing countries. In particular, indicators for environmental development are of interest which allow assessing the tolerance of the environment against harmful substances and the vulnerability of the environmental items to be protected. That means quantities have to be defined which characterize sustainability.

Mere protection concepts to avoid acute dangers are no longer sufficient to satisfy the needs for the actual international discussions in environmental protection. The concepts also have to be applicable to the demands of sustainability. With respect to the status of the abiotic environment the radiological aspects should be assessed in close connection with the status of other non-radioactive harmful substances and should allow for a generalization with the goal of a consistent concept of environmental protection.

Examples for human environmental impact which – at least at the time being – are not directly of radiological relevance but rather represent adverse effects by changing the environmental radioactivity are the emission of Kr-85 in the atmosphere and the release of long-lived radionuclides from human practices, such as H-3, C-14, Cl-36, Tc-99, I-129, Cs-135, and some long-lived actinides, into atmosphere or hydrosphere and their subsequent local or global dispersion. For these cases, internationally accepted measurable quantities and criteria are still missing.

It is evident that the development of internationally acceptable goals, criteria and finally constraints or limits based on doses or dose rates for the living environment and on activity concentrations for the abiotic environment is extremely difficult. It will take a long time to succeed. In spite of these difficulties one should at least try to develop indicators for the quantification of the radiological status of the abiotic environment. ICRP should consider taking the lead in future-oriented discussion of a comprehensive environmental protection.

3. INDICATORS OF A SUSTAINABLE DEVELOPMENT

The following indicators have been proposed: The degree by which radiation exposures are in compliance with dose limits, the average exposure to Rn-222 in homes, and the activities of long-lived radionuclides in air, water, and soil. However, these indicators do not cover all relevant endpoints. Rn-222 exposure in homes addresses an aspect of human well-being and is not a relevant indicator for the abiotic environment. The activity concentrations in air, water, and soil are more generally suitable for a quantification of the state of the environment though they are mostly used in terms of predicting the exposure of living organisms with respect to potential hazards or health risks.

To assess the impact on the environment of such substance, their amounts released to the environment (soil, water and air as compartments to be protected) appear to be better measures than the activity concentrations $C(t)$ in a compartment. However, the absolute releases do not take into account the dispersion in the environmental compartments and the dynamics of these departments. Therefore, better indicators of sustainability are the changes of the concentrations or specific activities of harmful substances in the relevant compartments weighed by the probability that they are removed from the respective environmental compartment. Consider to this end a simple single compartment model in which a substance is increasing the concentration at a rate R_{in} by releases into the compartment and in which the substance disappears with a probability λ. Then the concentration $C(t)$ can be described by:

$$\frac{\mathrm{d}C(t)}{\mathrm{d}t} = R_{\text{in}} - \lambda \times C(t) \tag{1}$$

with λ being the effective decay constant in this compartment. The effective decay constant λ is describing two effects. First, the physical (or chemical) decay of the substance and, second, the ecological decay, i.e. the probability for the substance to be removed from the compartment by environmental transport processes.

$$\lambda = \lambda_{\text{phys}} + \lambda_{\text{ecol}} \tag{2}$$

The physical and ecological decay probabilities are measures of the physical mean lifetime $1/\lambda_{phys}$ and of the ecological mean lifetime $1/\lambda_{ecol}$ in the respective compartment.

It has to be noted that transport means here the transfer into another environmental compartment which either can be a real sink in which the substance is no longer regarded as harmful or relevant or a compartment in which the consequently changing concentrations have to be further considered, e.g. with respect to an exposure via another exposure path.

It is trivial that the concentration $C(t)$ will decrease exponentially if there is no input (release):

$$C(t) = C(0) \cdot \exp(-\lambda \times t) \tag{3}$$

But it is important to consider that $1/\lambda$ is a measure of the mean time during which the substance is of relevance in the considered compartment. Therefore, an indicator for sustainability could be:

$$I = \frac{1}{\lambda} \times \frac{dC(t)}{dt} \ . \tag{4}$$

It is the rate at which the concentration of a potentially harmful substance $C(t)$ in a compartment is changing multiplied by its mean time of relevance $1/\lambda$. This approach holds for radioactive and stable substances, independent whether activity concentrations or element concentrations are used.

Examples which indicate the importance to consider the dynamics of environmental compartments are the decline of C-14 specific activities in various environmental compartments and the clearance of Cl-36 from the atmosphere after the atmospheric nuclear explosions in the 1960s. Also the environmental impact of Kr-85 releases to the atmosphere and of I-129 releases to the atmosphere and hydrosphere can only be adequately described using dynamic models, which in first order are taken care of by the indicator given in equation 4.

This quantity allows for an individual judgment about the different environmental compartments. Negative values are indicating improvements, positive values point to increases of the environmental burden. Actual exposures of humans or of non-human species can be easily derived in this concept since an exposure E to a harmful substance during a time span between t_0 and t_1 is given by summing up the exposures via the compartments i:

$$E(t_0, t_1) = \sum_i DF_i \cdot \int_{t_0}^{t_1} \frac{dC_i(t)}{dt} dt \tag{5}$$

making use of aggregated dose factors DF_i which describe the exposure in the environmental compartment i per unit concentration of the substance. Likewise, risk assessments can be performed by weighing an indicator I with relative or absolute risk factors for a particular endpoint. The potential indicator according to equation 4 is an example for discussion pointing to the relevance of the dynamics of the system environment.

Independent of their final formulation such indicators are needed as a quantitative basis in the competition for sustainable development. At present, they can be developed without giving already the final answers with respect to constraints or limits. But, they can serve as a basis for a system of minimization and optimization in environmental protection taking into account the precautionary principle. Tasks in which they are needed are e.g. the bookkeeping of emissions and of the input into the oceans within the OSPAR convention [5] in the context of the Sintra Statement [6], the distinction of the input in the hydrosphere of natural radionuclides by natural weathering from that originating from human practices or work activities. They can be used to address the question of sustainability in the context of the clearance of long-lived radioactive materials from practices and of the release of residues containing elevated levels of long-lived natural radionuclides from work activities. Further, they can provide the tools to compare different options of human action and their inferred risks in the final disposure of conventional and radioactive wastes.

4. CONCLUSIONS

ICRP should take an active role in the development of operable indicators for sustainable development. Such indicators should not only consider the radiation exposures of man and of other living species. Also the human impact on the abiotic part of the environment by releases of radioactivity has to be accounted for. As for other environmental pollutants (e.g. heavy metals) the establishment of meaningful indicators in the context of radiation protection is not trivial and should be performed under consideration of the international discussions. Radiological protection should become a consistent part of a general scheme of the protection of the environment. Indicators for sustainability should take into account the dynamics of the abiotic compartments of the environment and should be likewise applicable to radioactive and non-radioactive environmental pollutants.

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Protecting the fast breeders:
Problem formulation and effects analysis

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Abstract. Recent debates on protection of the environment from ionising radiation have reached reasonable agreement over the ethical and philosophical basis of environmental protection and a recognition that a practical system of protection will need to support (at a minimum) the principles of sustainable development, biodiversity, and conservation. However, there is still some controversy over the use of dose assessment tools within risk evaluation and management. The paper uses the case of the Dounreay "radioactive rabbits" to discuss the advantages and limitations of proposed systems, focusing primarily on the interaction between ecological risk assessment (ERA) and the reference flora and fauna approach. It concludes that the reference approach is a valuable tool for the analysis of environmental effects, but that there is a problem if it becomes the driving force of the protection framework. In particular, there is a need for a clearer focus on non-technical issues within the problem formulation stage of ERA, particularly the social, ethical, political and economic issues, and there should be a strong commitment to stakeholder involvement at this stage. The problem formulation stage should identify the relevant assessment tools; the assessment tool should dictate neither the problem formulation nor the risk management.

1. INTRODUCTION

In June 2003, a Scottish Environment Protection Agency (SEPA) inspector on a routine visit to the Dounreay site noticed a rabbit in the vicinity of the low-level waste-pits. Subsequent checks revealed burrows and rabbit droppings in the waste-pit area, leading to SEPA serving an enforcement order requiring the UK Atomic Energy Authority (UKAEA) to take action to prevent animals and birds entering the waste-pit area amid fears that they could spread radioactivity. The news lead to the Food Standards Agency (FSA) being called in to test the rabbits and ensure that contaminated meat was not entering the food chain. Dounreay took measures to "secure the area against further intrusion", and announced that "We are also arranging a cull of rabbits on the site and reviewing what measures may be taken to curb other wildlife". Local activists responded with: "Rabbits now, who's next?" [1-3]

In the light of the recent interest in radiological protection of the environment, one might question the logic of developing a system to provide protection of non-human species from the effects of ionising radiation if the practical solution is to shoot those affected. On another level, the case illustrates the inadequacy of the position that "if man is sufficiently protected then so is the environment", as well as the enormous challenge in developing a framework that might be capable of a providing a rational input to environmental protection policy. The following paper uses the above case to give a short evaluation of the practical and ethical issues that any system will have to address, highlighting some of the advantages, disadvantages and potential pitfalls of some proposed approaches.

2. PROPOSED FRAMEWORKS AND SYSTEMS

Over the past five years there has been a growing international consensus that radiation protection needs to address environmental effects. There is also a reasonable agreement over the ethical and philosophical basis of environmental protection, an awareness of the limitations in the outdated anthropocentric approach, and a recognition that a practical system of protection will need to corroborate (at a minimum) the principles of sustainable development, maintenance of biodiversity, and conservation of species and habitat [4-6]. Most proposals acknowledge that some analysis of radiation doses and effects on the environment will have to be undertaken, however there is still disagreement (and controversy) as to the role and applicability of this assessment to risk evaluation and management [7,8]. In particular, it is not clear how the "reference flora and fauna approach" proposed by ICRP might interface with the Ecological Risk Assessment (ERA) and the setting and compliance with standards and/or criteria.

3. CHALLENGES

3.1. Problem formulation

In the Ecological Risk Assessment (ERA) paradigm, problem formulation is defined as the first step of any risk assessment and is intended to identify the context and purpose of the assessment framework [9]. This includes the process of choosing appropriate assessment endpoints, identifying sources and describing the environment, as well as ecological and political issues related to the questions being addressed [10]. There are a number of "problems" one might formulate connected to Dounreay, which was built in the 1950s as the nuclear research centre of Britain's programme to develop a fast breeder reactor. Today the programme has been abandoned; the last of the three breeder reactors was closed in 1994, reprocessing halted in 1996, and in 2001 the government decided to decommission the complex at an estimated cost of £4 billion and time of 60 yrs. Hence, future environmental protection questions could be expected to relate largely to decommissioning options. The pits invaded by the rabbits were used between the 1950s and 1990s and contain low-level waste stored in drums, encased in concrete, covered with one metre of soil and turfed over, then surrounded by a 6ft high, barb-wire topped fence. It seems safe to say that future policy will now have to address the question of protecting the site from rabbits. But what about protecting rabbits from the site? Should the future "green site" aim to remove the wildlife or the humans?

One might argue that considering the rabbits only as potential spreaders of radionuclides and a source of radioactivity to human beings exemplifies the anthropocentric attitude to radiation protection that we are trying to avoid. However, the case illustrates some pertinent differences between the two main non-anthropocentric alternatives: the ecocentric and biocentric approach. An ecocentrist, prioritising consequences for the whole ecosystem, might argue that since rabbits are an acknowledged environmental pest in themselves (remember Australia) radiological effects on individuals or whole populations would not carry much weight in an evaluation. Indeed, getting rid of the rabbits would protect both plants and other higher predators such as buzzards, providing there was a sufficient source of food elsewhere. A biocentrist, extending moral standing to individual organisms, could be against harming any animal to satisfy non-essential human needs, be it culling or causing detrimental effects from radiation. The two would, however, find more agreement over protection of endangered species. They may both find a cull unacceptable had it been badgers burrowing in the pits.

The tension between different ethical standpoints is well recognised in environmental philosophy [11], and the eco/biocentric example above is just one case of how value judgements can influence practical environmental protection. Since the differing opinions reflect disagreements about ethics rather than facts, a robust and justifiable system should ensure that value judgements are transparent. The problem formulation phase of ERA needs to be expanded to accommodate a more systematic evaluation of ethical, social, economic and political aspects. In addition to technical issues, one might expect a consideration of the relevance of the principles of biodiversity and species and habitat conservation, the available alternatives as well as consideration of other protection principles such as BAT and the precautionary principle. Ecological knowledge would be needed on the potential impact on all wildlife and plants, including questions related to endangered species and the habitat itself. Such an evaluation would require a commitment to interaction between risk assessors, risk managers and stakeholders.

3.2. Assessment and management of effects

The reference flora and fauna approach first proposed by Pentreath [12], and supported by IUR [5] and ICRP [6] has similarities with other assessment systems such as the screening or tiered approach [13]. They share many of the same assumptions, dosimetry and transfer models, and reference entities. The main differences are arguably how the systems are used in management rather than assessment. Two areas have attracted attention in discussion of the approaches, both of which have been used as examples of the conflict between "top-down" and "bottom-up" paradigms in ecotoxicology. However, those paradigms differ somewhat depending on whether one is talking about the analysis of effects or the evaluation and management of environmental risk.

The reference flora and fauna system has been identified with a "bottom-up" or reductionist method of dose-response analysis [14]. The approach focuses explicitly on effects (or various biological endpoints) observable in individuals, and extrapolates potential effects on populations (and ecosystems) from that data. The alternative "top-down" method focuses first on the complex interactions within the ecosystem and attempts to gather knowledge on endpoints relevant to the overall functioning of the ecosystem—e.g., stability, resilience, biodiversity [5]. In support of the bottom-up approach, one might point out that the individual is often the highest level at which scientific experiment and hypothesis testing can be directed. Observed biological or physiological effects on an individual organism (or its cells, DNA, etc) may be reduced causally to the radiation exposure; subsequent effects on a population or ecosystem level require more complicated ecological modelling. Furthermore, knowledge on individual effects might be necessary from the point of protection, as in the case of protected species. As an exercise in systematically gathering the knowledge available on radiation exposures and effects in non-human species, the reference flora and fauna approach has practical advantages over some of the more simplistic screening systems. However, any assessment framework needs to acknowledge that actions can have a variety of effects on the environment (from DNA to ecosystems), and consider a range of biological endpoints, changes and causes. Both the "top-down and "bottom-up" approaches will be necessary for a comprehensive evaluation of environmental effects; neither will be sufficient alone.

A distinction between "top-down" and "bottom-up" is also made in a risk management context. Here the issue is not so much how effects' analysis is performed, but how the evaluation and management of the risks is carried out. In risk management, a "bottom-up" approach asks what the possible environmental effects of our actions might be: "What harm might we cause and how might we avoid doing it?" On the other hand, a "top down" system depends on predetermined constraints, standards and compliance, usually derived from "no observed affect (NOEL)" or "critical load" criteria. In this case the question often asked by risk management is: "How much can we release?" The "top-down" system presupposes that there is sufficient knowledge to set dose limits. The reference flora and fauna system (if used correctly) has the potential to accommodate a more cautious input to decision-making, and one that may encourage a more transparent debate on the acceptability of the consequences of actions. A simple statement that the "releases are below dose limits" tends to beg the question of where those limits came from and whether or not they are correct.

Perhaps a better focus on the interaction between problem formulation and assessment options would help to illuminate the advantages and limitations of the various systems, for example, to underline how the required knowledge on effects will depend on the context and alternatives being considered. If we wanted to compare the *radiobiological* effect of effluent treatment options, it may be sufficient to consider only concentrations of radionuclides in biota and/or abiota. On the other hand, comparing the *full ecological environmental impact* of alternative actions (e.g., to address substitution and BAT) would need some method of assessing doses and responses for radionuclides, chemicals and other environmental stressors, as well as ecological knowledge on ecosystem interactions. Regulations to ensure compliance may by sufficiently served by dose limit derived concentrations. However, monitoring may need to include a measurement of more than concentrations, particularly if one were to test the assumption of no effect by checking that no biological changes were observed.

3.3. Uncertainties and variability

Any tool used for assessment of doses and effects will need to acknowledge and inform on the uncertainties in predictions. These uncertainties have been well documented [5] and include statistical errors associated with all stages of the effect analysis, namely dose estimations, dose-response relationships and severity of effects, as well as source-specific, site-specific, species-specific and individual-specific variability. In addition to assessing a source's impact on the environment, one should consider the way the environment components (e.g., rabbits) influence the transfer of radionuclides to other species. There are of course large uncertainties associated with the prediction of environmental effects from any human activities. Rabbit culls might also alter the genetic make-up of populations. Being based on dose calculations, the reference flora and fauna approach might be accused of adding an additional level of uncertainty to the analysis of effects, particularly when linked to the various endpoints that it is attempting to document. This complexity does have the disadvantage of making life difficult, and there might well be cases where a simple approach is sufficient. But until better scientific evidence can be provided to back up such judgements, oversimplification should be avoided. What is arguably more important is that the system does endeavour to provide relevant information on uncertainties and avoid some of the shortcomings of the human protection system regarding averaging, distribution and variability. This is particularly important if the information is to help in balanced consideration of the applicability of the precautionary principle—a precautionary approach does not automatically lead to a demand for zero release. We might not be so concerned over uncertainties in estimations of doses and effects in rabbits—the consequences of errors may not be irreversible or ecologically damaging, but we may want to exercise more precaution when judging the possible consequences on the red-throated diver. In some cases, concern for the habitat itself (i.e., the abiotic environment) might call for prevention of contamination irrespective of any consequences on living organisms. Again, the level of precaution deemed reasonable will depend on the value judgements made in the problem formulation.

4. CONCLUSIONS

Problem formulation is a critical step in ecological risk assessment and one that deserves more attention in developing a system of protection of the environment from ionising radiation. In addition to consideration of technical issues such as identifying assessment endpoints and the ways in which these can be measured and evaluated, the step should consider social, ethical and political issues related to the questions being addressed. Communication between stakeholders, managers and assessors should take place at the start of the process. The reference flora and fauna approach can provide a systematic approach to the collation and analysis of available knowledge on radiation effects in non-human species. And as a tool to assess such effects, one might argue that it is at least as good as the alternatives, and possibly better suited to addressing the uncertainties associated with the available knowledge. However, there is a danger that the tool will be used to shape both the problem and the solution. The information provided by the reference flora and fauna approach is only one input to an evaluation of consequences, and although a necessary input it should not be taken to be sufficient.

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Cytogenetic effects of low dose ionizing radiation in plants and a problem of the radiological protection of the environment

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Abstract. Differences between the principles of the radiological protection of man and the environment are compared. Cytogenetic effects in plants after an exposure to ionizing radiation at low doses alone and in combination with other factors are discussed.

1. DIFFERENCE BETWEEN PRINCIPLES OF THE RADIOLOGICAL PROTECTION OF MAN AND THE ENVIRONMENT

Based on the long experience, a system of radiation consequences assessments and adverse effects prevention has been developed concerning the population safety. At the same time, a notion of the "radiation protection of the environment" has remained uncertain up to now. There is no uniform opinion so far even concerning a level of biological organization, alterations on which should be admitted essential and demanding an immediate response from man. Nowadays accumulated experience of radioecological investigations [1-5] testifies that a large-scale release of radionuclides in the environment can result in serious consequences for biota, including abnormalities on an ecosystem level. The South Urals (1957) and Chernobyl (1986) accidents are the most dramatic examples.

For the last three decades, a scientific background for radiation protection of the environment has been based on principles developed by the ICRP. The most important of them can be formulated as 'if man is protected then the environment is also adequately protected'. Although standards derived from these principles guarantee the protection for a human, they are not always able to pledge the equal protection for other environmental objects. This approach implicitly assumes that doses absorbed by human are close to doses absorbed by flora and fauna and, hence, if low doses to a population is assured then other organisms will be safe. But this is not necessarily true. Thus, doses to plants and animals from radioactive fallouts per unit of radioactive contamination density can exceed values for humans by a factor of 10-300 [1, 2]. Taking into account a close radiosensitivity of human and some species determining functioning and stability of ecosystems, such a relationship between doses absorbed by human and environmental objects makes clear that a special attention is to be paid to the protection of plants, animals and their communities. Therefore, notwithstanding the merits of the system that has been developed for the radiological protection of man, there is still a need to develop criteria and frameworks to the system for protecting the environment from ionizing radiation.

A hierarchic structural-functional organization of the living matter determines a multi-level response to an external impact. An important problem is a choice of the level of biological organization to be used with the purposes of the radiological regulation of effects on the environment. As a matter of fact, this means a choice of parameters, which modification should be admitted essential and demanding implementation of protective measures. The radiological protection practice most often uses genetic tests as the most sensitive indices of radiation exposure that are simultaneously reliable enough, or abnormalities on a population level as an indicator of serious alterations occurring in an ecosystem. A mode of plants and animals' existence in the environment is a population, therefore, when one switches from the anthropocentric to ecological principles, modifications not only should occur in the indicators applied but also in the object of the indicators application itself. A role of such an object is now occupied not with individuals, as it was in the case of the anthropocentric approach but with systems of the above-organismic levels, that are populations, ecosystems, biogeocenosis.

A range of dose rates at which biota irradiation actually takes place covers five orders of magnitude [3]. For all that, the irradiation with dose rate of up to 10 Gy/year does not lead to any irreversible modifications in environmental plant communities [1-5]. The corresponding doses exceed the maximum permissible dose to the population of 1 mSv/year 10^4 times, that certainly exceeds possible differences in doses obtained by man and biota. Therefore, the principle of the radiation protection of the environment developed by the ICRP - "if man is protected then the environment is also adequately protected" - is not broken in an overwhelming majority of the ecological situations. But at the same time, in radioactively contaminated territories, there were actual cases when clear signs of radiation damage were registered in plant and animal individuals, populations and even communities [2, 3, 5].

On the other hand, being necessary for protecting the biota at the population level, the principle suggested by the ICRP, is not sufficient to rule out possible significant changes within a population, that cause a breaking of relations between individual elements of ecosystems and, at the end, a destruction or radical rearrangement of natural communities. Moreover, the most basic, testable piece of information for which a dose-effect probability factor could be derived [6] applies to an individual organism. In this context, the question is: changes at what level of biological organization are the first to indicate negative effects in the environment? The results described in [4] showed that genetic effects and alterations of genetic structure in wild plants and animals' populations are observed at dose rates that are below the level considered by the IAEA as safe for biota. Nevertheless, it is changes at the molecular-cellular level that give [7] the earliest and confident information about any negative changes in the environment. In addition, the genetic tests reveal the most substantial effects of environmental pollution related to an increasing mutagenic pressure on the biosphere. This appears in an increasing incidence of cancer and hereditary diseases, genetic load in humans, animals, and plants, and a changing genetic and species structure of biocenoses. Therefore, it is the genetic test-systems that should be used for an early diagnosis of the alterations caused by the human industrial activity.

2. CYTOGENETIC EFFECTS OF LOW DOSE IONIZING RADIATION IN PLANTS

Biological effect of low-level radiation has some important features that don't follow from well-known effects of high and moderate doses are:

- nonlinearity of dose response;
- increased efficiency of irradiation at low dose rates;
- synergetic and antagonistic effects of combined action of different nature factors;
- radiation-induced replicative instability of genome;
- phenomenon of radioadaptation.

The analysis of cells reactions observed to low level irradiation showed [8] that the regularities of cytogenetic disturbances yield in this range are characterized by a sound nonlinearity and have an universal character. Figure 1 presents our data on cytogenetic disturbance frequency in *Hordeum vulgare* germs [9]. The piecewise linear model (line 2 in Figure 1) based on our concept of biological effect of low-level ionizing radiation [8] fits the data much better than the linear one. The improvement of the quality of approximation is not reached on account of the model complicating but because it is possible to achieve a mutual conformity between a phenomenon and its mathematical model.

In 1987-1989, an experimental study of the cytogenetic variability in three successive generations of winter rye and wheat, grown at four plots with different levels of radioactive contamination, was carried out within the 10-km ChNPP zone [10]. Figure 2 shows that, in autumn of 1989, aberrant cell frequencies in leaf meristem of winter rye and wheat of the second and third generations significantly exceeded these parameters for the first generations. The distinctions between cytogenetic indices obtained for the second and third generations were insignificant. In autumn of 1989, plants of all three generations were developing in the identical conditions and were exposed to the identical doses so that the most probable explanation of the registered phenomenon is related to a genome destabilization in plants grown from seeds affected by radiation. From these viewpoints, these data may sign a cytogenetic adaptation beginning; that is, the chronic low-dose irradiation appears to be an ecological factor creating preconditions for possible changes in the genetic structure of a population. Possible trends of such modifications were discussed elsewhere [11].

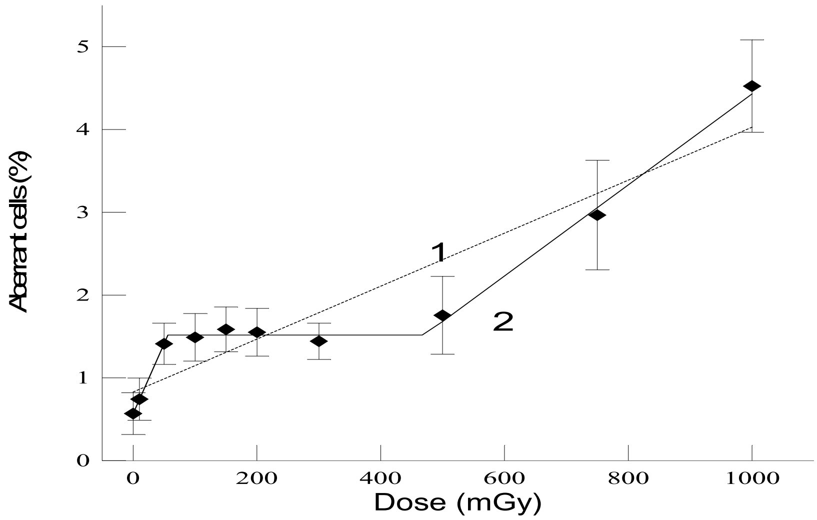

FIG.1. Frequency of aberrant cells in barley germs exposed to low radiation doses and its approximation with linear (1) and piecewise linear (2) models.

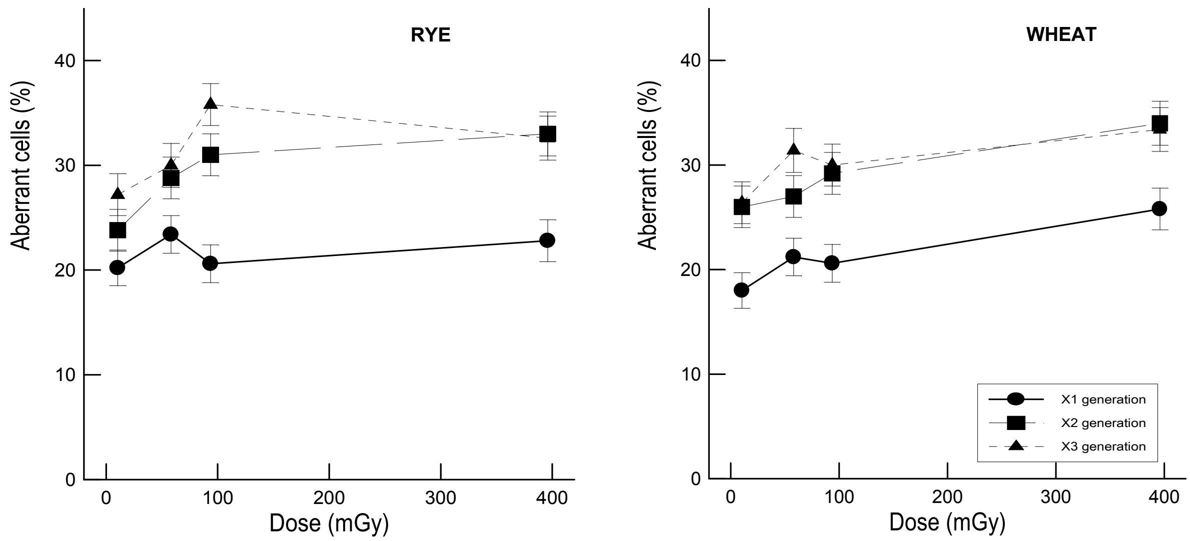

FIG.2. Aberrant cell frequency in intercalar meristem of winter rye and wheat in three generations grown on radioactively contaminated plots.

3. CONCLUSION

The further evolution of investigations in this field should issue in a development of theoretical bases and practical procedures for protecting the environment from ionizing radiation, taking into account the new experimentally confirmed facts about the presence of such essentially important singularities of the biological effect of low ionizing radiation doses as the nonlinearity of a dose-effect relationship, increased efficiency of irradiation at low dose rates, radiation induced replicative instability of genome, phenomenon of radioadaptation, increased probability of synergetic and antagonistic effects of combined action of different nature factors. A development of a new concept of radiation protection for a human and biota should be based on the clear understanding of these effects and their contribution to the response of biological objects to single low-level and combined with other anthropogenic factors radiation exposure.

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Approaches to research of key objects of radiation monitoring of forest ecosystems in ultimate period after nuclear accident

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Abstract. The main approaches to apportionment of 'key compartments' of radiation monitoring have been analyzed for forest ecosystems in ultimate period after nuclear accident. Criteria for apportionment of such 'key compartments' have been elucidated. The example of apportionment of these 'key compartments' has been done on the results of radioecological study in certain forest ecosystem.

1. INTRODUCTION

Behaviour of technogenous radionuclides essentially varies in forest ecosystems in different periods after nuclear accident, therefore radiation monitoring should be divided in time on two parts: 1. upon reaching of quasi-equilibrium in soil-plant cover; 2. after reaching this equilibrium. Elucidated two periods also essentially differs with the raw of important parameters such as test-objects of monitoring – ecosystem compartments, where the maximum of radionuclides concentration is observed which caused dose loading on various biota species; ways of radionuclides intake to biota; parameters of radionuclides migration in food chains; staining of dose effects etc.

2. GENERAL APPROACHES

From the experience of research of consequences of Chernobyl accident in the environment the necessity of apportionment of different bioindicators became obvious for adequate evaluation of radiation effects of different levels of biota organization – from cell to ecosystem – in certain period after the accident. For example, Scotch pine meristematic tissues and needles were the good test-objects in the first acute postaccidental period because of high dose loading on it has been caused by application radiocontamination. But in ultimate period after nuclear accident redistribution of radionuclides can be observed among compartments in forest ecosystem that caused an essential change of dose fields as in ecosystem as a whole and in its' separate compartments as well. It is expediently to point out some special compartments in forest ecosystem, in dependence of its age, species composition and vegetation structure etc., where radionuclides concentration will be in several times more than in the rest ones. Also there will be increased dose loadings on associated with this compartment species of biota (using it as a substrate for growing, as environment or source of food). Such compartments according to P. Strand and C.-M. Larsson [1] we also propose to call 'key compartments' that influences as well as on radionuclides migration in ecosystem and on exposure dose of biota. For example, in ecosystem of 60-years old Scotch pine forest of cenosis Molinio-Pinetum through 15 years after Chernobyl accident especially high ^{137}Cs content was associated with layer of humified forest litter (O_h), that also means – with higher exposure dose of plants whose roots are situated mainly in this layer (Vaccinium myrtillus L., Trientalis europaea L., Melampyrum pratense L. etc.), and also of soil fauna, in particular soil ticks (Oribatei, Mezostigmata).

Another important approach – research of doses increase of internal exposure of organisms in complicated food chains. For example, in dry Scotch pine forest growing on sandy dunes of cenosis Cladonio-Pinetum epigeious lichens form the dense layer of vegetation and are used as a source of food by saprotrophe Oribatei and the last, in their turn, as a food for predator forms of soil ticks. In this case exactly for last ones exposure doses will be the maximum and species, in dependence of their radiosensitivity, will be more or less suitable bioindicators of radiocontamination.

Thus in process of radiobiological and ecotoxicological research in forest ecosystems should clearly divide biological test-objects into two categories:

- species-concentrators of radionuclides in which radionuclides content can exceed an analogues indexes of environment or substrates in many times (fungi, mosses, lichens etc.);
- species-indicators – species of biota in which radionuclides content can be not so high but due to association with highly radioactively contaminated ecosystem compartments and low radioresistance of organisms numerical biological effects can be observed as an answer on increase of dose loading.

Biological effects can reveal on different levels of biota that organisation as a specified radiobioligical effects: in plants – morphoses, oppression of meristematic processes etc., in mammals – breach of homeopoesis, specified cancers etc. In irradiated ecosystem (with high doses) can be observed radiosuccessions (disappearance of less radioresistant species and intensification of dominance of more radioresistant ones), changes in population structure of certain species (quantity of individuals, age structure, fertility and so on), interpopulation interconnections (change of dominance index etc.). But it should be underlined that many from the mentioned above biological effects can be find in some species of biota on statistically trustworthy level even under relative low level of additional irradiation due to artificial sources. From this point of view both concepts – threshold and non-threshold – should be seriously discussed as well as criteria of endangered species and whole ecosystems in conditions of additional technogeneous irradiation.

Also special attention should be paid to investigation of radionuclides redistribution in plants and animals taking into account their peculiarities. For example, in arboreal plants ^{90}Sr accumulation is taking place through the bast to wood and further back stream of this radionuclide doesn't observe from this tissue thus with time exposure dose on it increases from internal exposure. Another example: ^{137}Cs can be find in root system on average in relatively low concentrations but in the meristematic zone its content exceeds the average value of whole root more than 10 times [2, 3]. Thus just these lowest radioresistant root tissue obtains highest exposure doses which can cause radiobiological effects and should be an object of special attention.

3. AN EXAMPLE OF APPORTIONMENT OF 'KEY COMPARTMENTS' IN FOREST ECOSYSTEM

With the purpose of finding of 'key compartments' in forest ecosystem research was carried out in Polessye of Ukraine, in pine forest of cenosis Cladonio-Pinetum in the poorest conditions on dry sandy dunes. Tree canopy consisted of Pinus sylvestris in age of 40 years. Grass–dwarf-shrub layer was rarefied, had total projective cover 10-12%. Its basis was created by Corynephorus canescens (L.) P.Beauv (5-7%), Thymus serpyllum L. (1-3%) and Calluna vulgaris (L.) Hull (3-5%). Lichen layer was represented by two sublayers – epigeious and epiphytic. Sublayer of epigeious lichens was dense, with total projective cover 85-90% and phytomass about 0,2 kg/m^2 of absolute dry weight. Its basis created Cladina arbuscula ssp. mitis (Sandst.) Ruoss (60-65%) and Cl. uncialis (L.) F.Weber ex F.H.Wigg. (10-15%). The layer of macromycetes was represented mainly by Lactarius rufus, Siullus variegatus, Boletus badius and Paxillus involutus. It was necessary to evaluate ^{137}Cs specific activity as well as mass of each compartment on the square unit (1 ha). Obtained results on distribution of ^{137}Cs specific activity and total activity in this forest ecosystem are generalized in Table 1.

Let's analyze the data of Table 1 proceeding from mentioned above proposal of apportionment of 'key compartments' in forest ecosystem. As it can be seen the maximum of ^{137}Cs content was character for mushrooms, after this group in descending order followed the moss layer, lichen layer and thin root of pine. But taking into account the weight characteristics of each from mentioned above compartments in this ecosystem we apportionment such 'key compartments'*:* from biota – pine root (27% from ^{137}Cs total stock in biota) and lichen layer (44% from biota) and from the soil compartment – forest litter (14% from ^{137}Cs total activity in soil) and superficial mineral soil horizon (HEI 0-2 cm – 54% from ^{137}Cs total activity in soil).

TABLE 1. DISTRIBUTION OF ^{137}Cs ACTIVITY IN ECOSYSTEM OF CLADONIO-PINETUM, DENSITY OF GROUND DEPOSITION OF ^{137}Cs – 46 kBq/m^2

Forest ecosystem compartment	Mass, kg/ha (absolutely dry weight)	^{137}Cs specific activity, Bq/kg (d.w.)	^{137}Cs activity, MBq/ha	Part of ^{137}Cs total activity, %
Tree canopy:	23402	418*	9.791	50.56
Wood	8547	126	1.077	5.56
Bark	1155	700	0.809	4.18
Branches	4774	350	1.671	8.63
Annual shoots	150	1210	0.182	0.94
Needles	2079	385	0.800	4.13
Roots thick	6110	730	4.460	23.03
Roots thin	587	1350	0.792	4.09
Juvenile tree layer	5.9	800	0.004	0.026
Undergrowth layer	22	290	0.006	0.031
Grass–dwarf-shrub layer	288	418*	0.120	0.62
Lichen layer:	1995.8	4107*	8.197	43.586
Sublayer of epigeious lichens	1995	4106*	8.192	43.56
Cladina arbuscula ssp. mitis	1400	4180	5.852	30.22
Cladonia uncialis	360	4300	1.548	7.99
Cladonia gracilis	180	4400	0.792	4.09
Cladonia subulata	55	4440	0.244	1.26
Sublayer of epiphytic lichens (Hypogymnia physodes)	0.8	5800	0.005	0.026
Moss layer *(Polytrichum piliferum)*	70.5	6800	0.479	2.48
Layer of Macromycetes (fruitbodies):	4.76	10084*	0.524	2.708
Cortinarius sanguineus	0.84	160000	0.134	0.69
Lactarius rufus	0.96	100000	0.096	0.50
Cantharellus cibarius	0.14	11120	0.002	0.008
Siullus variegatus	0.87	79200	0.07	0.36
Siullus bovinus	0.66	58700	0.04	0.20
Boletus badius	0.41	75700	0.03	0.16
Paxillus involutus	0.88	174000	0.153	0.79
TOTAL	–	–	19.367	100.00
SOIL TOGETHER WITH MYCELIUM OF FUNGI				
Forest litter	16400	4653*	76.310	14.44
Forest litter semi-decomposed	3200	293	0.938	0.18
Forest litter decomposed	13200	5710	75.372	14.26
Mineral soil layers:	4935600	91*	452.274	85.57
HEI 0-2 cm	228000	1250	285.000	53.92
HEI 2-4 cm	290800	330	95.964	18.16
HEI 4-6 cm	288400	110	31.724	6.00
HEI 6-8 cm	338000	35	11.830	2.24
HEI 8-10 cm	289600	20	5.792	1.10
HEI 10-12 cm	317200	14	4.444	0.84
Pi 12-30 cm	3183600	5*	17.48	3.31
including mycelium (in all soil layers)	1200	10030*	132.035	24.98
TOTAL (soil with fungi mycelium)	4012800	132*	528.543	100.00

**Note:* average-weighted data on whole layer.

Apportionment exactly of these ecosystem components as the ‘key compartments’ were caused by application of such criteria as:

— high values of ^{137}Cs content;

— significant part of the retained total radionuclide activity;

— defining influence on forming and distribution of dose loadings inside the ecosystem;

— numerical species of biota associated with these compartments;

— significant influence on intensity of biogeochemical circle of radionuclide in ecosystem.

It’s rightfully to suppose that the most important biological reactions will be observed on additional exposure doses in this ecosystem exactly in species of biota associated with these compartments. Using compartment models of ^{137}Cs migration in forest ecosystems [4, 5] prognosis of redistribution of radionuclide activity in ecosystem can predict what will happen on changing of ‘key compartments’ but also changing of dose loading to separate species of biota. In such case scientific monitoring accents should be moved to the new ‘key compartments’.

After apportionment of ‘key compartments’ their study is carried out by all methods suitable for ecotoxicological research – methodology of radiobiology, radioecology, anatomy, physiology, biochemistry, population ecology etc.

4. CONCLUSION

Thus apportionment of ‘key compartments’ in forest ecosystem helps us to focus the efforts of scientists on the most important problems from the point of view ecotoxicology of non-human biota. Unfortunately it has to ascertain that in spite of significant success in study in the field of ecotoxicology, anthropocentric approach which is character for radiobiological research as a whole, doesn’t allow us to answer many of vital problems the main of which has become the possibility of sustainable development of ecosystems in conditions of increase of irradiation of different ecosystems as well as numerical species of biota; changes of indexes of biodiversity; interconnections of ionizing radiation with another negative anthropogenic factors of non-radioactive origin.

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4.2. ESTABLISHMENT OF STANDARDS AND CRITERIA AND COMPLIANCE ISSUES

Risk based approach to environmental monitoring program requirements for nuclear facilities in Canada

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Abstract. In Canada the promulgation of the Nuclear Safety and Control Act and its associated regulations raised concerns about the Canadian Nuclear Safety Commission's (CNSC) new mandate on environmental protection. Licensees raised the issue that environmental protection requirements designed to regulate high effluent release facilities would create an unfair and onerous burden on low effluent release facilities. The CNSC addressed the issue by adopting a risk management approach which uses site-specific ecological risk assessments (ERA) to define the scope and complexity of the licensee's environmental monitoring program. The ERA classifies a nuclear facility in three risk categories: low, medium or high. Increasing risk spurs an increase in the frequency and complexity of environmental monitoring program requirements. This ensures that costs and efforts of both licensees and the CNSC are commensurate with the risk associated with the facility.

1. INTRODUCTION

Nuclear facilities are regulated by the Canadian Nuclear Safety Commission (CNSC) under the Nuclear Safety and Control Act (the "Act") in Canada. The "Act" replaced the Atomic Energy Control Act, and unlike its predecessor, specifically mandates the protection of the environment in addition to the health and safety of persons. Applicants for a license to operate a Class I nuclear facility (e.g., nuclear power plants, uranium refineries) or a uranium mine and mill are expected to provide information on the effects on the environment and the health and safety of persons that may result from the operation of the facility. Licensees are also required to submit a proposed environmental monitoring program. The environmental monitoring program design is expected to confirm that releases of hazardous and radioactive nuclear substances to the environment do not result in unreasonable risk to the environment or to the health and safety of persons.

Licensees expressed concerns that environmental protection requirements designed to regulate high release facilities would create an unfair and onerous burden on low release facilities. Currently, there is a wide variability among nuclear facilities regarding the quantity and quality of their effluent releases and their subsequent impact on the environment. The CNSC addressed the issue by adopting a risk management approach which uses site-specific ecological risk assessments (ERA) [1] to define the scope and complexity of the environmental monitoring program required by a licensee. This approach allows consistent and comparable risk assessments from exposure to both hazardous and radioactive nuclear substances. As ERAs predict potential effects on valued ecosystem components (VECs) associated with a facility, it is the logical starting point for identifying operational environmental monitoring program requirements. Consequently, the operational environmental monitoring program is designed to confirm that the facilities' environmental performance meet licence conditions and remains within the effects boundaries predicted by the ERA and accepted at the time that the license was issued.

2. LINKAGE BETWEEN THE ERA AND THE ENVIRONMENTAL MONITORING PROGRAM

An ERA satisfies the requirements of the Class I Nuclear Facilities Regulations and the Uranium Mines and Mills Regulations. It serves as a predictive management tool to develop risk based operational environmental monitoring. The ERA identifies the environmental risks associated with the facility by providing the following information:

— identifying the environmental effects predicted for the facility, including the effects which the operational environmental monitoring program assesses;

— documenting predicted contaminant releases and loadings to the environment;
— identifying significant contaminants and exposure pathways for critical groups (human) and VECs (non-human biota);
— identifying sensitive compartments (i.e., air, water, soil, sediment) in significant exposure pathways;
— identifying appropriate sampling media to represent a sensitive compartment and;
— identifying appropriate sampling locations (e.g., near-field, far-field, reference or control) for monitoring sensitive compartments in significant pathways.

The magnitude of the environmental risk translates into a more comprehensive and complex environmental monitoring program. The results of the ERA will then be used to classify the facility into one of three risk categories: low, medium or high.

3. LOW RISK – IDENTIFICATION OF ENVIRONMENTAL MONITORING PROGRAM REQUIREMENTS

Licensed facilities with no releases of hazardous or radioactive nuclear substances into the environment are not required to have an environmental monitoring program. The time and resources required to implement such a program would outweigh its usefulness. An environmental monitoring program is also not required if the licensed activity meets *all* of the following conditions:

— environmental contaminant concentrations are below analytical levels of detection when method detection limits are below known toxic effect levels (with a margin of safety);
— contaminants released by the facility are not environmentally persistent or predicted to bioaccumulate or biomagnify;
— continuous exposure to undiluted effluent would not result in unreasonable risk to human health or biota;
— contaminant activity levels or concentrations are indistinguishable from the background.

The CNSC expects licensees of low risk facilities to demonstrate the adequacy of provisions to protect the environment and the health and safety of persons by other means, i.e, effluent monitoring and environmental modeling.

4. MEDIUM RISK – IDENTIFICATION OF ENVIRONMENTAL MONITORING PROGRAM REQUIREMENTS

Medium risk facilities have releases of hazardous or radioactive nuclear substances to the environment. These facilities do not meet all of the above conditions but are not anticipated to cause impacts on biota at the population or community level. These facilities require an environmental monitoring program which focuses on pathways confirmation monitoring (PCM). A PCM is contaminant monitoring involving the measurement of contaminant levels in abiotic (e.g., sediment, soil, water, air) and biotic (e.g., mammals, aquatic organisms, etc.) components of the receiving environment. The PCM confirms that contaminants released to the receiving environment are behaving as predicted. If the PCM confirms the contaminant transfer and exposure modeling in the ERA (or its conservativeness), then there is continued confidence in the effects predictions of the ERA. The PCM achieves this by:

— assessing levels, detecting changes and evaluating long-term trends in contaminant behaviour in the receiving environment and testing against modeled predictions;
— providing information on environmental contaminant levels and comparing them to ERA predictions;
— increasing the knowledge concerning site specific contaminant behaviour. This improves the basis for future predictions and the estimation of levels that might arise in the event of an emergency;

— verifying environmental transfer models by validation of model parameters and assumptions used in the calculation of the derived release limits and environmental model predictions, and;

— assessing performance against any applicable regulatory limits, standards, guidelines, operating limits (public dose limits, as low as reasonably achievable (ALARA), performance indicators, action levels, or other accepted criteria and objectives on exposures).

The scope (e.g., extent and frequency of monitoring) of the PCM is commensurate with the environmental risk posed by the facility. The PCM for higher risk facilities will collect field samples on a more frequent basis, sample more compartments within a pathway and monitor more pathways leading to a greater number of VECs. If the PCM results indicate that the parameterization of the previous ERA was inadequate, then the ERA may be revised using the accumulated site-specific monitoring data and knowledge. The updated ERA is then used to refine the environmental monitoring program and, if needed, change the facilities risk classification (low or high).

5. HIGH RISK – IDENTIFICATION OF ENVIRONMENTAL MONITORING PROGRAM REQUIREMENTS

High risk facilities release hazardous or radioactive nuclear substances into the environment in concentrations or under conditions that impact non-human biota at the population or community level. For these facilities, sole reliance on the PCM program is not considered acceptable due to the following weaknesses of environmental modeling:

— synergistic or antagonistic reactions that may occur from complex mixed effluents.

— cause and effect relationships between elevated environmental concentrations (in abiotic or biotic compartments) and population or community level effects.

— the need to extrapolate long-term chronic effects based on acute or short-term chronic studies.

— the need to extrapolate *in situ* field effects from laboratory toxicity tests on laboratory reared species.

Conservative assumptions in the ERA are typically used to address uncertainties arising from these weaknesses. Facilities not predicting significant population or community level effects can rely on a conservative ERA approach in combination with a PCM to ensure adequate protection of the environment. However, higher risk facilities with the potential for population and/or community level effects are required to add an additional Biological Effects Monitoring (BEM) component to their environmental monitoring program.

BEMs detect measurable biological effects for non-human biota exposed to hazardous and radioactive nuclear substances at the population or community level. Contaminant exposure can initiate changes in a wide range of biological levels. These range in an increasing order of ecological relevance from molecular to ecosystem levels. Selecting the biological level at which the BEM operates is a significant decision. Biomarkers operating at lower levels (e.g., molecular) provide early warning of potential impacts, but have questionable ecological relevance. Given an organism's inherent compensatory capabilities, biological responses (e.g., biochemical, histological) may occur well below exposure levels that impair reproduction or survival [2]. The probability that biomarkers at these exposure levels translate into detectable population effects is low. Operating BEMs at population and community levels ensure the monitoring of ecologically relevant biomarkers of environmental health. The major weakness with this approach is that impacts detected at these levels may be irreversible. One solution is to monitor communities and/or populations which have a strong potential for recovery after mitigation and are not an economic resource or a species/community at risk [2]. A common practice in Canada for monitoring aquatic releases has benthic macroinvertebrate community monitoring combined with population and health monitoring of a sentinel fish species using both direct (abundance and age structure) and indirect (e.g., gonado-somatic indices, egg counts) metrics [3, 4].

6. CONCLUSION

A risk-based approach allows environmental monitoring program requirements to increase as the risks pose by the nuclear facility increase. The use of ERAs allows site-specific flexibility in designing an environmental monitoring program. Ultimately, a risk-based approach ensures a cost-effective and resource-effective way for licensees to meet their regulatory requirements and for the CNSC to fulfill their mandate on environmental protection.

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Suitability of individual biological effects benchmarks for the protection of wild populations of mammals

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Abstract. In Canada, regulations developed under the Nuclear Safety and Control Act require that license applicants describe the effects on the environment of the nuclear facility to be licensed. For the purpose of assessing risks to the environment, the Canadian Nuclear Safety Commission recommends the use of an ecological risk assessment approach. It is based on toxicity benchmarks from chronic exposure studies of reproduction and survival in sensitive species. For mammals, an Expected No-Effects Value of 3 $mGy \cdot d^{-1}$ has been derived from limiting studies on fertility effects in squirrel monkeys exposed to tritium. This benchmark is adequate for regulatory purposes relative to data on wild populations of small rodents living in contaminated areas, or in areas artificially exposed to high levels of gamma radiation. At least for short-lived, prolific species of mammals, it has been impossible to define a Lowest-Observed-Effects-Level for population effects at chronic dose rates up to about 100 $mGy \cdot d^{-1}$ for lifetime doses up to about 10 Gy. Refinement of taxa-specific benchmarks to an appropriate level of conservatism will require more research on long-lived, slowly-reproducing species.

1. INTRODUCTION

In Canada, the regulation of nuclear facilities by the Canadian Nuclear Safety Commission under the Nuclear Safety and Control Act requires that licensed activities be evaluated in order to demonstrate adequate protection of the environment. An ecological risk assessment approach has been adopted for this purpose[1], because it allows the assessment of risks from exposure to both hazardous and radioactive substances in a consistent and comparable manner. This regulatory effort has required the derivation of benchmarks suitable for the protection of populations and communities under realistic conditions of chronic exposure.

In the absence of robust information on thresholds for radiation effects at high levels of biological organization, the approach has been to adopt limiting values based on readily-observable, individual effects in the most sensitive species for which data is available. Benchmarks (Expected No-Effects Values, ENEVs) have been derived from an updated literature review of relevant Critical Toxicity Values (CTVs), with safety factors incorporated as appropriate. In most cases, benchmarks have been derived from chronic exposure studies with endpoints of significance to reproduction or survival. For mammals, a median lethal dose of 3 $mGy \cdot d^{-1}$ (corrected for a Relative Biological Effectiveness of three for tritium) for immature oocytes of the squirrel monkey has been chosen as a CTV [2]. Given that this type of effect may not result in sterility or reduced fecundity, no safety factor was applied to derive an ENEV. This value of 3 $mGy \cdot d^{-1}$ is within the range of thresholds (1-10 $mGy \cdot d^{-1}$) where low-level effects are interpreted as being of potential environmental significance [3, 4].

2. INDIVIDUALS VERSUS POPULATIONS

The Canadian regulatory approach assumes that benchmarks based on individual-level effects in sensitive taxa will be protective of all taxa at higher levels of biological organization in the field. This assumption has rarely been tested due to the paucity of quantitative data on populations exposed to elevated levels of radionuclides. In mammals, historical observations and initial experimental studies in contaminated environments have indicated considerable resilience at the population level [5]. This overall conclusion has since been validated on a large scale by the the observation of a "vibrant ecosystem" within the heavily-contaminated zone at the Chernobyl Nuclear Power Plant in the Ukraine [6]. For example, small mammal populations and communities at Chernobyl cannot be distinguished from those in uncontaminated areas [7] at current dose rates of about 33 $mGy \cdot d^{-1}$ [8]. Even at the level of the individual, it has been impossible to demonstrate any long-term effects using sensitive biomarkers of radiation damage in a common small rodent, the bank vole (*Clethrionomys glareolus*) [9].

These results agree with the limited data available from experimental exposure of field populations of mammals to ionizing radiation. In large, natural enclosures in Nevada, USA, the pocket mouse (*Perognathus formusus*) maintained high populations with near-normal characteristics at a measured dose rate of 9 $mGy \cdot d^{-1}$ (0.5 $mGy \cdot d^{-1}$ in winter to 18 $mGy \cdot d^{-1}$ in summer, up to 100 $mGy \cdot d^{-1}$ potential short-term exposure) over 4 years of artificial irradiation [10]. Similar exposure of a free-ranging population of red-backed voles (*Clethrionomys gapperi*) at the Field Irradiator - Gamma (FIG) forest in Manitoba, Canada showed almost no evidence for radiation-specific effects over ten years at a nominal exposure rate of 21 $mGy \cdot d^{-1}$ (geometric mean of ground-level exposure rates at small mammal trapping stations, range 4-105 $mGy \cdot d^{-1}$) [11]. Along with data from less substantive studies [12], these field experiments suggest that populations of short-lived, prolific species can readily accomodate to exposure rates of up to about 100 $mGy \cdot d^{-1}$. Any effects of radiation at the individual level are presumably counterbalanced by compensatory mechanisms at the population level, or are simply undetectable relative to natural patterns of variation.

3. THE CANADIAN ZEUS EXPERIMENT

During the 1980's, large-scale field experiments were conducted in Manitoba at the ZEUS facility of Atomic Energy of Canada Limited to better define the threshold for population effects in mammals under natural conditions [13]. Free-ranging populations of meadow voles (*Microtus pennsylvanicus*) living on 1-ha "island meadows" were gamma-irradiated for 1-1.5 years in three experiments at measured dose rates of 0.4, 15, and 44 $mGy \cdot d^{-1}$. The experiments were designed to address deficiencies in previous studies with ^{137}Cs irradiators (enclosure artifacts, seasonal artifacts, variable spatial exposure rates, variable animal dose rates, poor information on reproduction or life history, minimal statistical power, lack of robust controls). Considerable effort was expended to obtain uniform, continuous animal exposures within well-defined, natural populations.

Even at the highest dose rate (44 $mGy \cdot d^{-1}$), no individual, population, or community effects were observed in a comprehensive program addressing the health and demography of all small mammals living within the ZEUS facility. Critical spring populations of resident voles accumulated individual doses up to 10 Gy without discernable effects on reproductive performance. Second or third generation offspring of these voles also showed no evidence of radiation effects at lifetime doses of about 4-6 Gy. These chronic doses are similar in magnitude to acute lethal doses in small mammals under laboratory conditions. However, as in previous field studies, it was impossible to define a population-based LOEL (Lowest- Observed-Effects-Level) over a realistic range of dose rates.

4. CONCLUSION

The weight of evidence suggests that small rodents are resilient to radiation effects at levels of exposure typical of anthropogenic activities, either routine or accidental. Population-based Critical Toxicity Values for realistic situations in the field appear to be an order of magnitude higher than individual-based benchmarks being considered as international standards for the protection of non-human biota. Adoption of a reproductive benchmark from chronic exposure of a radiosensitive mammal is therefore adequate in a regulatory context to ensure protection of wild mammals, particularly, large, slow-growing species that reproduce late in life. Until more studies are conducted on long-lived [14] or slowly-reproducing mammals, it is unlikely that new insights will be gained on the exact benchmark that would be protective of all mammals, under all circumstances, without being unduly conservative.

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Biological effects benchmarks for the protection of aquatic organisms against radiation

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Abstract. In Canada, regulations developed under the Nuclear Safety Control Act require that license applicants describe the effects on the environment of the nuclear facility to be licensed. For the purpose of assessing risks to the environment the Canadian Nuclear Safety Commission recommends the use of an ecological risk assessment approach. It is based on toxicity benchmarks from chronic exposure studies of reproduction and survival in sensitive species. The benchmarks or Estimated No Effect Values (ENEVs) for the various taxonomic groups are determined from literature data using an ecotoxicological approach. The ENEVs derived for radiation effects on aquatic biota are: 0.6 $mGy{\cdot}d^{-1}$ for fish, 2 $mGy{\cdot}d^{-1}$ for amphibians and reptiles, 2.4 $mGy{\cdot}d^{-1}$ for algae and macrophytes and 4.6 $mGy{\cdot}d^{-1}$ for benthic invertebrates.

1. INTRODUCTION

In Canada, the regulation of nuclear facilities falls under the Nuclear Safety and Control Act and requires that licensed activities describe their effects on the environment and that the effects be evaluated in order to determine whether or not adequate provisions have been made for protection of the environment. An ecological risk assessment (ERA) approach has been adopted for this purpose [1], because it allows the assessment of risks from exposure to radioactive nuclear substances and hazardous substances in a consistent and comparable manner. Risk characterization involves the calculation of a risk quotient (exposure/benchmark) or a comparison of exposure values with a distribution of dose-effects data. Benchmarks or Estimated-No-Effect Values (ENEVs) suitable for the protection of populations and communities under realistic conditions of chronic exposure are needed and are generally derived from critical toxicity values (CTVs).

2. IDENTIFICATION OF CRITICAL TOXICITY VALUES AND DERIVATION OF ESTIMATED NO EFFECT VALUES

The CTV is usually an estimate of low toxic effect (e.g. EC_{25}- concentration causing a 25% reduction in growth). In order of preference the CTV should be derived using: (1) LD_{25}, EC_{25}; (2) LOEL or NOEL; and (3) LD_{50}, EC_{50}. LOELs and NOELs are not the preferred dose-response measurements because estimation techniques will usually determine these values even for studies with poor dose-response data. Median toxic effects measurements (e.g., LD_{50}) are the least preferred, because larger application factors (AFs) are required to derive an ENEV from a LD_{50} than from a LD_{25}, or EC_{25} [2].

The ENEV should, ideally, be based on chronic endpoints measured in controlled field tests for a wide array of endpoints (e.g., growth, reproduction). However, such data sets are rarely available and ENEVs must often be derived using laboratory toxicity data on a limited number of species. The likelihood that a laboratory test conducted on a typical bioassay species will adequately represent an assessment endpoint is low. To account for such deficiencies in dose-response data, application factors (AF) (also called extrapolation, safety or uncertainty factors) are applied to the laboratory test results (i.e., CTV/AF = ENEV). This is generally referred to as the “ecotoxicological” approach.

Several extrapolations are needed to convert a CTV observed on a measurement endpoint (e.g., EC_{25} for growth of rainbow trout in the laboratory) to an ENEV for a corresponding assessment endpoint (e.g., no effect on pelagic fish species). Some of the necessary extrapolations relate to:

— intraspecies variability - results of toxicity tests vary within species due to differences in age and sex, conditions under which the animals are kept, etc. Sensitivity of an organism to radiation depends on the life stage at exposure. Embryos and juveniles are more sensitive than adults, e.g., the LD_{50} for an acute radiation dose is 11.2 Gy for juvenile and 20.5 Gy for adult *Fundulus heteroclitus*;

— interspecies variability - large variations in sensitivity can occur within phyla. For example, the dose at which fertility in fish exposed to chronic radiation is affected range from 0.2 to 37 $Gy{\cdot}a^{-1}$.

— aboratory to field - toxicity data derived from laboratory studies generally have limited relevance to field situations. Laboratory conditions do not allow for the integration of the numerous environmental influences on the environmental fate of toxicants and on the various organisms exposed to the toxicants. Laboratory studies do not include the multiple stresses usually present in the field (e.g., temperature fluctuations, competition, food limitation, disease, presence of other contaminants, etc.). In the case of radiation exposure, anomalies of the reproductive system, including sterility, to silvercarp occurred in the Chernobyl cooling pond at a dose rate of 0.2 $Gy{\cdot}a^{-1}$, whereas acute doses of about 10 Gy or more cause sterility in fish in the laboratory.

The limited ecological relevance of laboratory studies as well as inter- and intra-species differences in responses to a substance preclude generalizations among species concerning toxicological responses and support the use of application factors. Empirical data provide limited support for some of the application factors currently in use [3, 4, 5, 6, 7].

3. IDENTIFICATION OF CRITICAL TOXICITY VALUES AND DERIVATION OF ESTIMATED NO EFFECT VALUES FOR RADIATION EFFECTS

There are few studies involving chronic radiation exposure at environmentally relevant dose rates from which to derive benchmarks (ENEVs) for assessing radiation effects to aquatic biota. Most studies on the effects of radiation on survival are for acute exposures where the post-irradiation observation period is 30 days (30-d LD_{50}). For poikilothermic (cold-blooded) animals, temperature can control the time of expression of radiation effects. Extending the observation period usually lowers the dose causing 50% mortality [ref. in 1]. Therefore, for fish, amphibians and reptiles, which are poikilotherms, a 60- or 90-day study period is more appropriate. Bird et al. [1] present the radiation effects data reviewed for the purpose of deriving ENEVs for biota. The ENEVS for aquatic biota are presented below with the key references for the CTVs.

3.1. Amphibians and reptiles

Studies involving chronic exposure of desert lizards to ^{137}Cs at about 20 $mGy{\cdot}d^{-1}$ (7 $Gy{\cdot}a^{-1}$) for four years resulted in sterility at a cumulative dose of 15 Gy [8]. An ENEV was derived from this long-term field experiment by applying a safety factor of 10 to convert the CTV of 20 $mGy{\cdot}d^{-1}$ for sterility into an ENEV for effects on reproduction. This ENEV of 2 $mGy{\cdot}d^{-1}$ should provide a margin of safety for long-lived turtle species and is likely conservative for shorter-lived amphibian species that would accumulate lower life-time doses.

3.2. Fish

Under field conditions, low chronic exposure at 0.6 $mGy{\cdot}d^{-1}$ had no effect on the population of mosquito fish in White Oak Lake, whereas, reproductive effects were observed in carp subjected to chronic exposures of 0.6 $mGy{\cdot}d^{-1}$ in the Chernobyl cooling pond [9].

Because of the relatively few chronic radiation exposure studies with fish, and the high quality of the Chernobyl study [9], the value of 0.6 $mGy{\cdot}d^{-1}$ for reproductive effects in carp was chosen as the CTV. An application factor of one is used resulting in an ENEV of 0.6 $mGy{\cdot}d^{-1}$ for effects to populations of fish. This ENEV may be somewhat conservative, in that factors other than radiation (e.g., thermal and chemical pollution) may have contributed to the effects observed. On the other hand, this value is in the range where both effects (long-lived species) and no effects (short-lived species) are seen. It should ensure that long-lived (10 to >20 years), fish species that start reproducing at the age of 3-7 years or older, are reasonably protected.

3.3. Benthic invertebrates

Of the aquatic invertebrates studied, the polychaete worm *Neanthes arenaceodentata* is sensitive to radiation [10]. Therefore, the value of 4.6 mGy·d^{-1} for effects on reproductive indices for this polychaete worm was chosen as the CTV. An application factor of 1 was chosen because both the nature and magnitude of effects on reproductive indices were similar at 4.6 mGy·d^{-1} and 50.4 mGy·d^{-1}. The ENEV of 4.6 mGy·d^{-1} from this study is in the range of radiation doses experienced by *Chironomus tentans* larvae in White Oak Lake that resulted in an increased frequency of chromosomal aberations but no population level effects. Therefore, an ENEV of 4.6 mGy·d^{-1} should adequately protect both freshwater and marine benthic invertebrate species.

3.4. Aquatic plants

The lowest dose causing sublethal effects on aquatic plants (macrophytes and algae) is 0.07-0.12 mGy·d^{-1} (0.03-0.04 Gy·a^{-1}), which caused a loss of synchrony in growth of *Chlorella pyrenoidosa* cultures. Circumstantial evidence that low radiation doses may impede algal development also comes from analysis of algae species invading a ^{137}Cs contaminated man-made pond and a control pond. The control pond was covered with a thick bloom of algae, but surface algal growth was sparse in the contaminated pond. Green algae were dominant in the contaminated pond, whereas blue-green algae were dominant in the control pond. These studies were not of sufficient quality to provide a CTV.

Because of the scarcity of data for chronic radiation effects on aquatic plants, the ENEV for conifers (terrestrial plants) of 2.4 mGy·d^{-1} (a NOEL) [11] was adopted as the ENEV for both algae and macrophytes. Conifers are more sensitive to radiation than lichen and lichens are composed of fungus in symbiotic union with an alga. Therefore, the use of the conifer data for the ENEV for aquatic plants is likely conservative.

4. CONCLUSIONS

The IAEA [12] and UNSCEAR [13] concluded that a dose rate of approximately 10 mGy·d^{-1} to the most highly exposed individuals in a population (and most probably lower average dose rates to the whole population) would be protective of populations of aquatic plants and animals. The IAEA and UNSCEAR used the "experts review" approach to derive the 10 mGy·d^{-1} dose benchmark. The experts review approach does not provide explicit information on how areas of uncertainty in available data have been accounted for and is, therefore, perceived to be inconsistent with the transparent, open and participatory approach used in environmental protection decision-making [13]. In contrast, the ENEVS for aquatic biota derived using an ecotoxicological approach range from 0.6 mGy·d^{-1} (fish) to 4.6 mGy·d^{-1} (benthic invertebrates). These values represent an average population dose rate and are not widely different from the 10 mGy·d^{-1} value discussed above.

The ENEVs for radiation exposure developed using an ecotoxicological approach have the advantage, for environmental protection purposes, of having clearly stated assumptions and provide values that are consistent with ENEVs derived for non-radioactive hazardous substances.

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Environmental Protection from ionising radiation in the Arctic

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Abstract. The EU-funded project "EPIC" deals with the practical application of a system for environmental protection from ionising radiation to Arctic areas. In this paper, examples are given of selected key activities completed since the project began in October 2000. Having earlier selected a set of reference organisms that might suitably be used as focal points for impact assessments, models have been developed and employed to predict uptake to the bodies and transfer to the habitats of these biota. In the case of the terrestrial environment this has, in the case of radiocaesium and radiostrontium for example, involved the adaptation of an earlier-developed dynamic model that was used to consider doses to man through the ingestion of reindeer meat. In the case of the marine environment biokinetic models have been applied, which account for processes including biomass change, metabolic activity and depuration rates to simulate the biological behaviour of a suite of radionuclides. Dose models have been developed that allow absorbed fractions to be calculated for user-selected geometries and energies. The model can be applied to quite complex situations whereby differential activity concentrations are observed within the body of the biota. Finally, a dose-effects database has been constructed in Excel format mainly from Russian literature sources. Dose reconstruction, in the case where activity concentrations and effects data only are available, has been performed for aquatic organisms, mainly fish and fish eggs. The types of organisms considered in the database include aquatic organisms, terrestrial animals (mainly rodents), soil invertebrates, soil micro-organisms and trees.

1. INTRODUCTION

The debate concerning the requirement to transparently demonstrate environmental protection from ionizing radiation (see for example [1, 2]) has led to a proposal for the development of a system within which existing data could be considered, areas of information paucity identified and environmental protection criteria developed [3]. It has further been proposed [3] that any system for the radiological protection of the environment should include the development of quantities and units, defined reference organisms, reference dose models and tables of effects for reference organisms. In response to these recommendations, The EU-funded project "EPIC" - "Environmental Protection for Ionising Contaminants in the Arctic" adopts a practical perspective in developing a protection framework for the Arctic environment. There are several aspects to the rationale behind the choice of the Arctic for specific study. Firstly, low temperatures, extreme seasonal variations in light and lack of nutrients are some of the physical and chemical characteristics which cause environmental stress to organisms in the Arctic and make them potentially more vulnerable to contaminants [4]. Secondly, such environmental stresses restrict the number of organisms within the Arctic, making it easier to identify reference organisms for which generic methodologies can be developed. Finally, there is particular concern over potential radioactive contamination of the Arctic due to the wide range of nuclear sources, including nuclear power and reprocessing plants, civil and military nuclear powered vessels and nuclear weapons testing areas. At present little is known with respect to the potential effects on flora and fauna that would be observed in the Arctic following a significant release of radioactivity.

2. PROJECT OBJECTIVES

The EPIC project started in the autumn of 2000 will have a 3-year duration and involves collaboration of 4 institutes from 3 countries (Norway, Russia and the UK). The following objectives are stated for the project:

(1) collate information relating to the environmental transfer and fate of selected radionuclides through aquatic and terrestrial ecosystems in the Arctic;

(2) identify reference Arctic biota that can be used to evaluate potential dose rates to biota in different terrestrial, freshwater and marine environments;

(3) model the uptake of a suite of radionuclides, both natural and anthropogenic to reference Arctic biota;

(4) develop a reference set of dose models for reference Arctic biota;

(5) compile data on dose-effects relationships and assessments of potential radiological consequences for reference Arctic biota;

(6) integrate assessments of environmental impact from radionuclides with those for other contaminants.

3. RESULTS AND ACHIEVEMENTS TO DATE

Numerous tasks were scheduled within the EPIC project. Three of these have now been selected in order to give an insight into the tangible activities of the consortium. Additional information can be found in several papers and reports (e.g. [5–7]).

3.1. Modelling transfer in the environment

A review of available models for considering transfer of radionuclides in terrestrial Arctic ecosystems has been undertaken by the Centre for Ecology and Hydrology (CEH), UK. There are no models that have been specifically designed to simulate the transfer of a suite of radionuclides to various organisms within Arctic ecosystems. The Institute of Radiation Hygiene (IRH) developed the ECOMARC model to estimate exposure of Arctic human populations considering both radiocaesium and radiostrontium in the EC-Copernicus project AVAIL [8]. This semi-dynamic model is an adaptation of the ECOSYS-87 agricultural food chain model [9] with the inclusion of some Arctic-specific parameters. CEH have begun work to adapt and parameterize this model so that it may be used to derive concentrations in selected reference organisms. The model has been used to predict activity concentrations of ^{137}Cs and ^{90}Sr in reindeer muscle following a single deposition of 1 Bq m^{-2} of each isotope occurring in June. Predicted ^{137}Cs and ^{90}Sr activity concentrations in reindeer muscle have subsequently been used to predict the ^{137}Cs and ^{90}Sr activity concentrations in wolves, hypothetically consuming reindeer as their sole dietary intake.

In relation to modeling of uptake and transfer in food-chains in Arctic aquatic environments, biokinetic (dynamic) models have been discussed and applied. TYPHOON's model ECOMOD has been used to simulate the behaviour and fate of a suite of radionuclides (including radionuclides from the original list of selected radionuclides – Cs, Sr, I, Ra and some additional highly-assimilated radionuclides routinely assessed by TYPHOON - P, Mn, Co, Zn). The ECOMOD model considers the radionuclide as a tracer, identical in its properties to its stable (analogous) element. The major mechanism of accumulation of such nuclides as ^{137}Cs, ^{90}Sr, ^{32}P, ^{60}Co, ^{65}Zn, ^{54}Mn, ^{131}I, ^{226}Ra in aquatic organisms is bioassimilation. Bioassimilation involves active intake and assimilation of radionuclides, with radionuclides being incorporated in internal tissues and organs. The radionuclide assimilated by an organism goes to the production of new biomass, and to compensate for metabolic losses of the stable element.

3.2. Dosimetric models

IRH have now completed development of a model entitled DOSE3D which can be used to calculate internal and external doses (dose-rates) for user-defined geometries (Figure 1). The program is constructed from 2 component parts:

(1) Geometry module – This part of the program allow the user to create a geometry and subsequently manipulate and view this object. The module deals with a variety of shapes including ellipsoids, spheres, cylinders, conical cylinders and egg-shaped (i.e. irregular ellipsoid) objects. A 3-dimensional solid array is generated from the original mesh. A Monte Carlo algorithm is subsequently employed in order to calculate chord/segment distributions.

(2) Dose module – This part of the program uses chords data output from the geometry module, in the form of histograms, to derive absorbed fractions or dose rates. Absorbed fractions can be calculated for α, β and γ radiation types. The user is prompted to select the energy (monoenergetic α and γ or maximum and average for β) Scaling factors allow calculations to be performed for a phantom of larger size but the same shape.

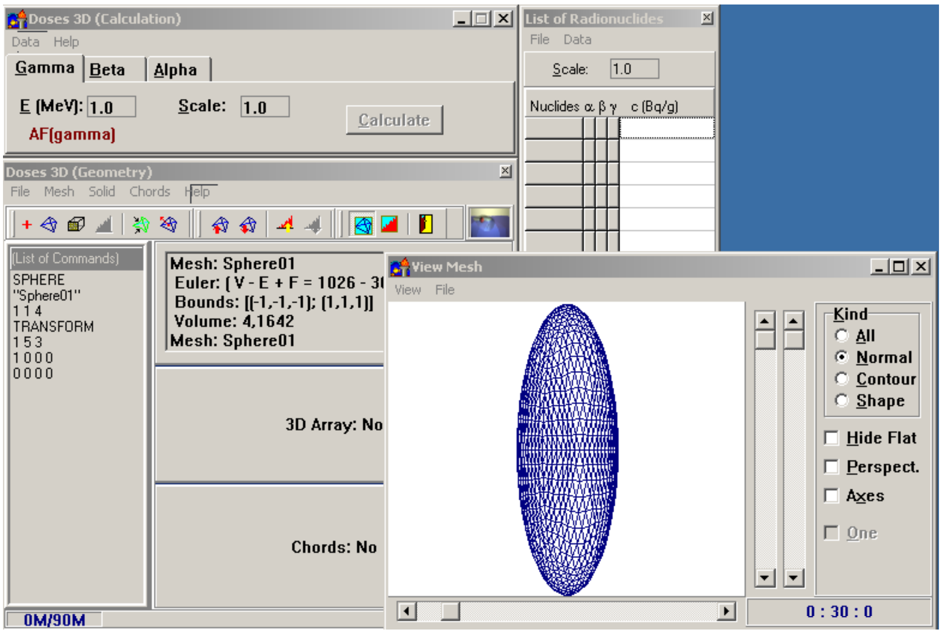

FIG. 1. Dose 3D User interface.

The program is currently available in 2 forms, one which can be used to carry out calculations for (1) simple situations whereby activity concentrations are uniformly and homogeneously distributed within the organisms and/or its environment and (2) more complex situations whereby differential activity concentrations between organs can be defined and absorbed fractions and dose rates calculated for the sets of organs involved.

3.3. Dose effects data base

Scientists from TYPHOON have taken the main responsibility of constructing a database (with emphasis on studies conducted in the former Soviet Union and reported within Russian language literature) on dose-effects relationships. The format for the data-base is Microsoft Excel, with accompanying text abstracts (Microsoft Word files providing detailed description of recorded effects) accessed using hyperlinks. Effects of long-term chronic radiation exposure have been considered in the database. The data base includes information from field studies and laboratory experiments. Efforts within Year 1 and 2 of the project have been focussed on literature review, data extraction and incorporation within the data base. In some cases, and where possible, i.e. in the case where detailed information is provided concerning activity concentrations within biota and their media, doses (dose-rates) have been reconstructed using TYPHOON methodologies. This type of dose reconstruction has been performed for aquatic organisms, mainly fish and fish eggs. The types of organisms considered include aquatic organisms, terrestrial animals (mainly rodents), soil invertebrates, soil micro-organisms, trees. Sources of possible misinterpretation of effects at low doses were considered. In addition to this work, TYPHOON is collecting the information on chemical toxicants, which can simulate the effects of ionising radiation in organisms. This information will be used for analysis of dose-effects relationships and evaluation of links between radiation and other chemical toxicants. Further details concerning the radiation dose-effects database can be sound in Sazykina and Kryshev (2003).

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Radioactivity and Wildlife:
Taking stock

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Abstract. Radiation protection standards were developed with the express aim of protecting human health asserting that these would therefore protect non-human biota, an assertion which did not derive from evidence. English Nature (then Nature Conservancy Council) commissioned a report in 1988 to examine the issue of radioactivity and wildlife and, again in 2001 with the Environment Agency, sought to update the original work in the light of new evidence. The organisation took an interest because of the persistence, capacity for bioaccumulation of some radionuclides and chronic toxic effects at low doses and the potential concerns for wildlife at the site-specific level as well as the distribution of radioisotopes more widely in the environment. In the last decade, biodiversity legislation such as the Habitats Directive and Regulations has been enacted which specifically prohibits damage to the "integrity" of internationally important habitats and species. The history of developments in knowledge about the health effects in humans and other species following exposure to radioactivity is reviewed. Currently, there is a dearth of information in certain areas including the effects of specific radionuclides, the behaviour and pathways of naturally occurring radionuclides, chronic as opposed to acute effects, interactions with other stressors, some taxa and alternative exposure scenarios. English Nature and the Environment Agency have adopted a precautionary principle in judging whether a significant effect is likely or not following a radioactive impact. Developments at the cellular level could provide a joint basis on which to base assessments of radiation damage regardless of the species affected. However, it will be some time before any possible links between such damage and the development of an associated pathology become clear whether in humans or wildlife species. Knowledge about the impacts of radiation continues to evolve and needs to be taken into account.

1. BACKGROUND

In 1988, a seminal article raised questions and the impacts of radioactivity on species other than human beings and identified a lack of research [1]. English Nature commissioned a report to find out what evidence did exist [2]. Radiation protection standards had been developed with the express aim of protecting human health and relied on an assertion rather than evidence about non-human species. The limits stated that if human beings were protected, then other species of flora and fauna would be safeguarded:

> *"the level of safety required for the protection of all individuals is thought likely to protect other species…The Commission therefore believes that if man* [sic] *is adequately protected then other living things are also likely to be sufficiently protected*." [3]

and more recently:

> *"Occasionally, individual members of non-human species might be harmed, but not to the extent of endangering whole species or creating imbalance between species."* [4]

2. HEALTH IMPACTS

2.1. Evolution of radiation protection standards

Concerns about the adverse health impacts of ionising radiation were first raised within a year of Roentgen's discovery of X-rays in 1895. Thomas Edison reported eye irritation from experiments with X rays and skin burns, hair loss and skin cancer were observed in 1901 [5]. It was among the radiographers, the profession developing and using X-rays that significant impacts were first noted. The first malpractice award for X-ray burns occurred in 1899.

2.2. Risks to human health: populations

Risk estimates have been derived from monitoring the Atomic Bomb Survivors, people who lived through the dropping of atomic bombs on Hiroshima and Nagasaki, or people born subsequently. For such studies to be applicable to any population, the group being studied needs to be representative of age, gender, social background and radiosensitivity. The standards derive from keeping various parts of the Life Span Study under continuous surveillance and recording who has died, when and from what. The LSS concluded that:

— cancer is the only late effect of radiation exposure;

— there was a greater risk of developing leukaemia than of solid tumours;

— the interval between exposure and death from leukaemia was relatively short (5-10 years);

— there was no risk at levels below those set for nuclear workers;

— young people were more radiosensitive than older people.

Other populations studied have arisen from the use of X-rays. In particular, the Oxford Survey of Childhood Cancers which compared that who died young from cancer with similarly aged children who did not [6]. The comparisons showed that the mothers of those children who had died had been X-rayed more often before birth than those children who did not die. Thus the Oxford Survey became an important source of information about the cancer effects of foetal irradiation and about the aetiology of childhood cancers. There have also been studies of nuclear workers, some of which have detected cancer risks at dose levels which have been regarded as safe and following the Three Mile Island accident [7-11]. Leukaemia studies around nuclear power stations have been summarised in many places (12, Comare Reports 1988 and 1996).

Following the Chernobyl accident, an increase in childhood thyroid cancer was observed. It was suggested that the latency period was too short and that the pathology had been faulty; that cases were being found because people were looking for them and that Iodine-131 was safe [13]. However, the increased incidence was verified by Professor Dilwyn Williams when he compared a map of the exposure to I-131 fallout with that of the distribution of cases. While I-131 is used to treat adults, such exposure is not benign for young people [10]. Williams is convinced of the need to study Chernobyl on a scientific basis in order that data arising from fallout after the accident can be incorporated into the risk estimates which evolved from the studies of the atomic bomb survivors (ABS) and which did not capture these early years.

2.3. Cellular developments

Recent developments in the laboratory have arisen in the identification of effects referred to as "Genomic instability" and that of the "bystander effect". In the former, a single alpha particle was introduced into a culture of mouse bone marrow stem cells [14]. No change was observed in the cells. However, ten cell division generations later, chromosome damage was observed which could only have occurred as a result of the original irradiation. The findings "imply that alpha particle exposure may induce damage that is transmitted to daughter cells and which may result in the expression of genetic instability in later generations". This phenomenon has been confirmed in human cells. It has also been observed that cells adjacent to those which have been irradiated can show the effects– this is known as the bystander effect [15-18].

Interest in pre paternal irradiation (PPI) was initiated by work done by the late Martyn Gardner when he found that PPI had influenced the appearance of leukaemia in the offspring of those who had been exposed to ionising radiation [19-20]. Recent studies with mice did not show a direct cause and effect but did reveal that PPI could potentially provide part of the answer. Where bone marrow damage occurred it could be shown that leukaemia incidence increased. The process of blood production could be slightly damaged so that it was more susceptible to malignant transformation [21].

2.4. Areas of controversy

Different approaches have been derived in order to assess the risks posed by radiation. The Linear No Threshold (LNT) approach has been adopted which means that radiation at any level is judged to be deleterious to health. The higher the exposure, the greater the chance of human beings developing cancer (22). The dose limits have been lowered five times reflecting a growing understanding of the effects of ionising radiation. The current dose limits are 1 mSv and 20 mSv for workers.

Some have argued that current methods for assessing radiation standards are flawed [23-24] and there continues to be a debate about effects at low doses of radiation and the radiation risk estimates. The link between genomic instability and any arising pathology is not yet known and cannot be predicted.

Historically, radiation protection has not adopted the approach adopted with regard to other pollutants, namely “Control and Contain” where they are assumed to be harmful unless shown otherwise. The authorities concerned with radiological protection have relied on a “Dilute and Disperse” philosophy thus transferring radionuclides throughout natural systems and the environment. Initially the organisations involved with ionising radiation concentrated, in the main, on artificial sources principally those of X-rays and atomic fallout whereas natural radiation sources only featured erratically in their deliberations.

3. EVOLUTION OF ENVIRONMENTAL INTEREST

To date, there has been little evidence of damage to wildlife populations in the UK as a result of exposure to naturally-occurring or anthropogenically-derived radiation. Concern arises because exposure to levels significantly higher than background can and certainly does occur and because the risks to wildlife have generally not been considered.

The requirement for site protection under the Habitats Regulations and the OSPAR agreement, and more generally for protection of populations of Biodiversity Action Plan (BAP) species, provide drivers for the consideration of potential risks to wildlife arising from exposure to artificial sources of radiation. Risks to wildlife have not traditionally been explicitly considered during risk assessment processes but differences in sensitivities between species and in routes of exposure suggest that they should be.

The research carried out had not been coordinated nor directed to the key species or habitats. The publication of Thompson’s article spurred a number of reviews collating the evidence [25-31]. The European Commssion established the FASSET project (Framework for Assessment of Environmental Impact, due to finish in October 2003) to examine environmental risks and the ICRP has set up a Task Group on protection of the environment.

3.1. History of radioecological research

The effects of ionising radiation on plants and animals have been studied since the start of the last century. The first experiments undertaken to examine the toxic effects of radon were carried out by Soviet researchers and by Pierre Curie in France in 1904. (32: London and Gol’berg, 1904, cited in Letavet, 1962). London found that frogs and white mice died when they had to breathe in an atmosphere of radon (the concentration was not specified). Gol’berg concluded that “*emanation is capable of killing higher animals by changing the composition of the blood and affecting the epithelium of respiratory organs*.” Animal studies have also often provided new insights. It was work on the fruit fly *Drosophila melanogaster* which led to an understanding of the mutagenic effect of radiation [33]. The first postdoctoral theses on the impacts of radioactivity following the invention of the atomic bomb were carried out in 1946 examining salmon and trout in the adjacent waters [cited in 34].

In the 1950s and 60s radioactive fallout was transported from the test sites in the Pacific Ocean to the feeding grounds of reindeer in the Arctic. Lichens absorbed the radioactivity which was transferred to the reindeer as they fed on the lichen. In the United Kingdom sheep movements are still restricted following the Chernobyl accident in 1986 when radioactive material fell onto nutrient poor upland soils which were unable to immobilise the radioactivity. Until then studies had concluded that radioisotopes would be immobilised but the research had been done on lowland soils containing more clay. Chernobyl showed the limitations of our knowledge.

Long term effects of exposure of wildlife to ionising radiation are generally poorly researched and understood and the consequences of current levels of exposure of sedentary localised populations within the Irish Sea and the wider roaming of predators such as the cetaceans cannot be safely predicted.

3.2. Recent review: main findings

English Nature commissioned a report together with the Environment Agency which set out to review the evidence for the effects of selected radionuclides. It includes spreadsheets for estimating dose rates in a range of habitats – Freshwater, Maritime and Grassland [31 op cit].

Its key findings revealed: a lack of information on effects of specific radionuclides; on behaviour and pathways of naturally occurring radionuclides; a need for data on chronic effects as opposed to acute; a need for understanding synergistic effects; some wildlife groups (birds, reptiles, amphibians) have been poorly studied and there has been hardly any research into aquatic mammals and the need to consider other exposure scenarios.

The key assumptions made in the impact assessments include: organisms being viewed as ellipsoids; that radionuclides uniformly distributed externally and are in equilibrium; Concentration factors (CF) are assumed where unknown; provisional weighting factors for RBE are assumed; some radionuclides (eg H-3 and Tc-99) need additional assessment and deterministic effects are more important in considering wildlife. It uses suggested dose rates (26 cited in 29).

3.3. Precautionary principle

The approach adopted seeks to be precautionary in estimating dose rates but concentration factors used to calculate internal exposure may be at least an order of magnitude out in either direction; they may be non-uniform distribution within organism and the assumptions are made for those radionuclides not considered in depth in the report. Therefore, in judging whether a significant effect is likely, current joint EA/EN guidance uses 5% of IAEA dose rates as a trigger for further investigations; assumes a worst case distribution of radionuclides in the vicinity of an outfall; calls for additional assessment where there is previously high loading of long-lived radionuclides and triggers a site specific assessment where there is particular uncertainty concerning the sensitivity of organisms/ecosystems.

4. CONCLUSION: AREAS FOR FUTURE CONSIDERATION

Developments at the cellular level could provide a joint basis on which to base assessments of radiation damage to cells regardless of the species affected. However, it will be some time before any possible links between such damage and the development of a direct link to pathology become clear. Both the individual and the population have proved to be useful tools in examining human populations but there are limitations to their usefulness for other species. The costs of undertaking the necessary research are very large and it could be useful to identify key species to use as “canaries” with regard to radiation doses [35].

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