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IAEA-TECDOC-2036

Modelling Approaches for Management and Remediation at Sites Affected by Past Activities

Report of Working Group 2
EMRAS II Topical Heading
Reference Approaches
for Human Dose Assessment

Environmental Modelling
for Radiation Safety (EMRAS II) Programme

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MODELLING APPROACHES FOR MANAGEMENT AND REMEDIATION AT SITES AFFECTED BY PAST ACTIVITIES

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MODELLING APPROACHES FOR MANAGEMENT AND REMEDIATION AT SITES AFFECTED BY PAST ACTIVITIES

REPORT OF WORKING GROUP 2
EMRAS II TOPICAL HEADING
REFERENCE APPROACHES
FOR HUMAN DOSE ASSESSMENT

ENVIRONMENTAL MODELLING
FOR RADIATION SAFETY (EMRAS II) PROGRAMME

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FOREWORD

A range of diverse sites around the world have been affected by past activities and events resulting in their contamination with residual radioactive material. Historically, activities at such sites have either been unregulated or have not been carried out in accordance with current standards. As a result, there is a need to evaluate the site specific radiological hazards and doses to people and the environment to determine whether remediation is justified and, if so, to develop and implement plans to address the situation. The IAEA has published IAEA Safety Standards Series No. GSG-15, Remediation Strategy and Process for Areas Affected by Past Activities or Events, which establishes a stepwise approach for the planning and implementation of remediation applying the principles of radiation protection.

Environmental impact assessment models can be used to assess the radiological impacts of actual and potential releases of radionuclides to the environment, serving as essential tools for use in evaluating remedial options and their effectiveness in support of remediation planning and implementation. It is therefore important to verify, to the extent possible, the reliability of the predictions of such models through comparison with measured values in the environment and/or with predictions of other models. Ideally, such systematic model testing and verification will be conducted in the context of practical examples and case studies. The IAEA has been organizing programmes of international model testing since the 1980s. These programmes have contributed to a general improvement in models, in the transfer of data and in the capabilities of modelers in Member States. IAEA publications on this subject over the past three decades demonstrate the comprehensive nature of the programmes and record the associated advances that have been made.

From 2009 to 2011 the IAEA organized an international programme entitled Environmental Modelling for Radiation Safety (EMRAS II), which focused on the improvement of environmental transfer models and the development of reference approaches to estimating the radiological impacts on humans, as well as on flora and fauna, arising from radionuclides in the environment. Different aspects were addressed by nine working groups covering three themes: reference approaches for human dose assessment; reference approaches for biota dose assessment; and approaches for assessing emergency situations.

Working Group 2 addressed Reference Approaches to Modelling for Management and Remediation at NORM and Legacy Sites. This publication considers modelling approaches for the management and remediation at sites affected by past activities and demonstrates the practical application of the stepwise approach presented in GSG-15 for the planning and implementation of remediation for sites affected by past activities.

The IAEA wishes to express its gratitude to all those who participated in the work of the EMRAS II programme and gratefully acknowledges the contributions of A. Liland (Norway). The IAEA officer responsible for this publication was T. Yankovich of the Division of Radiation, Transport and Waste Safety.

CONTENTS

SUMMARY 1

1. INTRODUCTION 3

1.1. BACKGROUND 3
1.2. OBJECTIVE 5
1.3. SCOPE 6
1.4. STRUCTURE 7

2. CHALLENGES RELATED TO SITES AFFECTED BY PAST ACTIVITIES 8

2.1. HISTORIC NORM SITES 10
2.1.1. Awareness issues 12
2.1.2. Current challenges in regulation 12
2.1.3. Retrospective assessment of historic wastes and other residues from past activities 13
2.1.4. Monitoring in support of assessment 15
2.1.5. Current trends in NORM residue management 15
2.2. HISTORIC NUCLEAR SITES 15
2.2.1. Types of historic nuclear sites 16
2.2.2. Safe management of historic nuclear sites 17
2.2.3. Types of assessment relating to historic nuclear sites 18
2.2.4. Current challenges in regulation 19

3. FRAMEWORK FOR REMEDIATION OF SITES AFFECTED BY PAST ACTIVITIES 21

3.1. REMEDIATION GOALS AND OBJECTIVES 21
3.2. GENERIC REMEDIATION FRAMEWORK 22
3.2.1. Application of a graded approach in assessment 25
3.2.2. Application of the generic remediation framework in assessment 25

4. GENERAL ASSESSMENT METHODOLOGY FOR THE RADIOLOGICAL RISK AND IMPACT ASSESSMENT OF SITES AFFECTED BY PAST ACTIVITIES 30

4.1. APPLICATION OF THE GENERAL ASSESSMENT METHODOLOGY 30
4.2. DEFINE REMEDIATION OBJECTIVES 30
4.3. ENGAGEMENT OF INTERESTED PARTIES IN ASSESSMENT 31
4.4. THE GENERAL ASSESSMENT METHODOLOGY 31
4.4.1. Identify the problem 35
4.4.2. Review of available information and limited characterization 35
4.4.3. Establish screening criterion 38
4.4.4. Screening assessment 38
4.4.5. Detailed assessment 40
4.4.6. Establish remediation criteria 43
4.4.7. Implementation of remedial actions and verification monitoring 45
4.4.8. Post-remediation management 47
4.5. GENERAL EXAMPLE SUMMARY 47

5. MODELS FOR RISK AND ENVIRONMENTAL IMPACT ASSESSMENT 50

5.1. APPROACHES TO MODEL DEVELOPMENT 50
5.2. ENVIRONMENTAL SYSTEMS AND PROCESSES TO BE MODELLED 52
5.3. MODELS AND MODELLING SYSTEMS THAT ARE AVAILABLE 55
5.3.1. Framework packages 55
5.3.2. Simulation packages 56
5.3.3. Process specific models 57

6. TESTING OF THE GENERAL ASSESSMENT METHODOLOGY ON ACTUAL SITES 61

6.1. GELA SITE (ITALY) 61
6.1.1. Identify the problem (and site description) 61
6.1.2. Review of available information and limited characterization 62
6.1.3. Define remediation objectives 66
6.1.4. Establish screening criterion 66
6.1.5. Screening assessment 67
6.1.6. Detailed assessment 69
6.1.7. Establish remediation criteria 73
6.1.8. Implementation of remedial actions 74
6.1.9. Verification of remediation against criteria 78
6.2. BOTUXIM SITE (BRAZIL) 79
6.2.1. Identify the problem (and site description) 79
6.2.2. Preliminary site evaluation and characterization 80
6.2.3. Establish screening criterion and remediation criteria 80
6.2.4. Screening assessment 80
6.2.5. Detailed assessment 86
6.2.6. Validation 86
6.2.7. Summary 86
6.3. SØVE MINING SITE (NORWAY) 86
6.3.1. Identify the problem, site investigation and characterization 86
6.3.2. Detailed assessment 88
6.3.3. Establishing criteria for remediation 90
6.4. THE FORMER ATOMIC WEAPONS TEST SITE AT MARALINGA (AUSTRALIA) 92
6.4.1. Identify the problem (and site description) 92
6.4.2. Define remediation objectives 94
6.4.3. Site characterization 94
6.4.4. Preliminary and detailed assessment 96
6.4.5. Establish criteria 96
6.4.6. Remedial action 97
6.4.7. Radiation protection issues during remediation 97
6.4.8. Verification monitoring 97
6.4.9. Summary 98
6.5. TESSENDERLO CAF2 SLUDGES DUMPSITE (BELGIUM) 99
6.5.1. Identify the problem 99
6.5.2. Site investigation and characterization 99
6.5.3. Screening assessment and screening criterion 99
6.5.4. More realistic assessment 99

6.5.5. Detailed assessment and remediation 100
6.6. THE FORMER RADIUM PRODUCTION FACILITY OF OLEN (BELGIUM) 100
6.6.1. Identify the problem 101
6.6.2. Screening assessment 101
6.6.3. More realistic assessment 101
6.6.4. Detailed assessment and remediation 101

7. MODEL–MODEL INTERCOMPARISON 102

7.1. GELA SITE (ITALY) 102
7.1.1. Description of the site 102
7.1.2. Objectives of the model–model intercomparison 102
7.1.3. Screening criterion and other relevant criteria 102
7.1.4. Screening assessment 102
7.1.5. More realistic assessment 103
7.1.6. Detailed assessment 104
7.2. BELLEZANE SITE (FRANCE) 104
7.2.1. Description of the site 104
7.2.2. Scenario descriptions 112
7.2.3. Available environmental measurement data 113
7.2.4. Available site specific data 115
7.2.5. Radiological environmental impact assessment 118
7.2.6. Summary of the Bellezane modelling exercise 141

8. SUMMARY 143

9. FURTHER WORK 145

APPENDIX I. SITES TO ILLUSTRATE TYPICAL ISSUES AND FOR POSSIBLE MODELLING EXERCISES 147

APPENDIX II. HISTORIC NUCLEAR SITES IN THE RUSSIAN FEDERATION 149

APPENDIX III. HISTORIC URANIUM MINING AND MILLING SITES 157

APPENDIX IV. HISTORIC NORM SITES 173

APPENDIX V. AVAILABLE MODELS FOR RISK AND ENVIRONMENTAL IMPACT ASSESSMENTS 207

REFERENCES 241

ANNEX I. EXAMPLES OF CURRENT REGULATIONS RELEVANT TO THE MANAGEMENT AND REMEDIATION OF SITES AFFECTED BY PAST ACTIVITIES 259

REFERENCES TO ANNEX 263

BIBLIOGRAPHY 265

CONTRIBUTORS TO DRAFTING AND REVIEW 269

LIST OF WORKING GROUP 2 PARTICIPANTS 271

SUMMARY

A broad range of sites with diverse site specific environmental conditions have been affected as a result of past activities worldwide. To address the resultant range of prevailing circumstances and conditions at such sites, it is necessary to apply a systematic, stepwise approach to assess the magnitude of site specific hazards and risks to people and the environment currently and in the future, and to establish a pragmatic plan for site remediation to then address identified risks. Such a systematic, multi-phased approach is provided in IAEA Safety Standards Series No. GSG-15 [1], Remediation Strategy and Process for Areas Affected by Past Activities or Events, which may be used to determine whether or not remediation is justified based on radiological criteria. In cases where it is, to undertake optimization of protection and safety of radiological and non-radiological impacts and risks in support of planning and implementation of remediation. Environmental impact assessment, dose assessment and risk assessment models can serve as useful tools to evaluate the effectiveness of remedial actions undertaken in accordance with the approved remediation plan. This publication considers the outputs of modelling exercises conducted as part of IAEA's Environmental Modelling for RAdiation Safety (EMRAS II) international model validation programme for case studies that consider actual sites affected by past activities, which are discussed in the context of the stepwise remediation process described in GSG-15 [1].

EMRAS II Working Group 2 (WG2) evolved from the Naturally Occurring Radioactive Material (NORM) Working Group during the first phase of the EMRAS Programme[1] to the NORM and Legacy Sites Working Group in EMRAS II. This extension of the field of interest was made in recognition that the issues being considered previously regarding NORM were also applicable to some historical nuclear sites affected by past activities.

In EMRAS I, the NORM Working Group was initially limited to working with hypothetical scenarios [2]. In EMRAS II, the contribution of actual scenarios from participants of the Working Group and the availability of appropriate field data allowed the scope to be extended. This has resulted in a wealth of information to select from when comparing models for use in remediation planning and implementation.

The main types of NORM sites of interest include those linked to uranium mining and milling; the phosphate industry and phosphogypsum use and disposal; other mining and ore processing activities; coal mining; coal burning; NORM from the oil and gas industries; and NORM contaminated land (as opposed to factory or mine premises).

The main types of sites affected by past activities (referred to as 'historic sites' in this publication) are former nuclear technology facilities contaminated with anthropogenic radionuclides.

WG2 was established by creating a joint forum of regulatory authorities, modelers, and assessment specialists where the needs and possibilities of each contributor could be discussed, with a focus on selected sites affected by past activities (e.g. by NORM or other past activities) as case studies. The goal of the forum was to use actual site characteristics and management challenges to develop a reference approach for risk assessment and environmental impact assessment for the management and remediation of sites affected by past activities ('historic sites').

[1] Hereinafter referred to as EMRAS I.

A range of models have been developed over the years to perform risk assessments and radiological environmental impact assessments pertaining to radioactive waste or releases under planned and emergency conditions. These range from simple screening tools to sophisticated, complex modelling codes and have various applications. Some are available free of charge, whereas others are only commercially available. WG2 incorporated the use of 10 such models within the modelling exercises undertaken within EMRAS II. Descriptions of the models, including the key mechanisms modelled, application, and availability have been provided. Details of some other models that are potentially useful for modelling of historic sites have also been included to provide a source of reference for model users. Specific model–model intercomparisons were performed on the data and information available for the Gela site in Italy and the Bellezane site in France.

In this report, WG2 has provided a general assessment methodology for evaluating the radiological impact of historic sites in support of the management of such sites. This general assessment methodology provides more detailed steps, specifically related to assessment of impacts and risks in support of the implementation of GSG-15 [1]. The methodology is intended for assessors who are asked to provide evidence based advice on such sites to decision makers.

The methodology described can be used as part of a graded approach to optimize the type of assessment with respect to regulatory requirements and oversight, radiological protection, socioeconomic factors, cost, and other factors, for example, in support of selection of appropriate remedial options.

The general assessment methodology described in this publication was tested for several actual scenarios of exposure involving historic sites with NORM contamination, and with artificial radioactivity due to atmospheric nuclear weapons testing. These sites are in Australia, Belgium, Brazil, Italy and Norway.

EMRAS II WG2 used measured data from a wide range of contaminated sites to validate models, while demonstrating the practical application of IAEA recommendations relating to remediation planning and implementation. In addition, several scenarios were offered that have not yet been modelled by the group, but that are listed in the appendices to this publication. Such scenarios can be used to broaden the applicability of available models and the general assessment methodology described in this publication through providing examples for academia, industry, and regulatory authorities for testing of models in the future within the framework of current IAEA safety standards.

1. INTRODUCTION

1.1. BACKGROUND

A broad range of environmentally diverse sites have been affected by past activities globally. As a result, there is a need to evaluate the site specific radiological hazards and risks to people and the environment to determine whether remediation is justified, and if so, to develop and implement plans to address the situation. Historically, such sites, which have been affected by past activities, have either not been regulated or have not been managed in accordance with current standards. Due to the wide range of site specific environmental conditions, combined with the diversity of facilities and activities that have led to impacts on such sites, this has resulted in a wide range of prevailing circumstances and conditions, and the need for a systematic approach to evaluate and prioritize potential impacts and risks relating to past activities. To address this issue, the IAEA has developed guidance in IAEA Safety Standards Series No. GSG-15, Remediation Strategy and Process for Areas Affected by Past Activities or Events [1], which establishes a stepwise, multi-phased process to determine whether remediation is justified at a given site based on radiological criteria. In cases where remediation is deemed justified, GSG-15 provides recommendations on how to conduct optimization of protection and safety, with consideration of radiological and non-radiological impacts and risks, in support of planning and implementation of remediation.

In addition to such guidance, tools are needed to evaluate site specific impacts and doses, to identify practical remedial actions that can be implemented to mitigate risk, and to later verify the effectiveness of remediation to ensure that it has been implemented in accordance with the approved remediation plan. Environmental impact assessment, dose assessment and risk assessment models provide effective tools for use in remediation planning and implementation to mitigate hazards and risks relating to sites affected by past activities. As part of quality control, it is beneficial to independently test such models to validate model predictions, ideally, under conditions prevailing at actual sites.

From 2009 to 2011, the IAEA organized an international programme entitled ‘Environmental Modelling for RAdiation Safety’ (EMRAS II) that concentrated on the improvement of environmental transfer models and the development of reference approaches to estimate the radiological impacts on humans, as well as on flora and fauna, arising from radionuclides in the environment. The work of the EMRAS II programme provides technical information that can be used by relevant authorities to develop and improve radiological and environmental impact assessment approaches and tools to support them in meeting the requirements of IAEA Safety Standards Series No. GSR Part 3, Radiation Protection and Safety of Radiation Sources: International Basic Safety Standards [3], with respect to safety assessment for facilities and activities.

The following topics were addressed in nine working groups:

Reference Approaches for Human Dose Assessment

- Working Group 1: Reference Methodologies for Controlling Discharges of Routine Releases

— Working Group 2: Reference Approaches to Modelling for Management and Remediation at NORM and Legacy Sites[2]
— Working Group 3: Reference Models for Waste Disposal

Reference Approaches for Biota Dose Assessment

— Working Group 4: Biota Modelling
— Working Group 5: Wildlife Transfer Coefficient Handbook
— Working Group 6: Biota Dose Effects Modelling

Approaches for Assessing Emergency Situations

— Working Group 7: Tritium Accidents
— Working Group 8: Environmental Sensitivity
— Working Group 9: Urban Areas

The results achieved by the Working Groups are described in individual IAEA Technical Documents (IAEA TECDOCs). This publication applies the work of the Reference Approaches to Modelling for Management and Remediation at NORM and Legacy Sites Working Group in the context of recently published IAEA recommendations on remediation to address existing exposure situations, with specific reference to GSG-15 [1].

Prior to the work reported herein, the IAEA has run several international programmes that assessed the transport of radionuclides in the environment. These included the BIOMOVS (BIOspheric Model Validation Study), BIOMOVS II, VAMP (VAlidation of Model Predictions) and BIOMASS (BIOspheric Modelling and ASSessment) programmes. These programmes evaluated and used various models appropriate for simulating the behaviour of radionuclides in the environment. Although the Remediation Assessment Working Group under Theme 2 of the BIOMASS programme considered the application of models to a site contaminated with radium, the emphasis in these programmes was on those radionuclides produced in the nuclear fuel cycle. Some of the models developed for simulating the behaviour of these anthropogenic radionuclides in the environment need considerable adaptation for use in assessing the impact of NORM in the environment, particularly for naturally occurring radionuclides with long half-lives and long decay chains.

While many of the earlier programmes focused on nuclear accidents and radioactive fallout and the environmental transfer and impact of anthropogenic radionuclides, they have not focused on assessment methodologies and modelling for management and remediation of different types of sites affected by past activities using data from actual sites. In EMRAS I, there was a specific working group on NORM; however, unlike EMRAS II WG2, the scenarios considered in EMRAS I were hypothetical in nature, as opposed to cases studies for actual sites, and the EMRAS I Working Group did not cover the wide range of sites dealt with in this publication [2, 4]. Historic NORM sites that might be of interest include those linked to the phosphate industry and phosphogypsum use and disposal; other mining and ore processing activities; coal

[2] Due to the wide range of definitions of 'legacy sites', although the title of Working Group 2 originally referred to such sites, this term has been replaced with the broader term 'historic sites' in this publication, consistent with the case studies considered here within. In addition, the scope has been clarified to indicate that focus is limited to existing exposure situations (e.g. for NORM sites) and that planned exposure situations are not within the scope.

mining; coal burning; NORM from the oil and gas industries; and NORM-contaminated land (as opposed to factory or mine premises).

Historic nuclear sites that might be of interest include those used for storage of spent nuclear fuel (SNF) and radioactive waste, decommissioned or abandoned nuclear facilities which had not been operated in accordance with current standards, areas used for nuclear weapons development and testing, areas contaminated by a nuclear or radiological emergency, uranium mining and processing sites, and radium extraction and processing facilities.

The models necessary for performing risk assessments[3] and radiological environmental impact assessments (REIAs)[4] are developed by researchers and modelers. Operating organizations, technical support organizations, assessment practitioners, the regulatory body and other relevant authorities use such models in support of sound decision making, for example, in approval of remediation plans [1]. The regulatory body and other relevant authorities ensure that the remediation process, end point criteria and the end state criterion comply with national regulations and authorization conditions, consistent with international recommendations and guidance on health, safety, and the environment (see Sections 3.1 and 3.2 for discussion of remediation criteria). In addition, the operating organization for a given site has the detailed data on waste and site characteristics, which are crucial for performing site specific predictive modelling. In developing adequate models, researchers need to be familiar with:

(1) The legal and regulatory requirements that operating organizations need to comply with;
(2) Actual situations that operating organizations are dealing with.

The regulatory body and other relevant authorities, on the other hand, need information on how to use models when reviewing or performing independent risk assessments and environmental impact assessments; this includes both guidance on which parameters to include, how suitable a model is for a given site and the uncertainties associated with the model predictions. Experience has shown that effective community and consultation between regulatory authorities, operating organizations, and modelers and assessment specialists on this topic can be beneficial, recognizing the need to maintain the effective independence of the regulatory body, as established in Requirement 17 of IAEA Safety Standards Series No. GSR Part 1 (Rev. 1), Governmental, Legal and Regulatory Framework for Safety [5].

1.2. OBJECTIVE

EMRAS II WG2 was established to develop or improve assessment methodologies and modelling for the management and remediation of different types of sites affected by past activities (including historic NORM sites and nuclear legacy sites). This was implemented by creating a joint forum of regulatory authorities, operating organizations, modelers, and assessment specialists, and by using selected sites as case studies. The objective was to use actual site characteristics and challenges in the management of such sites within a reference approach for risk assessment and environmental impact assessment appropriate to the

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[3] A risk assessment is defined as an "Assessment of the radiation risks and other risks associated with normal operation and possible accidents involving facilities and activities. This will normally include consequence assessment, together with some assessment of the probability of those consequences arising" [6]. The risk assessment is typically part of a safety assessment, which is defined as "The systematic process that is carried out throughout the design process (and throughout the lifetime of the facility or the activity) to ensure that all the relevant safety requirements are met by the proposed (or actual) design" [6].
[4] A radiological environmental impact assessment is defined as "Assessment of the expected radiological impacts of facilities and activities on the environment for the purposes of protection of the public and protection of the environment against radiation risks" [6].

management and remediation of actual sites affected by past activities. The objective of this publication is to present the results generated from modelling of case studies by EMRAS II WG2 in the context of the stepwise remediation process presented in GSG-15 [1] to demonstrate its practical application.

1.3. SCOPE

This publication presents an approach that has been developed for application in risk assessment and REIA in support of management and remediation of sites that have been affected by past activities. Such sites fall within the definition of existing exposure situations [3] and may or may not still be operating. Although the approach presented in this publication might be applied to areas affected by emergencies, such situations are not the focus; instead, this publication focuses on sites affected by activities that were not undertaken in accordance with current standards. Planned exposure situations are also out of the scope of this publication.

The approach presented in this publication can help to ensure that such assessments are based on sound science, suitable for regulatory purposes and consistent with IAEA recommendations and guidance. With the exception of planned activities relating to an approved remediation plan (e.g. including radiation protection of remediation workers), which has been developed to address an existing exposure situation, facilities and activities relating to planned exposure situations (e.g. the operation of planned NORM facilities) are out of the scope of this publication.

WG2 recognized that even though generic international recommendations are available, the development of a reference approach for management and remediation that allows for national or local adaptation, taking account of country specific and site specific factors, would be advantageous. Therefore, WG2 considered international recommendations, national regulations, actual sites, and available modelling tools in developing a reference approach, and in subsequently testing the methodology for actual sites around the world. This work has subsequently been placed within the context of the systematic remediation process presented in GSG-15 [1] to demonstrate its practical application for real situations.

This publication and the assessment methodology that it presents are intended as guidance for assessors who are asked to provide scientifically based advice to decision makers on radiological issues associated with historic sites. The methodology is not intended to include the decision making process, but it is recognized that it needs be integrated into that process.

This publication focuses on radiological considerations, recognizing that the decision maker needs to consider many factors, in addition to the radiological issues, including:

- — Legal and regulatory requirements;
- — Possible recycling and reuse of residues generated during operation of a nuclear facility or activity;
- — Non-radiological issues, such as the presence of other toxic substances (such as heavy metals, wastewater, process water and chemicals, biohazards), physical hazards (such as friable asbestos, unstable buildings and structures, unstable slows, mine openings);
- — National policies and strategies that can differ depending upon socioeconomic and political considerations;
- — Public perceptions, attitudes and expectations.

The latter three factors highlight the great importance of communication and consultation with interested parties, such as between the responsible party or operating organization, the regulatory body and other relevant authorities, the public and special interest groups, as part of the process of remediation of contaminated sites and areas (see Ref. [1]).

1.4. STRUCTURE

Section 2 of this publication describes the challenges related to sites affected by past activities, including historic NORM and historic nuclear sites. Section 3 provides the framework for planning and implementation of remediation to address such sites, considering recommendations in the IAEA safety standards.

In support of remediation, Section 4 describes a general assessment methodology for performing risk assessment and environmental impact assessment.

Available models for radiological risk assessment and environmental impact assessment are described in Section 5. These range from simple screening tools to sophisticated modelling codes and simulation systems.

Section 6 provides examples of the application of the general assessment methodology, described in Section 4, to various actual sites (Gela in Italy, Botuxim in Brazil, Soeve in Norway, Maralinga in Australia, and Tessenderlo and Olen in Belgium) applying the modelling approaches described in Section 5.

Model–model intercomparisons for the Gela phosphogypsum stack in Italy and the remediated uranium mining site of Bellezane in France are described in Section 7.

Appendices I–IV provide a detailed description of a number historic nuclear and historic NORM sites that have been affected by past activities around the world, highlighting the variety of challenges and conditions that exist. Specifically, Appendix I provides an overview of the structure used to present information for the specific cases described in appendices that follow. Appendix II describes a number of examples of historic nuclear sites, Appendix III describes examples of historic uranium mining and milling sites, and Appendix IV describes a number of historic NORM sites other than those related to uranium.

Appendix V provides details relating to the various modelling tools summarized in Section 5.

Annex I provides examples of current regulations relevant to the management and remediation of sites affected by past activities.

The bibliography provides a list of reference publications that contain additional information on specific technical, organizational, financial and safety issues relating to remediation.

2. CHALLENGES RELATED TO SITES AFFECTED BY PAST ACTIVITIES

Different types of sites have been affected by past activities, including historic NORM sites (Section 2.1) and nuclear legacy sites[5] (Section 2.2), which are the focus of this publication. Many such sites exist around the world and there is concern about how to make sound decisions on the future management and necessary remediation of these sites to ensure protection of people and the environment now and in the future.

The term 'historic sites' is taken in this publication to mean sites that have been operated in the past not in accordance with current standards, resulting in a loss of control of radioactive material, and need, or are expected to need, some form of remediation to adequately protect people and the environment. In the IAEA safety standards, such sites are classified as 'sites affected by past activities' [1]. The sites may be contaminated by anthropogenic radionuclides or by NORM. Some of these sites might still be operational or under some form of control, or might have been completely abandoned. A common feature of such sites is that they were formerly operated without adequate controls, leading to radioactive contamination of larger areas, including beyond the site perimeter. The sites themselves and these larger areas might need remediation and fall within IAEA's definition of 'existing exposure situations'[6] [3] (see also ICRP 103 [7]).

Table 1 provides an overview of the actual sites presented and discussed by EMRAS II WG2. The sites identified in *italics* are described in more detail in Appendices I–IV. The phosphogypsum stack of Gela (Italy) is described in Sections 6.1 and 7.1 and the uranium tailings repository of Bellezane (France) is described in detail in Section 7.2 as part of the model–model intercomparisons. The Megalopolis and Kavala sites in Greece were described in the EMRAS I report [4]. The Soeve historic NORM site in Norway and the Maralinga nuclear test site in Australia are described in Sections 6.3 and 6.4, respectively, covering the application of the general assessment methodology presented in this publication to selected sites.

[5] Different countries use different definitions to describe 'legacy sites'. In a number of countries, such sites have been abandoned and have no owner.
[6] A situation of exposure that already exists when a decision on the need for control needs to be taken.

TABLE 1. OVERVIEW OF ACTUAL SITES DISCUSSED BY EMRAS II WORKING GROUP 2

Country	Category	Name of site[a]
Argentina	*Historic uranium mining and milling site*	*Los Gigantes*
Australia	Historic nuclear site	Maralinga nuclear test site
Belgium	Historic nuclear site	Olen (radium production)
	Historic NORM site	*Tessenderlo (phosphate processing)*
	Historic NORM site	Other Belgian historic NORM sites (phosphogypsum stacks, ferro-niobium processing)
Brazil	*Historic uranium mining and milling site*	*Poços de Caldas*
	Historic NORM site	*Botuxim site (storage of residues from monazite processing)*
Bulgaria	Historic uranium mining and milling site	’Iskra’ site (Katina)
		Uranium milling plant ’Zvezda’
China	*Historic NORM site*	*Baotou sites (rare earth extraction and steel industry)*
Estonia	Historic nuclear site	Paldiski (former nuclear submarine training facility)
	Historic uranium mining and milling site / legacy	Sillamäe (Uranium mine tailings + former nuclear material production facility)
France	Historic uranium mining and milling site	Uranium mining sites of the Limousin region
Greece	Historic NORM site	Megalopolis – Coal fired power plant
	Historic NORM site	Kavala phosphogypsum stack
Italy	Historic NORM site	Gela phosphogypsum stack
Norway	Historic NORM site	Soeve site (Niobium mining and processing)
Poland	*Historic NORM site*	*Sites in the Upper Silesia coal basin*
Russia	*Historic nuclear site*	*Andreeva Bay, the Lepse vessel*
Slovenia	*Historic uranium mining and milling site*	*Žirovski vrh wastes piles*
Spain	*Historic NORM site*	*Compostilla – Coal fired power plant*
Ukraine	Historic uranium mining and milling site	Pridneprovsky plant (Dniprodzerzhinsk)
USA	Historic uranium mining and milling site	Abandoned uranium mine sites on Navajo Nation territories

[a] The sites listed in *italics* are described in more detail in Appendices I–IV.

2.1. HISTORIC NORM SITES

Of all anthropogenic sources, NORM contributes the largest collective doses to the world's population and can, therefore, result in radiological risk, the magnitude of which is dependent upon the characteristics at a given site [8−9]. For the purposes of this publication, and in accordance with the IAEA Safety Glossary [6], NORM is defined as "radioactive material containing no significant amounts of radionuclides other than naturally occurring radionuclides"[7], and for which the radionuclide activity concentrations and corresponding exposures have been modified by anthropogenic activities. Many countries have historic sites contaminated due to past NORM activities. These sites resulted from past activities that were never subject to regulatory control, or that were subject to regulatory control but not in accordance with current standards, for example, in situations where no decommissioning or remediation was carried out after active operations ceased where it could now be justified (see GSG-15 [1]). The focus in this publication is exposures exceeding natural background radiation levels relating to sites affected by past activities.

For the assessment of impact of NORM on human health and the environment, notable features include the presence of:

- A large number of different radionuclides;
- A very broad range of radioactive half-lives;
- A range of physical forms and chemical properties.

There are several additional factors that are important for the management of historic NORM sites:

- Regulatory aspects, where the emphasis is shifting from limitation of dose to setting the reference level, along with additional criteria through the process of optimization of protection and safety to avoid exceeding what is considered to be 'acceptable risk';
- Ensuring adequate characterization of NORM, including radiological and non-radiological contaminants, as input to remediation planning and implementation;
- NORM can consist of small volumes with high activity concentrations and large volumes with low activity concentrations and low levels of radiation exposure;
- Many minerals give rise to NORM-containing products and by-products that are not exploited for their radionuclide content, as well as wastes and other residues. Radionuclides in these materials, resulting from the processing of these minerals, might complicate their management and potential for reuse;
- Radon gas produced by decay of primary radionuclides in NORM creates a secondary hazard that has to be considered in the management of wastes and other residues;
- Reuse of materials contaminated with NORM is frequent (and makes use of large volumes of these materials);

[7] These materials are referred to in some countries as TENORM (technologically enhanced naturally occurring radioactive materials).

— Large areas of land are needed to store NORM residues, and projected land use is an important factor when discussing possible remedial actions[8].

It was noted in the EMRAS I report on NORM that all countries probably have industries that produce NORM wastes and other residues [4]. Waste is any material for which no further use is foreseen. Nowadays, there is pressure to reuse or recycle residual materials from non-product streams to minimize waste generation and keep operating costs low to the extent possible. Such residual materials are not considered as wastes (also see [1]).

The industries and processes that produce residual materials (including wastes) containing NORM, in some cases, leading to sites affected by past activities, are listed in the EMRAS I report on NORM [4]. References to detailed descriptions of most of the relevant industries were provided in Ref. [4] and a summary is given in Table 2, reproduced from Ref. [4] (see also Refs [10−21]). Detailed descriptions of many of these industries are also given by UNSCEAR [8−9].

In cases where such industries and processes have not been managed in accordance with current standards, when planning and implementing remediation, it is necessary to consider the types and forms of wastes and other residual materials that might have been generated historically. During remediation, the characterization and classification of residual materials on a site can provide important information in support of identification of appropriate remedial actions, possibly involving recycling and/or reuse of residual materials (e.g. slightly contaminated rock) to limit the generation of waste that would require long term management and oversight [1].

TABLE 2. LIST OF THE MAJOR INDUSTRIAL PROCESSES THAT GENERATE NORM RESIDUES AND THE TYPES OF PRODUCTS, WASTES AND OTHER RESIDUES GENERATED (reproduced from Ref. [4])[a].

Practice	**Products**	**Form of wastes or other residues**	**NORM wastes or other residues**
Mining and milling	Mineral	Liquids and solids	Tailings, process water, scales, evaporates
Mineral processing	Metal	Scales, sludges, volatiles	Slags, tailings
Phosphate industry	Fertilizer, phosphoric acid	Liquids and solids	Phosphogypsum, scales
Power generation (fossil fuels)	Electricity	Solids and gases	Ash, mine water
Oil and gas production	Oil, gas	Liquids and solids	Scales, sludges, process water
Water treatment	Potable water	Liquids and solids	Sludges, bio-solids, scales, contaminated filters and residuals

[a] Additional information relating to practices involving NORM is available in Refs [12−21].

[8] A remedial action is “The removal of a source or the reduction of its magnitude (in terms of activity or amount) for the purposes of preventing or reducing exposures that might otherwise occur in an emergency or in an existing exposure situation. Remedial actions could also be termed protective actions, but protective actions are not necessarily remedial actions” [6].

2.1.1. Awareness issues

Past activities were often undertaken with little awareness of the issues surrounding NORM. The operations of many NORM industries were not regulated or assessed for the potential radiological impacts of NORM on human health and the environment, which leads to three principal considerations:

- Many countries have challenges with the management of historic wastes, including those that arose from mining and mineral processing;
- Monitoring of many historic sites and their surroundings was not necessarily required in the past, so that currently available data do not always provide sufficient detail for modelling studies;
- Misuse of NORM wastes or historic sites can result in ‘unacceptable’ radiation exposures to members of the public (e.g. building houses from uranium mining or milling wastes or on sites containing NORM).

2.1.2. Current challenges in regulation

More than two decades ago, the US National Academy of Sciences evaluated international guidance for NORM, including its scientific basis, and resulting risk management recommendations [12]. It was found that nearly all guidance was based on the same set of epidemiological studies (e.g. the cohort study on underground uranium miners). However, the risk management guidelines depended on organizational or national policy concerning acceptability of risk. Thus, the risk of long-term mortality from exposure to a source of NORM deemed to be acceptable for a member of the public varied from 1 in 1000 (1×10^{-3}) to 1 in 10 000 (1×10^{-4}). It was noted that this might lead to different regulatory requirements between countries for the same industry (zircon producers, for example).

Although legal and regulatory frameworks have now been put in place in many countries to address NORM sites affected by past activities (in existing exposure situations), the regulatory situation varies from country to country, and work is ongoing to implement recent international standards for managing wastes and other residues containing NORM (e.g., see IAEA Safety Standards Series No. SSG-60, Management of Residues Containing Naturally Occurring Radioactive Material from Uranium Production and Other Activities [10]). Lack of uniformity has implications for assessing the behaviour of NORM in the environment and its potential adverse impacts, as well as for international trade [11]. In countries where there is regulatory oversight of NORM facilities and activities, there is likely to be a systematic approach to monitoring and assessment, including when and how mitigative measures may be needed. However, in cases where there is a lack of regulation, the generation of wastes and other residues containing NORM is unlikely to be adequately controlled, potentially leading to additional contaminated sites coming into existence in the future.

Nowadays, in order to prevent future NORM legacies, an environmental impact statement is often required for any application for a licence or permit to handle materials containing NORM, particularly for surface or near-surface waste and/or residue disposal. This is ideally based on the known characteristics of the disposal site (e.g. based on measured data), good knowledge of the materials involved, and a technically robust assessment of the impacts of the facility on the health of the workforce, the surrounding environment and members of the public who live near the site. However, at the start of operation of a facility, an environmental impact assessment may be only provisional in nature. Once the operation of the facility has been initiated and site specific monitoring data become available, the assessment can be refined and adjusted to ensure that the impacts on human health and the environment remain within acceptable criteria and consistent with overall principle of optimization of protection and safety. At the start of an activity involving NORM residue management or waste disposal, mostly generic models are used for predictive assessment. As site specific monitoring data become available, the models can be applied to take account of site specific factors, for example in the environmental transfer processes simulated and the parameter values used, improving predictiveness of the models with respect to the specific site and activity under consideration. This iterative improvement process is important because of the long half-lives involved compared to the operational (and institutional) lifetime of a managed facility.

2.1.3. Retrospective assessment of historic wastes and other residues from past activities

Past activities involving the handling of wastes and other residues containing NORM were often not under regulatory control or were not regulated in accordance with current standards. This resulted in the existence of so-called ‘historic sites’ that have been affected because of past activities and that have not been assessed (or robustly assessed) in terms of potential hazards, risks and impacts. Models can be used in evaluating options for management of such sites, for example:

— As an aid in assessing impacts of the site to human health and the environment in its current state, and in deciding if remediation may be justified (i.e. during the ‘preliminary evaluation’ and ‘detailed evaluation’ phases of remediation; see GSG-15 [1]);
— In assessing the effectiveness of proposed remediation strategies and remedial options[9] (as part of optimization of protection and safety in remediation planning; see GSG-15 [1]), if the initial assessment (i.e. the preliminary evaluation and detailed evaluation) suggests some form of remediation is justified;
— In assessing the impacts of future use of the site and of possible misuse of NORM residues (e.g. in building materials, potentially resulting in public exposure).

However, because for many sites, there may have been little or no attention paid to site characterization or the design of facilities where the wastes and other residues were placed, site specific conditions, such as the layout, geology and hydrogeology of many of these sites are not available. In such cases, generic models are not easily applicable.

In line with current practice, monitoring and prediction of environmental impacts are included as regulatory conditions in support of the design and the operation of any new facility for management of waste and other residues. In support of such activities, models may be applied to select appropriate sites for facilities for the storage and disposal of waste and other residues, taking account of site characteristics (e.g. soil type, geology, hydrogeology, climatic conditions). Through site specific application of models, assessment of impacts of stored or disposed of material to people and the environmental then becomes possible.

[9] “A remedial option is a possible action or set of actions that might be undertaken to implement the remediation plan. Options that have been selected for implementation become remedial actions. Active remedial options involve earth-moving or other physical work, and passive options involve allowing natural dispersion and decay (e.g. natural attenuation) to reduce the hazards and include land use control. Both active and passive options involve monitoring and surveillance to verify the option is behaving as expected, in accordance with the remediation plan and the authorization” [1].

In adopting such an approach, two classes of models are needed for use in human health and environmental impact assessments:

(1) Generic (or screening) models used where available data do not allow detailed site specific assessment;
(2) Site specific models that can be updated as more data are collected from site specific and area specific monitoring programmes.

Many currently available models can be used for both screening and site specific assessments: default values for the model parameters can be used to begin with and later updated when site specific data become available from monitoring programmes. In addition, models implemented in general purpose simulation packages (see Section 5) can be altered structurally as more information and data become available concerning a specific site. Thus, it is possible to adapt both the structure and the parameterization of a model to provide a more appropriate representation of site specific characteristics.

2.1.3.1. Examples of typical scenarios resulting in NORM exposure

Historic NORM sites are those for which contamination occurs due to naturally occurring radioactive material and that were never subject to regulatory control or have not been operated in accordance with current standards. Some typical situations that can result in exposure to NORM include:

- Decommissioned, remediated or abandoned nuclear facilities, including nuclear fuel cycle facilties;
- Uranium mining and milling sites, where the purpose was to extract uranium for use in the nuclear fuel cycle;
- Radium extraction and processing facilities;
- Storage of wastes and other residues in waste stacks, waste rock piles, and tailings disposal facilities, including ponds and solid deposits in formerly ponded areas;
- Cleaning of scales from pipes and other production equipment;
- Recycling of scrap metal;
- Reuse of NORM residues in building materials (fly ash, phosphogypsum);
- Reuse of NORM residues for landfill (fly ash and waste rock) and land spreading (phosphogypsum and red mud), taking account of the future land use[10] when assessing the potential radiological risk associated with reuse;
- Releases of ^{210}Pb and ^{210}Po from fossil fuel power stations (these are volatile at common temperatures in conventional power stations and can be released up the stack and to the surrounding environment).

For such situations, a range of possible pathways of exposure might need to be considered. Selection of scenarios of exposure that are appropriate for the site and the prevailing circumstances under consideration is an important part of the assessment process. The approach to scenario selection and development is described in detail in the Improvement of Safety

[10] Typically, land can be used for residential, industrial, recreational, or agricultural purposes.

Assessment Methodologies for Near Surface Disposal Facilities ('ISAM') reports11 (see Refs [22–23]). This approach is based on a systematic analysis of the features, events and processes (FEPs) associated with each site being assessed (see Ref. [24].

2.1.4. Monitoring in support of assessment

The current practice in many countries obliges the responsible party to perform routine monitoring programmes when handling, processing, storing or disposing of materials that might contain elevated activity concentrations of radionuclides and other contaminants related to NORM facilities and activities. Monitoring data can be used to verify the ongoing effectiveness of management procedures for waste and other residues and in computer models to assess human health and environmental impacts over the lifetime of the facility or activity.

2.1.4.1. On-site monitoring and assessments

On-site monitoring and assessments are primarily performed to assess the impacts of a NORM facility or activity on the workers at the site and the environment (e.g. impacts to wildlife and their habitat, groundwater and surface water quality, soil quality).

2.1.4.2. Off-site monitoring and assessments

Off-site monitoring and assessments are performed to determine the impacts of a NORM facility or activity on the health of members of the public and on the surrounding environment.

2.1.5. Current trends in NORM residue management

Emphasis is currently placed on the development of methodologies for management of contaminated residues and sites [25] aiming to:

(1) Establish an iterative process for managing the impacts of NORM facilities and activities that allows for changes to be made as more information and/or data become available (e.g. on health impacts or from source and environmental monitoring);
(2) Engage and build confidence amongst all interested parties (sometimes called 'stakeholders'), ensuring that the management process is understood and that they are involved in the decision making process.

2.2. HISTORIC NUCLEAR SITES

Past development of commercial and military uses of radioactive material led to the development of numerous nuclear facilities worldwide. In many cases, facilities were built and operated before adequate legal and regulatory infrastructure was in place to ensure they were safely decommissioned with the radioactive and other wastes safely and securely managed and disposed of at the end of their useful life. As a result, many countries have contaminated lands and abandoned nuclear facilities. This includes sites with inadequate facilities for storing SNF and radioactive waste, or areas where uncontrolled releases (e.g. due to spills or accidents) have occurred resulting in radioactive contamination that has created risks to human health and

[11] All publications of the Improvement of Safety Assessment Methodologies for Near Surface Disposal Facilities (ISAM) Co-ordinated Research Project (CRP) are available for download from: http://www-pub.iaea.org/books/IAEABooks/6971/Safety-Assessment-Methodologies-for-Near-Surface-Disposal-Facilities [22–23].

environmental impacts. In many cases, due to their history, for such contaminated lands and abandoned facilities, there is no responsible party (e.g. no owner).

2.2.1. Types of historic nuclear sites

Historic nuclear sites are those for which contamination occurs due to artificial radionuclides and that were never subject to regulatory control or have not been operated in accordance with current standards. Such sites might include:

— Decommissioned, remediated or abandoned nuclear facilities, including research facilities, research reactors, or power plants;
— Sites for storage and/or disposal of SNF and radioactive waste with inadequate or badly degraded containment, already causing unacceptable leaks of radionuclides, or being likely to so do in future if not improved;
— Areas affected to a level that is considered to be 'unacceptable' level by interested parties due to nuclear weapons development and testing, or nuclear test explosions;
— Areas contaminated by a past nuclear or radiological emergency on a site.

These types of sites typically can pose significant challenges to the responsible party with operational and/or management responsibility for their remediation because of a wide range of issues and challenges arising, such as:

— Need for physical protection of areas from a few hundred square meters up to tens of square kilometers, depending on the nature of the conditions relating to the site;
— Poor physical condition of facilities, degraded containment systems, obsolete or faulty engineering facilities;
— Proximity to communities, sometimes close to major cities, sometimes in very remote areas, both of which are problematic and can lead to public exposure, as well as non-radiological hazards (e.g. relating to travel to remote sites);
— Poorly characterized radioactive materials, for example SNF in degraded stores, or large volumes of low-level contamination, which has been dispersed and sometimes is not contained at all;
— Poorly characterized chemotoxic and other dangerous material (e.g. friable asbestos);
— Physical hazards;
— Need for security, as necessary, depending on the site and the conditions, for example, to prevent site access by unauthorized persons to prevent injury due to site specific hazards;
— General radiological protection, as necessary, depending on the site and the conditions, for example, during maintenance work or monitoring by authorized individuals;
— Optimization of protection and safety during the recovery of hazardous materials and the planning and implementation of remediation;
— Control of materials and prevention of accidents during the recovery of hazardous materials and the planning and implementation of remediation;
— Transfer of former military facilities from military to civilian regulatory supervision, involving security and safeguards;
— Geographic conditions of climate, geology, hydrology, and landscape all subject to environmental change;
— Land use planning within the wider social and economic context (e.g. including the need for processes for funding and financing remediation);

- Multiple expectations and concerns of interested parties, short and long term;
- Coherence of management of existing exposure situations (in this case, sites affected by past activities) with the national radioactive waste management programme and other relevant programmes (e.g. national policy, strategy and priorities on remediation), and availability of radioactive waste disposal routes.

Descriptions of four different types of historic nuclear sites, illustrating the range of issues above, are provided in this publication. These include:

- The nuclear test site of Maralinga in Australia (Section 6.4);
- The former radium production facility in Olen, Belgium (Section 6.6);
- The Lepse Technical Support Vessel used to store SNF and solid and liquid radioactive waste, currently located in the north-western part of Russian Federation and awaiting dismantling (Appendix II);
- The site of temporary storage for SNF and radioactive waste at Andreeva Bay, in the Kola Peninsula, Russian Federation (Appendix II).

2.2.2. Safe management of historic nuclear sites

Safe management of historic nuclear sites arising from past activities is an important issue in maintaining confidence in the continuing and future use of radioactive material. Effective and efficient regulatory supervision of historic nuclear sites and their management is a key part of this process (e.g., see GSG-15 [1]).

Regulatory supervision over the management of historic sites and facilities needs to address site specific and facility specific challenges, with adequate consideration of many different issues in support of decision making. In doing so, it is necessary to take a balanced approach that identifies and takes into account relevant factors, while ensuring protection of human health (workers, members of the public) and the environment, in planned and unplanned situations, and in both the short and long term.

Although international recommendations and guidance exist on planning and implementation of remediation (e.g., see Section 5 of GSR Part 3 [3]; GSG-15 [1]; GSG-11 [26]), currently, further elaboration is needed to develop comprehensive and consistent international technical guidance that can be applied holistically to implement international recommendations and to address the complex set of issues discussed above (e.g., see Section 2.1 in relation to historic NORM sites). Therefore, it is beneficial to systematically compile and review relevant cases, summarizing successes and lessons learned in dealing with existing exposure situations under a range of prevailing circumstances and conditions. Such examples could be particularly beneficial in cases where Member States are establishing or strengthening the national legal and regulatory framework to address historic sites in line with current international standards and good practice.

Substantial progress in managing historic nuclear sites within national legal and regulatory frameworks, in line with current IAEA safety standards, has been achieved, for example, in the USA and the Russian Federation, although this is an ongoing process internationally [27].

Some examples of current legal and regulatory frameworks in Member States are presented in Annex I. These examples are only illustrative and do not provide a comprehensive analysis.

2.2.3. Types of assessment relating to historic nuclear sites

The needs of, and radiological protection objectives for, historic nuclear sites might be expected to be similar to those for historic NORM sites, since the issues are broadly similar. A number of references describing different aspects of assessments are mentioned in Section 5, for example, those related to the BIOMOVS [28], BIOMOVS II [29] and BIOMASS studies [30]. Among the key differences is the set of radionuclides involved, with anthropogenic radionuclides representing a specific set of radioactive decay characteristics, chemical forms and environmental behaviours, which differ from those of naturally occurring radionuclides.

National guidance on assessments is provided in some countries. For example, the Belgian Federal Agency for Nuclear Control (FANC) has issued two technical guides regarding what needs to be covered in characterization studies for a contaminated site and regarding the intervention levels used, as follows:

- ‘Generic content of an orientation and descriptive study’ [31];
- ‘Intervention levels for prolonged exposure situations’ [32].

The environmental impact assessment in Belgium is a two step process, which involves:

(1) An ‘orientation study’ to validate the presence of contamination and to provide an initial indication (screening) of the associated risk.[12]
(2) In cases where the presence of contamination is confirmed, a ‘descriptive study’ to assess the radiological risk.[13]

The orientation study contains at least the following elements:

- Administrative data for the site: identifying the areas of contamination, the identity and contact details of the owner and user of the site, the current use of the site, map of the site, and other information, as relevant;
- Description of the history of the site (successive activities carried out on site, description of the processes used in these activities, copies of former authorizations, and other information, as relevant);
- Geological and hydrogeological data.

Based on these elements, initial estimates of the contamination can be made. These can be used to guide a basic radiological characterization, which is also part of the orientation study.

The descriptive study provides a detailed characterization of the contamination, including:

- Horizontal mapping and vertical profile of the contamination;
- Localization of hot spots;
- Study of the possible migration of radionuclides, in particular with respect to groundwater contamination;

[12] The ‘orientation study’ is equivalent to the ‘Preliminary Evaluation’ phase of remediation in GSG-15 [1].
[13] The ‘descriptive study’ is equivalent to the ‘Detailed Evaluation’ phase of remediation in GSG-15 [1].

— A dose assessment (see Ref. [33]).[14]

The dose assessment that is carried out in the descriptive study, needs to take into account at least three scenarios of exposure:

— The current use of the site;
— A worst-case scenario, i.e., a realistic scenario that leads to the highest exposure (often an intrusion scenario, for example, involving building dwellings on the site);
— A 'likely' scenario that corresponds to a likely evolution in the use of the site from its current use.

Based on the dose assessment that has been carried out by the responsible party in the descriptive study, the regulatory body or other relevant authorities will decide on whether or not remediation is justified and whether the proposed remediation plan will be approved. The intervention levels, which are applied in some countries (e.g. Belgium) have been described in Section 6.5.4. These are equivalent to the concept of the screening criterion[15] (to determine whether remediation might be justified) and the reference level (to determine whether remediation is justified) in GSG-15 [1] and indicate whether or not remediation is justified, based on radiological criteria.

2.2.4. Current challenges in regulation

Some key technical issues relevant to regulation of radiation protection for historic nuclear sites are:

— Characterization of existing and potential accident source terms for releases of radioactive material;
— Mechanisms for planning and control of radiation exposure during activities involving hazardous conditions;
— Emergency preparedness and response;
— Gaining understanding of local radionuclide migration and accumulation;
— Evaluation of potential public exposures and environmental impacts in the short and long term.

National authorities might already have appropriate policy, strategy, legal and regulatory framework, and corresponding guidance documents for dose assessment, risk assessment and risk management for contaminated sites in place. Where such infrastructure has been established, these guidance documents can be used to determine which standards and criteria are appropriate for site modelling, evaluation, and decisions on remedial action. In those cases where there is no national infrastructure and corresponding guidance on management of contaminated sites are available, these need to be developed; in the interim, the requirements

[14] 'Dose assessment' is defined as an "Assessment of the dose(s) to an individual or group of people. … The term *exposure assessment* is also sometimes used" [6]. A dose assessment is undertaken as part of a radiological environmental impact assessment in the "Assessment of the expected radiological impacts of facilities and activities on the environment for the purposes of protection of the public and protection of the environment against radiation risks" [6] (see also GSG-10 [34]). Radiological environmental impact assessment is part of an overall environmental impact assessment.

[15] The screening criterion is a conservative criterion that is applied during the preliminary evaluation phase of remediation to determine whether or not remediation might be justified (e.g. the lower level of the reference level range, as established in the national strategy for remediation) (see GSG-15 [1]). It can be established by the responsible party or the regulatory body, depending on the national legal and regulatory framework, and needs to be approved by the regulatory body and documented in the approved remediation plan.

established in Section 5 of GSR Part 3 [3] and in GSR Part 1 (Rev. 1) [5], and the recommendations provided in GSG-15 [1] may be applied.

The relationship between setting of standards, prognostic assessment of impacts and the approval of remediation activities is ideally explored within the context of an overall strategy for management of historic sites and radioactive waste management, including disposal. Consideration is also ideally given to arrangement of effective communication and consultation interfaces, important in the support of good decision making and the avoidance of creation of future existing exposure situations (including nuclear legacy sites). International cooperation activities, such as the IAEA's Regulatory Supervision of Legacy Sites (RSLS)[16] programme [35] and similar initiatives (e.g., see Refs [36-47]), also in addition to bilateral programmes, play an increasing role in resolving historic issues. For example, the objective of RSLS is to promote effective and efficient regulatory supervision for the management of historic sites (e.g. including legacy sites), consistent with IAEA guidance [1, 3−4, 22, 48] and good international practice. This is realized through exchange of information, experiences and expertise relating to historic nuclear sites, and discussion regarding how regulatory supervision can be implemented and maintained.

Activities of RSLS [35] are organized under three working groups as follows:

- *Working Group 1, Enhancing the Regulatory Infrastructure*: This Working Group is focused on the review of the experience of regulatory authorities in planning the management and regulatory supervision of sites affected by past activities in order to make recommendations for enhancement of the regulatory infrastructure. Such recommendations may be developed from working with responsible organizations carrying out practical activities at actual sites.
- *Working Group 2, Safety Assessment Methods and Environmental Impact Assessment*: This Working Group is focused on the application of methods for safety assessment and environmental impact assessment for application in support of the management of sites affected by past activities.
- *Working Group 3, Professional Development for Regulatory Authorities*: This Working Group is focused on the professional development and training of regulatory staff for supervision of historic sites (including legacy sites).

RSLS is designed to be inclusive regarding the scope of issues relating to past activities (existing exposure situations), using a working definition of a historic site as a facility or area that has not completed remediation and is radioactively contaminated at a level that is of concern to regulatory authorities. The results in this publication are anticipated to provide useful input to the RSLS, particularly to Working Group 2 on Safety Assessment Methods and Environmental Impact Assessment. Regulatory decisions clearly need to be backed up and supported by safety assessments and environmental impact assessments. However, it is recognized that the detailed nature of the type and level of assessment that is necessary will depend on the specific aspects, the prevailing circumstances, and conditions relating to a historic site.

International programmes, such as RSLS, are not only useful through the sharing of knowledge and experience but are necessary because sites affected by past activities and their management can have transboundary impacts.

[16] http://www-ns.iaea.org/projects/rsls/legacy-sites-reg-supervision-tm.asp

3. FRAMEWORK FOR REMEDIATION OF SITES AFFECTED BY PAST ACTIVITIES

A diverse range of prevailing circumstances and conditions exist for sites affected by past activities. Due to their diversity, an important step in addressing such sites involves the evaluation of site specific hazards[17] and risks relating to the impacted site or area(s) where remediation is being considered to address the situation. In accordance with the IAEA Safety Glossary [6], remediation includes "Any measures that may be carried out to reduce the radiation exposure due to existing contamination of land areas through actions applied to the contamination itself (the source) or to the exposure pathways to humans".

Clear remediation goals and objectives need to be defined on the basis of the site specific evaluation. Modelling approaches and tools, such as those described in this publication, can provide important insights in assessing whether or not remediation is justified, and if so, to identify appropriate remedial actions to be taken to reduce exposure, and to later evaluate the effectiveness of remediation.

3.1. REMEDIATION GOALS AND OBJECTIVES

In cases where risk is considered to be 'unacceptable' (i.e. remediation is deemed to be 'justified'), for example, based on risk assessment and consultation with interested parties, an overarching goal of remediation is the reduction or elimination of risk to people and the environment now and in the future (e.g., see Ref. [48]). This may be accomplished through the removal of source(s) of contamination, the modification and disruption of exposure pathways, and where this is not possible, the establishment of restrictions or advisories to limit exposure [1, 4, 36, 49−54].

From the perspective of the IAEA safety standards, the decision regarding justification of remediation is taken on the basis of radiological criteria, and specifically, the 'reference level', in the context of radiation exposure relating to the site under consideration [1, 3]. In an existing exposure situation (e.g. for sites affected by past activities), the reference level is "the level of dose, risk or activity concentration above which it is not appropriate to plan to allow exposures to occur and below which optimization of protection and safety would continue to be implemented" [6]. In cases where the reference level is not exceeded, remediation would not be considered justified on the basis of radiological criteria [1]. Although such sites would fall outside of the scope of the IAEA safety standards, which are focused on radiation protection, it remains possible that remediation might be needed to address non-radiological hazards and risks associated with the site (physical, chemical, biological), which would then fall under the jurisdiction of other authorities than the regulatory body (as defined in the IAEA safety standards; see Refs [3, 5, 6]).

In cases where remediation is deemed to be justified (i.e. the reference level is exceeded), remediation objectives and corresponding criteria will then need to be defined in consultation with interested parties who have legitimate interest in the outcome of the remediation. Such criteria will include end point criteria to verify that a specific remedial action or combination of remedial actions have been implemented as planned, as well as an end state criterion, which

[17] For the purposes of the current report, hazard is defined as the "potential for harm or other detriment, especially for radiation risks; a factor or condition that might operate against safety" [6].

indicates that the overall remediation has been completed in accordance with the approved remediation plan and the approved end state for the site has been achieved [1].

Remediation objectives are an integral part of the remediation strategy for a given site and prior to selection of technically acceptable remedial actions and their implementation, remediation objectives need to be agreed [55]. In general, remediation objectives are site specific and take account of prevailing circumstances and conditions at the site of interest. Such considerations include the contaminants present and their levels, site location, physicochemical attributes of the environment (e.g. topography, pH), the density of the local population, expectations and perceptions of interested parties, financial constraints and other factors.

3.2. GENERIC REMEDIATION FRAMEWORK

Given the wide diversity of sites and the broad range of conditions that might be present at sites affected by past activities, in order to achieve remediation goals and objectives, a systematic, stepwise approach needs to be applied.

A generic framework for remediation is provided in GSG-15 [1].

The remediation process, as presented in GSG-15 [1], consists of five phases (see Fig. 1), as follows:

(1) **Phase 1: Preliminary evaluation**, involving review of available information and limited characterization of a site to define the issue, and preliminary screening of radiological exposure relative to a defined ‘screening criterion’ to evaluate whether remediation ‘might be justified’ (Section 3.2.2.1);

(2) **Phase 2: Detailed evaluation**, which is conducted if preliminary evaluation suggests that the anticipated radiological exposure exceeds the screening criterion[18]. In such cases, detailed site surveying and additional site characterization is undertaken to assess risk and to determine whether remediation ‘is’ justified by comparing anticipated radiation exposure against the agreed reference level. The data generated through characterization undertaken during detailed evaluation will serve as a pre-remediation baseline against which to evaluate the effectiveness of remedial actions should remediation be deemed justified (Section 3.2.2.1);

(3) **Phase 3: Planning of remediation**, which involves the following activities (Section 3.2.2.2):

    (a) Identifying and evaluating remedial options;
    (b) Agreeing on an end state criterion, that is appropriate to the prevailing circumstances and that, when achieved, will indicate that it is appropriate to transition the site from implementation of remediation into post-remediation management. The end state criterion will need to be established with consideration of radiological, as well as non-radiological factors (e.g. physical, chemical, biological);

[18] The screening criterion is a conservative criterion that is applied during the Preliminary Evaluation phase of remediation to determine whether or not remediation might be justified (e.g. the lower level of the reference level range, as established in the national strategy for remediation; see Section 3.1) (see also GSG-15 [1]). It can be established by the responsible party or the regulatory body, depending on the national legal and regulatory framework, and needs to be approved by the regulatory body and documented in the approved Remediation Plan.

(c) Establishing end point criteria and easily measured ‘derived criteria’[19] (e.g. dose rate, activity concentration), which indicate that a specific remedial action or combination of remedial actions have been carried out as planned;
(d) Assessing cost of remediation;
(e) Evaluating remedial options and selecting an optimized remedial action or set of remedial actions;
(f) Establishing a remediation plan (including contingencies), taking account of past experience, for example, in terms of the feasibility, ability to implement, effectiveness and uncertainties relating to specific remedial actions (e.g. technologies) within the plan;
(g) Gaining regulatory approval to implement this plan.

(4) **Phase 4: Implementation and verification**, involving conducting remediation according to the approved remediation plan, assessing effectiveness of remedial actions, determining whether restrictions are necessary to ensure adherence to established remediation criteria (end point criteria, end state criterion, and corresponding ‘derived criteria’), and determining of whether further optimization is beneficial (Section 3.3.3.3);

(5) **Phase 5: Post-remediation management**[20], involving implementation of restrictions, as needed, and monitoring and surveillance to assess and demonstrate long-term effectiveness and performance of remediation and post-remediation stewardship (Section 3.3.3.4).

Throughout the remediation process, it is necessary to perform characterization (as necessary) and monitoring, as well as communication[21] and consultation[22] with interested parties at appropriate times commensurate with actual and perceived risk.

This publication focuses on the application of a general assessment methodology (described in Section 4), which provides more detailed steps in support of the practical implementation of the phases of remediation recommended in GSG-15 [1] and demonstrates modelling tools that can be used to determine whether remediation is justified, and if so, to support remediation planning and implementation.

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[19] “The term ‘derived criteria’ is related to the concept of ‘derived reference levels’ established by the ICRP 126 [56]. According to ICRP 126 [56], a ‘derived reference level’ is defined as a “numerical value expressed in an operational or measurable quantity, corresponding to the reference level set in dose.” A ‘derived criterion’ is more generic than a ‘derived reference level’ and refers to a numerical value expressed in an operational or measurable quantity, corresponding to a given criterion, such as a reference level, screening criterion, end point criterion or end state criterion” [1].

[20] Sometimes called ‘institutional control’.

[21] For the purposes of this publication, communication is defined as the *“exchange of information between an organization and its interested parties for the purpose of informing, influencing, persuading or developing a common understanding in pursuit of an organization’s long term objectives, and of serving the public interest for safety”* [57].

[22] For the purposes of this publication, consultation refers to processes through which a party seeks or, in accordance with the (national) legal framework, has to seek the views of interested parties on matters that affect the decision making process, that affect interested parties directly or in which they have a significant interest” (based on GSG-6 [57]).

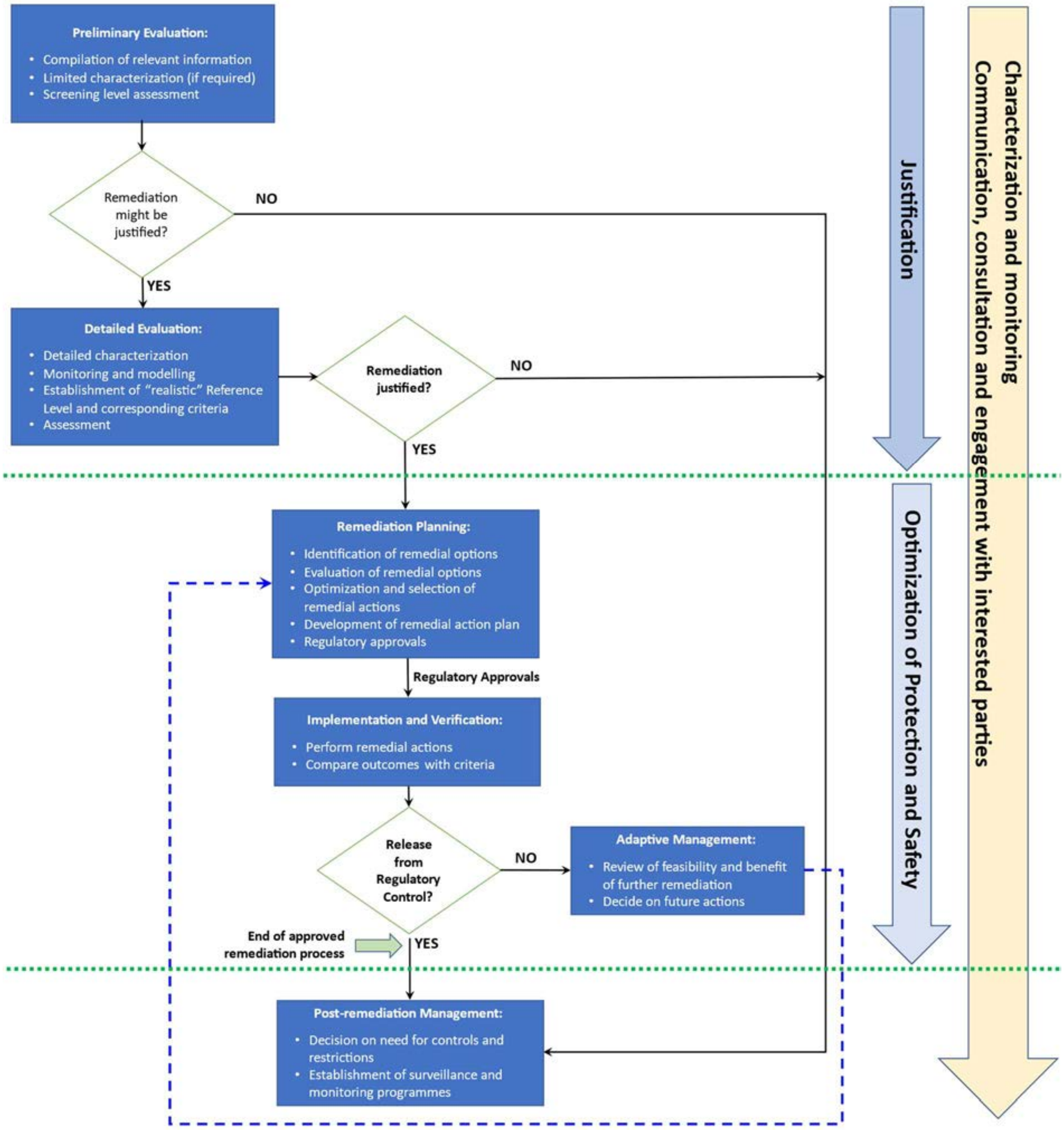

FIG. 1. Representative scheme for the phases involved in the remediation of a contaminated site or area (modified from Fig. 1 in Ref. [1]).

3.2.1. Application of a graded approach in assessment

The methodology presented in this publication emphasizes the need for a graded approach in the REIA of a site, with the level of assessment increasing (if necessary, commensurate with risk) from a simple screening assessment, which involve a desk based review of existing data, to more complex assessments based increasingly on site specific data. As the complexity of the assessment increases, the use of site specific data can become more important. Such a graded approach optimizes both the work that needs to be undertaken and the REIA process, as it allows the assessors and decision maker(s) to stop at the point where it becomes clear that further work will not change the outcome of the assessment (e.g. in terms of addressing identified risks).

A graded approach is generally applied, placing focus on the information needed for each phase of the remediation process. A lack of prioritization of risk can compromise safety, for example, if disproportionate focus is placed on low risk, instead of to high risk facilities or activities. In addition, it is not cost effective, for example, to carry out a detailed measurement programme before undertaking a screening assessment, if the results of the screening assessment indicate that the screening criterion is satisfied and, therefore, on the basis of technical considerations, remediation is not justified. The graded approach also implies that measurement and assessment will be carried out in parallel.

The emphasis on dose assessment, which is focused on radiological risk, versus a broader risk assessment that considers radiological and non-radiological risks (chemical, physical, biological), will vary between countries, depending on national policy and strategy, and the legal and regulatory framework. The balancing of different types of risk at a given site will be undertaken as part of optimization of protection and safety, applying a graded approach, such that the level of regulatory oversight and effort is commensurate with risk.

3.2.2. Application of the generic remediation framework in assessment

The REIA of a contaminated site or area typically includes an evaluation of doses and risks to:

- Workers at the site;
- Members of the public;
- The environment, for example, impacts on wildlife (flora and fauna) in the context of relevant ecosystems, possibly including a groundwater body.

The REIA is part of the overall environmental impact assessment, which covers radiological and non-radiological risks, including those related to human health, ecological risk, and socioeconomic factors.

Assessment of the radiological impacts of a facility (e.g. a historic site) or activity (such as implementation of a remedial action) involves the development of a ‘conceptual site model’ and a ‘quantitative site model’ and an analysis of exposure pathways and scenarios of exposure. The evaluation of the site and the surrounding environment, which involves characterization and assessment of impacts (e.g. during the ‘preliminary evaluation’ and ‘detailed evaluation’ phases of remediation; see GSG-15 [1]; see also Fig. 1), enables scenarios of exposure and exposure pathways to be identified, and facilitates the development of a conceptual site model.

In accordance with Footnote 28 of GSG-15 [1]:

> "The 'conceptual site model' provides a qualitative overview of the key aspects for consideration during remediation and the connection of these aspects to the site or area being considered for remediation. The conceptual site model identifies relevant sources of contamination, contaminant transport pathways, receiving environments and receptors of exposure (e.g. human populations) to facilitate a broad estimation of the possible activity concentrations of radionuclides in the environment and the associated levels of exposure of persons. This model synthesizes and confirms what is known about a site in support of decision making. Development of the conceptual site model is an iterative process. For screening purposes (e.g. during preliminary evaluation), the assessment of impacts typically involves conservative assumptions, for example, to estimate doses to the public. In cases where detailed evaluation is necessary, the radiological environmental impact assessment incorporates site or area specific data, and the conceptual site model is reviewed and updated, as necessary, to capture the site or area specific conditions."

Underpinning the conceptual site model and providing the basis for a detailed evaluation of radiological impacts, a quantitative site model is typically developed.

Footnote 30 of GSG-15 [1] states:

> "The 'quantitative site or area model' is a representation that uses a mathematical formulation to describe the movement of radionuclides in the environment and to estimate the resulting exposures. The development of the quantitative site or area model is an iterative process."

Models, such as those applied in this publication, can serve as a basis for the quantitative site or area model for a historic site.

3.2.2.1. Preliminary evaluation and detailed evaluation

The general approach to assessment is to start with a simple, conservative assessment of dose relative to the screening criterion (i.e. a 'preliminary evaluation'). If the screening criterion is not satisfied, the level of detail in the assessment is increased and a 'detailed evaluation' is undertaken [1]. This will usually involve more detailed site specific measurements, the specification of more realistic exposure pathways and/or parameter values for use with already specified pathways, and the use of more complex models, as necessary, to achieve a more realistic representation of both the site and the exposure pathways. If the results of the detailed evaluation (using site specific data) do not satisfy the relevant criteria (e.g. including the reference level), the decision maker would be expected to determine that remediation (which might include restrictions on the use of the site and/or food and drinking water consumption advisories) is justified (i.e. the benefits of remediation outweigh the risks).

Justification [23] of remediation involves demonstration that the benefits of implementing remediation outweigh the detriments. Computer models can be applied to predict future impacts, recognizing that the quality of their outputs is dependent on the availability and quality of data. The data might be generic in the early stages of an assessment (e.g. during 'preliminary

[23] Justification is "The process of determining for an emergency exposure situation or an existing exposure situation whether a proposed protective action or remedial action is likely, overall, to be beneficial; that is, whether the expected benefits to individuals and to society (including the reduction in radiation detriment) from introducing or continuing the protective action or remedial action outweigh the cost of such action and any harm or damage caused by the action." [6].

evaluation' [1]), but will likely become more detailed and more site specific, as the assessment becomes more realistic (less conservative) (e.g. during 'detailed evaluation' [1]).

3.2.2.2. Planning of remediation

Once a decision has been made that remediation is justified, it is necessary to undertake optimization of protection and safety[24]. Optimization of protection and safety within the remediation process (and in particular, in remediation planning and implementation) involves evaluating possible remedial options to determine which is preferred, taking all relevant benefits and detriments into account, also considering the 'no action' alternative amongst the options considered. There might be risks associated with the presence of physical hazards, non-radiological contaminants and/or biological hazards at any specific site. Such risks and hazards need to be considered as part of optimization of protection and safety to weigh out and evaluate the remedial options being considered for protection of people and the environment and their implementation [51−54]. In addition, decision maker(s) will need to take all such risks and hazards into consideration when deciding how to manage the site. Examples of how to evaluate different aspects as part of optimization of protection and safety are provided by MODARIA I WG1 on 'Remediation strategies and decision aiding techniques' and MODARIA II WG1 on 'Assessment and Decision Making of Existing Exposure Situations for NORM and Nuclear Legacy Sites' (e.g., see Refs [49, 59−60]), and are not the focus of this publication.

The IAEA safety standards (e.g., see Section 5 of GSR Part 3 [3] and GSG-15 [1]) and ICRP 103 [7] establish requirements and provide recommendations and guidance on the goals of assessment in relation to remediation planning and implementation; case studies in the annexes of GSG-15 [1] provide more detailed technical considerations. Additionally, several countries have already developed publications with methodologies to undertake assessments to address sites affected by past activities (e.g., see the bibliography).

Information and data collected through characterization, monitoring and assessment can provide input into the process of estimating the financial cost of remediation. Estimation of cost can be an iterative process, particularly for site which have been abandoned with no 'owner', resulting in a lack of clarity regarding who is responsible to pay for remediation, or for which little to no information is available, resulting in uncertainty relating to the nature of the hazards and risks. After remedial options have been evaluated on the basis of technical information, the technical and financial evaluation of remedial options can be undertaken together, as part of optimization of protection and safety, for example, to identified preferred options, although – consistent with good practice – this needs to be conducted independently of the technical assessment of remedial options.

3.2.2.3. Implementation and verification monitoring

Over the course of implementation of remediation, it will be necessary to conduct monitoring (source, environmental, individual) to verify that exposure and risks (including radiological risks) fall within those predicted in the environmental impact assessment. For example, such measurements are needed to validate model predictions (if possible) and to verify the effectiveness of any remedial action that is carried out relative to remediation criteria that relate

[24] Optimization of protection and safety is "The process of determining what level of protection and safety would result in the magnitude of individual doses, the number of individuals (workers and members of the public) subject to exposure and the likelihood of exposure being as low as reasonably achievable, economic and social factors being taken into account (ALARA)." [6].

to the remediation objectives, as set during remediation planning (see Section 3.2.2.2). The inclusion of specific exposure pathways and scenarios will depend on the purpose and the context of the assessment. In addition, the timeframe of the assessment also needs to be considered, particularly for sites contaminated with long lived radionuclides and progeny.

During remediation, measurements on and near the site of interest can be carried out to:

(1) Show that relevant criteria are being met, which enables a given remedial action to be modified, if necessary;
(2) Verify that doses to the workers do not exceed the required dose limits and that measures are in place to ensure that those doses are being optimized and are in compliance with any required dose constraints;
(3) Verify the performance of measurement equipment;
(4) Verify the methods used for analysing data;
(5) Verify the performance of computer codes used for estimating and predicting health and environmental impacts;
(6) Keep the decision maker informed of the status and effectiveness of the remediation to ensure work is being undertaken in accordance with the approved remediation plan and to facilitate communication and consultation with other interested parties.

More detailed site characterization, monitoring and assessment can be carried out, as needed, during each assessment stage to:

(1) Identify the main exposure pathways;
(2) Provide input for the different stages of assessment, the results of which can be compared against the relevant criterion (screening criterion in the Preliminary evaluation phase of remediation, reference level in the Detailed evaluation phase; see Fig. 1);
(3) Estimate and later verify the effectiveness and impacts of different remedial options (including the option of leaving the site 'as is' without remediation) on members of the public, workers and the environment,
(4) Establish the end point criteria and ultimately, the end state criterion, which include regulatory criteria, and acceptable risk levels (or the corresponding derived criteria, such as concentration levels or reference levels), that have to be met for remedial action to be regarded as effective and complete (see Sections 3.1 and 3.2). Criteria are needed to ensure that any actions taken to remediate the site will reduce radiological and non-radiological risks to acceptable levels;
(5) Verify the suitability of measuring equipment;
(6) To gain additional understanding of site specific conditions, for example, in cases where remediation is not proceeding as planned (e.g. if remedial actions are not as effective as initially anticipated);
(7) Verify that remediation has been implemented in accordance with the approved remediation plan.

It is important to establish and maintain effective communication and consultation between those undertaking modelling and those measuring samples to streamline the assessment process and to ensure appropriate data are being generated. Ideally, a single, integrated team of field staff and modelers may be established, with at least some individuals within the team having expertise in conducting both field investigations and modelling; however, it is important to recognize that, often, it is not feasible to establish a large team of practitioners, making it necessary to establish innovative approaches to gain access to the necessary expertise for assessments.

3.2.2.4. Post-remediation management

Once a site or part of a site has been released from regulatory control, it is necessary to establish and implement appropriate restrictions and associated passive and/or active controls, as appropriate, taking account of the final end state of the site and its approved end use. In addition, food and drinking water advisories might need to be established, for example, to prevent installation of drinking water wells on footprints of engineered tailings facilities or waste disposal facilities.

A subset of locations and measurements from the source and environmental monitoring programmes that were established during earlier phases of the remediation process (see Sections 3.2.2.2 and 3.2.2.3) will likely need to be carried forward into post-remediation management to facilitate continuity in data and understanding.

Such data can then be inputted into models, such as those described in this publication (see Section 5 and Appendix V), to ensure that remedial actions continue to function as designed and that engineered structures are maintained and continue to serve as barriers to releases of contaminants (including radionuclides) to the environment.

4. GENERAL ASSESSMENT METHODOLOGY FOR THE RADIOLOGICAL RISK AND IMPACT ASSESSMENT OF SITES AFFECTED BY PAST ACTIVITIES

This section provides a general methodology for assessing the radiological impacts and risks associated with sites affected by past activities (including those that are still operational, e.g. Chalk River Laboratories, Hanford), along with technical guidance in support of the management of such sites [58]. This methodology provides more detailed steps than those that describe the five phases in the overall remediation process (see Section 3.2). It is specifically focused on assessment of impacts and risks for use in support of the implementation of GSG-15 [1], and in particular, Fig. 1 of GSG-15 [1] (see also Section 3.2).

4.1. APPLICATION OF THE GENERAL ASSESSMENT METHODOLOGY

The general assessment methodology is intended for use by assessors who are asked to provide evidence based advice to decision makers on sites affected by past activities. The assessment methodology described in this publication is applicable to sites contaminated as a result of past activities (e.g. mining operations and/or processing of mineral ores, historic research reactor sites) ('historic sites'), or as a result of releases of radioactive material to the environment relating to a nuclear or radiological emergency. In this methodology, a distinction is made between the 'decision maker(s)' and other interested parties. The methodology does not include the decision making process. The provision of expert advice on specific technical issues, on the basis of sound and defensible assessment procedures, is an important component of the overall decision making process. However, this process includes wider considerations than those presented here, such as evaluation of the beneficial and detrimental impacts of alternative options for the management of a site and its remediation.

There are several possibilities for setting up a general assessment methodology, but it is important to keep the overall approach as simple as possible, while ensuring its applicability to address a wide range of issues. Unrestricted use of historic sites is often not possible due to the level of contamination, the presence of areas ('footprints') with some level of impact remaining after remediation, and the presence of areas with engineered structures that require inspection by qualified experts and maintenance into the future. A decision to leave a site 'as is' without remediation or to undertake limited remediation might not be an optimum option, for example, due to the costs of prolonged monitoring and surveillance measures and the restrictions it places on the current and future uses of the site. In addition, if areas affected by past activities are not dealt with, this may lead to a lack of trust with local communities and, due to local opposition, might limit the possibility to undertake future activities, such as mining, should such opportunities arise.

4.2. DEFINE REMEDIATION OBJECTIVES

It is necessary to clearly define the relevant objectives in support of assessment of hazards and impacts for each phase of the overall remediation process (see Fig. 2) and to ensure that they are discussed and understood by all interested parties. The objectives will depend on government policy and strategy, legal and regulatory requirements, the nature and location of the site, the anticipated future use of the site and other factors depending on the prevailing circumstances. In cases where international assistance is needed to remediate a site or area that has been affected by past activities, remediation objectives might depend on international standards, for example, if an adequate national policy and strategy, and/or legal and regulatory framework is not in place in the country (see GSR Part 1 (Rev. 1) [5]).

In general, remedial actions need to be selected, applying the principle of optimization of protection and safety [1]. For example, the potential doses to workers involved in remediation, along with other occupational hazards (e.g. exposure to non-radiological contaminants, biohazards and/or physical hazards), need to be considered as factors when evaluating the adverse impacts relative to the benefits associated with the remedial action. More generally, the radiological impacts of each alternative remedial option need to be taken into account to evaluate the overall benefits and detriments associated with each remedial option and to identify the preferred remedial option (or set of remedial options), recognizing that the radiological impacts alone are not, in general, a sufficient basis for determining the preferred remedial option.

4.3. ENGAGEMENT OF INTERESTED PARTIES IN ASSESSMENT

While this publication focuses on radiological considerations, for example, in determining whether remediation is justified (see Section 3.1), the decision maker needs to consider many factors in addition to the radiological issues, including non-radiological issues such as the presence of non-radiological contaminants (e.g. heavy metals, wastewater, process water and chemicals), physical hazards, Government policies, socioeconomic and political considerations, and public attitudes and expectations.

Communication and consultation between the responsible party and other interested parties is an important part of the process of remediation of contaminated sites or areas, in all phases of remediation (i.e. preliminary evaluation and detailed evaluation of the site, planning and implementation of remediation and post-remediation management) (Fig. 1) (see also Section 3.2 and GSG-15 [1]).

In most situations, the decision maker from the perspective of providing final approval of the proposed remediation plan, which has been submitted by the responsible party for a site affected by past activities, will be the regulatory body or other relevant authority; however, in some countries or political jurisdictions, depending on the situation, the decision maker may be the national, regional, or local government. There may be situations where there are several authorities involved in the decision making process because there are several legislative and/or regulatory requirements, administered by different authorities and/or jurisdictions, that need to be met. Other interested parties might include operating organizations, local residents, Non-Governmental Organizations, and special interest groups.

Engaging with and evaluating input from a range of interested parties can be an important part of the information gathering and decision making process, depending on the national policy and strategy and the legal and regulatory framework, the nature of the site and its history.

The relationship between decision makers and other interested parties will influence the decision making process, taking account of the science based environmental impact assessment process to ensure regulatory compliance is achieved.

4.4. THE GENERAL ASSESSMENT METHODOLOGY

The general approach to assessment proposed here, as it relates to the remediation process recommended in the IAEA safety standards (in particular, GSG-15 [1]), is presented in Fig. 2.

The main phases in the overall assessment process shown in Fig. 2 are the following:

Preliminary evaluation (including screening assessment):

(1) Identify the problem.

(2) Review available information and conduct limited characterization (if necessary) and develop the conceptual site model (see Section 3.2.2).

(3) Carry out a screening assessment[25]:

- (a) Establish a screening criterion;
- (b) Estimate the impact using conservative assumptions and scenarios of exposure, using a simple model [61];
- (c) Compare the model outputs with the screening criterion;
- (d) If the model outputs exceed the screening criterion, consider using site based exposure scenarios and data;
- (e) Notify the decision maker(s) and other interested parties of the results of the screening assessment.

Detailed evaluation:

(4) If the model outputs exceeds the screening criterion, carry out a detailed (more realistic) assessment[26]:

- (a) Develop a quantitative site model that corresponds to the conceptual site model at an appropriate level of detail, taking account of the situation and conditions;
- (b) Establish reference level and other criteria (derived measurement parameters) based on the reference level;
- (c) Estimate the impact using more realistic assumptions and scenarios of exposure, collect more data to improve estimation of transfer parameters and of the source term, and use complex models, if appropriate;
- (d) Compare the model outputs with the reference level and any other criteria that have been established based on the reference level (i.e. ‘derived criteria’);
- (e) If the relevant criterion (e.g. the reference level) is still not met, carry out a more detailed evaluation;
- (f) Use assumptions and scenarios of exposure which are as realistic as possible;
- (g) Use sensitivity analysis methods to identify sensitive parameters, as appropriate;
- (h) Collect data on sensitive parameters and data needed to improve the estimation of the source term and transfer parameters, as appropriate;
- (i) Use more complex models, if appropriate;
- (j) Notify the decision maker(s) of the results of the detailed evaluation.

Planning of remediation:

(5) If the reference level is exceeded and, therefore, remediation is deemed to be justified, it is appropriate for the decision maker to initiate planning of remediation, based on the following steps:

[25] The ‘screening assessment’ is equivalent to the ‘Preliminary evaluation’ phase of the remediation process in GSG-15 [1]. For the purposes of facilities and activities, this involves a conducting screening level dose assessment.

[26] The ‘detailed assessment’ is equivalent to the ‘Detailed evaluation’ phase of the remediation process in GSG-15 [1].

(a) Define remediation objectives of remediation and criteria, including end point criteria (e.g. soil concentration levels for contaminants) and end state criterion (see Sections 3.1 and 3.2). Depending on the scale of the remediation, establish appropriate end point criteria for different phases. The remediation criteria will be used to verify that remediation has been completed in accordance with the approved remediation plan (including the approved end state and end use);
(b) Collect additional data, as necessary (e.g. in cases where unexpected contamination or hazards are found during remediation);
(c) Establish appropriate scenarios of exposure, for the workers involved in remediation and for the future use of the site and surrounding area, for example, by members of the public;
(d) Assess the remedial options available (including the 'no action' alternative), taking into account relevant factors, and select one that yields the optimum result, taking into account the benefits and detriments of each option, including those associated with post-remediation radiological impacts on the public and the environment, and radiological impacts to workers, the public and the environment while the work is being carried out;
(e) Identify and prioritize remedial actions;
(f) Prepare a remediation plan, based on the information that has been compiled, and submit it for regulatory approval.

Implementation and verification monitoring:

(6) Implementation of remediation and verification monitoring:

(a) Implement the approved remediation plan that is necessary to meet the relevant remediation criteria (end point criteria, and ultimately, the end state criterion);
(b) Collect data from the remediated areas to verify that the relevant criteria (e.g. end point criteria, end state criterion) have been met;
(c) Where residual material remains, carry out monitoring to ensure remediation is effective, in accordance with the approved remediation plan;
(d) Review whether further remediation is justified, and the preferred remedial option for such remediation, if it is necessary;
(e) Verify that the end state criterion has been met, before initiating post-remediation management;
(f) Request release of site from regulatory control.

Post-remediation management:

(7) Once it has been verified that the approved end state criterion has been met and the site (or part of the site) has been released from regulatory control, initiate post-remediation management, in accordance with the national legal and regulatory requirements.

Throughout this process, it is necessary for assessors to maintain effective communication and consultation with the decision maker(s) and other interested parties, as relevant.

Each of the steps described above is discussed in more detail in the following Sections 4.4.1–4.4.8.

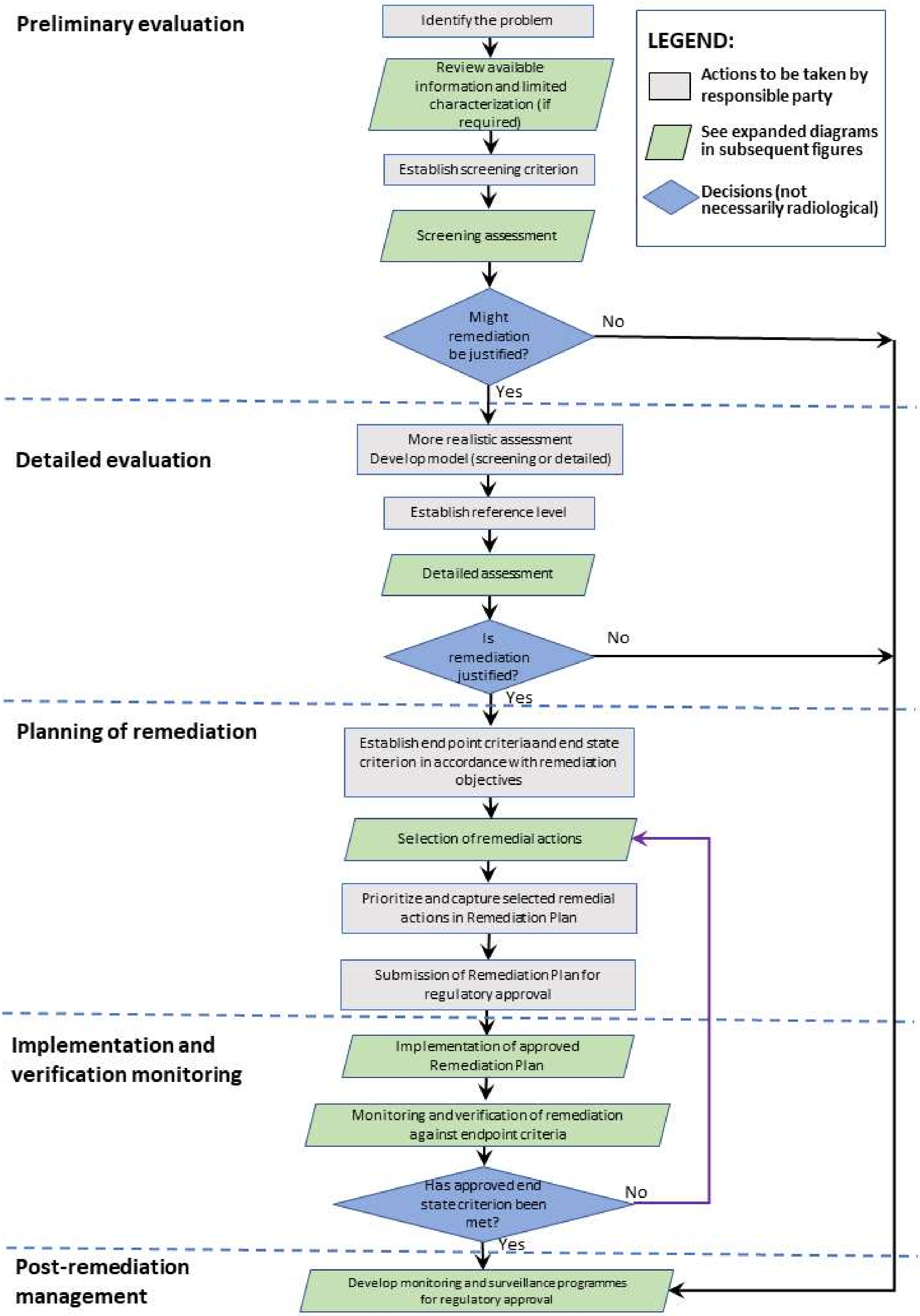

FIG. 2. The overall assessment and management process within the five phases of remediation (modified from Fig. 1 in Ref.[1]).

4.4.1. Identify the problem

Information that may be relevant to this step can be obtained from records of the operating organization (if the operating organization is still in business, or the records are still available) from the responsible party (e.g. in cases where there is no longer a 'site owner' or operator), from records of the regulatory body or other relevant authority, the local government, or from members of the public who lived near or worked at the contaminated site when it was operational. In cases where there is limited or no information or data available, it might be necessary to conduct consultation with interested parties and characterization to gain further understanding of the site and conditions. This might include review of record archives, meeting with members of the local community (e.g. including those who might have worked on the site in the past), visiting the site, taking measurements or samples, and conducting dose assessments based on available data (see also Section 4.4.2). An important aspect of identifying the problem is to identify receptors of exposure[27] (people and non-human species, as applicable), including, for example, the representative person[28].

Checking national regulations and standards regarding site remediation, and concentration criteria for radioactive and chemical substances and physical hazards of concern will help with identification of the problem. In cases where national regulations are not available, international recommendations (e.g. IAEA safety standards) or regulations and guidance from other nations could be helpful, recognizing the need for harmonization with the national legal and regulatory framework.

4.4.2. Review of available information and limited characterization

The steps involved in the review of available information and limited site characterization, if necessary, which is undertaken within the Preliminary Evaluation phase of remediation, is summarized in Fig. 3. The aim of this phase is to establish the features of the site that are important for the assessment relating to Preliminary Evaluation. This phase of the process might be carried out interactively with the identification of the problem (see Section 4.4.1).

The first part of the review of available information and site characterization step involves the compilation of available information regarding the hazards and impacts associated with the site. As before, information that might be relevant to this step can be obtained from available records (see Section 4.4.1). Technical information on the activities formerly carried out at the site (e.g. mining; processing of ore), or on material deposited at the site (e.g. through its use as a landfill) is also important. For a former operational site, this information typically includes site maps, if available, showing the layout of buildings, structures (e.g. utilidors that might contain friable asbestos; septic tanks that might contain biological hazards), stockpiles, storage tanks, tailings dams and/or other relevant site features (e.g., see Appendix II, Gunnar case study), and technical descriptions of materials and processing methods. Input to capture key site information and infrastructure, as depicted in Fig. 3, can be provided by interested parties. In cases where such information is not available, data from similar sites elsewhere (nationally or internationally) can be obtained or limited site characterization might be undertaken to fill key gaps. If available, information about the levels of environmental contamination is compiled and

[27] A receptor of exposure is "the entity exposed to the stressor of concern. This term may refer to humans, plants and animals (including endangered and threatened species), habitats, or ecosystems." [1]

[28] The representative person is defined as an "individual receiving a dose that is representative of the doses to the more highly exposed individuals in the population" [3].

evaluated and, as necessary, limited characterization of contamination can be undertaken for screening purposes.

The next step is to identify the hazards and impacts (see Fig. 3). This step of the process might necessitate measurements, for example:

(1) To fill gaps in existing data records (where necessary);
(2) To establish radionuclide activity concentrations in air, soil, surface water (including sediments) and groundwater within the site boundaries (e.g. to characterize the source term);
(3) To determine whether contaminated material has migrated from the site to the surrounding environment;
(4) To determine the physical and chemical characteristics of the contaminated materials;
(5) To provide information to address questions or concerns raised by interested parties, such as local communities, special interest groups, members of the public or the regulatory body or other relevant authority.

Information on the hazards and impacts will, ultimately, need to be evaluated in the context of the pathways of exposure and relevant scenarios that might result in exposure, as well as possible receptors of exposure.

Therefore, once the hazards and impacts have been identified, the FEPs relevant to the site can be established. This FEP analysis also provides a mechanism for the possible exposure pathways and scenarios of exposure to be established and documented, and for a conceptual site model to be developed (see Fig. 3).

If a suitable model is already available, this might provide useful insight in developing a conceptual site model. In doing so, it will be important to audit the available model to verify that it includes a suitable representation of all the relevant FEPs. Model suitability will depend on several factors, and in particular, experience gained from assessing similar sites. This is discussed in more detail in Section 4.4.5 on detailed assessment. Guidance is available on the selection of scenarios of exposure in Refs [22–23, 62], on the identification of hazards in Ref. [23], and on the design and implementation of measurement programmes in Refs [23, 63–66].

Documenting and storing the information gathered during this phase of the remediation process, as part of an integrated management system, is extremely important. Requirements for a management system are established in IAEA Safety Standards Series No. GSR Part 2, Leadership and Management for Safety [67], and associated recommendations are provided in IAEA Safety Standards Series No. GSG-16, Leadership, Management and Culture for Safety in Radioactive Waste Management [68].

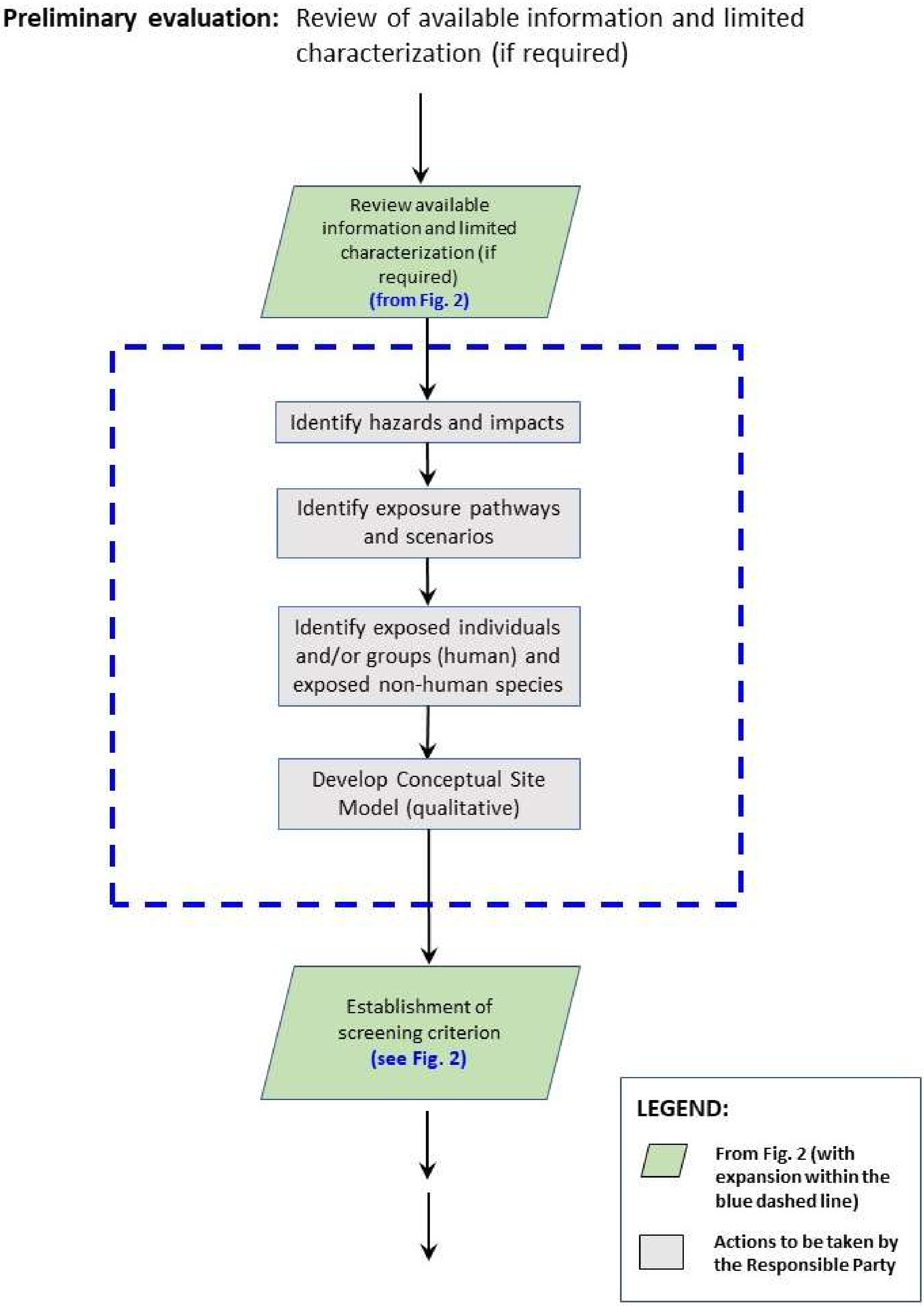

FIG. 3. The review of available information and limited characterization stage (expanded from Fig. 2).

4.4.3. Establish screening criterion

A national screening criterion might be determined and expressed in practical terms as total effective dose (or total risk), annual effective dose (internal, external), or radionuclide activity concentrations in air, soil, water, sediment or other environmental media or materials. The screening criterion needs to be established such that if the results of the assessment do not exceed this criterion (Section 4.4.3), the radiological risk is of no regulatory concern, both in the present and the future. The screening criterion needs to take account of the likelihood of exposure and the number of people potentially exposed, in accordance with the principle of optimization of protection and safety [3]. This implies that unless the screening criterion is exceeded, there is no radiological justification for carrying out remediation or additional assessments. This does not preclude the possibility that the decision maker may decide that remediation is needed for other reasons (including non-radiological, socioeconomic, political and public acceptability considerations), or that remediation may be needed because due to other non-radiological hazards associated with the site (e.g. physical hazards, chemicals, heavy metals). Also, it does not preclude the carrying out of actions by the responsible party or the operating organization that will have the effect of reducing radiological impacts, even though those actions are not required by the regulatory body or other relevant authorities.

4.4.4. Screening assessment

Once the review of available information and limited characterization step has been completed (Section 4.4.2) and the remediation objectives (Section 4.2) and screening criterion (Section 4.4.3) have been defined, a screening assessment can be carried out. The screening assessment methodology is summarized in Fig. 4. The purpose of the screening assessment is to eliminate from further consideration any site with only a low level of radiological concern and that does not pose significant risk to people or the environment.

The identification of hazards and impacts, scenarios of exposure, exposure pathways, and exposed groups (human and non-human species) is largely generic at this screening assessment stage and is combined with a conservative (cautious) quantitative assessment, consistent with the application of a graded approach to site assessment and management.

Thus, a screening assessment using a conservative model that simulates the major features and exposure pathways of the site is performed. The model can be generic, or a simple simulation can be used; in either case, the assessment needs to be demonstrably conservative. For example:

- Assuming that all exposures are continuous (i.e. that the humans or non-human species spend all their time in the contaminated area);
- Assuming that the radionuclide activity concentrations are uniform across the site and equal to the highest measured values;
- Assuming that all food and water consumed are contaminated;
- Gathering adequate data to enable a reasonable approximation of the source to be used (e.g. total activity rather than a detailed radionuclide inventory);
- Using the basic characteristics of the source material (e.g. solubility);
- Using default values for the site transfer parameters.

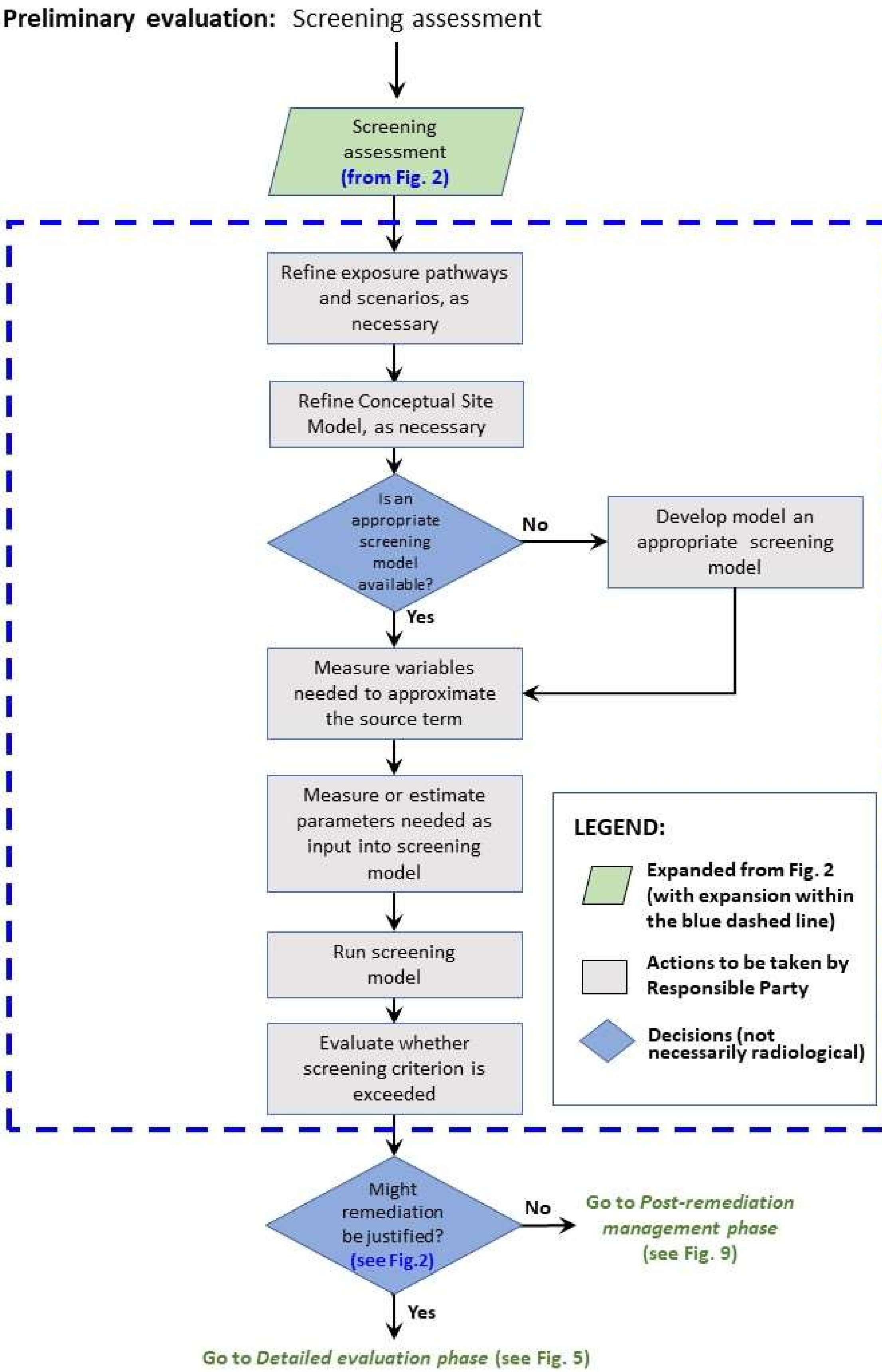

FIG. 4. The screening assessment methodology (expanded from Fig. 2).

There is little point in gathering detailed data for a screening assessment, as site specific data, other than the source term, are not necessary for a screening assessment. It might be useful, however, to have some meteorological and hydrological data, so that dispersion and dilution factors for radionuclide releases can be estimated appropriately. Default values can be used for transfer parameters and species-related factors, such as diet and occupancy time, provided they can be shown to be consistent with the conservative approach [69−70]. The use of a conservative model allows the assessor to assert, where appropriate, that the estimated doses or radionuclide activity concentrations will not exceed the relevant screening criterion. If the screening criterion is satisfied, the decision maker may decide to request approval to terminate regulatory control of the site, or may need to carry out some remedial action, in view of other non-radiological considerations.

4.4.4.1. More realistic screening assessment

If the results of the conservative screening assessment do not satisfy the screening criterion, a more realistic assessment is performed, which is intermediate in nature. This may necessitate one or more of the following:

— Use of more realistic assumptions and scenarios of exposure;
— Use of more realistic exposure times;
— Use of more realistic values (based on discussions with interested parties) for the fractions of contaminated food and water, without necessarily considering the full details of the local diet;
— Use of a more realistic source inventory;
— Carrying out more work on site investigation and characterization, particularly if values for local parameters and variables needed for modelling are not already available;
— Use of more realistic models.

If the results of this assessment satisfy the screening criterion, then the decision maker needs decide whether to request authorization to release the site from regulatory control or to proceed with detailed assessment (see Section 4.4.5), for example, to take account of non-radiological considerations.

4.4.5. Detailed assessment

If the results of the more realistic screening assessment do not satisfy the screening criterion, then a detailed assessment is performed and the outcomes are compared to the reference level (i.e. "… the level of dose, risk or activity concentration above which it is not appropriate to plan to allow exposures to occur and below which optimization of protection and safety would continue to be implemented" [6]). In some cases, a derived reference level might be used, which is "a numerical value expressed in an operational or measurable quantity" that corresponds to the reference level (see GSG-15 [1]; see also Refs [49, 71]). The key steps within a detailed assessment are shown in Fig. 5. A detailed assessment typically involves:

— An extensive radiological survey to establish the source term(s) and the values of site specific transfer parameters for the contaminated area and surrounding environment for use in the model, and for validating the model predictions, where feasible;
— The use of realistic scenarios of exposure based on local habits, diet and other factors for both humans and non-human species;

— The use of a detailed source inventory;
— The use of complex models.

The use of realistic scenarios of exposure and site specific data avoids unduly overestimating the risk and then having to conduct unnecessary remedial action, often at considerable cost in terms of time and other resources, with possible loss of confidence in site management by interested parties. Guidance on which parameters are likely to be most important is available in Ref. [72]. If a specific transfer parameter is shown to affect a critical exposure pathway, for example, as indicated by a sensitivity analysis, a local determination of that parameter is often needed. Where site specific values for transfer parameters cannot be derived from measurements, default values can be used (e.g., see Refs [73−76]).

As the assessment process becomes more detailed, discussion and/or estimation of uncertainties[29] and variabilities[30] in the results, and communication with the decision maker on the interpretation of such results, along with model assumptions, becomes more important [66, 70].

Characterization of a particular site, together with information about potential future uses of the site, will generate a set of exposure pathways and possible scenarios of exposure for that site. The choice of appropriate scenarios of exposure for a detailed REIA is not always straightforward. In general, the total risk of harm for any site depends on the product of the probability that a particular scenario of exposure will occur and the consequences (the probability of harm if the scenario does occur), summed over all possible scenarios. One approach to evaluating this total risk of harm is to attempt to include all potential scenarios of exposure in the assessment. However, this is an open-ended process, as it is not usually possible to clearly demonstrate that all potential scenarios have been included. Therefore, in practice, it is often necessary to rely on experience and judgement in choosing the scenarios that are to be included. Discussions between assessors, responsible party or the operating organization (as relevant), the regulatory body and other relevant authorities, and other interested parties are essential when selecting the scenarios to be included. Documentation and justification of this selection process to interested parties are also important. Where it does become necessary to modify the assessment process at some later time (for example, because more data become available or the risk associated with a particular scenario has been reassessed based on scientific studies), the nature of and reasons for the changes in the assessment need to be documented and communicated to all decision makers and other interested parties. Key decisions and their justification, as they relate to remedial actions to be taken (e.g. based on running model scenarios), need to be captured in support of the implementation of the remediation plan, which is developed and approved during the planning of the remediation (see Figs 2 and 6).

The detailed assessment needs to consider the current characteristics of the site and the source material, possible future uses of the site (e.g. recreational, agricultural, industrial, residential), the natural evolution of the site, and possible future changes in scenarios of exposure, site characteristics and source term characteristics.

[29] Uncertainty refers to an incomplete understanding of the context of a risk assessment or a lack of data and is quantified using a probability distribution.
[30] Variability is quantified using a frequency distribution to represent the range or spread of a set of values derived from observed data.

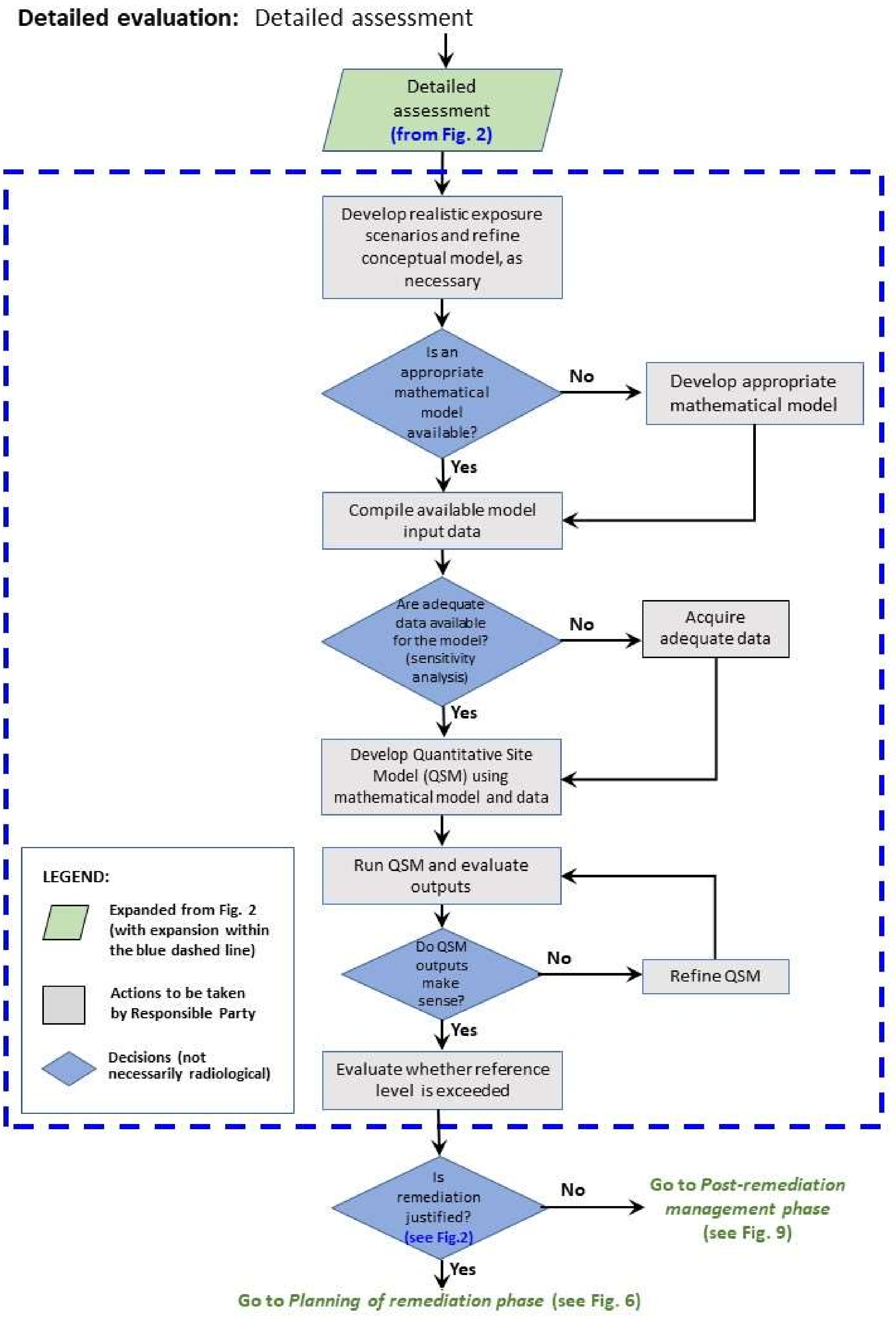

FIG. 5. Steps underlying the detailed assessment methodology and model development (expanded from Fig. 2).

The methodology to develop a quantitative site model, in the context of the detailed evaluation, is shown in Fig. 5. If suitable modelling packages are available (freely or commercially), they can be used. The criteria that apply to the process of selecting and using such packages (i.e. determining their suitability) [70] include the following:

- The packages are readily available;
- The packages are well documented and easy to use;
- The mathematical model(s) have been verified;
- The mathematical model(s) are appropriate for the work to be undertaken;
- The model(s) has(ve) been validated for a range of conditions that includes the current situation;
- The values assumed for the model parameters for detailed assessment have been validated to the extent possible.

If packages are not available, or if the above criteria are not met for those packages that are available, the assessment process will include model development, testing and validation for the specific site under consideration. If the model is not appropriate for the problem to be evaluated, the results could be misleading or meaningless.

4.4.6. Establish remediation criteria

The establishment of appropriate ('fit for purpose') interim end point criteria and ultimately, a final end state criterion for application during implementation of remedial actions is an integral part of remediation planning (see Fig. 6). If the decision maker decides to proceed further, either on policy grounds, to comply with legal, regulatory, or other requirements, or because the reference level is not satisfied by the detailed assessment, end point criteria that indicate the effectiveness and completion of individual remedial actions or combinations of remedial actions, or that indicate that a given phase of the remediation process has been completed (see Fig. 1), need to be established [1]. In addition, the end state criterion, which is "a set of conditions that need to be met to verify that remediation has been completed and the defined end state has been achieved" [1], needs to be set. It is important that all interested parties have input to this step, and that the reasons for the established criteria are clearly communicated to all interested parties [69, 77−78]. Such criteria are set during remediation planning (see Fig. 6). For the purposes of checking the status or progress of remedial actions, secondary criteria[31] can be derived in terms of concentrations that are amenable to measurement. The reasons for choosing criteria (both primary and secondary) need to be documented and communicated to all decision makers and other interested parties, and as relevant, such criteria need to be established in consultation with relevant parties.

It Is preferrable if the remediation criteria are independent of the remedial action selected. This allows the process to be modified, if necessary, without having to revise the overall objectives. Furthermore, the use of option independent criteria facilitates comparisons to be made between remedial options, in order to evaluate and compare them when identifying the preferred option.

[31] 'Secondary criteria' are equivalent to 'derived criteria' as defined in GSG-15 [1] (see Section 3.2).

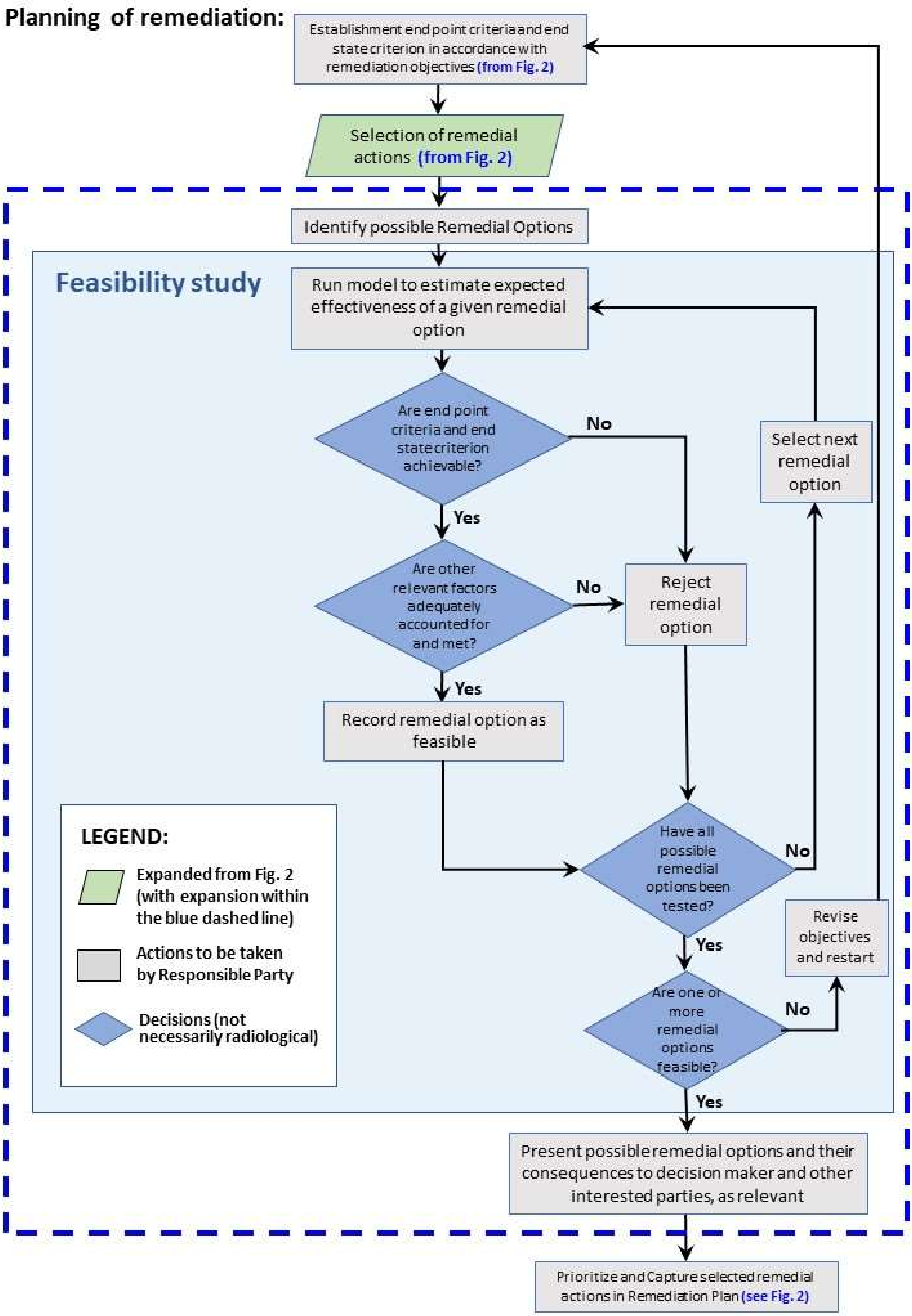

FIG. 6. Steps underlying planning of remediation (expanded from Fig. 2).

4.4.7. Implementation of remedial actions and verification monitoring

Remedial action is the "removal of a source or the reduction of its magnitude (in terms of activity or amount) for the purposes of preventing or reducing exposures that might otherwise occur in an emergency or in an existing exposure situation" [6] and consists of several steps. The first step is to identify and assess the possible remedial options. This step will consider (at least) the scale of the problem, contaminated media, radionuclides involved, relative locations and spatial extent of the contamination on the site and the location of the local residents, both for present and potential future uses of the site and the surrounding area. Guidance on this stage of the process is available in Ref. [63].

The Implementation of the selected remedial actions will be carried out in accordance with the approved remediation plan, which has been tested (by modelling) to determine whether the predicted outcome is expected to achieve the end point criteria and end state criterion (see Figs 7 and 8; see also Section 4.4.6). The remediation plan also allows for periodic reassessment (monitoring, modelling) of the impact of the site on workers, members of the public, and the environment, for example, if the source term remains with controls in place.

A methodology that can be applied for developing monitoring programmes to support the implementation of remedial actions is presented in Fig. 7. If evaluation of monitoring and surveillance data, and reassessment indicate that changes are needed in the remediation plan, then these changes need to be discussed with interested parties, possibly including the regulatory body and other relevant authorities depending on the nature of the change (e.g. relative to the approved remediation plan), before any decision is made. Any changes and the reasons for the changes need to be communicated to all interested parties and documented. This is similar to the iterative improvement process used for operational sites [61] and helps to build and maintain confidence in the process.

In some situations, remedial action might be limited to restrictions on the use of the site (or on the use of groundwater, surface water or other resources around the site) without doing any physical remediation 'work' on the site. In other cases, remediation will be implemented, after which there may or may not be a need to impose restrictions.

The selection of a remedial action from a range of different remedial options is subject to optimization of protection and safety, which includes consideration of doses to the workers, impacts to the public and the environment now and in the future, the cost of the remediation, available resources, and other factors.

For many sites, there may be more than one possible remedial option. These options need to be evaluated systematically [79−80]. This can be done by applying the detailed assessment methodology outlined in Fig. 5 to each possible option, using the approach shown in Fig. 2.

The assessment provides an estimate of the uncertainties and variabilities in the predicted doses and risks for each possible remedial option. It is also important to carry out a sensitivity analysis, determining, to the extent possible, the parameters and variables that have the greatest effect on the predictions. This information can help the decision maker in the task of choosing which parameters and variables will need to be monitored and to what extent once the implementation phase commences. It can also assist in the development of remedial options, by focusing attention on the more sensitive parameters and how they would be affected by the implementation of the alternative remedial options.

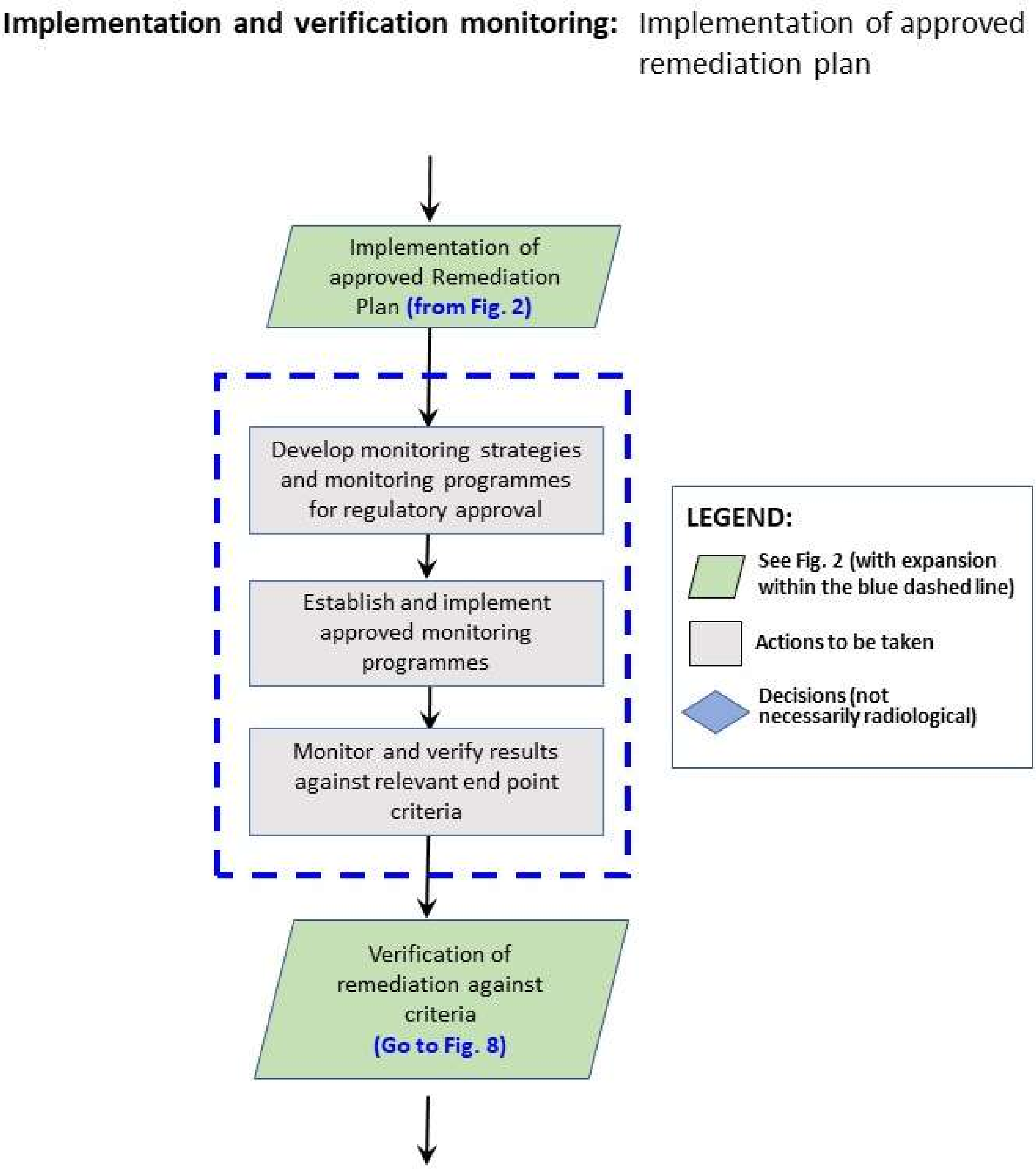

FIG. 7. Implementation of remedial actions (expanded from Fig. 2).

It is also important for assessments and model predictions to be independently checked, whenever possible. This helps to maintain the objectivity of the scientific aspects of the assessment and remediation, and provides other scientists and assessors with a record of the experience gained at a particular site. This, in turn, leads to a process of continuous improvement of assessment and modelling techniques.

Other factors that need to be considered include an assessment of economic costs and benefits (e.g. costs of remedial actions, waste disposal, monitoring, reduction of the need for security and/or control, increased economic value of the site) and social factors (e.g. reassurance of the public, the negative effects of concerns and anxiety relating to a contaminated site or its remediation) [79−80]. By considering these costs and benefits, a ranking and optimization of relevant remedial options can be achieved using a multi-attribute decision analysis (e.g., see Refs [36, 53−54, 81−84]). This ranking can then provide additional guidance to the decision makers.

In evaluating different remedial options, the detailed assessment may have to be modified to consider the effect of engineering works on the future impact of the site. In addition, the decision maker(s) will need to consider the economic cost and public acceptability of the proposed work relative to leaving the site 'as is', the estimated doses to the workers involved in remediation, and the predicted dose reduction and other benefits of each remedial option.

If this evaluation process does not lead to the selection of any feasible remedial options, the remediation objectives can be revised and the evaluation process repeated (see also Fig. 6, which depicts a methodology for identifying and evaluating possible remedial options).

4.4.8. Post-remediation management

Once the end point criteria and end state criterion have been met, a request for release of all or part of the site from regulatory control can be made, and if approved, the final disposition of the site can be implemented during the post-remediation management phase of remediation (see Fig. 9).

4.5. GENERAL EXAMPLE SUMMARY

The preceding discussion is long and detailed. Therefore, the following summary is intended to highlight the main ideas. A typical assessment might proceed as follows:

- Identification of problem:
  - Historical records and anecdotal evidence suggest that a site may be contaminated and may pose a radiological risk to the local population.
- Site characterization:
  - Preliminary measurements (e.g. external dose rates, radionuclide activity concentrations in soil and groundwater) might lead to a better formulation of the problem and could provide an early indication that a more detailed assessment might be justified.
- Screening assessment:
  - Assume that all exposures are continuous;
  - Assume that all food and water consumed are contaminated;
  - Gather enough data to enable a reasonable approximation of the source to be used (e.g. total activity rather than a detailed inventory);
  - Use the basic characteristics of the source material (e.g. solubility);
  - Use default values for the site transfer parameters.
- Intermediate assessment:
  - Use more realistic exposure times;
  - Use more realistic values (based on discussions with interested parties) for the fractions of contaminated food and water, without necessarily considering the full details of the local diet; and/or
  - Use a more realistic source inventory.
- Detailed assessment:
  - Use realistic scenarios of exposure;
  - Use a detailed source inventory;
  - Use a realistic diet; and/or
  - Use site specific values for the transfer parameters.

The same model might be suitable to be used for all three stages of an assessment, or increasingly complex models can be used at each stage.

The measurement programme will be designed to provide adequate data to allow each stage of the assessment to be carried out. This takes account of the need to make the process as cost effective as possible once safety requirements have been met.

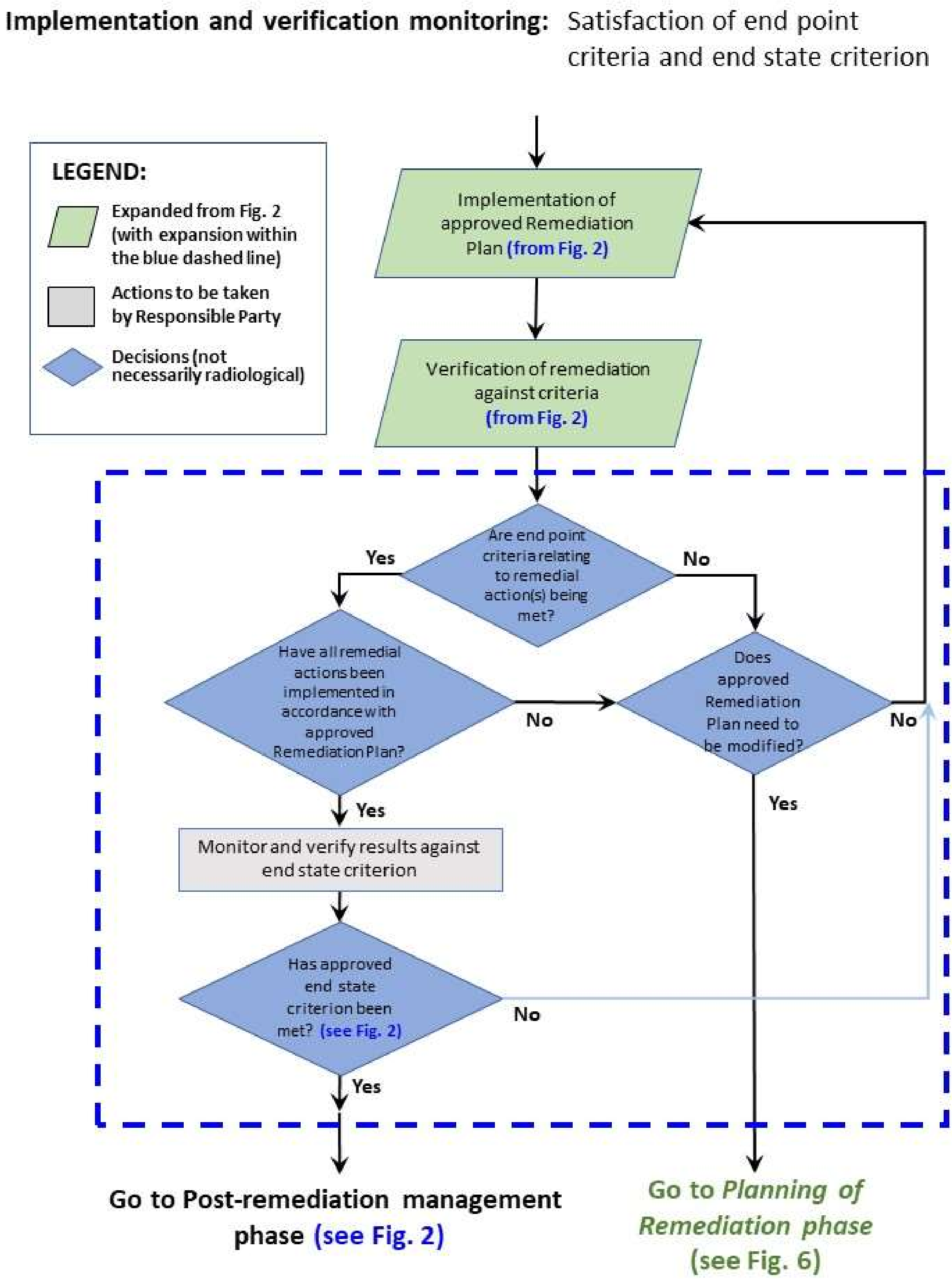

FIG. 8. The methodology for evaluation of possible remedial options (expanded from Fig. 2).

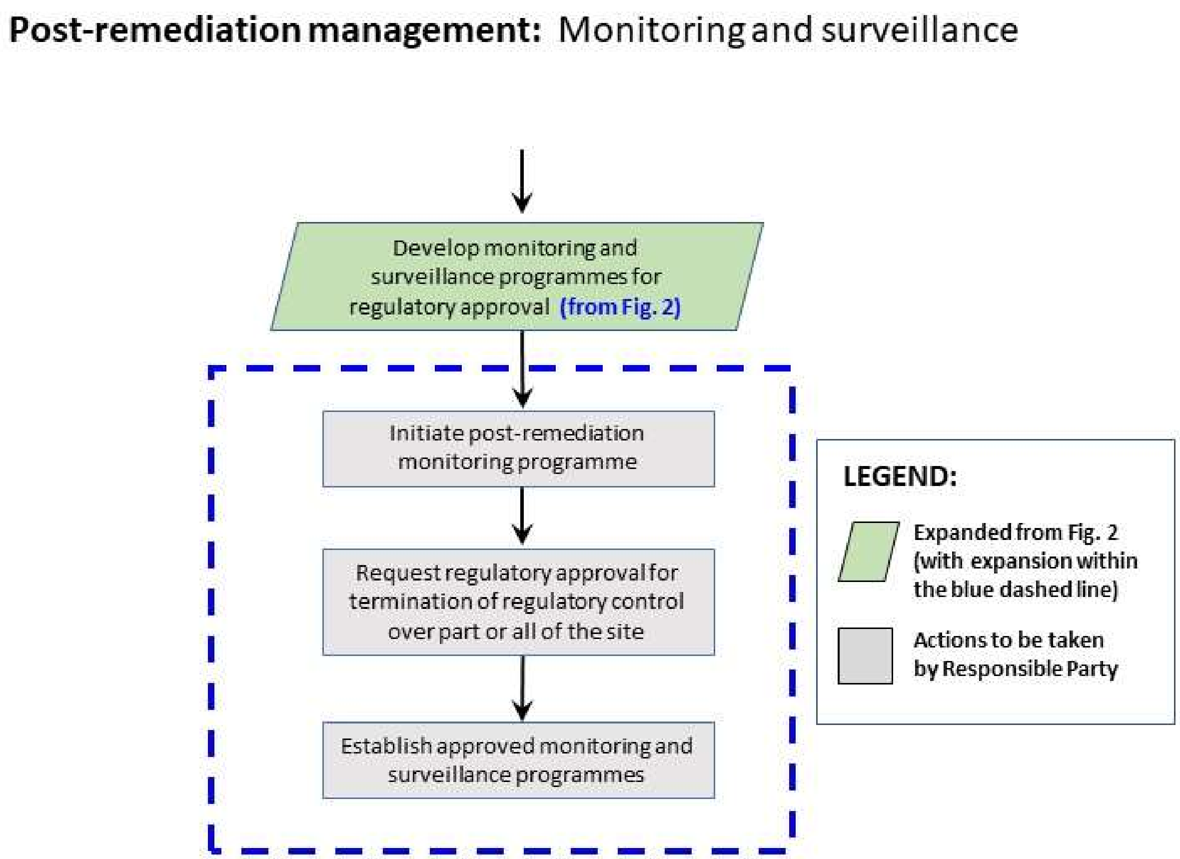

FIG. 9. The methodology to develop monitoring and surveillance programmes for regulatory approval in post-remediation management (expanded from Fig. 2).

5. MODELS FOR RISK AND ENVIRONMENTAL IMPACT ASSESSMENT

5.1. APPROACHES TO MODEL DEVELOPMENT

In the past, computational models were generally implemented by coding the underlying mathematical model using a high-level programming language such as FORTRAN or C++, and for some applications, this remains the preferred approach. However, this approach typically implies that the structure of the model is 'hard wired' and precludes changes to that structure to adapt the model to site specific conditions or new information resulting either from site characterization or research activities. Thus, adaptation of the model was typically limited to changing parameter values or parameter value distributions in probabilistic modelling. In more recent years, there has been extensive development of simulation packages that can solve a wide variety of models, if those models can be represented mathematically as sets of analytic functions and ordinary differential equations (ODEs). Compartmental models and discretized models of advective and dispersive contaminant transport can readily be represented in this form [85].

Furthermore, at one time, assessment models did not take into account the time-varying spatial distribution[32] of radionuclides and other contaminants, except to the extent that individual compartments in the model were taken to represent geographically distinct elements of the environment. For example, topsoil and subsoil in the area under consideration might be represented as two vertically stacked compartments or a river system might be represented as a linear chain of compartments. With point scale models, only a limited number of input parameters are needed, and it is often appropriate to specify their values or distributions directly. However, there is increasing reliance on spatially distributed models in which the parameters are defined on 2D or 3D grids. This is particularly the case for hydrological and hydrogeological models, which are naturally defined over surface water or groundwater catchments, with finite-element or finite-difference methods being used to simulate water flows and the transport of sediments and contaminants. For such models, it is often appropriate to use a geographic information system (often referred to as 'GIS') to enter input data and display the results obtained [86].

The degree of generality of packages for simulating ODEs varies substantially. Some packages (e.g. ModelMaker4[33]) can be used to simulate a wide variety of ODEs arising in different fields, including ODEs that represent severe discontinuities or gradients in the underlying relationships. However, for such systems, the expertise of the user is of importance, as few tools are available to assist the user in constructing systems of equations or in checking that the equations implemented are physically reasonable. In contrast, other systems (e.g. AMBER[34]; GoldSim[35]; and Ecolego[36]) are tailored for specific applications and provide a diversity of tools that facilitate the coding and checking of the systems of equations being implemented. Such tools include automatic checking of the consistency of units, vector and array processing that allows the transport of multiple contaminants between multiple compartments to be coded in a simple representation that is then duplicated, and the embedding of specialist calculational routines or lookup tables within the overall modelling structure, for example, by using dynamic

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[32] Sometimes called 'point scale' modelling, which accounts for the true horizontal ground distance of a point on a surface of the Earth, accounting for the same distance when projected onto a grid or a mapping plane (https://www.surveyorpockettools.org/how-to-use/point-scale-factor).

[33] https://www.apbenson.com/about-modelmaker-4

[34] https://www.quintessa.org/software/AMBER

[35] https://www.goldsim.com/web/home/

[36] http://ecolego.facilia.se/ecolego/show/HomePage

link libraries. Increases in computational power and the sophistication of numerical methods over the last few decades mean that the speed, stability, and accuracy of the routines used to solve stiff and non-linear systems of ODEs are now seldom an issue in model implementation. However, where complex, 3D models that couple multiple processes are employed (e.g. where hydrogeological models are coupled to representations of the kinetics of chemical speciation), computational resources can still be limiting, as is also the case for complex models embedded in a probabilistic simulation.

The simulation packages generally help the user in constructing a model through the provision of flexible ‘drag and drop’ facilities. Thus, systems of equations are represented by dragging ‘boxes’ into the model domain to represent variables and dragging in ‘connections’ between the boxes to represent the associated functional relationships. Thus, for example, the boxes could represent compartmental contents and the connections could be rate coefficients for the transport of those contents between compartments. Having structured a model by this ‘drag and drop’ technique, drop-down menus are used to specify the characteristics of both the boxes and the connections.

The flexibility and ease of use that such simulation packages provide mean that it is easy to both create and elaborate complex models. However, such models are also readily modified, meaning that it can be difficult to ensure quality control of a developed model, for example, in cases where the model has been changed in a way that is not consistent with its intended use. In such cases, it can be difficult for the regulatory body or other relevant authority to verify whether a model being used in support of a submitted safety case is fit-for-purpose. This issue is ameliorated by the functionalities provided in such packages for internal documentation and for partitioning of the model into multiple linked pages that display different aspects of its structure. Thus, modularity, continuing documentation and version control are encouraged. Nevertheless, these aspects cannot readily be fully automated, meaning that the model developer retains substantial responsibility for producing a model that is readily scrutable and has been subject to appropriate quality control throughout its development.

In some instances, an assessment will involve the combined use of several models that were developed separately. In this context, outputs from one model will be used as inputs to one or more other models and there may be feedback loops in which some of the outputs from the second model feed back into the first. Such loops are often described as ‘loose coupling’ between the models. In addition, the outputs from one model, although needed by a second, may not be in an appropriate form; for example, they may need conversion of units, or interpolation of values, either in space or time. Thus, the models may have to be augmented by interface modules that address these conversions. Such complexes of models and interfaces may be best handled within a specialized framework that facilitates the import of the necessary components, checks their consistency, and controls the overall simulations.

In this section, a description is provided of various specialist models, simulation packages and simulation frameworks for risk and environmental impact assessments. More details of these various models, packages and frameworks are provided in Appendix V, which comprises a compilation of information provided by participants as input to the study. Neither the information in this section nor the material in Appendix V is intended to be comprehensive. Rather, the aim is to provide an indication of a diversity of tools and range of applications to which they may be applied.

5.2. ENVIRONMENTAL SYSTEMS AND PROCESSES TO BE MODELLED

Radionuclides present in sources will be subject to environmental transport of the contamination away from its original location. Site specific conditions relating to the physicochemical attributes of the environment (e.g. pH, Eh, topography, soil type, concentrations of major cations and anions) can affect the form and speciation radionuclides and their transport. Such transport has two broad consequences: First, it alters the spatial distribution of the contamination over time, such that previously uncontaminated areas or environmental media may become contaminated; and secondly, because of their transport, contaminants may be transferred to humans and other biota, or to the environmental media that they use, potentially resulting in adverse radiological impacts on human health and the environment.

Both pathways of transport and radiological impacts on human health and the environment may be identified by describing the source-pathway-receptor linkage. This aims to identify the various sources of contaminants in the environment, the pathways by which those contaminants are transported, and the ways in which the transported contaminants impact on human health and the environment. It may also be helpful to undertake such an analysis in reverse, i.e., first identifying the pathways by which a receptor may receive exposure and then addressing which of those pathways could be subject to contamination by the source.

In general, transport will occur via the atmosphere, in the flow of surface waters, in the flow of groundwaters or by mass movement of solid materials [86]. Transport in the atmosphere can be of radioactive gases (e.g. ^{222}Rn or $^{14}CO_2$) or of liquid and solid aerosols contaminated with radionuclides. The two classes are not entirely distinct, considering that ^{222}Rn decays to short-lived progeny that become attached to ambient aerosols and are transported with them. Similarly, transport in surface waters can be in solution or as suspended particulate material. With transport in the atmosphere and in surface waters, deposition and resuspension processes need to be taken into account, and it may also be important to address transfers across the air-water interface, e.g., the outgassing of $^{14}CO_2$ from surface water bodies [85−86].

In the case of groundwaters, contaminants are generally transported in solution or in colloidal form. In either case, transport is due to advection and diffusion, with advection, inclusive of dispersion, typically dominant. Thus, modelling of the groundwater flow system is fundamental. In such cases, different modelling techniques may be adopted depending upon whether the system is saturated or unsaturated. Under saturated conditions, flow velocities vary linearly with hydraulic head differences and inversely with porosities. In contrast, under unsaturated conditions, complex non-linear relationships relate soil water potential and hydraulic conductivity to the degree of saturation present [87−88].

In the past, groundwater flow has typically been modelled using an approach that treats the system as a continuous porous medium (CPM). This is generally adequate for granular media, such as soils, though refinements, such as the use of a dual porosity representation, may be necessary. However, the CPM approach necessitates that at some spatial scale, smaller than the overall scale of the model, a representative elementary volume can be defined to which CPM properties can properly be assigned. However, in fractured media, the concept of a representative elementary volume may not apply because transport structures (fractures) exist at a wide range of different spatial scales. Thus, to represent flow and contaminant transport in fractured media, the last twenty-five years has seen the development of discrete fracture network models. Some of these models can also accommodate flows of multiple fluids through a fracture system, e.g., those developed in the context of oil reservoir engineering [89].

In comparison with water flow, contaminant transport in both porous and fractured media can be retarded. Retardation arises due to two broad classes of processes. These are sorption to surfaces and diffusion into the rock matrix. Sorption is generally treated as a reversible process and steady state is also assumed, such that the concentration on solids, C_s (mol/kg dry mass) and the dissolved concentration in groundwater, C_d (mol/m^3) are related by Eq. (1):

$$C_s = K \times C_d^n \qquad (1)$$

where K and n are empirical coefficients.

This relationship is known as the Freundlich Isotherm. In practice, radionuclides are typically present in trace amounts, so this approach can be linearized to give Eq. (2):

$$C_s = K_d \times C_d \qquad (2)$$

where K_d (m^3/kg dry mass) is known as the solid–liquid distribution coefficient.

With this linearization, it is valid and often useful to adopt units of Bq/kg dry mass and Bq/m^3 for C_s and C_d, respectively [86].

Diffusion into the rock matrix retards transport because much of the porewater within the matrix is trapped in dead-end pores or fractures. Therefore, a contaminant that diffuses into the matrix can only resume advective transport when it diffuses out again.

Advective and diffusive processes also govern transport in the atmosphere, with advection including turbulent transport, which is generally represented as a quasi-diffusive (Fickian) process. Typically, assessment models have used a Gaussian plume approach, with the contaminant advected downwind at the speed of the wind at a reference height (usually 10 m) [90]. Dispersion, which includes the effects of both diffusion and advective dispersion, is represented by treating the concentration of a pulse of contaminant as being normally distributed in both the horizontal and vertical directions, with the variance of the normal distribution dependent on atmospheric stability conditions and increasing with increasing distance downwind. In continuous release models, dispersion in the direction of advection is often ignored. However, more sophisticated models treat dispersion in 3D and, by modelling the plume as a sequence of pulses, can also handle changes in wind direction during a release [91−92]. To simulate long-distance transport of contaminants on distance scales of tens to thousands of kilometres, use is made of wind fields obtained from meteorological observations or from meteorological models used in predictive mode [93]. The various types of dispersion models may also represent other effects, for example, including dispersion into a building wake and both thermally- and momentum-driven plume rise.

Both atmospheric and groundwater transport can result in radionuclides entering terrestrial, freshwater, estuarine, and marine environments. In the terrestrial environment, agricultural systems tend to be of greatest interest because they are often key in assessing human exposures. However, other types of environments (e.g. wetlands) may also be of interest in terms of impacts on other biota. Deposition from the atmosphere to soils and plants may occur by either dry or wet deposition. Dry deposition of aerosols arises because of processes, such as gravitational settling, inertial impaction, or interception. It is typically represented using a deposition velocity that relates the deposition rate (Bq m^{-2} s^{-1}) to the radionuclide concentration in ground-level air (Bq/m^3). Although it has units of m/s, the deposition velocity is an empirical coefficient and is not related to a measurable velocity [86].

Wet deposition arises through the attachment of aerosols to precipitation (occurring as mist, rain or snow). Thus, the deposition rate is determined by the vertically integrated concentration of a radionuclide in air (Bq/m^2) rather than the ground-level concentration. This is related to the deposition rate ($Bq\ m^{-2}\ s^{-1}$) through a washout coefficient with units of s^{-1} [86].

Both dry and wet deposition are typically partitioned between vegetation and soils using a fractional interception factor that is typically taken to be a function of the standing biomass density or of the Leaf-Area Index of the vegetation. Activity deposited on the aboveground part of vegetation is subject to losses by weathering and leaf fall, and its concentration may also decrease by dilution as the plant mass increases during growth. In addition, some of the deposited activity is taken up by the plant and internally translocated to specific organs and tissues [86].

Radionuclides deposited directly on the soil or indirectly deposited due to losses from plants tend to be transported downward through the soil column in soil water by advective and diffusive processes, and by bioturbation. Some key radionuclides (e.g. ^{137}Cs) are strongly bound to soils. This limits their vertical migration to a fraction of a metre over several decades, placing particular emphasis on bioturbation and human disturbance of the soil as a transport mechanism for these radionuclides [73−74].

Roots take up radionuclides from the soil, with the degree of uptake determined both by the degree of sorption of the radionuclide to the soil and by its plant-determined bioavailability. Some radionuclides are efficiently taken up by plants, from example, ^{36}Cl, because chloride is only sorbed to soils to a very limited degree and because plants maintain high concentrations of chloride relative to soil solution. In contrast, other radionuclides (e.g. ^{239}Pu) are both strongly sorbed to soils and strongly bioexcluded from plants [73−74].

Radionuclides present in both soils and plants may be taken up by both wild and domesticated animals. For radionuclides that are strongly excluded from plants, intakes in inadvertently consumed soil may dominate, though uptake from the gastrointestinal tract of a radionuclide sorbed to soil may be much less than the uptake of the same radionuclide incorporated in the diet [86].

Both uptake of radionuclides from soil by plants and transfer of radionuclides from diet to animal tissues are typically represented by steady state transfer factors. However, in recent years, there has been increasing use of biokinetic models, typically comprising one or more well-mixed compartments, to represent these processes [86].

Representation of the transport of radionuclides entering estuarine or marine systems is often based on systems of well-mixed compartments. These are typically vertically stacked, with one or more compartments representing the water column in a particular area overlying compartments representing several sediment layers that may be distinguished by depth, susceptibility to deposition/resuspension, degree of oxygenation, or susceptibility to bioturbation [85]. Such approaches consider lateral transport of activity in solution within the water column and associated transport with suspended solids. Rates of transport are determined by tidal flows at a small scale and by residual currents at larger scales. Because of the complex geometry of ocean basins and the high spatial and temporal variability of flows, increasing reliance is being placed on 3D numerical representations of flow and transport, using finite-difference and finite-element techniques, including, in some cases, adaptive gridding to enhance simulation efficiency [94]. Sorption to sediments is typically represented using steady state distribution coefficients, as discussed above in the context of groundwater transport in soils and rocks [95]. Similar methods are increasingly being used in modelling the transport of radionuclides in surface water catchments in 3D, taking transport in both groundwater and surface water into account (for an early study, see Ref. [75]).

Radionuclide uptake by freshwater, estuarine and marine organisms is typically modelled using empirical concentration ratios[37], assuming steady state conditions [95]. However, in some specialist applications, biokinetic models of retention may be used. This can be applicable where organisms can range widely through an environment in which spatial and temporal variations in radionuclide concentrations are significant.

5.3. MODELS AND MODELLING SYSTEMS THAT ARE AVAILABLE

The information in Sections 5.1 and 5.2 does not provide a comprehensive description of approaches to modelling, but it does provide a context for discussing the various models identified as relevant by participants in this study and described in detail in Appendix V, where references to websites providing details of the models are given. Following the discussion in Section 5.1, this subsection covers, in turn:

- Framework packages that provide a structure or architecture within which to models developed in various contexts can be applied in an integrated manner through the provision of interfaces converting outputs from one model, such that they are suitable for use as inputs to one or more other models;
- Simulation packages that facilitate the implementation and modification of models characterized in terms of interlinked sets of analytical equations and ODEs;
- Models for specific components of the environment or sets of processes, with the equations of relevance either implemented directly in the model or in an underlying simulation package that is typically only accessed by an advanced user.

5.3.1. Framework packages

The characteristics of a framework package are illustrated by FRAMES (Framework for Risk Analysis Multimedia Environmental Systems). This is software that can use several separate environmental transport and risk assessment models with user-developed databases and accommodate user-designed scenarios of exposure and exposure pathways. It uses a common application programming interface for data transfer between models, either built-in or those defined by the user. It also supports sensitivity and uncertainty analyses on data from deterministic models.

Once a model is set-up for the scenario, the user can change a parameter or database and rerun it. Problems involving radionuclides, chemicals, groundwater and surface water flow, and atmospheric dispersion can be handled in an integrated operating package. Specialized programming knowledge is not needed, so that the user can concentrate on the environmental problem, and not the programme development.

The software can provide risk and dose assessment for radiation and/or hazardous substances in environmental settings. It does this by using multiple ‘medium specific’ models (e.g. air, water, and human impacts) with a database of chemical properties with associated environmental parameters to solve risk analysis problems.

[37] The application, uncertainties and limitations of such steady state ratios are described in Ref. [74].

5.3.2. Simulation packages

Simulation packages are exemplified by ModelMaker4, AMBER, GoldSim and Ecolego.

ModelMaker4 defines components and relationships between them in a generalized way that allows almost any physically plausible system of ODEs to be represented. Thus, for example, by defining displacement and velocity as the two components and relating them through acceleration calculated from their instantaneous values, a damped, non-linear, harmonic oscillator is readily simulated. Similarly, non-linear changes in the abundance of populations in a contaminated area in response to changes in resources, variations in radiation dose rates, in- and out-migration, and alterations in the relative sizes of groups within the population with different radiosensitivities can be readily simulated.

AMBER defines components and relationships in a way similar to ModelMaker4, but the package is designed primarily for compartmental modelling. However, recent enhancements in model specification and display mean that it is now relatively easy to generate representations of spatially distributed 3D systems as sets of compartments with well-defined spatial boundaries between them. A particular advantage of AMBER is the incorporation of vector and matrix multiplicity. Thus, variables and coefficients can be defined as arrays, allowing multiple, interlinked calculations to be readily coded. The package has facilities for the creation of sub-models with those sub-models shown and documented on separate pages, facilitating internal documentation. AMBER has been used to implement the United Kingdom Food Standard Agency's PRISM package for modelling radionuclide transport in agricultural systems. Standard users of PRISM do not see the underlying AMBER implementation. However, advanced users can set up a model via PRISM and then modify or interrogate it via the underlying AMBER encoding.

GoldSim has a similar range of functionalities to AMBER. However, the models are structured rather differently. In AMBER, the emphasis is on an overall model with sub-models, analogous to a main routine and subroutines in a high-level language program, such as FORTRAN. In contrast, GoldSim models are based on 'containers'. A container can hold, for example, all the material properties needed in a model, or all the input probability density functions used. Of course, a container could be like a sub-model in AMBER, if it held the compartments between which transfers were calculated. GoldSim also places greater emphasis on facilitating simulations of advective-dispersive transport than the other packages described in this publication.

Ecolego is used for creating dynamic models and performing simulations and for conducting risk assessments of complex dynamic systems that evolve over time. To make complicated models with many interconnections easier to map out and understand, models are represented by interaction matrices instead of traditional flow diagrams. Combined with hierarchical containers (sub-systems), these greatly simplify construction and documentation of large and complex models. The reports generated from Ecolego contain all the details used in the model definition and its parameters. As with AMBER and GoldSim, although Ecolego is a commercial package, it includes a free player. Such players typically allow models to be run and parameter values adjusted, but do not allow the integral structure of the model to be changed.

All the above packages allow either deterministic or probabilistic simulations to be performed, facilitating both sensitivity and uncertainty analyses.

5.3.3. Process specific models

Process specific models include both integrated system models with explicit representations of a wide variety of processes and specialized models representing only a few processes. System models applied in this study are summarized below.

CROM is a generic environmental code that was developed by the Centre for Energy, Environmental and Technological Research (CIEMAT), Spain, to implement models recommended by the IAEA [96−98]. It includes a Gaussian plume atmospheric dispersion model and a surface water model that can represent transport by advection and dispersion in rivers, small and large lakes, estuaries, and coastal waters. Contamination of surface water bodies from an atmospheric release can be represented and the surface water component can be adapted to also represent sewerage systems. The terrestrial model accepts radionuclide inputs from the atmosphere and from surface waters and represents build-up in soil over a 30 year period, taking radioactive decay and ingrowth into account. Transfers of radionuclides to both terrestrial and aquatic biota is represented using steady state, element specific bioaccumulation factors. Effective doses to humans are evaluated for external exposure, and for radionuclide intakes by ingestion and inhalation.

PC-CREAM 08 [99] is also a generic code comprising a suite of models and data appropriate to assessing the radiological impacts of routine, continuous discharges of radionuclides in atmospheric and liquid effluents. It includes a Gaussian plume model for atmospheric dispersion (PLUME), a resuspension model (RESUS), a model for calculating external exposure from radionuclides deposited on the ground and transported through the soil (GRANIS) and a set of models to represent radionuclide transfers into terrestrial foods (FARMLAND). The various models transfer their results to a model that calculates external and internal doses to humans (ASSESSOR). A marine dispersion model (DORIS) is also included in the suite of codes, together with two alternative river models.

DandD is a rather more specialized system model [100−101]. It is specific to decontamination and decommissioning scenarios and was developed by Sandia National Laboratories, USA. It simulates a building occupancy scenario and a residential scenario. The building occupancy scenario can be used to estimate effective doses due to external exposure, inhalation, and secondary ingestion of radionuclides from occupancy of a contaminated building. The residential scenario relates to soil contamination and uses a stack of three compartments to represent surface soil, underlying unsaturated soil and an aquifer. Exposure pathways include ingestion of drinking water and various foods contaminated via irrigation water, and dust inhalation. The code operates probabilistically, facilitating its use for sensitivity and uncertainty analyses.

DOSDIM is a multi-compartmental model of the biosphere that has been used several times in international code comparison exercises [102]. It includes a Gaussian plume model of atmospheric dispersion, but is mainly of interest because it can be used to model radionuclide transport in the unsaturated zone underlying the contamination. For this, it relies on calculations performed with the HYDRUS−1D and HYDRUS−2D codes [103−104]. These codes can be used to solve Richards' equation for variably saturated water flow and the advection-dispersion equation for radionuclide transport in both the unsaturated and saturated zone. If 3D flow fields need to be simulated at a catchment scale for surface waters, the unsaturated zone, and the saturated zone, the MIKE-SHE/MIKE11 suite of codes from the Danish Hydrological Institute may be used. These codes can accept as input detailed meteorological data obtained on a sub-daily timescale.

ReCLAIM is a spreadsheet based tool that can be used to address various land use scenarios for contaminated land [105]. It addresses external irradiation from contaminated land or water bodies, dermal contact with contaminated ground, inhalation of dust and ingestion of contaminated water, soil and various plant and animal food products, including freshwater fish. The scenarios include those that relate to agricultural workers and their families, recreational use of the land, construction workers, householders and persons present in schools and offices. Residential, allotment and commercial or industrial uses of the land can also be represented.

RESRAD is not a single code, but rather a series of codes developed at the Argonne National Laboratory for the US Department of Energy (DOE) [72, 100−101, 106−113]. The original RESRAD code is now named 'RESRAD (onsite)'. It can be used to model external irradiation from contaminated soil, inhalation of airborne radionuclides including radon progeny, ingestion of plant and animal products contaminated with radionuclides originating from soil and irrigation water, ingestion of contaminated drinking water and soil, and consumption of fish from a contaminated pond. RESRAD (onsite) has limited source region geometry capabilities and is not designed to estimate off-site impacts. However, these can be assessed using RESRAD-OFFSITE, which addresses a similar range of exposure pathways to those addressed by RESRAD (onsite) and has a map interface to specify the primary contamination and off-site areas. RESRAD-OFFSITE simulates leaching of the contamination and subsequent groundwater transport using numerical methods of solution and adopts a Gaussian plume model that includes buoyancy driven plume rise for atmospheric dispersion. Scenarios of exposure that can be modelled include a rural resident farmer, an urban resident, an industrial worker, and various recreational activities. These RESRAD codes can be used in probabilistic mode, as needed, for sensitivity and uncertainty analyses. In addition, RESRAD-BUILD can be used in either deterministic or probabilistic mode to estimate radiation doses from buildings contaminated with radioactive material when they are being remediated or inhabited. Exposure routes modelled include external exposure, inhalation of dust and ^{222}Rn plus its progeny, and ingestion of dust.

SATURN is an integrated model for assessing the radiological impacts of contaminated land that has been implemented using the Ecolego simulation package. As with RESRAD-OFFSITE, it includes both an atmospheric dispersion transport module and a model for leaching from the contaminated land, downward migration in the unsaturated zone and downstream transport in the underlying saturated, groundwater system. Transport in the saturated zone is described using a 1D, flow-tube-based approach, with the geometry and flow properties obtained from field observations combined with hydrogeological modelling. Advection, dispersion, sorption, and radioactive decay are all represented in the 1D model of groundwater transport. Receptor environments for groundwater include a well and various surface water bodies, including rivers, lakes and coastal waters. The exposure pathways modelled are like those included in the RESRAD codes, but specifically include radionuclide concentrations in indoor air and the associated doses from inhalation and external exposure.

AMCARE is also an integrated system model developed under the CARE (Common Approach for REstoration of contaminated sites) project of the European Commission. However, it was developed primarily to provide a common approach to the ranking of different remedial options and adopts a generally cautious approach to modelling transport pathways and human exposures. It can be used with the separate, proprietary CRYSTAL BALL sampling software [114] for undertaking uncertainty analyses. Internal components include GWAM for modelling radionuclide transport in groundwater and GASAM for modelling atmospheric dispersion using a Gaussian plume model. These modules provide inputs to a relatively simple, equilibrium-based spreadsheet model (DOSEAM) that is used for dose assessment.

Turning to specialized models used to represent a limited number of processes, the ERICA Tool [115] is a widely adopted approach to assessing radiation doses to non-human biota present in terrestrial or aquatic habitats. The user specifies radionuclide concentrations in the environmental media of interest and the code uses a database of concentration ratios to calculate concentrations in various types of biota. External and internal dose rates to the biota are then calculated using a simplified approach in which the biota are treated as uniformly contaminated ellipsoids and the dose rates are estimated as averages over the whole organism. A database provides reference concentration ratios and dose coefficients[38] for various standard types of organisms, and the user can define additional types of organism, as necessary. An underlying database provides information on the sensitivity of various types of organism to exposures to ionizing radiations.

The MicroShield tool can be used for estimating external exposures in various geometrical configurations. This allows the definition of surface and volume contaminated sources of various idealized geometries to be specified. Self-absorption of photons in the source can be simulated, as can attenuation in shielding materials located between the source and the dose point. MicroShield is a useful supplementary tool in situations for which it is not adequate to specify the source of external radiation as an infinite plane or slab of specified thickness. Thus, it may be particularly useful in assessing dose rates local to contaminated buildings or in the vicinity of waste packages. The ROOM model [116] can also be used to estimate indoor gamma doses from building materials containing naturally occurring radionuclides for rooms of user-specified geometry built of materials of user-specified composition.

CARAIBE is specific to estimating ^{222}Rn concentrations within buildings using a 1D vertical dispersion model, with the source located in the underlying soil and a model of ventilation that accounts for airflows into and out of the building. Similarly, CITRON is focused on calculations of concentrations of ^{222}Rn and its short-lived progeny in the open air due to dispersion of ^{222}Rn from the underlying ground. It accounts for both wet and dry deposition of those progeny.

CHAIN and CHAIN 2D are models for representing radionuclide transport in unsaturated soils [117]. Both codes can represent chain decay. CHAIN uses an analytic approach under steady state conditions to solve the advective-dispersive transport equation for chains of up to four members. CHAIN 2D can be used to solve Richards' equation for flow and models advective-dispersive transport of either contaminants or heat. The contaminant transport equation can be modified to represent non-linear, non-steady state reactions between the solid and liquid phases, and to include linear, equilibrium reactions between liquid and gas phases. Decay chains of up to six members can be represented. FECTUZ is also a 1D transport model applicable to the unsaturated zone [118−119]. It assumes steady state flow and can handle decay chains with up to seven sequential members. MULTIMED_DP was initially developed as a multi-media model that could be used to simulate contamination migration through different pathways in air, surface water, soil, and groundwater [120−122]. It represents transport in the unsaturated zone using a 1D approach with uniform, steady flow. However, time-variable infiltration can be specified, and the 1D unsaturated zone transport model is coupled to a 1D saturated zone model that represents 3D dispersion, as well as linear absorption, first-order decay, and dilution due to recharge. MULTIMED_DP can handle chain decay for chains of up to three members.

[38] Sometimes called 'dose conversion factors' or 'dose conversion coefficients'.

MILDOS AREA is specifically focused on atmospheric releases from sites used for uranium mining and milling [123−124]. For releases in particulates, application is restricted to ^{238}U, ^{230}Th, ^{226}Ra and ^{210}Pb. In addition, gaseous releases of ^{222}Rn and its short-lived decay products can be modelled. A Gaussian plume model is used for atmospheric dispersion. Exposure pathways include inhalation, external exposure from ground deposits and from cloud immersion, and ingestion of contaminated foods.

Finally, it is noted that preliminary assessments of sites may use screening models, rather than the more detailed process-based models described above. The US EPA has developed Preliminary Remediation Goals (PRG) and Dose Compliance Concentrations (DCC) calculators that allow the user to modify default exposure parameters to calculate preliminary remediation goals that are site specific. Similarly, Oak Ridge National Laboratory (ORNL) has developed Building Preliminary Remediation Goals (BPRG) and Building Dose Compliance Concentrations (BDCC) to define preliminary estimates of remediation requirements for buildings defined as activity per unit area and/or activity per unit mass. These codes are used by the US EPA, as are the Surface Preliminary Remediation Goals (SPRG) and Surface Dose Compliance Concentrations (SDCC) calculators developed by ORNL for situations where the contamination of outdoor hard surfaces is an issue. In the United Kingdom, the RCLEA (Radioactively Contaminated Land Exposure Assessment) methodology can be used similarly to calculate potential doses to compare with regulatory criteria or to estimate limiting radionuclide concentrations against which future measurements may be compared.

6. TESTING OF THE GENERAL ASSESSMENT METHODOLOGY ON ACTUAL SITES

Parts of the general assessment methodology have been tested in several actual scenarios of exposure involving contamination either with NORM or with artificial radioactivity due to atmospheric nuclear weapons testing. Some of these scenarios relate to sites that have been partially remediated, making it possible to check the validity of some of the assumptions adopted in the modelling work can be checked.

The actual sites and models used for the testing of the General Assessment Methodology that are presented here are:

- Gela site (Italy) – RESRAD (onsite), RESRAD-OFFSITE, DandD and ReCLAIM;
- Botuxim site (Brazil) – RESRAD-OFFSITE and ERICA;
- Soeve site (Norway) – MicroShield, Ecolego, ERICA and RESRAD-OFFSITE;
- Maralinga site (Australia) – a description of the process is presented, but no detailed models were used;
- Tessenderlo site (Belgium) – AMCARE;
- Olen site (Belgium) – DOSDIM.

6.1. GELA SITE (ITALY)

6.1.1. Identify the problem (and site description)

The Gela site consists of a single phosphogypsum stack in Sicily, Italy, at approximately 1–2 km from the sea. Site layout and sampling points are shown in Fig. 10.

Between 1967 and 1981, phosphogypsum was directly released into the sea, and during the period from 1981 to 1992, residues were accumulated in a landfill, which consists of four basins (one empty). In 2002, an external wall, made with bentonite concrete, was constructed.

The disposal site is inside the boundary of a large industrial area. It was remediated under the control of the Italian Environment Ministry in compliance with regulations for chemical pollutants.

For any kind of industrial waste disposal facility, hydraulic control (for extraction, monitoring and treatment) of underground water inside and outside the boundary of the facility is mandatory to limit the migration of hazardous pollutants. For the Gela phosphogypsum stack, this hydraulic control is accomplished at the sampling points. The control can be stopped when the monitoring data indicate that the concentrations of the contaminants are below the regulatory limits over an extended period of time and are likely to remain so for the foreseeable future.

The designated future use of the area is a solar power plant installation. Capping of the phosphogypsum stack with a plastic liner and covering with clean soil has been carried out.

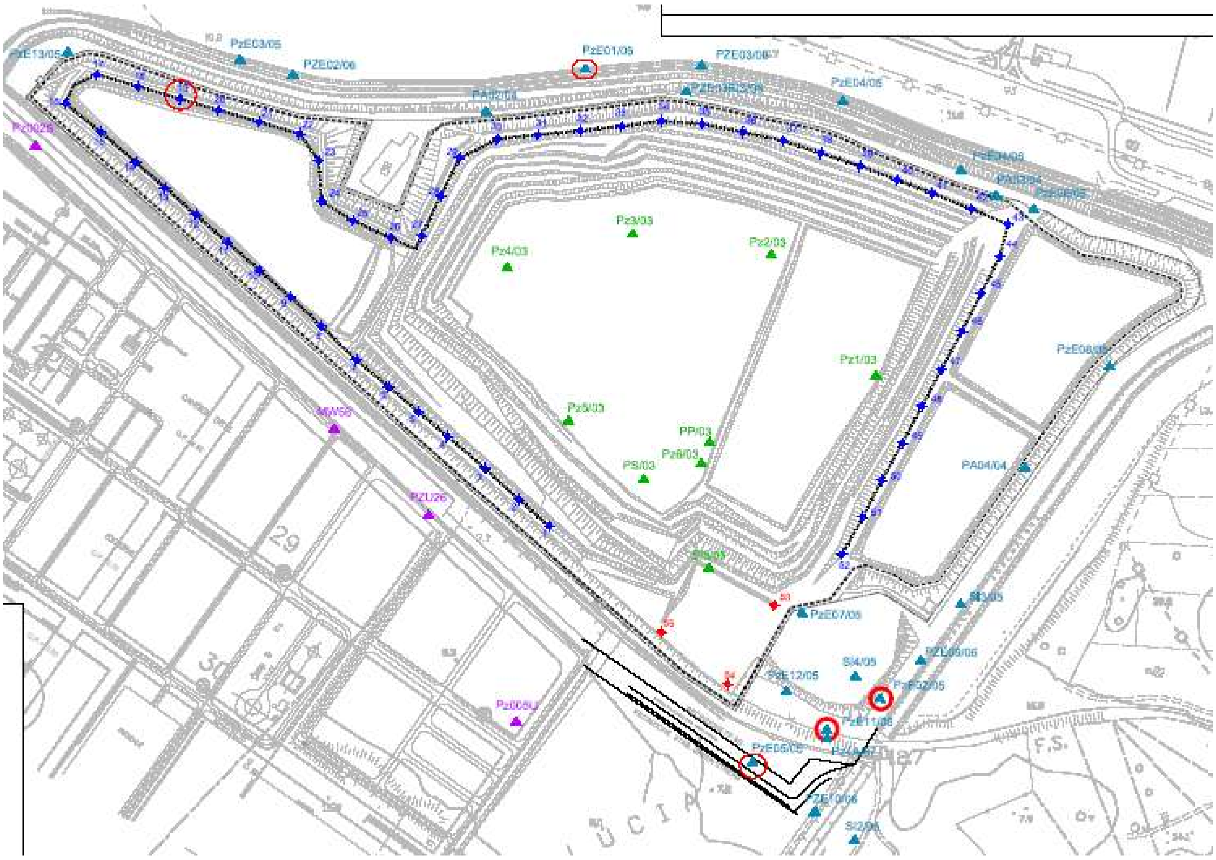


FIG. 10. Phosphogypsum stack at the Gela site and locations of sampling points.

6.1.2. Review of available information and limited characterization

The stack is surrounded by a concrete retaining wall. The layout of the retaining barriers is shown in Fig. 11. The wall penetrates 3 m into the underground clay. All the information available for the phosphogypsum, clay, and other materials is presented below.

The information provided on the industrial process is:

- Phosphorite consumption: 350–400 × 10^3 t/a;
- Type of process: mainly the Prayon phosphoric acid wet process;
- Production of phosphoric acid: 60–100 × 10^3 t/a;
- Production of slurry: 300 × 10^3 t/a;
- The concentration of phosphogypsum in the slurry was 10–20%;
- Until 2000, there were no radiological regulatory requirements applicable.

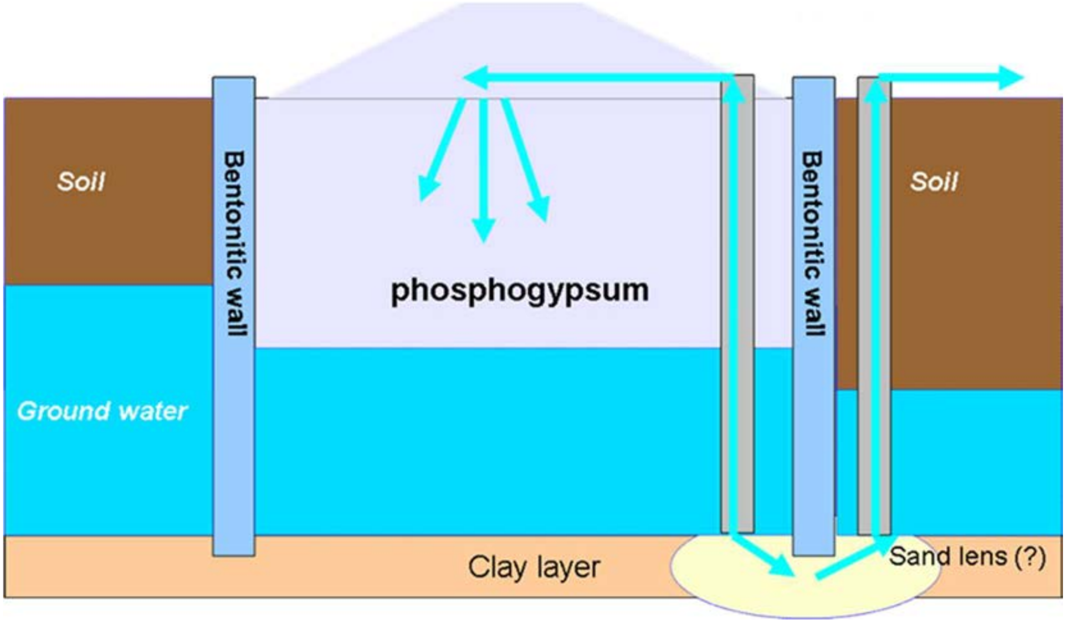


FIG. 11. Layout of the barriers for the containment of leachates from the phosphogypsum stack.

The characteristics of the phosphogypsum stack are as follows:

- Total area: 55 hectares;
- Average depth of phosphogypsum: 14.5 m;
- Hydraulic conductivity: 5×10^{-6} m/s.

The characteristics of the clay bed below the phosphogypsum stack are as follows:

- Total area: hundreds of hectares;
- Thickness: 20–30 m;
- Hydraulic conductivity: $10–12 \times 10^{-11}$ m/s.

The characteristics of the surrounding concrete retaining wall are as follows:

- 2–3 m inserted in the clay bed;
- A retaining wall surrounds the stack at a distance of about 5 m from the heap of residues to restrict the flow of leachate. It is 60 cm thick and 3550 m long;
- A drainage trench was built between the wall and the heap. A series of wells to collects rainfall percolate. The whole trench was lined with waterproof materials;
- A lens of sand receives water from the superficial groundwater from the surrounding hills. It is located at the southeast corner outside the stack wall, where the sampling points Pz 02/05 and Pz 11/05 are.

The characteristics of the lens of sand are as follows:

- Hydraulic conductivity: 2×10^{-4} m/s;
- Darcy velocity: 5 m/a.

Groundwater data are as follows:

- Normal groundwater flow direction: northwest to southeast (towards the sea);
- No data on the groundwater below the clay bed;
- Surface groundwater depends on very rare but intense rainfall events and is not permanent;
- For long periods during the year, piezometers and wells outside the N-NW side of the stack are dry. For that reason, groundwater samples are not regularly collected;
- The depth of water is 6.25 m in Pz 11/05 (12.3 m above sea level);
- The depth of water is 4 m in Pz 02/05 (10.4 m above sea level);
- Leachate extracted from these wells is pumped back to the top of the stack.

From this hydrogeological information, it is reasonable to conclude that leaching of NORM from the phosphogypsum stack will be highly unlikely after the completion of remediation. Radionuclides measured at sampling points near the southeast corner could result from contamination that occurred prior to construction of the containing wall. Table 3 presents the activity concentrations measured in phosphogypsum and phosphorite for some radionuclides from the natural decay chains of ^{238}U and ^{232}Th.

The measurements of the ^{234}U/^{238}U ratio downstream can indicate whether the retaining wall is functioning as designed and anticipated. Table 4 presents the measurements in groundwater and percolate of both radioisotopes and their ratio for the different measurement points and different days.

TABLE 3. MEASURED ACTIVITY CONCENTRATIONS OF RADIONUCLIDES IN PHOSPHOGYPSUM AND PHOSPHORITES

Phosphogypsum	
Radionuclide	**Bq/kg dry mass**
Ra-226	418 ± 27
Pb-214	313 ± 15
Bi-214	272 ± 12
Pb-212	19 ± 1
Bi-212	19 ± 2
Pa-234m	25 ± 4
Ra-226	410 ± 35
Pb-214	293 ± 27
Bi-214	248 ± 18
Pb-212	18 ± 1
Bi-212	19 ± 2
Pa-234m	<10
Phosphorites	
Radionuclide	**Bq/kg dry mass**
Ra-226	1249
Pb-214	1261
Bi-214	1170
Pb-212	40
Bi-212	41
U-235	65
Pa-234m	1415
Ra-226	1237
Pb-214	1230
Bi-214	1093
Pb-212	39
Bi-212	40
U-235	66
Pa-234m	1459

TABLE 4. MEASUREMENTS OF ^{234}U AND ^{238}U ACTIVITY CONCENTRATIONS (mBq/L) AT THE DIFFERENT POINTS

Sampling date	Sampling point	Sample kind	U-234	U-238	U-234/U-238
15/03/2006	PzE01/05	Groundwater	452±40	316±40	1.43±0.22
30/01/2007	PzE01/05	Groundwater	566±70	385±52	1.47±0.27
15/03/2006	PzE05/05	Groundwater	1045±119	908±104	1.15±0.19
30/01/2007	PzE05/05	Groundwater	194±28	179±27	1.08±0.23
30/01/2007	PzE11/05	Groundwater	70±16	49±14	1.41±0.52
15/03/2006	WELL#19	Percolate	15560±1710	14330±1570	1.09±0.17
30/01/2007	WELL#19	Percolate	15428±1693	14083±1547	1.10±0.17
30/01/2007	PzE02/05	Groundwater	397±50	326±43	1.22±0.22

6.1.3. Define remediation objectives

The main objectives of the radiological study that was considered in this case study were focused on the following objectives:

- **Primary objective:** To determine whether the remediation performed on the stack for chemical pollutant control also provides safe conditions from a radiological protection point of view.
- **Secondary objective:** To evaluate whether a significant change in the source term would occur if the old phosphoric acid plant were to be dismantled and disposed of in the confined landfill. To determine if the remediation would still be sufficient under these conditions and whether a new authorization for radioactive release could be be issued.

Two additional objectives were defined during the work:

- To perform an evaluation of the evolution in time of the migration of the relevant radionuclides in the absence of any physical barrier, in order to represent the situation before remediation (current measurements in groundwater show contamination that is probably due to the previous landfill configuration).
- To simulate a test with radioactive tracers for the integrity of the wall and clay system (actual test requested by the Italian Ministry).

6.1.4. Establish screening criterion

During the Preliminary Evaluation phase of remediation, a screening criterion can be set using one of several approaches and will depend on several factors. The most important factors are government policy and strategy, laws and regulations, location and major features of the site, anticipated future use of the site and surrounding areas, and the objective(s) of the assessment (see Section 6.1.3).

Two of the different types of criteria that can be applied are:

- **Type 1 screening criterion (based on activity concentrations):** Several international publications discuss the issue of the maximum concentrations of natural radionuclides in materials both for disposal or for reuse, that produce an increase in the exposure to humans that can be considered below regulatory concern, i.e. maximum concentrations that are broadly acceptable though not necessarily negligible. Two publications treat the issue starting from different approaches and reach activity concentrations of natural radionuclides (known as the exemption levels and clearance levels) that provide an acceptable increase in the effective dose [11, 125]. The clearance level recommended for ^{40}K is 10 000 Bq/kg (10 Bq/g), whereas for the rest of the radionuclides from the natural decay chains, it is 1000 Bq/kg (1 Bq/g) [126]. These levels can be reduced in national regulatory regimes based on a process of optimization of protection and safety by the decision makers (including the regulatory body, who would need to approve clearance levels and exemption levels).
- **Type 2 screening criterion (based on the increase of effective dose over the natural background):** Different levels of increase of effective dose over the natural background can be applied. In the case of NORM, 0.3 mSv was assumed in the case study to be an acceptable annual effective dose above background (see Ref. [125]). In addition, the annual effective dose limit of 1 mSv, recommended by the IAEA in GSR Part 3 [3] and

the ICRP in ICRP 103 [7], could be proposed as an acceptable level, depending on the factors that have been previously discussed, also recognizing the need for optimization of protection and safety below this value.

6.1.5. Screening assessment

A screening assessment is a simple, usually very conservative, assessment that is carried out to provide an overall indication of the level of concern. It is based on the idea that if the result of the screening assessment is below the screening criterion, then no further work needs to be done, whereas if the result equals or exceeds the screening criterion, a more realistic assessment is needed. The IAEA describes the process of performing screening assessments for a particular situation (i.e. in a planned exposure situation) [61] and the concepts presented are applicable in this situation.

If the Type 1 screening criterion is used (as described in Section 6.1.4), then, as the activity concentration values have been derived for the purposes of this case study using conservative assumptions, a direct comparison with actual measurements can be made. In the case studied here, all the measurements in the phosphogypsum are below the exemption and clearance levels. This is not the case for the phosphorites. However, the industrial activity itself was not evaluated, only the disposal of residual phosphogypsum. Based on this criterion and using the general assessment methodology described in this publication (see Section 4), neither remediation nor further studies are justified for the phosphogypsum piles for radiological protection purposes. That said, this is only applicable after a positive decision of the decision makers, who, in turn, need to consider the opinions of other interested parties.

Using the Type 2 screening criterion (see Section 6.1.4), based on the increase in effective dose over the natural background, a range of pathways and scenarios of exposure need to be discussed and considered. Inhalation of resuspended material and ingestion of foods cultivated in the area have been identified as possible pathways of exposure under the present conditions (after the remediation for chemical pollutants). If the integrity of the plastic liner and soil cover is degraded, other pathways, such as inhalation of radon, might become significant. Also, if there are no retaining walls or if their integrity is in doubt, groundwater, as a source of drinking water, could be an important pathway to consider. The most restrictive scenario would be for people settling on the site in the future. The scenario under consideration also defines the representative person (human) to be evaluated.

A first evaluation was carried out to quickly assess the radiological impact on the representative person. In addition, several tools were selected for assessing the annual effective dose under different assumptions. Specifically, DandD 2.1.0, RESRAD 6.5, RESRAD-OFFSITE, and ReCLAIM v3.0 were used in the assessment. Evaluations considering the retaining walls and covers were performed. The consequences of damage to the barriers were also evaluated. The screening evaluations were performed using default parameter values in the different tools. The default values were not always selected in the evaluations due to their conservatism, and instead typical examples were used in the software. Therefore, the results need to be checked to ensure that they are conservative.

A first evaluation was made for an extremely conservative case. For screening purposes, a rural residential scenario was considered, assuming that all the food ingested by the representative person was grown directly on the stack and no remedial actions were undertaken. If the only contribution to exposure is the consumption of vegetables grown on-site, radium in phosphogypsum transferred to plants will produce an annual effective dose, E, of 1.92 mSv.

This value was calculated using the following equation:

$$E = C_{soil} \cdot F_v \cdot H \cdot DCF \quad (3)$$

where:

E is the annual effective dose (mSv);
C_{soil} is the activity concentration of the radium in soil (418 Bq/kg dry mass);
F_v is a conservative transfer factor from soil to vegetables of 0.04 [61];
H is the quantity of vegetables consumed annually, which in Europe can be considered as 410 kg/a [61];
DCF is the committed effective dose conversion factor, which is 2.8×10^{-7} Sv/Bq, according to Ref. [3].

This result indicates that the annual effective dose exceeds the proposed screening criterion, even if only radium from one pathway of exposure is considered and it is assumed that a hypothetical individual eats only vegetables grown on the stack. This indicates the need to carry out an intermediate assessment, as suggested in the general assessment methodology presented in Section 4.

A different approach for the screening assessment was carried out using other software tools. One option was the application of ReCLAIM [105], which applied the default values for the parameters in the model. ReCLAIM is an Excel spreadsheet-based software created for screening assessments. The result obtained in this case was E = 0.38 mSv. This result was obtained using only the radium activity concentration in soil. Although the result falls below the annual public dose limit of 1 mSv, it exceeded an annual dose constraint for public exposure of 0.3 mSv [3]. Therefore, whether or not the situation is acceptable radiologically depends on the value that is adopted as a screening criterion (i.e. whether it is one of these values or another value that has been agreed, based on optimization of protection and safety, recognizing that 0.3 mSv is conservative and falls below the recommended range of 1–20 mSv for the reference level in an existing exposure situation). Moreover, the simplifications used in the modelling suggest that more detailed evaluations need to be undertaken.

The second software tool used for the screening assessment was the DandD 2.1.0 computer code developed by Sandia National Laboratories and used by the United States Nuclear Regulatory Commission (US NRC) [106]. A residential scenario on the site was modelled, which assumed that remediation involving placement of a cover on the stack had been undertaken. The following exposure pathways were considered: external exposure, dust inhalation, soil ingestion, drinking water and agricultural products. DandD does not consider the inhalation of radon gas, nor does it distinguish between the ingestion of animal products or vegetables. The activity concentrations applied in the model were 0.42 Bq/g for the ^{238}U decay chain (^{226}Ra, ^{210}Pb, ^{238}U, ^{234}U and ^{230}Th) and 0.04 Bq/g for the ^{232}Th decay chain (^{228}Ra, ^{212}Pb, ^{228}Th and ^{232}Th). The contaminated area was 50 000 m^2 and the thickness of the unsaturated zone was 6 m. The remaining parameters were assigned their default values. In this case, the result generated was a probability distribution (see Fig. 12), which showed that in almost every case, the annual effective dose exceeds at least one of the following criteria of 0.3 mSv (which is likely too conservative even for screening) or 1 mSv. In this case, the dominant pathway was ingestion of food produced on site from a small holding that provided food to a single household. The next most important pathway was external exposure of this household. Less important pathways were ingestion of drinking water, inhalation, and secondary ingestion.

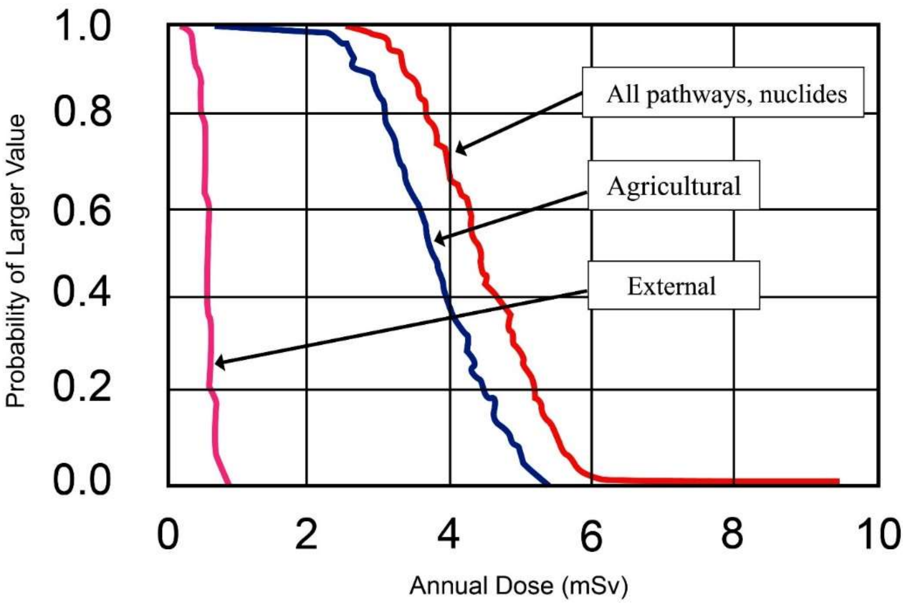


FIG. 12. Results obtained using DandD software.

6.1.6. Detailed assessment

All the screening assessments showed results above the screening criterion. Therefore, more detailed intermediate assessments were carried out in an iterative manner using less conservative (i.e. more realistic) assumptions. The same screening criterion, based on annual effective dose, was used.

6.1.6.1. Estimation of dominant exposure pathways contributing to dose

One of the conservative assumptions made for the screening assessment was that all the food ingested by the representative person is produced on the site and no remediation has been undertaken on the site. For the intermediate assessment, two scenarios of exposure were considered:

- A recreational scenario;
- A scenario that assumed 50% of animal feed is produced on the phosphogypsum stack. Again, it was assumed no remediation was undertaken.

For the first recreational scenario, the assumptions were as follows:

- The representative person is exposed for one hour per day;
- The exposure pathways are external exposure and inhalation of resuspended material;
- For resuspension, a conservative value of 10 mg/m^3 was assumed for the dust loading, as it is consistent with values measured under similar climatic conditions [127];
- For the dose coefficients[39] for external exposure, derived factors based on Ref. [128] were used, whereas for internal exposure, IAEA inhalation factors given in GSR Part 3 [3] were used.

[39] Previously called ‘dose conversion factors’.

Reference [128] is a rather dated reference and more recent compilations of external dose coefficients are available and would now be used (e.g., see Ref. [129]). Considering only ^{226}Ra, the scenario pathway specific results for annual effective dose were 13.8 μSv for inhalation, 130 μSv for external exposure from the soil, and 1.7×10^{-6} μSv for immersion in resuspended material. If the external exposure pathway is applied to two additional radionuclides (^{214}Pb and ^{214}Bi), assumed to be in secular equilibrium with ^{226}Ra, the annual effective dose calculated for this exposure pathway and these radionuclides is E = 299 μSv, which is approximately equal to the 0.3 mSv established screening criterion. If inhalation of ^{226}Ra in dust is included, values for the annual effective dose exceed the screening criterion. This assessment does not include the longer lived progeny of ^{226}Ra (^{210}Pb and ^{210}Po). These radioisotopes would not necessarily be in secular equilibrium but could make a significant additional contribution to the annual effective dose received.

For the second scenario, again it was assumed that there was no cover on the stack, and again a simplification of the source term was assumed, considering only ^{226}Ra. It was assumed that 50% of the feed consumed by animals was produced on the site, and that the activity concentration of natural radioisotopes in the rest of the food could be considered negligible. This feed consumption rate is almost certainly a conservative estimate because without a cover and the addition of topsoil on the stack, the amount of feed that could be grown on the waste pile would likely be limited. Indeed, it might not be possible to grow any food for humans or any feed for animals in such a case. Transfer factors and quantity of feed consumed by animals and food consumed by humans compiled in Ref. [61] were used to calculate an annual effective dose, E, of 304 μSv, which is again equal to the screening criterion of 0.3 mSv even without considering all radionuclides.

Another calculation was carried out using the RESRAD 6.5 software, applying the default values included in the code and considering all the exposure pathways (external exposure, inhalation, soil ingestion, vegetable ingestion, radon and drinking water) in a residential scenario. The activity concentrations used were 0.42 Bq/g for the ^{238}U decay chain (^{226}Ra, ^{210}Pb, ^{238}U, ^{234}U and ^{230}Th) and 0.04 Bq/g for the ^{232}Th decay chain (^{228}Ra, ^{212}Pb, ^{228}Th and ^{232}Th). It was assumed that there was no cover on the stack. The result obtained for E was 8.7 mSv, the main exposure being due to radon inhalation, as is shown in Fig. 13. Figure 14 depicts the evolution of exposure over time. After radon inhalation, the main exposure pathway is again the consumption of vegetables grown on-site, which might not be possible, as discussed above.

A second user also carried out a calculation using RESRAD 6.5. In this case, the user considered the scenario with and without the influence of radon inhalation. For the worst case, including radon inhalation, an annual effective dose of E = 9.8 mSv after 40–50 years was estimated, with estimated values always exceeding 9.45 mSv. When radon inhalation was not considered, the annual effective dose exceeded 1.6 mSv. The results are presented in Fig. 15.

None of the calculations performed show results that fell below the dose criteria discussed above, indicating that an unremediated site would have the potential to result in exposure exceeding an annual effective dose of 0.3 or 1 mSv. Accordingly, the exposure scenarios considered are highly conservative and unlikely to occur, particularly with proper controls, and the criteria against which exposure was compared are relevant for planned exposure situations, as opposed to existing exposure situations for which remediation would be undertaken. Therefore, although remediation performed due to chemical considerations could be justified, this is not necessarily the case on radiological grounds, since GSR Part 3 [3] recommends reference levels of 1–20 mSv below which optimization of protection and safety is undertaken in an existing exposure situation. Thus, in an existing situation, in terms of the radiological criterion, the annual effective dose can exceed 1 mSv. Whether these remedial actions also reduce the radiological risk below the approved screening criterion of 0.3 mSv is discussed in the following subsections.

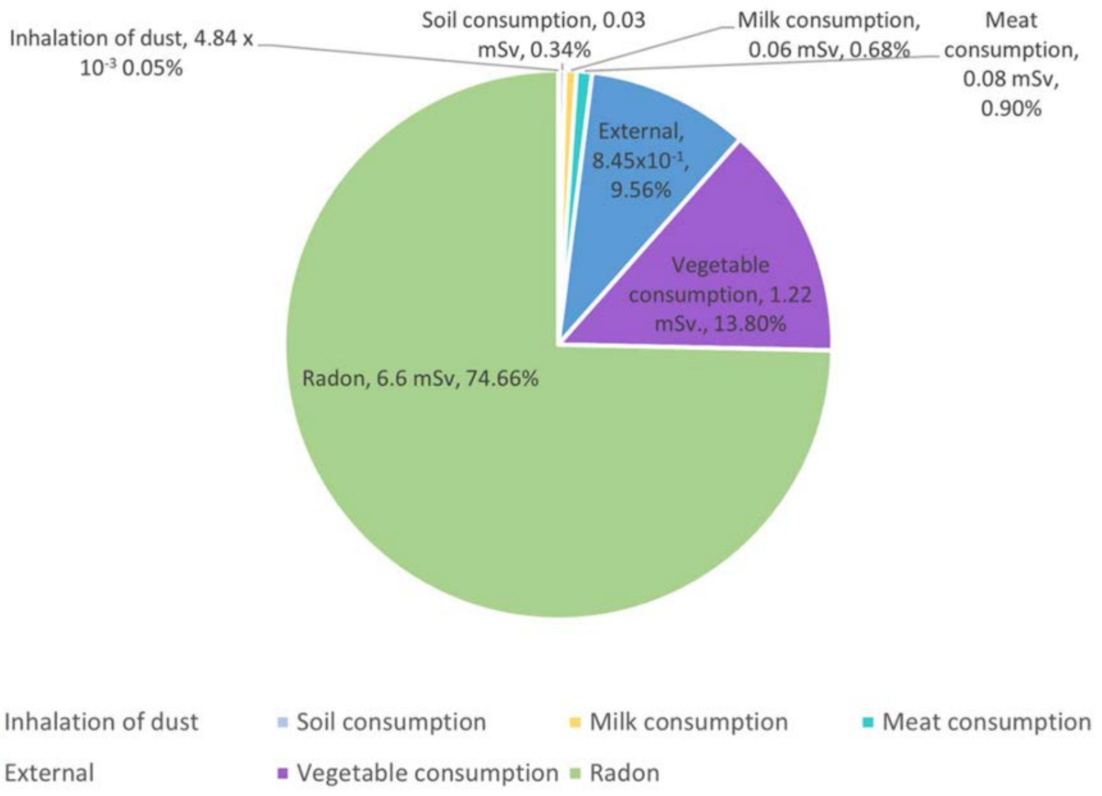


FIG. 13. Distribution of annual effective doses (mSv) for different exposure pathways in a residential scenario on the Gela site. Calculations carried out using RESRAD 6.5. No cover considered.

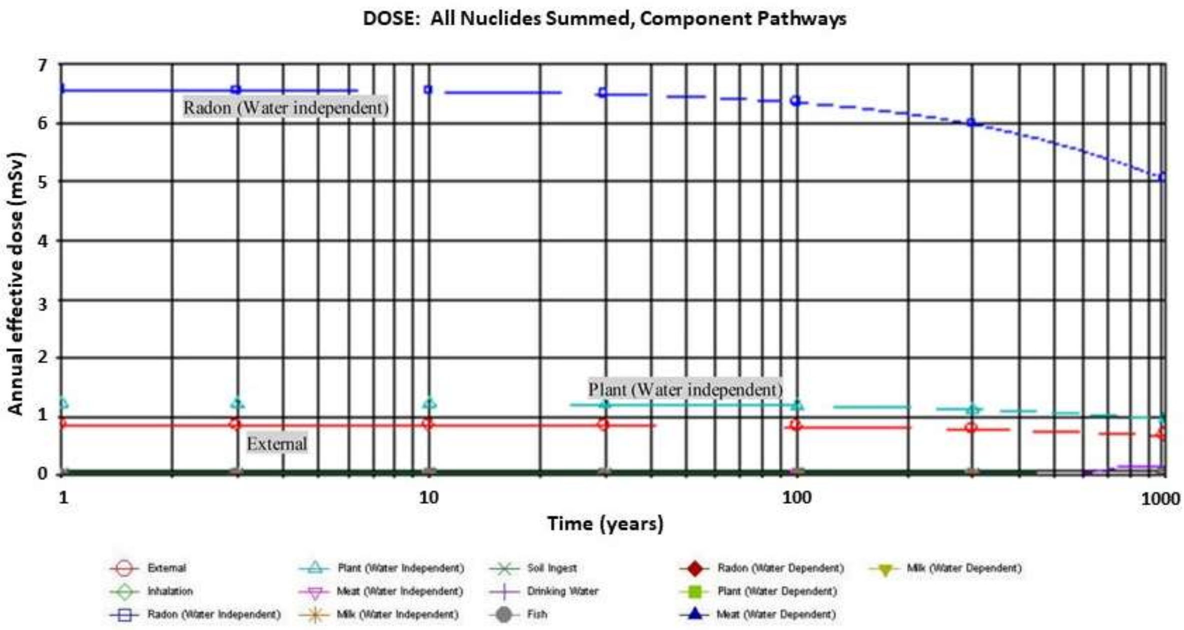


FIG. 14. Evolution of exposure with time.

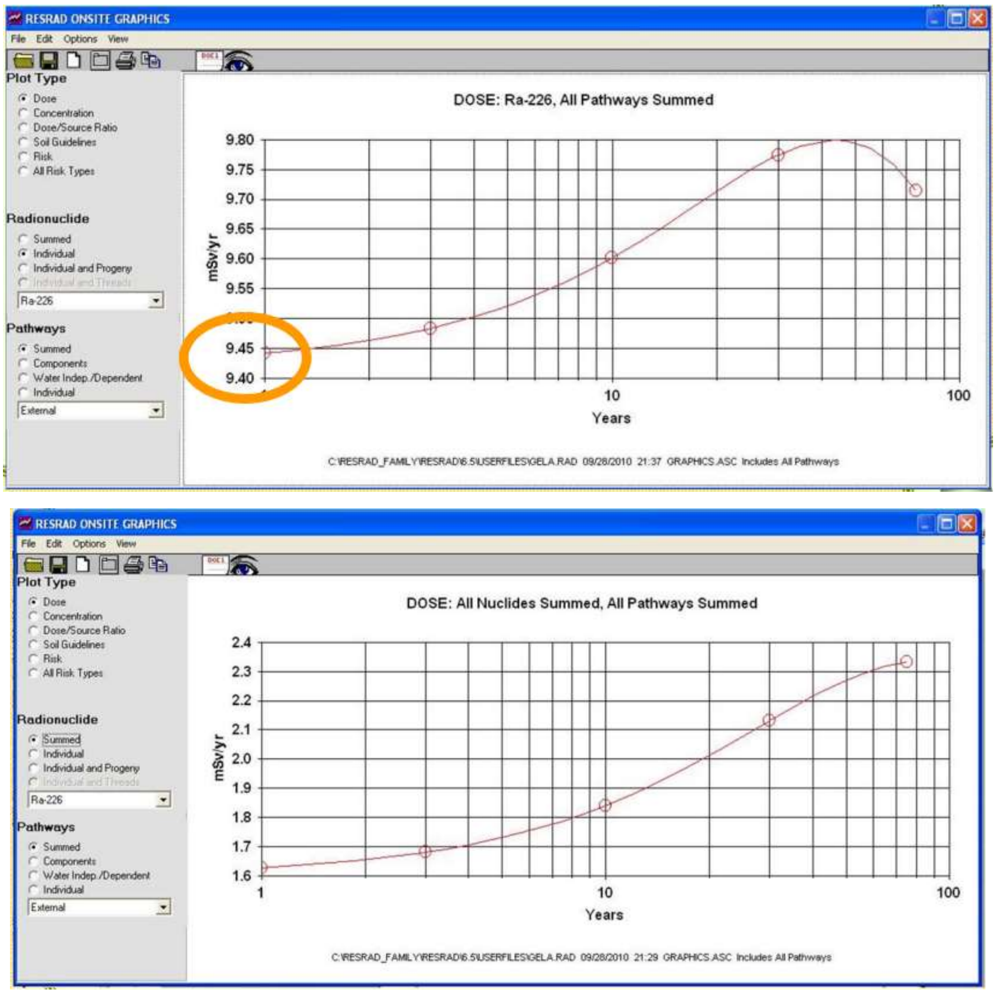


FIG. 15. Annual effective dose (mSv) evolution with consideration of radon exposure through inhalation (top) and without consideration of radon exposure (bottom).

6.1.6.2. Estimation of radium concentration in well water

The migration of radium from the waste in the stack into the environment was estimated using the HYDRUS−2D code. The waste layer, aquifer and bottom clay layer, as shown in Fig. 11, are considered in this code, along with their specific hydraulic conductivities, as provided in the site description. It was assumed that:

— The height of the waste is 14 m;
— There is a sandy soil layer of 2 m below the waste and that this layer is partly saturated;
— The net precipitation rate (precipitation minus evapotranspiration) is 0.368 m/year;
— The ^{226}Ra activity concentration of the waste is 1250 Bq/kg dry mass.

The radium concentration in well water was calculated. In doing so, it was conservatively assumed that the well is located next to the waste (i.e. the distance between the well and the waste disposal facility is zero) and water is extracted from the 1 m layer of soil just below the waste. It is emphasized that this is a very conservative approach because it is not usual (and sometimes not even possible) to extract water from the unsaturated zone. A well or borehole would usually be lined with an unperforated liner in the unsaturated zone and with a perforated liner in the saturated zone. In a dug well, the water level would typically correspond to the phreatic surface at the top of the saturated zone. Thus, in either case, the abstracted water would come from the saturated zone rather than the unsaturated zone. The results, assuming that abstraction from the unsaturated zone occurs, are provided in Fig. 16.

The effect of the bentonite retaining wall was also assessed. This could not be done using the code to directly calculate the rate of groundwater transport, probably because such transport for radium is slow even in the absence of a vertical barrier. However, by comparing the activity concentration of radium beneath the waste in the absence of a barrier, it was shown that the bentonite wall does have an effect. The activity concentration of radium below the waste after 200 years is predicted to be approximately 10% higher in the presence of a bentonite wall than in the absence of such a barrier. This is an interesting result, as it establishes that the bentonite wall might prevent some ^{226}Ra from moving horizontally off-site via groundwater. Therefore, the activity concentration of ^{226}Ra in water from a well located just outside the position of the bentonite wall could be lower or higher if there is a barrier in place, i.e., the higher activity concentration below the waste could be compensated for by reduced lateral transport.

6.1.7. Establish remediation criteria

Prior to implementation of site remediation, a previous step in the general assessment methodology, which is undertaken as part of remediation planning, is to establish relevant criteria, including the reference level, end point criteria and a final end state criterion (see GSG--15 [1]). The latter is typically set as an annual effective dose that can also be used to establish 'derived criteria'[40] in terms of activity concentrations that can be compared directly with measurement results, for example, from monitoring programmes. In this case study, an annual effective dose of 0.3 mSv was used as the end state criterion, recognizing that this is highly conservative and relevant to planned exposure situations, as opposed to existing exposure situations; for an existing exposure situation, a reference level within the range of 1–20 mSv is recommended by the ICRP in ICRP 103 [7] and in GSR Part 3 [3].

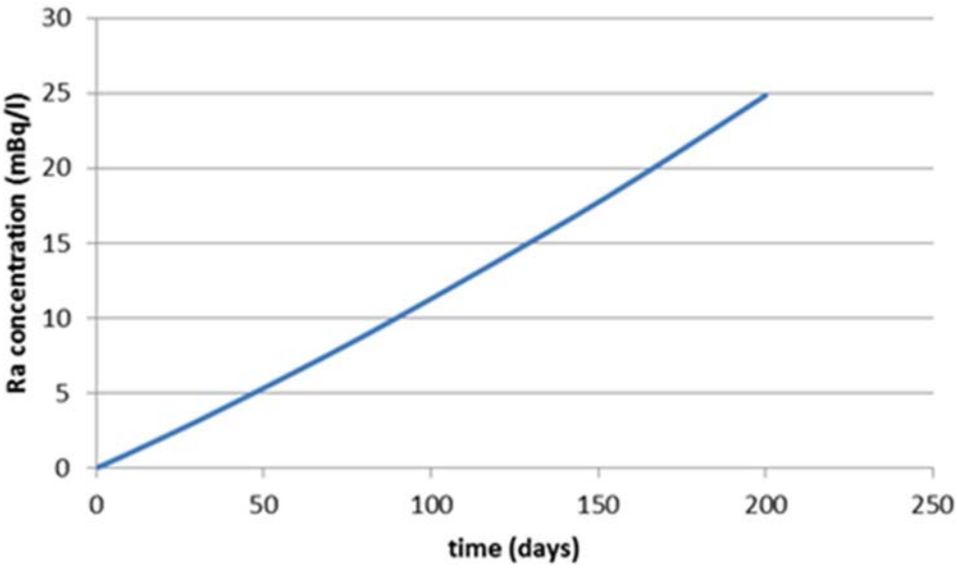


FIG. 16. Radium activity concentration in well water.

[40] Sometimes called 'secondary criteria' (see Section 4.4.6).

6.1.8. Implementation of remedial actions

As part of the remediation of the site, a plastic liner and several meters of clean soil covering have been installed on the stack at the site. The integrity of the plastic liner in the prevention of radon releases and the effect of soil depth on reduction of dose to individuals were tested as objectives of the radiological study. Human and animal intrusion scenarios were considered.

The integrity of the plastic liner affects radon releases, although, it is unlikely that a plastic liner will degrade for at least 30 years [130]. Therefore, an assumption of the degradation of the liner is considered a conservative scenario. However, the degree of conservatism may be less than it appears because when plastic liners or geotextiles are laid over extensive areas, there can be defects, puncture damage and joints, which can affect liner integrity, resulting in some water penetration from the outset. The effects of liner integrity were included in the assessments by considering the total dose with and without the radon inhalation pathway.

To calculate the necessary thickness of clean soil, the following codes were used:

- MicroShield was used for the case for which radon inhalation is assumed not to contribute to the dose (i.e. the plastic liner is emplaced and remains intact);
- RESRAD 6.5 calculations were made when radon inhalation was included.

In the case of the MicroShield predictions, the chemical composition and density of phosphogypsum and of a typical soil [128] were used. The soil composition is provided in Table 5. All the radionuclides in the decay chains were assumed to be in secular equilibrium and Fig. 17 presents the activity concentrations calculated using MicroShield for each radionuclide in the natural decay chain of ^{226}Ra after 30 years of decay.

Figure 18 shows the reduction in external exposure for different thicknesses of soil cover. As shown, a soil cover depth of less than 1 m is expected to result in a reduction in external exposure by three orders of magnitude, with an annual external dose that meets the end state criterion, which has been conservatively assumed to be 0.3 mSv.

The results generated using RESRAD 6.5, which assumed a 2 m thick cover of clean soil, are shown in Fig. 18. In this case, when it was assumed that a plastic liner was not installed, almost 100% of the dose was due to radon inhalation. The use of a clean soil cover is expected to reduce the effective dose to $E = 2.3$ mSv. It was also shown that a 4 m thick cover of clean soil would reduce the effective dose to less than 0.3 mSv. An increase in dose would be estimated to occur after several hundred years due to the erosion of the soil cover (a default soil erosion rate of 4×10^{-3} m/a was assumed in RESRAD). In practice, however, to achieve 2 to 4 m of total cover, a composite cap would probably be emplaced. Such a cap would be likely to comprise topsoil, clay layers, drainage layers and anti-intrusion layers to prevent intrusion by deep-rooted plants and burrowing animals. The topsoil layer might be 0.6 to 1.0 m thick. It would also be usual to adopt small slope angles and possibly armoured sides to counteract potential erosion on timescales of hundreds of years.

For a residential scenario, not considering radon exhalation, a maximum annual effective dose of greater than 1 mSv is estimated to occur after several hundred years. This is due to the ingestion of plants cultivated on-site and external exposure after the erosion of the soil cover.

If radon is considered, assuming there is no cover in place, a maximum annual effective dose exceeding 3 mSv is estimated to occur after several hundred years, as shown in Figs 19 and 20.

Using the default RESRAD 6.5 parameter values, the contribution to the annual effective dose, due to use of groundwater as drinking water, would exceed 5.5 mSv. In this case, the contribution of this pathway would be expected to decrease with time, as depicted in Figs 21 and 22. For this calculation, it was assumed that there is no physical barrier (no retaining wall or clay) and that all of the contaminated material is below the water table. Thus, the groundwater abstracted for use as drinking water is assumed to be drawn from this layer. However, if this layer is assumed to have a hydraulic conductivity appropriate for a typical clay, it is debatable whether it could, or would, be used as a source of abstracted water. Detailed studies would be needed to determine whether there is a range of hydraulic conductivity values that would result in radionuclide retention in this clay layer, coupled with a water yield sufficient to support abstraction demand.

TABLE 5. COMPOSITION OF A TYPICAL SOIL [128]

Element	Mass fraction
H	0.021
C	0.016
O	0.577
Al	0.050
Si	0.271
K	0.013
Ca	0.041
Fe	0.011
Total	**1.000**

Nuclide	curies	becquerels	µCi/cm³	Bq/cm³
Bi-210	4.7663e-002	1.7635e+009	1.0867e-006	4.0206e-002
Bi-214	7.8152e-002	2.8916e+009	1.7817e-006	6.5925e-002
Pb-210	4.7682e-002	1.7642e+009	1.0871e-006	4.0222e-002
Pb-214	7.8152e-002	2.8916e+009	1.7817e-006	6.5925e-002
Po-210	4.7135e-002	1.7440e+009	1.0746e-006	3.9761e-002
Po-214	7.8136e-002	2.8910e+009	1.7814e-006	6.5911e-002
Po-218	7.8168e-002	2.8922e+009	1.7821e-006	6.5938e-002
Ra-226	7.8167e-002	2.8922e+009	1.7821e-006	6.5937e-002
Rn-222	7.8168e-002	2.8922e+009	1.7821e-006	6.5938e-002

FIG. 17. Screen capture of RESRAD outputs indicating the estimated activity concentration of each radioisotope in the natural decay chain of ^{226}Ra after 30 years of decay calculated using MicroShield.

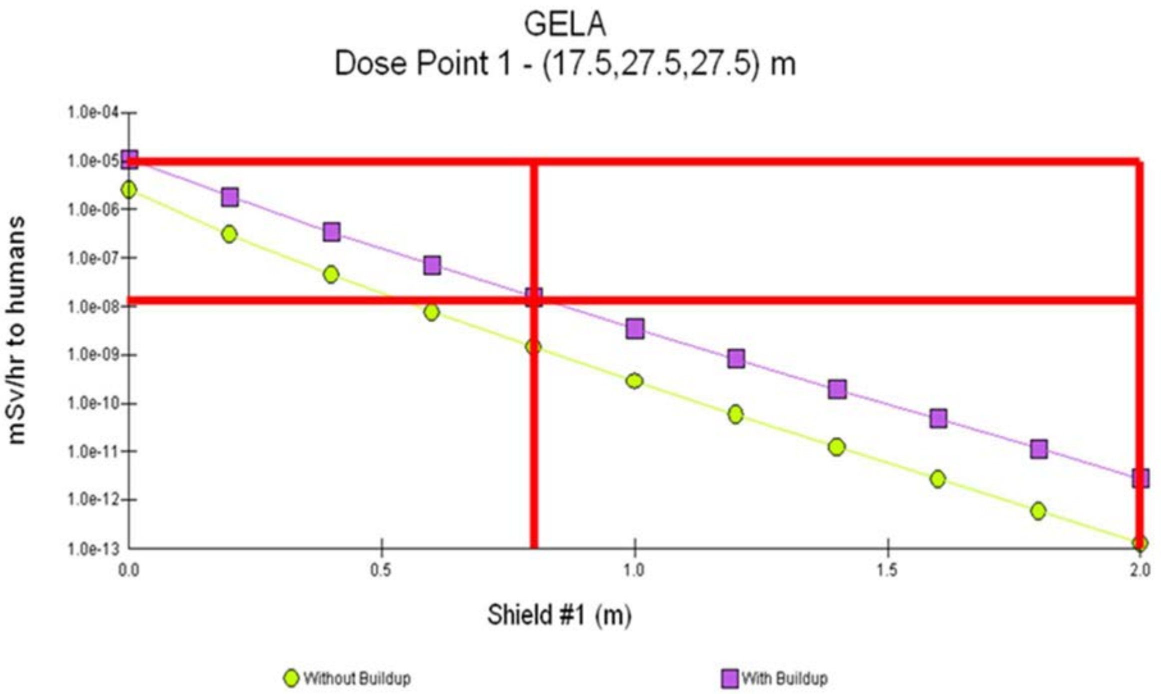


FIG. 18. Reduction of external exposure for different thicknesses of soil cover.

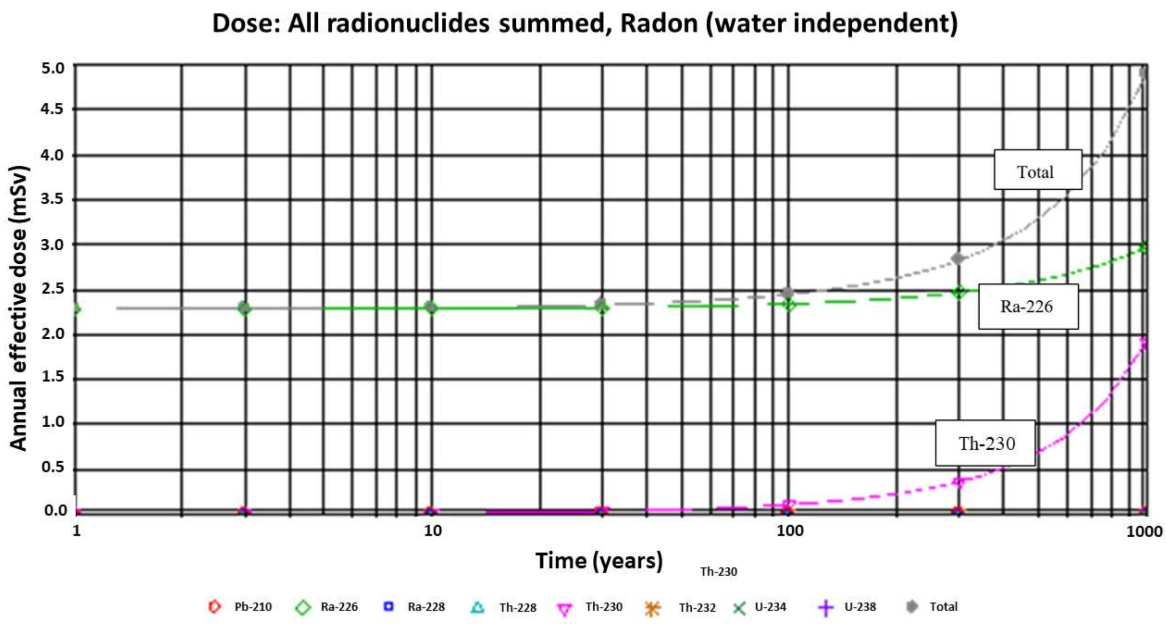


FIG. 19. Estimation of annual effective doses over time from radionuclides after covering the contaminated material with 2 metres of clean soil. The effects of erosion are expected to be observable beyond 100 years. However, by applying suitable engineering measures relating to cap design, the timescale for erosion effects to manifest could probably be greatly increased.

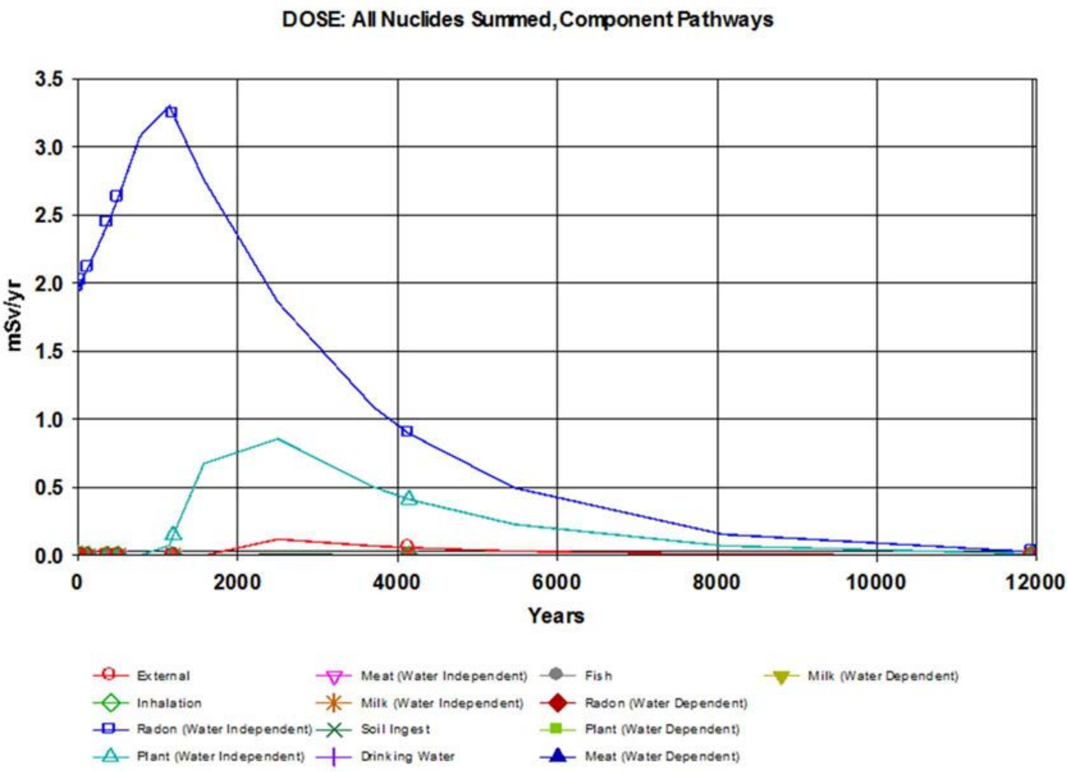


FIG. 20. Evolution of annual effective dose over time from different pathways of exposure assuming no remediation (i.e. no plastic liner has been installed).

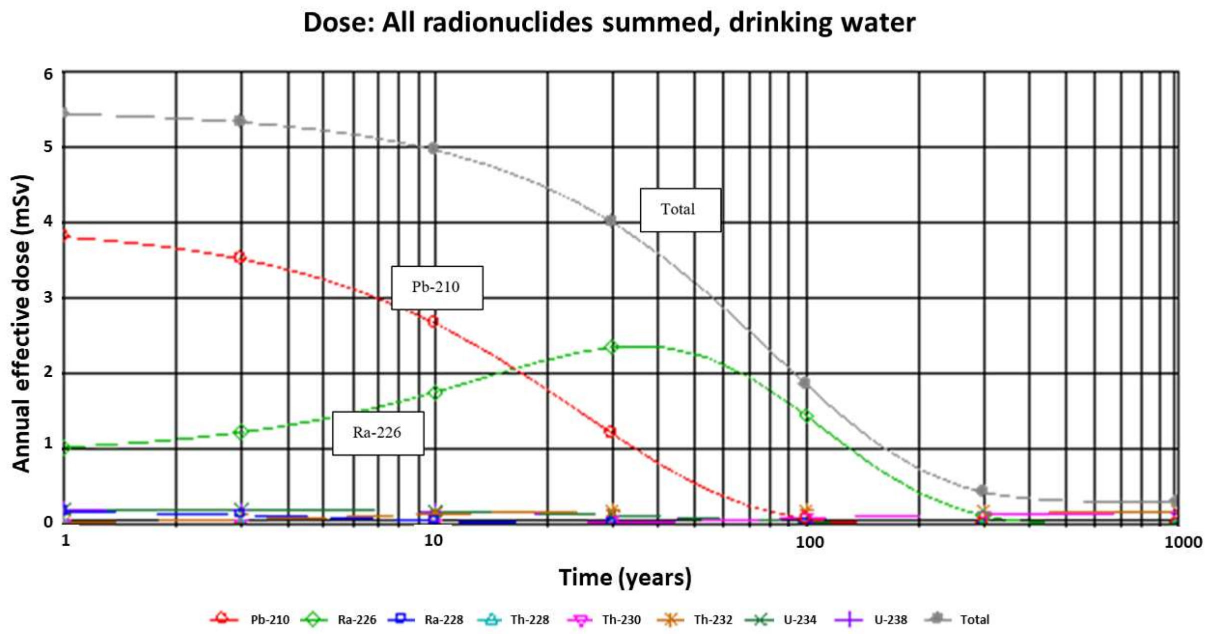


FIG. 21. Contribution of ingestion of drinking water to the annual effective dose over time. The total dose from all radionuclides is represented by the grey curve.

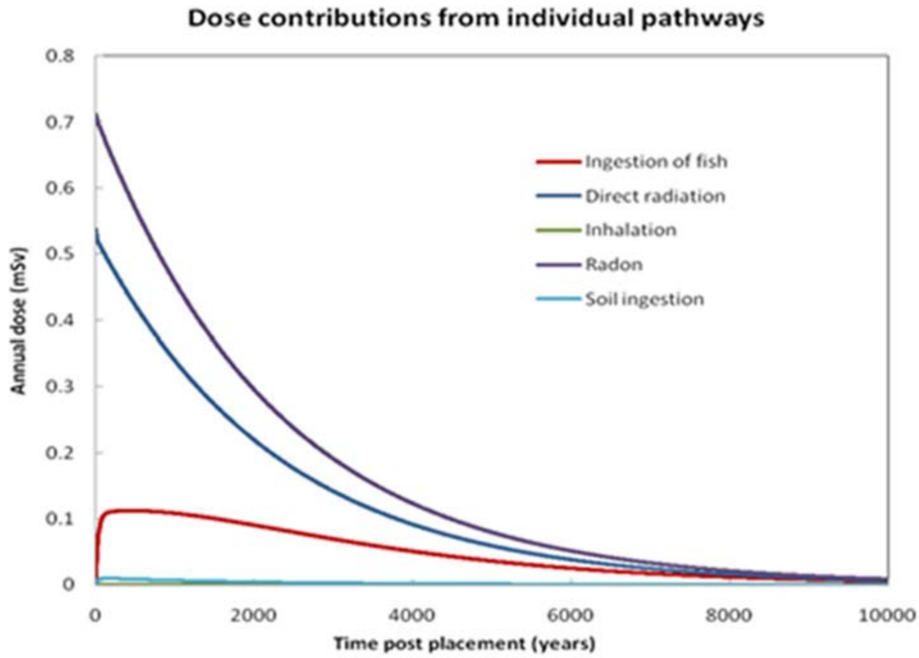


FIG. 22. Contributions of individual exposure pathways to annual effective dose due to groundwater.

It was shown that although several assumptions were needed to perform these assessments, several aspects of the remediation merit further investigation. Changes in the integrity of the plastic liner over time will affect radon releases, which were shown to be the main contributor to the annual effective dose in all cases. The integrity of the plastic will degrade over time (e.g. due to weathering or intrusions by humans or animals). If it is assumed that the integrity of the plastic liner is maintained over time, external exposure and groundwater would be the main exposure pathways. In the case of external exposure, a clean soil cover of 2 m would provide adequate shielding to reduce the exposure below the end state criterion, recognizing that the criterion that has been selected for the purposes of this case study is highly conservative and it is often not practicable to install a 2 m cover, for example, due to limited availability of clean cover material and/or the difficulty in justifying the resources (costs) that would be needed for a soil cover that is 2 m thick. In practice, rather than 2 m of soil cover, it is likely that a structured, multi-layer cap of similar thickness would be appropriate.

If maintenance of the integrity of the plastic liner cannot be assumed, radon will become the main contributor to the annual effective dose for all the cases considered.

6.1.9. Verification of remediation against criteria

Once remedial actions have been implemented in accordance with the approved remediation plan, verification of remediation outcomes against remediation criteria is necessary. Further detailed assessment is only needed in those cases where the intermediate assessment generates results that exceed the end state criterion. Based on the intermediate assessment, several questions were raised concerning the Gela site, the main issues being:

(1) The degradation of the plastic liner over time is a critical issue. This issue could be further investigated by means of a more detailed assessment of an engineered design for a realistic, multi-layer cap.
(2) The efficiency of the retaining wall against leaching is expected to greatly affect the magnitude of the annual effective dose. This could be investigated in detail using actual measured data from the site, if available. The loss of control of the site would likely result in a loss of integrity of the retaining wall, which in turn, would lead to an increase in the annual effective dose to levels that would exceed the end state criterion. One possible preventive measure that could be implemented is that the retaining wall could be keyed into the underlying clay layer. However, this would create a situation in which meteoric water infiltrating into the waste pile would have nowhere to go, likely leading to a bathtub effect with an increase in the water table in the waste pile and possible overtopping of the surrounding bentonite walls. This could also occur even with some leakage beneath the wall. Such a 'bath-tubbing' scenario was not addressed in this context, although it has been considered in other assessment contexts [131].
(3) The modelling indicates that preventing the influx of rainwater into the phosphogypsum stack significantly reduces the transfer of radionuclides to the surrounding environment. This can be achieved by covering the stack; however, further modelling might be necessary to determine the optimum cover thickness and the structure of the cover, with possible consideration of installation of an engineered multi-layered cap.
(4) Any groundwater flow occurs above the underlying clay layer, and is inhibited by the retaining wall. This reinforces the conclusion that the doses are likely to remain very low well beyond the projected institutional control period, provided the integrity of the retaining wall is maintained.
(5) In the calculations, residential occupancy is assumed either above the phosphogypsum stack or at the edge of it, and the annual effective dose to the representative person would be of concern after several hundred years. If the land is released for other uses, such as recreational or industrial use, the occupancy factor would be significantly decreased, which would significantly reduce the estimated doses to the representative person.
(6) Calculations suggest that any ^{232}Th in the stack will have negligible off-site impact, so the main efforts in refining the available data will need to focus on ^{238}U and daughters.

6.2. BOTUXIM SITE (BRAZIL)

6.2.1. Identify the problem (and site description)

Botuxim is a site where monazite ore was processed from 1945 to July 1992 to extract rare earth oxides. During processing, a residue containing Th and U oxide was produced. Figure 23 provides an illustration of the percentage levels of Th and U oxide that could occur from the processing.

A total of 3500 tons of residues known as 'Cake II' was stored on the Botuxim site in seven pools or silos (see Figs 23–24). Cake II is a mixture of 0.9% of U_3O_8 and 22% of ThO_2, with an activity concentration of approximately 1820 Bq/g dry mass.

The pools are 3 m deep, surrounded by 30 cm thick concrete walls and floors. Each pool is 0.5 m above the ground surface and 2.5 m underground. Pools are capped with concrete (see Figs 24–25).

A guard well is sited north of the silos (Fig. 25). High activity concentrations of radionuclides have been found in the well water. Some measurements carried out in the guard well and at two reference points are presented in Table 6 (Section 6.2.4).

The heavy rain that occurs in the rainy season seems to play an important role in determining the high concentrations of radionuclides being detected in guard well water.

6.2.2. Preliminary site evaluation and characterization

Some previous measurements were carried out around the site (Figs 26–27). Values measured during a total gamma radiation survey that was carried out ranged from 50 to 1000 counts per second.

Activity concentrations of several radionuclides from the ^{232}Th and ^{238}U decay chains that were measured in soil during 1993 were as follows:

- ^{228}Ra: 0.03 to 70 Bq/g dry mass;
- ^{226}Ra: 0.02 to 0.9 Bq/g dry mass;
- ^{238}U: 0.02 to 13 Bq/g dry mass.

6.2.3. Establish screening criterion and remediation criteria

The radiological criterion (i.e. the annual effective dose) for screening was set at 1 mSv (i.e. therefore, representing the screening criterion). This value was also established as the cleanup criterion, recognizing that this is highly conservative.

6.2.4. Screening assessment

RESRAD was used for the screening assessment. The scenario of exposure within RESRAD is shown in Fig. 28 and the silos are represented by a radioactive symbol in this figure. The guard well is to the north of the Botuxim site, and the river is represented as a blue line indicating surface water in Fig. 25 which is providing a water supply to Itu city. Vegetable production has been considered in this case.

Total annual effective dose for the highest activity concentrations of radionuclides in soil was calculated. Figure 29 indicates the contribution of each radionuclide to the annual effective dose and in particular, the dominant contribution of ^{232}Th to the total annual effective dose. The maximum value of the annual effective dose is $E = 0.44$ mSv.

TABLE 6. TOTAL BETA ACTIVITY CONCENTRATIONS IN SOME WATERS OF THE REGION (Bq/L)

Local	N	Minimum	Maximum	Geometric mean
Guard well	77	0.01	4.00	0.30
Mojolinho Creek	8	0.04	0.40	0.15
Itu town	6	0.05	0.20	0.11

Data supplied by Companhia Ambiental do Estado de São Paulo (CETESB).
N = number of measurements. A guard well is a monitoring well between the source(s) and the point(s) of exposure with concentrations below the site specific target level.

FIG. 23. 'Cake II' residue from monazite processing.

FIG. 24. Silos on the Botuxim site where residues from monazite processing are stored.

FIG. 25. Location of the Botuxim site (within the yellow box).

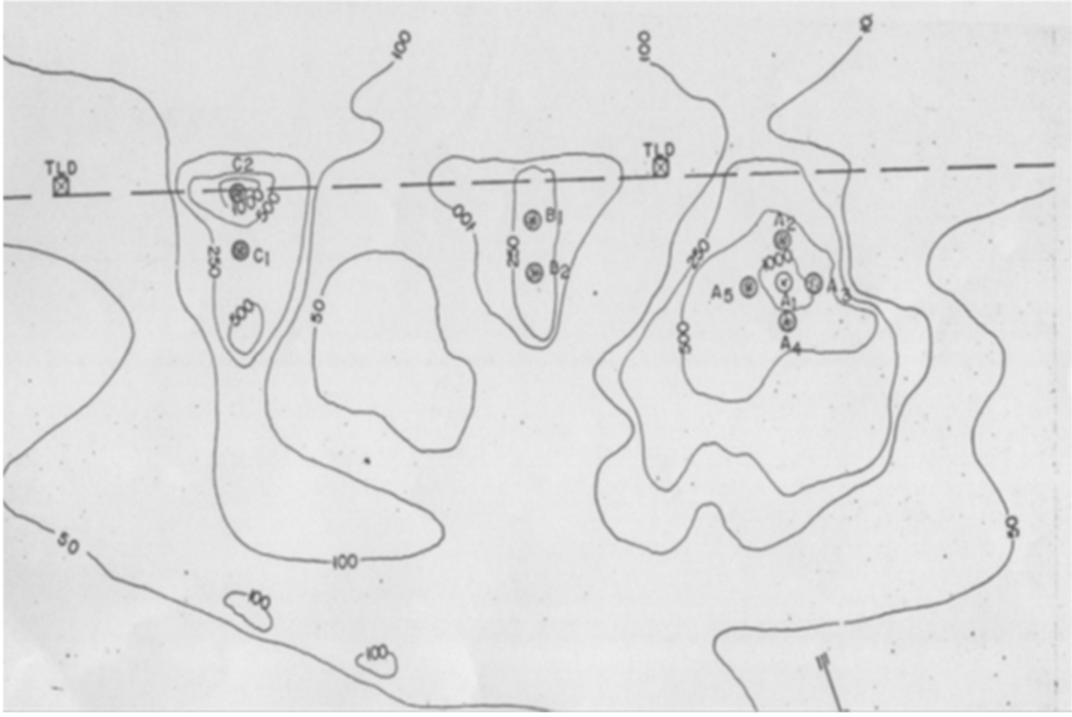


FIG. 26. Radiological measurements in counts per second for the area outside the fence around the silos.

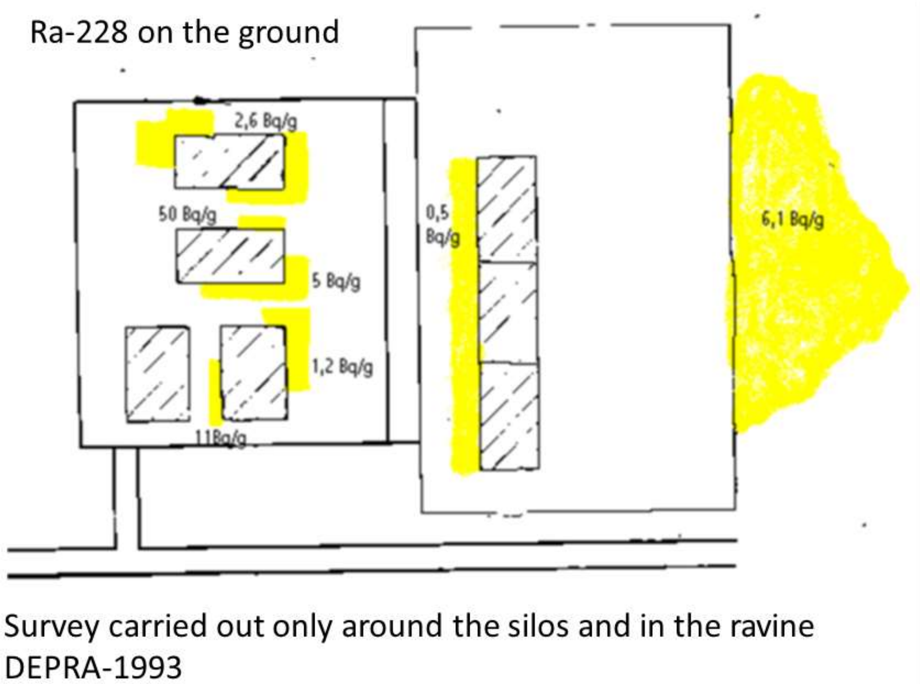


FIG. 27. Distribution of ^{228}Ra in soil around the silos.

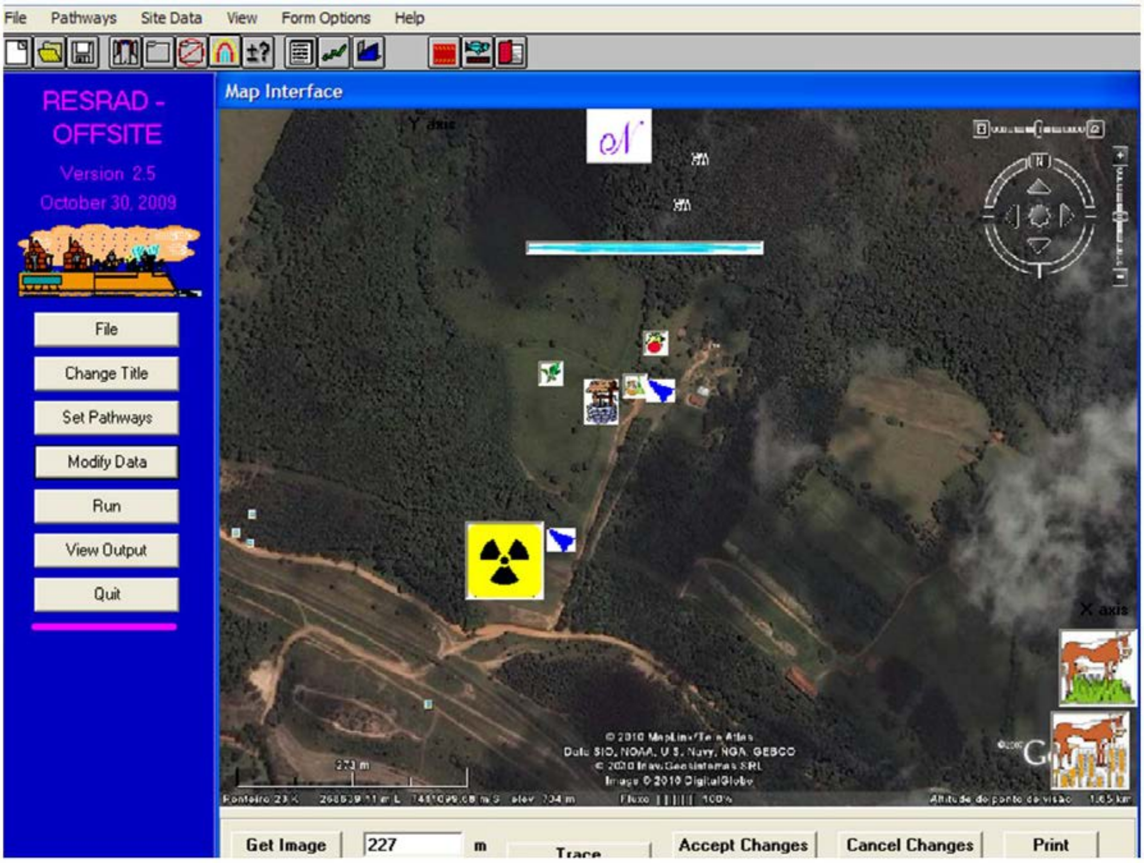


FIG. 28. Representation of the Botuxim case using RESRAD-OFFSITE.

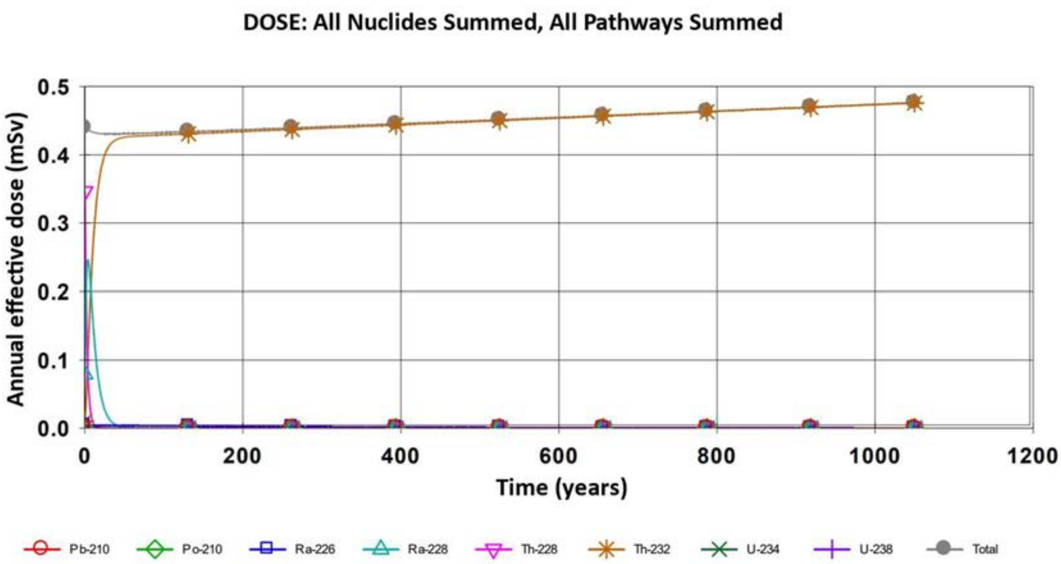


FIG. 29. Contribution of each radionuclide to the annual effective dose indicating ^{232}Th as the dominant contributor to dose.

For the most part, the default parameter values in RESRAD were used as input in the calculations, although the volume of surface water was modified to 300 m^3 and the mean residence time of the surface water was updated to 0.003 years. Because ^{228}Ra is assumed to dominate over ^{226}Ra in the source term, the main pathway is ^{220}Rn inhalation (see Fig. 30). To study this contribution, a sensitivity analysis was performed, considering the emanation of radon and the effective radon diffusion coefficient (see Fig. 31).

If the emanation of ^{220}Rn is changed by a factor of two, the dose also varies by a factor of two. However, if the effective radon diffusion coefficient in the contaminated area is varied by a factor of two, there is a much greater variation in the results. This arises because of the short half-life (55 s) of ^{220}Rn and the competition between diffusion and radioactive decay. Therefore, if possible, both the emanation and the effective diffusion coefficient needs to be locally determined.

A simple exercise was also carried out to assess impacts to non-human species (and specifically, impacts to fish) because, although a low effective dose to humans due to fish ingestion was expected, a high activity concentration of ^{228}Ra in surface water was expected, which might lead to bioaccumulation in fishes and exposure. The estimated activity concentrations of radionuclides in surface water after 0.5 years are as follows:

$^{228}Ra = ^{228}Th = ^{230}Th = 0.1$ Bq/L; $^{238}U = ^{234}U = 0.02$ Bq/L.

The activity concentrations of radionuclides estimated in fish flesh after 0.5 years are as follows:

— $^{228}Ra = 6.1$ Bq/kg dry mass;
— $^{228}Th = ^{232}Th = 30$ Bq/kg dry mass;
— $^{238}U = ^{234}U = 0.22$ Bq/kg dry mass.

Absorbed dose rates to non-human species were assessed using the ERICA tool (see Refs [93−94]). An absorbed dose rate of 10 µGy/h was set as a screening criterion[41] for protection of fish and this criterion was exceeded for benthic fish (Table 7). This exceedance was due to the high activity concentrations calculated in bottom sediments using the ERICA tool [132−133], with an activity concentration of 1840 Bq/g dry mass in the case of progeny of the ^{232}Th decay chain.

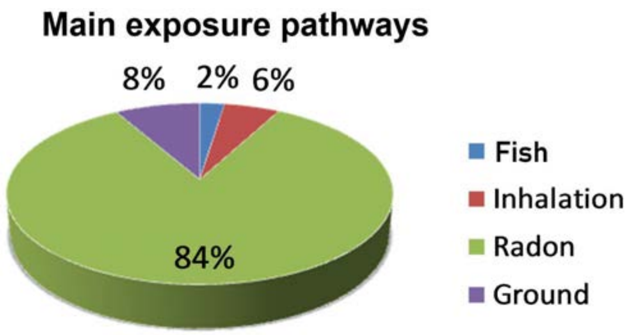
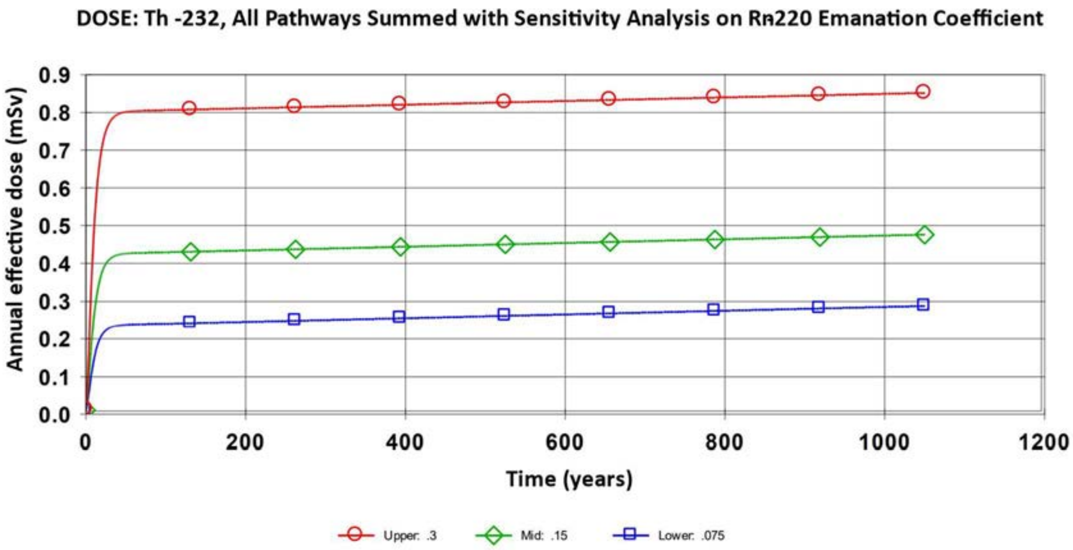


FIG. 30. Contribution of each exposure pathway to the annual effective dose.

[41] Other organizations, such as UNSCEAR (see Refs [134−135] and US-DOE (see Refs [136−137], have defined a screening (threshold) criterion for absorbed dose rate of 400 µGy/h (10 mGy/d) as the maximum dose rate to any individual in aquatic populations that would unlikely to have any detrimental effect at the population level.



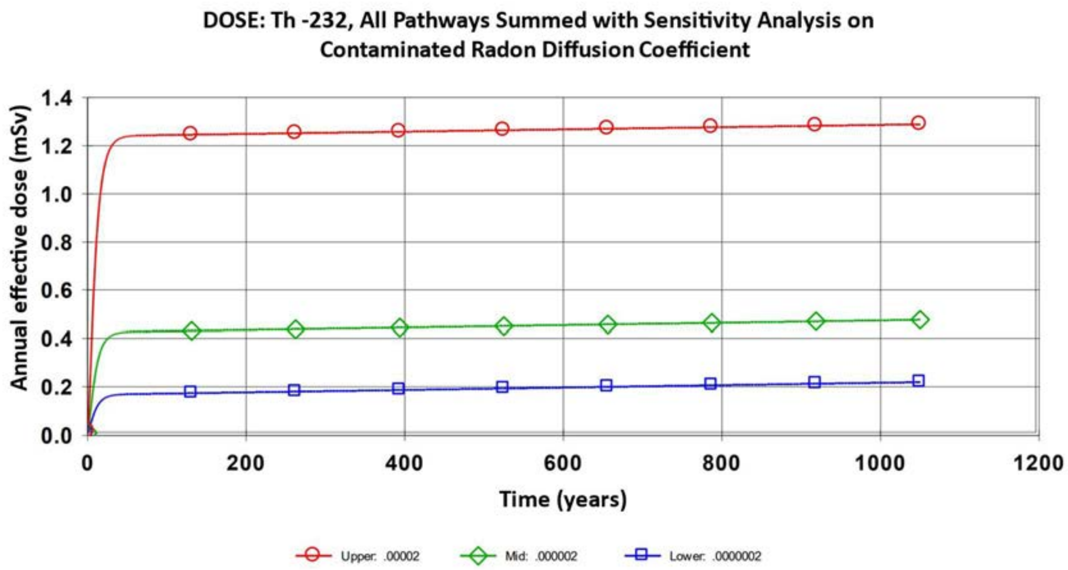


FIG. 31. Sensitivity analysis of the influence of the ^{220}Rn emanation coefficient (top) and the radon diffusion coefficient (bottom) on the annual effective dose from ^{232}Th considering all exposure pathways.

TABLE 7. ABSORBED DOSE RATE TO TWO ILLUSTRATIVE AQUATIC ORGANISMS ASSESSED WITH THE ERICA TOOL

Organism	**Total absorbed dose rate (µGy/h)**
Benthic fish	763
Pelagic fish	2.3

6.2.5. Detailed assessment

A further detailed assessment would need to be based on a survey of the surface and subsurface contamination, and the determination of some specific parameters, such as radon emanation and diffusion coefficients, and the residence time of water in the creek.

6.2.6. Validation

It is possible to validate the model using measurements of sediments and data from monitoring of ^{220}Rn.

6.2.7. Summary

The establishment and evaluation of scenarios of exposure provides important insight and understanding of exposure and dose through space and time, and is important in short-term and long-term management of contaminated sites, particularly for long lived radionuclides in the longer term.

In cases where default parameter values are used in modelling, for example, in conducting a screening assessment, it it beneficial to undertake a sensitivity analysis, in order to avoid misinterpretation. In such cases, the sensitivity analysis can be used to identify the specific parameters that need to be determined (measured) in a detailed assessment. The most sensitive parameters identified in a screening assessment (e.g. those that have a significant influence on dose) need to be included in the final report.

Key exposure pathways for humans can be very different from the key exposure pathways for non-human species. Modelling of exposure and dose to human and non-human species might need to be considered in further, more detailed evaluations, taking account of the prevailing circumstances and the situation.

6.3. SØVE MINING SITE (NORWAY)

6.3.1. Identify the problem, site investigation and characterization

Three main areas have been identified as hazards at the Søve mining complex [138] (see Fig. 32):

- **Sludge disposal site (‘SLAMDAM’)** – A brook runs through the sludge disposal site, which resembles a floodplain. The brook contains mainly fine-grained material (<1 mm), with some surficial slag lumps.
- **Wash House soils (‘Vaskerijord’)** – The sandy top layer of the Wash House soils consists of explosion-pulverised stone (from wall rocks) and crushed concrete, which come from the preliminary activities conducted when the mine was decommissioned. Removal of this material is expected to cause an increase in dose rate.
- **Slag heap (‘SLAGG’)** – Following the end of mining activity, slag that had been dumped on the site was covered with fine-grained carbonate sands. This sand has low variability in texture and, therefore, exhibits little resistance to erosion. There are large erosion scars at the top of the incline caused by running water. The surface layer is made of decomposing vegetative material (tree leaves and branches) and some rubbish.

Once objectives have been defined for remediation, a screening criterion needs to be set. Consistent with GSG-15 [1], the underlying general assessment methodology presented in Section 4 suggests that the screening criterion may be determined nationally, and expressed in practical terms, e.g., as total effective dose (or total risk), a dose rate, or in terms of radionuclide activity concentrations in environmental media.

In Norway, preliminary values defining the level at which material is classified as radioactive waste (necessitating consideration under the relevant regulations) have been formulated in the regulation FOR 2010-11-01 nr 1394[42]. For ^{238}U, ^{226}Ra, ^{210}Pb, Th_{nat} (including ^{232}Th), ^{228}Ra and ^{228}Th, the screening criterion is 1 Bq/g dry mass.

An additional alternative screening benchmark of 1 mSv/a, based on the basic recommendation of the ICRP in ICRP 103 [7], has been defined as a reference level for the representative person, in cases of situations where individuals receive exposures (usually planned exposure situations) that may be of no direct benefit to them, while recognizing that the exposure situation may be of benefit to society. For the Søve mining site, a limited number of samples were collected from the site and activity concentrations of natural decay series radionuclides were measured (Table 8). The risk quotient (RQ) is calculated by dividing the measured activity concentration by the screening benchmark. A more detailed assessment is needed if this value (individually or as a sum of all radionuclides) exceeds 1 (i.e. the screening criterion is exceeded). For such a screening assessment, the highest measured values would normally be used (thus ensuring conservatism). This is a slight divergence from the general assessment methodology (Section 4), which suggests that a screening assessment be carried out using a conservative model that simulates the major features of the site. As direct measurements were available, these data could be compared directly with screening activity concentrations, negating the need for a model, and strict adherence to the general assessment methodology (Section 4) was not considered essential.

Clearly, if RQ > 1 in almost all cases, the screening criterion is not met, and, in accordance with the procedure, it is necessary to proceed to a more realistic (intermediate) assessment. A similar simplified screening assessment can be performed using dose rate criteria and measured values for ambient dose equivalents. The RQs derived from this also exceed unity, even for quite moderate occupancy times.

[42] https://leap.unep.org/countries/no/national-legislation/regulation-no-1394-implementation-pollution-act-regard

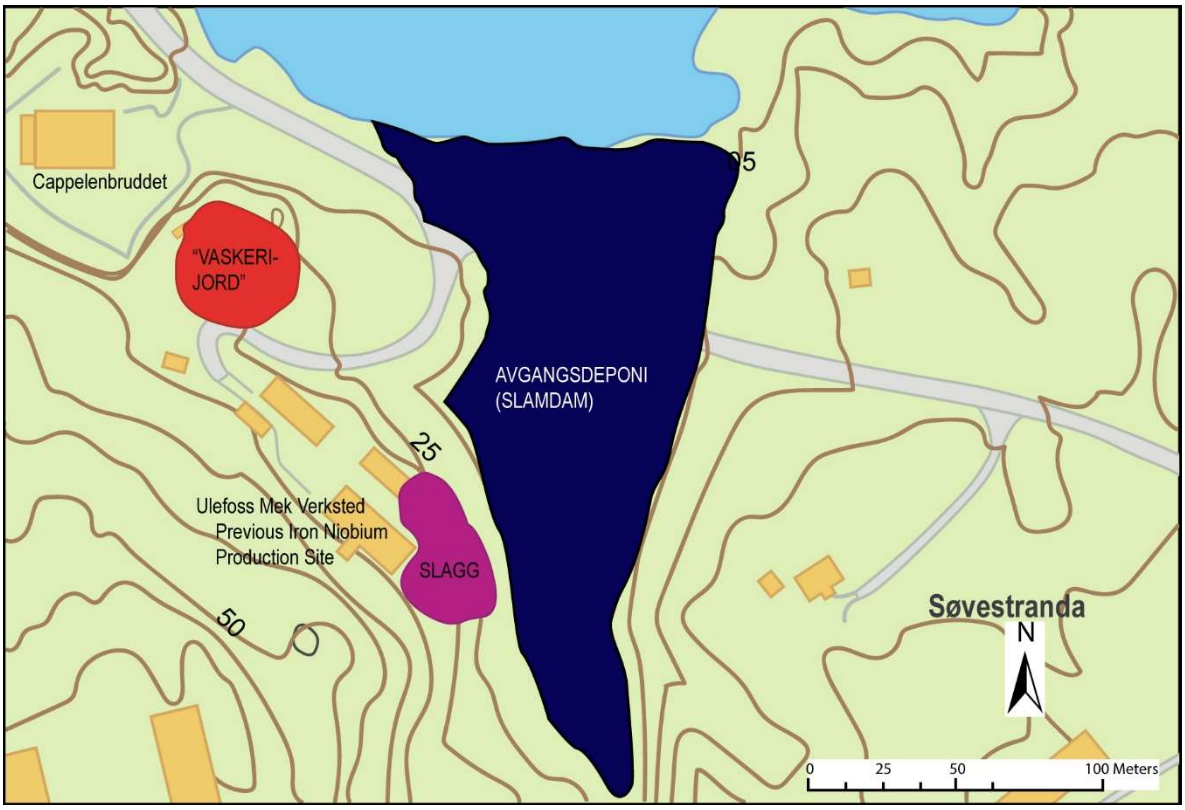


FIG. 32. Contaminated areas at the Søve mining site, and in particular, the three hazards identified on the site, which are the sludge disposal site ('SLAMDAM'), the wash house soils ('VASKERIJORD') and the slag heap ('SLAGG') [138].

TABLE 8. MEASURED ACTIVITY CONCENTRATIONS OF SOME NATURAL DECAY SERIES RADIONUCLIDES AT THE SØVE SITE [138−139]

Sample location	Activity concentration (Bq/g dry mass)						
	U-238	Ra-226	Pb-210	Th-232	Ra-228	Th-228	RQ
Wash House soils a	10.9	12.5	13.6	15.4	18.8	19.5	91
Wash House soils b	5.2	3.3	2.7	7.7	8.7	8.3	36
Wash House soils c	1	0.88	0.89	1	0.99	1.1	6
Slag heap – a	n.d.	5.4	n.d.	n.d.	5.2	n.d.	>10
Slag heap – b	n.d.	5	n.d.	n.d.	5.2	n.d.	>10
Slag heap – c	n.d.	0.04	n.d.	n.d.	0.04	n.d.	0.08

n.d. – not determined; RQ – Risk quotient

6.3.2. Detailed assessment

The general assessment methodology (Section 4) suggests that in cases where the results of a conservative screening assessment do not satisfy the screening criterion, a more realistic detailed assessment needs to be undertaken.

The RESRAD-OFFSITE model was used for a more detailed assessment [140]. This code can be used to model the radiological consequences to a receptor either on-site or outside the area of primary contamination. It can calculate radiation dose for the radionuclide activity concentrations predicted in the environment. An advective-dispersive groundwater transport sub-model predicts the transport of progeny.

For the Søve site, to develop a conceptual site model the exposure pathways to be addressed within the general assessment methodology were restricted to:

- External exposure from the primary contamination – The outdoor exposure time was set to 0.1 years per year, equivalent to just over 1 month (36 days), and it was assumed that individuals spent 0.3 years per year (110 days) indoors over the waste. These values, used for the representative person (working in a building on site) can be considered very conservative, as individuals would have to be working >50 hours per week for 52 weeks per year to attain these levels;
- Radon inhalation (subject to the same assumed occupancy);
- Ingestion of crustacea and fish from the nearby lake.

Although data did not always exist for many of the parameters needed for the site, some aspects of the assessment were performed in a site specific manner, including the size of the source and the position of receptors around it. Activity concentrations used as input data were assumed to be equal to the maximum values reported for the site in Ref. [138] (see Table 8). Activity concentrations for ^{210}Pb equal to those of ^{210}Po were included. Site specific values for hydraulic conductivity and gradient, dry bulk densities, precipitation and irrigation rate were used. For all other parameters, default values within RESRAD were used. Output from the model is shown in Fig. 33. Models have also been applied to simulate:

(1) Infiltration, vertical migration and groundwater flow of contaminants;
(2) External dose rates.

A transect through the contaminated area using the maps of the Søve mining site was used to produce a two-dimensional customized model for the site. The rate constants between compartments in the unsaturated zone are based on infiltration rates, retardation coefficients (determined from distribution coefficients), compartment depths and porosities. For compartments in the saturated zone, rate constants are a function of the Darcy velocity, porosity, retardation coefficient, and compartment size. To the extent possible, site specific data were used to parameterize the Ecolego model construction [141].

Rigorous validation of this model was not possible. This was a first attempt to develop a customized model for the Søve mining site. However, a simple comparison was made for selected radionuclides and time points between results from the widely tested RESRAD-OFFSITE model and this simplified model. At $t = 1000$ years, RESRAD-OFFSITE predicts an activity concentration in the primary contamination of ca. 7 Bq/g dry mass ^{226}Ra. This can be compared to the predicted value of 7.3 Bq/g dry mass from the customized model in 'slag heap 1', where identical initial activity concentrations were assumed as those of the primary contamination in the RESRAD-OFFSITE model runs. Predicted activity concentrations remain above the radionuclide specific criteria of the screening criterion for certain radionuclides even for $t > 1000$ years where remedial actions have not been implemented.

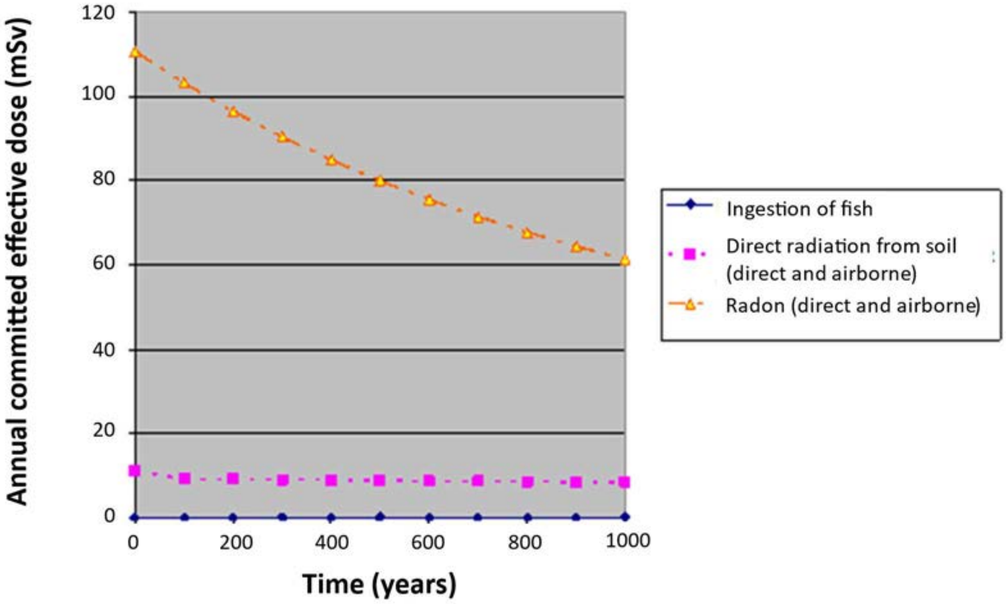


FIG. 33. Annual committed effective dose as a function of time at the Søve mining site for the existing situation, modelled using RESRAD-OFFSITE.

6.3.3. Establishing criteria for remediation

If the decision maker decides to proceed further with remediation, for example, on policy grounds or because the reference level is not satisfied by the detailed assessment, then end point criteria for each remedial action and phase in the remediation process, along with an overall end point criterion for the remediation, need to be established. In establishing such criteria, it is important to apply the process of optimization of protection and safety, also taking account of factors other than purely radiological considerations. For example, such factors might include cost effectiveness, end use and end state of the site, and others. In general, criteria need to be chosen so as to optimize dose and risk during and after remediation. Non-radiological criteria were not included in this assessment. Some examples of remedial actions and corresponding end point criteria might be to:

- Remove contaminated material until the activity concentrations in the surface soil are less than the level where it would be regulated as radioactive waste;
- Reduce the annual effective dose to the representative person, such that it falls below the reference level and the end state criterion.

Three remedial options have been suggested [138], which are:

- **Remedial Option 1:** Null alternative – do nothing;
- **Remedial Option 2:** Development of a local remediation plan involving removal and redisposal of materials in the area originally allocated for waste treatment;
- **Remedial Option 3:** Removal and disposal of radioactive waste to an approved disposal facility.

Remedial Option 1 was prospectively modelled using the RESRAD-OFFSITE code (see Fig. 33). Detailed exploration of the external exposure pathway associated with Remedial Option 2 through application of an external dose rate model using the highest activity concentration of the data presented in Table 8. MicroShield has been used to assess the annual effective dose in different external exposure configurations [142]. The use of an uncontaminated soil cover was studied using MicroShield, considering several thicknesses from 1 to 100 cm, to generate the results presented in Fig. 34 indicating the decrease in effective dose equivalent. Applying ca. 50 cm of topsoil reduces the effective dose equivalent by approximately 100-fold, corresponding to the reduction in exposure from waste to a value of around 0.1 µSv/h, i.e. to ca. 1% of the level without shielding. At such a dose rate, the end point criterion (1 mSv) would be satisfied even using extremely conservative occupancy factors. Nonetheless, erosion of the shielding layers of topsoil could lead to substantially increased exposure in the future and was not considered in these models.

Remedial Option 3 was modified slightly to reduce the contamination source to the benchmark activity concentration presented in Section 6.3.1, i.e., assuming that all activity concentrations are equal to 1 Bq/g dry mass, and that the land is used for agriculture, with all available exposure pathways 'open'. A cover of 0.1 m of clean soil was assumed, representative of the set-up of the current situation. The results from RESRAD-OFFSITE are given in Fig. 35.

This scenario shows that, theoretically, the annual effective dose to the representative person might still exceed 1 mSv, due mainly to radon but also to significant contributions arising from drinking contaminated water (assuming an extraction well near the site), ingestion of crops and direct external exposures. This highlights the importance of defining the post-remediation land use to ensure appropriate exposure pathways are scrutinized, and that secondary checks can be made on predicted prospective doses relative to established dose criteria.

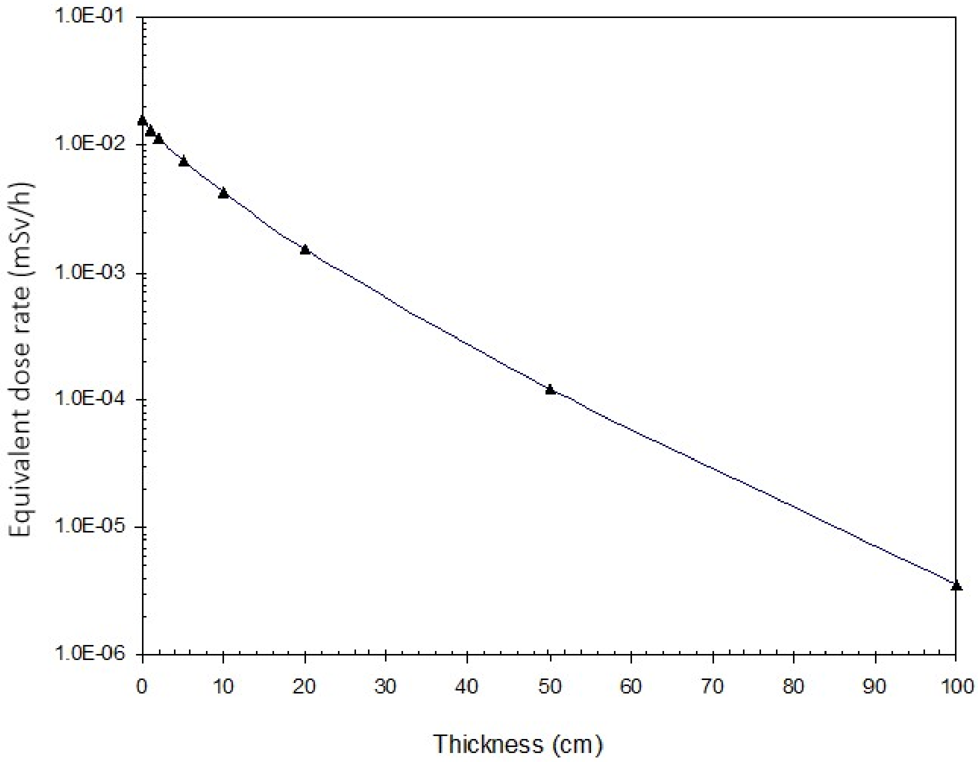


FIG. 34. Influence of the thickness of a clean soil cover on external exposure in terms of equivalent dose rate.

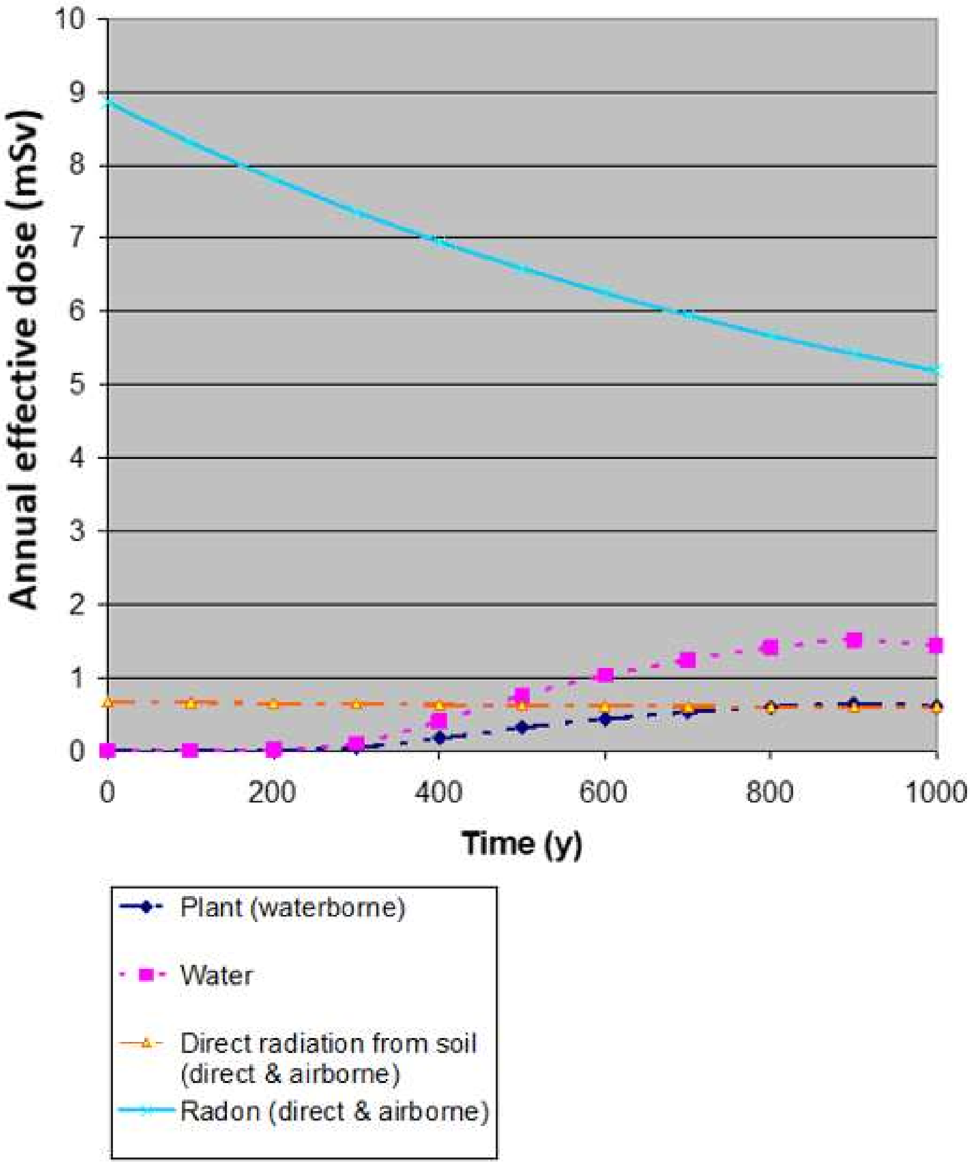


FIG. 35. Annual effective doses for scenario exploring remedial option 3: Removal of radioactive waste to a level below the screening criterion and opening of land to agricultural use.

6.4. THE FORMER ATOMIC WEAPONS TEST SITE AT MARALINGA (AUSTRALIA)

6.4.1. Identify the problem (and site description)

A series of nuclear weapons tests was conducted in Australia, from 1952 to 1963. The first such test was conducted at the Monte Bello islands, off the northwest coast of Western Australia. In 1953, two atomic explosions were detonated at Emu in South Australia. In 1956, two more tests were conducted at the Monte Bello islands, and from 1956 to 1957 a series of seven atomic tests was conducted at Maralinga, in South Australia (see Fig. 36).

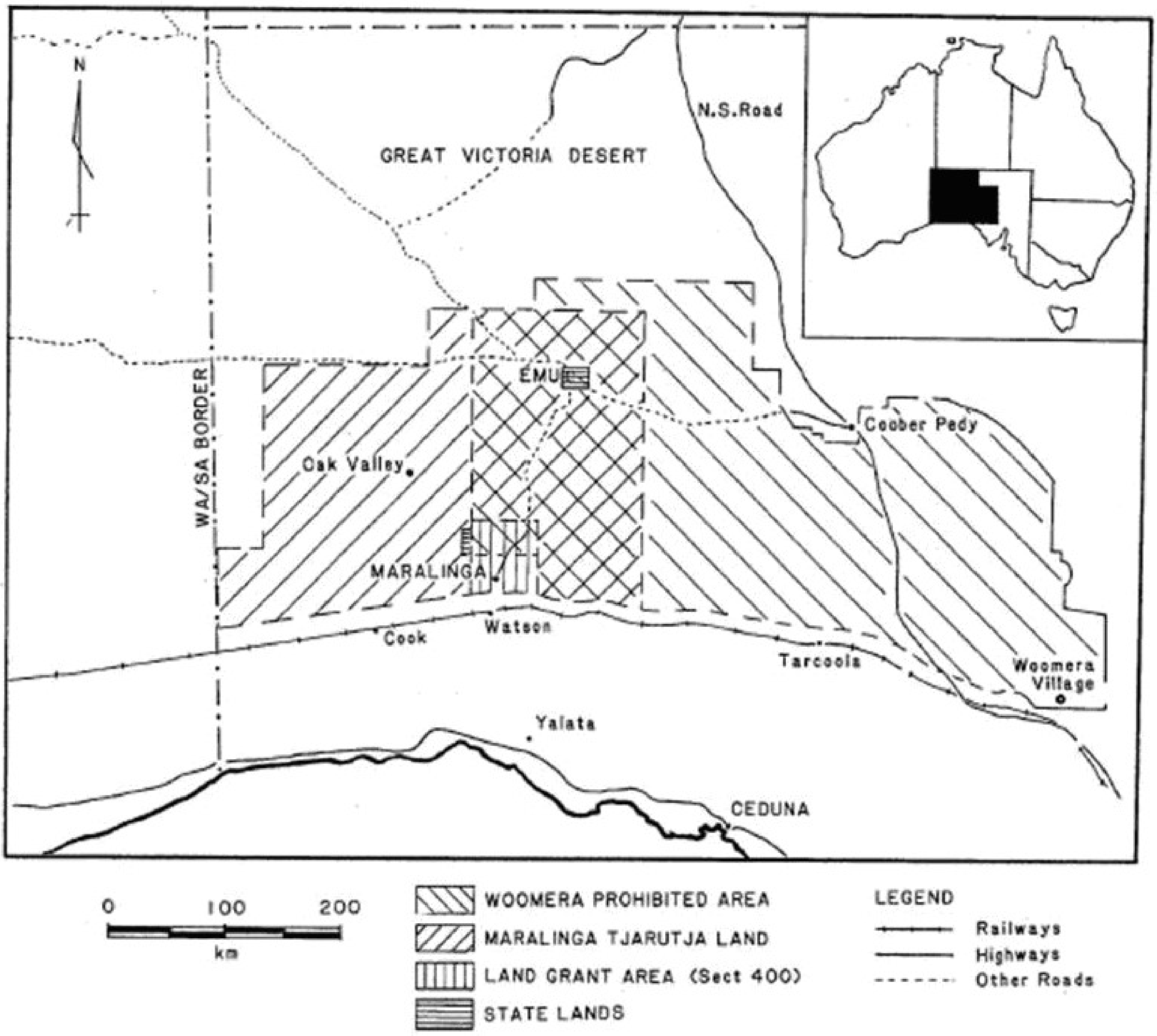


FIG. 36. Map showing the location of the Maralinga and Emu test sites in South Australia.

Most of the contamination that resulted from these tests (apart from fallout) was in the form of contaminated vehicles and aircraft that were placed in the test areas to study the effects of the atomic blast on military equipment.

From 1957 to 1963, hundreds of trials of triggering devices were also conducted at Maralinga. These so-called ‘minor trials’ contaminated the environment with plutonium and other radioactive debris (fragments of contaminated equipment).

In 1966, the British conducted the first cleanup (operation ‘Brumby’). This consisted of ploughing the surface material into wind rows, with the aim of burying the plutonium. This was supposed to have restored the site to a condition where it could be returned to the traditional owners, the Maralinga Tjarutja people.

6.4.2. **Define remediation objectives**

Measurement surveys by the Australian Radiation Laboratory during 1984 and 1985 showed that contamination levels were significantly higher than those reported previously. While not explicitly stated as part of the investigation, the criteria were as follows:

- Mean radioactive surface contamination level over a 1 hectare area was not to exceed a site specific ‘stated level’ of ^{241}Am, which was set taking account of varying Pu-to-Am ratios at a given site, and varying soil and resuspension characteristics;
- No fragment or particle would be present, which exceeded 100 kBq ^{241}Am;
- Particles with an activity of greater than 20 kBq ^{241}Am would not exceed a surface density of 0.1 per m^2.

6.4.3. Site characterization

In 1986, a Technical Assessment Group was established to oversee technical site studies and to advise on remedial options. More detailed studies, which were conducted in the late 1980s, revealed extensive surface plutonium contamination in well-defined ‘plumes’, determined by the wind direction during each minor trial, at two sites known as Taranaki and the TM site. The plumes at the Taranaki site are shown in Fig. 37.

A systematic programme of measurements indicated that the majority of the contamination remained within 10–20 cm of the surface (due to low rainfall) and consisted of three components:

- Visibly identifiable fragments of plutonium-contaminated debris;
- Potentially inhalable, fine material, consisting of plutonium oxide grains or contaminated soil, more or less uniformly distributed;
- ‘Hot’ particles of soil or other material (sub-millimetre sized), randomly distributed.

Many fragments had been previously placed in 22 burial pits capped with concrete.

The Maralinga area is characterized by limestone (karst), with extensive underground cave networks. Vegetation mostly consists of desert grasses, scrub and small trees (mostly eucalyptus and acacia). Rainfall is approximately 100 mm per year, but is highly irregular. With few exceptions, the groundwater is not potable. The traditional owners of the land (the Maralinga Tjarutja people) lead a semi-nomadic lifestyle, camping for extended periods (days to weeks) in the area.

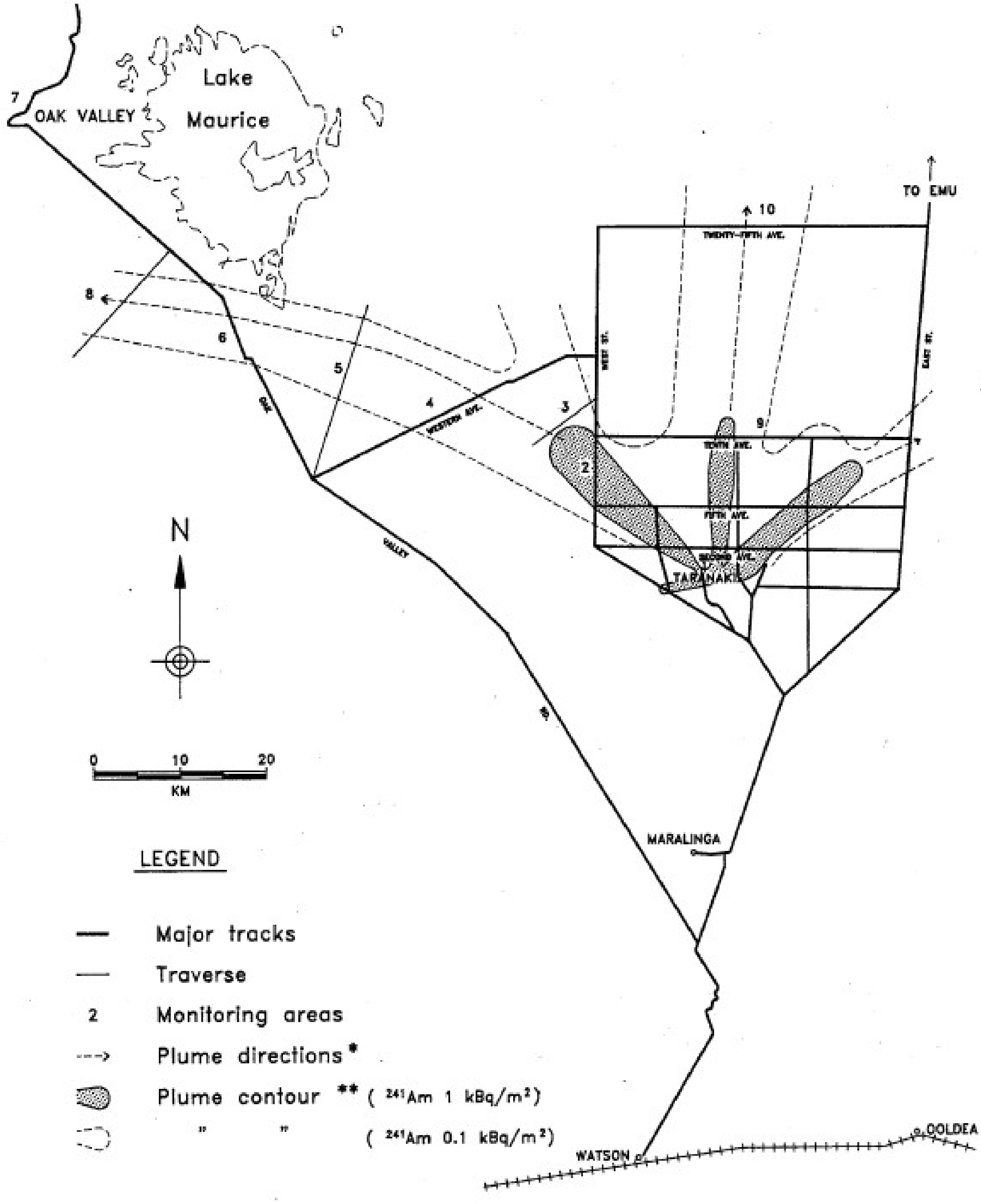


FIG. 37. The locations and orientations of the contaminated areas at the Taranaki test area. The TM site is approximately 10 km East–Southeast of Taranaki; the extent of contamination at TM is much smaller than at Taranaki, but the same well-defined plume distribution occurs at both sites.

6.4.4. Preliminary and detailed assessment

In 1993, the Maralinga Technical Assessment Committee (MARTAC) with experts from Australia, the USA, and Great Britain was created to evaluate the risks and determine acceptable remediation criteria.

The initial MARTAC assessment determined that indigenous people would be the group most at risk passing through and camping and possibly hunting on the site. The risk assessment was based on a study of the diet, habits and other relevant behaviours and attributes of these people. The exposure pathway of greatest concern was found to be inhalation of dust by children playing around camp sites.

Large scale surveys of the Maralinga site were then undertaken to:

- Characterize the nature of the hazard, including the radionuclides present, external exposure (i.e. dose rates), inhalation risks and presence of contaminated particles;
- Conduct field and laboratory studies to characterize the contamination (i.e. dust loadings, particle size distributions, chemical properties and solubilities of the radionuclides present on the site);
- Evaluation of the spatial extent of contamination through mapping of data generated by ground-based and aerial surveys;
- Evaluation of environmental impacts (e.g. contaminant uptake by animals and plants).

The preliminary and detailed assessments indicated that neither the screening criterion nor the reference level were met, and therefore, remediation was deemed to be justified.

6.4.5. Establish criteria

MARTAC established that the area would need to be remediated to a level at which residual contamination would result in a maximum annual dose of 5 mSv to an individual, for full-time occupancy of indigenous people living an ‘outstation lifestyle’ (camping). This dose criterion represented the reference level; it corresponds to a risk of fatal cancer of 1 in 10 000 by the 50^{th} year of life.

A detailed dose assessment was carried out to establish the remediation criteria that would enable the 5 mSv reference level to be met. The remediation criteria derived from this assessment were:

- A maximum activity concentration of plutonium per square metre in finely divided material;
- A maximum number of particles per square metre;
- Visible fragments to be collected and treated.

A derived criterion for soil contamination of 3 kBq/m^2 of ^{241}Am (as a surrogate for plutonium contamination) was established, which was considered equivalent to the reference level (see Annex IV of GSG-15 [1]).

6.4.6. Remedial action

Remedial actions were carried out in several steps:

- The extent of the area(s) to be remediated was determined from the detailed survey measurements;
- Trees and bushes were removed and seeds (grasses, trees, bushes) were collected, catalogued and stored for later replanting;
- The top 10–20 cm of soil was scraped off;
- This material was buried in large pits and covered with 5 m of clean soil;
- Eleven of the burial pits were treated by in situ vitrification;
- Due to a major accident, which severely damaged the in situ vitrification equipment, material from the remaining burial pits was retrieved and placed in a single large burial pit, which was then capped.

6.4.7. Radiation protection issues during remediation

The main radiological issue during the remediation was inhalation of plutonium attached to airborne dust particles. This necessitated dust suppression procedures, involving dust suppression by spraying water on the haulage routes (the roadways used for transporting the removed soil to the burial pits).

A strict health physics and occupational hygiene regime was put in place and undertaken by all personnel working in the contaminated areas to minimize the probability of ingestion or inhalation of contaminated material. To minimize contamination transfer, vehicles and personnel were checked for surface contamination before being permitted to leave contaminated areas.

6.4.8. Verification monitoring

As soon as an area had been scraped, measurements were made to check whether the remediation criteria had been met. Two measurement systems, built by the Australian Radiation Laboratory in the early 1990s, were needed to verify the two main remediation criteria (average plutonium activity concentration and number of particles per square metre). These systems were computer controlled and mounted on vehicles, as shown in Figs 38 and 39. Where it was found that the remediation criteria were not met, extra scraping was carried out, and the checks repeated.

Measurements of plutonium in suspended dust to verify that the airborne activity concentrations of plutonium were at acceptable levels. These measurements were carried out by personnel from Lawrence Livermore National Laboratory (LLNL) in the US and from Australian Radiation Laboratory using equipment from LLNL.

A post-remediation assessment, based on the verification measurements, suggested that the estimated doses after remediation were approximately a factor of five lower than the doses on which the remediation criteria were based.

Similar procedures to those developed and applied at Maralinga have been used in other countries.

FIG. 38. The purpose-built system used for testing the area density of finely divided contamination.

FIG. 39. The system used for measuring the number of particles per square metre.

6.4.9. Summary

The procedures used at Maralinga were applied well before any attempt was made to develop a formal methodology for addressing remediation of historic sites. However, the main features (identification of the problem, site characterization, screening assessment, detailed assessment and development of remediation criteria based on input of interested parties, and verification) are consistent with the General Assessment Methodology and with GSG-15 [1]. A case study for the Maralinga site is also included in GSG-15 [1].

6.5. TESSENDERLO CAF2 SLUDGES DUMPSITE (BELGIUM)

This section is concerned with the 'Veldhoven' landfill at the Tessenderlo site in northeast Belgium, where approximately two million tons of calcium fluoride sludge generated during the processing of sedimentary phosphate ores, have been dumped. More information on this site is provided in Section IV.1 of Appendix IV.

6.5.1. Identify the problem

The Belgian radiation protection regulations explicitly list several work activities involving NORM that may be of concern from a radiation protection perspective. Phosphate ore processing is one of these work activities. In accordance with the regulations, each company belonging to the phosphate sector is required to submit a notification to the radiation protection authority (see Annex I). This notification will allow the authority to identify possible risks of exposure for the workers and the public. The operating organization for the Tessenderlo site has fulfilled its obligations to submit a notification.

In the 1990s, the Geological Service of Belgium had carried out an aerial gamma spectrometry survey of the entire Belgian territory. This survey clearly revealed elevated levels of radiation compared with the local background for industrial landfills of the phosphate industry.

6.5.2. Site investigation and characterization

Most of the data regarding the site were directly provided by the operating organization through its notification to the authority and are described in more detail in Section IV.1 of Appendix IV. A short summary of the key aspects of the site is as follows:

- Until 1995, the ^{226}Ra concentration in the CaF_2 sludge was approximately 3.5 Bq/g dry mass. At this time, a process of co-precipitation of radium in the wastewater was applied, which led to an increase in the ^{226}Ra activity concentration in the residual sludge of up to approximately 11 Bq/g dry mass;
- The external dose rate at the surface of the dumpsite may reach 2.5 μSv/h;
- The outdoor radon activity concentration is monitored on and around the landfill, with annual average values of up to 160 Bq/m^3 being recorded.

6.5.3. Screening assessment and screening criterion

Several criteria in support of the overall screening criterion may be used to evaluate whether it might be justified to implement remedial actions on the Tessenderlo site. Belgian recommendations with respect to the management of NORM residues suggest a screening criterion of 0.2 Bq/g dry mass for the average activity concentration for the entire volume of a landfill where NORM residues are disposed of. If this average level of activity concentration is exceeded, a site specific risk assessment is to be carried out.

Other criteria that might be applied during a screening assessment might be based on the radon activity concentration. The WHO recommendations propose a reference level of 100 Bq/m^3 for indoor radon [143]. Both criteria are exceeded, indicating that remediation might be justified.

6.5.4. More realistic assessment

A REIA for this site has been performed within the framework of the CARE report of the European Commission [144]. Two scenarios of exposure were considered in this publication:

— A ‘normal evolution’ scenario (with farmers residing and working close to the site), which leads to an annual effective dose to the representative person of approximately 0.5 mSv;
— An intrusion scenario (with people living in dwellings built on site), which, according to the assumptions of the report, leads to an annual effective dose to the representative person of 357 mSv (essentially, from exposure to radon).

At this stage, the decision criterion is an annual effective dose (E) criterion. In Belgium, the following intervention levels (expressed as incremental dose with respect to natural background) are recommended by the radiation protection authority:

— $E < 0.3$ mSv $\Rightarrow$ intervention is not justified;
— 0.3 mSv $< E <$ 1 mSv $\Rightarrow$ intervention is rarely justified;
— 1 mSv $< E <$ 3 mSv $\Rightarrow$ intervention is generally justified;
— $E > 3$ mSv $\Rightarrow$ intervention is always justified.

Moreover, setting intervention levels is only meaningful in combination with scenarios of exposure. In addition, specification is necessary regarding whether the levels apply only to the current impact of the site or if they will also apply to possible future impacts, considering the evolution of the site, especially relating to the evolution of the use of the site. In Belgium, three scenarios of exposure are generally considered:

— The current use of the site;
— A worst-case scenario, representing the scenario that leads to the highest exposure (often an intrusion scenario, for example, involving building dwellings on the site);
— A ‘likely’ scenario describing a probable evolution in site use (e.g. new industrial activities).

For the site considered here, the intervention level of 3 mSv (based on the annual effective dose, as above), which is equivalent to the reference level, is exceeded for the intrusion scenario. It is important to note that an intervention does not necessarily involve remediation. In some cases, it may be sufficient to impose restrictions on the use of the site to prevent the intrusion scenario from occurring.

6.5.5. Detailed assessment and remediation

As this site is still being operated, a detailed assessment and the remediation of the site is not yet being planned. Remediation, in accordance with environmental regulations that are not focused on radiation protection, will be needed at the end of site operation. It has not yet been decided whether radiation protection considerations will cause further constraints on remediation.

6.6. THE FORMER RADIUM PRODUCTION FACILITY OF OLEN (BELGIUM)

The town of Olen in Belgium hosts a metallurgical company that was active in the extraction of radium and the production of radium sources from 1922 to 1969. This section describes remediation of the banks of a river near the D1 dump site. A more detailed site description and a more comprehensive summary of the assessment carried out is provided in Ref. [145].

6.6.1. Identify the problem

Historical records have reported radium contamination at the former dumpsites and some patchy contamination has also been found on the neighbouring streets. This is due to the past practice of using production residues as basement material in road construction. As there are no historical records for this practice, the resulting patchy contamination may only be identified through measurements.

6.6.2. Screening assessment

The average ^{226}Ra activity concentration in the D1 dump is approximately 20 Bq/g dry mass. Outdoor radon measurements at the surface of the dump may reach 1000 Bq/m^3. As in the case of Tessenderlo in the previous section, further investigation of these values is needed.

6.6.3. More realistic assessment

The dose assessment was performed by the Belgian Nuclear Study Centre (SCK-CEN) [146]. The normal evolution scenario (i.e. no major changes in use of the dumpsite or surrounding areas) leads to an annual effective dose of ca. 2 mSv. As the dose from groundwater contamination was negligible, this dose is almost completely due to radon. A dose assessment was also done for two intrusion scenarios (construction on site, residential on site). For the scenario 'living in a house built on-site', the annual effective dose was estimated to be 56 mSv, with radon as the dominant contributor, representing 44 mSv. By comparison, the annual effective dose from external radiation was 6.5 mSv and from consumption of vegetables grown on site was 5 mSv.

6.6.4. Detailed assessment and remediation

Remedial actions have already been undertaken along the banks of the nearby river (Bankloop). During production of radium, the wastewaters from the radium production facility were released into this river. This caused contamination of the sediments and the riverbank, because of dredging.

During the remediation of the riverbank, which took place between 2006 and 2008, operational cleanup criteria[43] were established to guide the excavation work. These criteria where then compared with both dose rate measurements and activity concentration measurements to demonstrate regulatory compliance.

First, the dose rate was measured and if the value of the dose rate was lower than 0.2 μSv/h (which corresponds to a value that is approximately twice that of the local background), no further excavation needed to be performed.

If the dose rate exceeded 0.2 μSv/h, the ^{226}Ra activity concentration had to be measured. If the activity concentration at a depth inferior (respectively superior) to 1 m was less than 0.5 Bq/g (respectively 1 Bq/g dry mass), no further excavation was necessary. If these values of activity concentration were exceeded, the soil had to be excavated and transported to a specific disposal site.

[43] These are equivalent to 'derived criteria' as defined in GSG-15 [1] (see Section 3.2).

7. MODEL–MODEL INTERCOMPARISON

Model–model intercomparisons were carried out for two sites. The first was the Gela site, which is discussed in Section 6.1 in terms of the general assessment methodology. The second site was the Bellezane site, which is situated near Limoges in the Limousin region of central France. The Gela intercomparison is discussed in Section 7.1, and the Bellezane intercomparison is discussed in Section 7.2.

7.1. GELA SITE (ITALY)

7.1.1. Description of the site

The Gela site is described in Section 6.1. The site layout and locations of groundwater sampling points are depicted in Figs 10 and 11, respectively.

The details regarding the retaining wall and the natural barriers at the site are presented in Fig. 11 in Section 6.1.2, along with the input data for the modelling exercise.

7.1.2. Objectives of the model–model intercomparison

The objective of this intercomparison was to identify and discuss any significant differences between the predictions of the different modelling packages and the approaches taken by each.

The model–model intercomparison for the different stages of the assessment process is discussed in the following Sections 7.1.3–7.1.6.

7.1.3. Screening criterion and other relevant criteria

As noted in Section 6.1.4, an exemption level of 10 000 Bq/kg dry mass was suggested for ^{40}K, whereas for the remaining radionuclides from natural decay chains, a value is 1000 Bq/kg was suggested [126].

In Section 6.1.4, it was also suggested that an annual effective dose of 0.3 mSv for natural radioisotopes was considered acceptable as a screening criterion. It is important to recognize, however, that this criterion is quite conservative, although the approach taken applying the overall framework is relevant. It was also noted that the annual effective dose limit (1 mSv according to GSR Part 3 [3] might also be proposed as a screening criterion).

7.1.4. Screening assessment

A screening assessment is designed to generate conservative estimates. A hand calculation and four modelling packages (DandD 2.1.0, RESRAD (onsite), RESRAD-OFFSITE, and ReCLAIM v3.0) were used for the screening assessment calculations described in this publication. The details of the calculations are presented in Section 6.1.5.

The screening calculations were performed using the default values for the parameters used in the different packages. For screening purposes, a rural residential scenario was considered, where all the food ingested by the representative person is grown directly on the stack. No remedial options were considered.

A hand calculation was done for an extremely conservative case. The dose in this case was estimated using a simple equation, as described in Section 6.1.5 (see Eq. (3)). The results (total doses only) for the screening calculations are presented in Table 9.

TABLE 9. RESULTS OF THE SCREENING CALCULATIONS

Modelling package	Hand calculation	DandD 2.1.0	RESRAD	RESRAD-OFFSITE	ReClaim v3.0
Total annual effective dose (mSv)	1.92	5.7	9	Not available	0.38
Screening criterion	0.3	0.3	0.3	0.3	0.3

The lowest model prediction was generated using the ReCLAIM methodology. However, as noted in Section 6.1.5, the ReCLAIM calculations only considered the impact due to radium in soil, and in general, for a screening assessment, which is intended to be conservative, all relevant radionuclides and exposure pathways need to be considered. In addition, the hand calculation, which considered only the ingestion of vegetables grown on the contaminated site, generated a higher predicted dose than ReCLAIM. This suggests that both the hand calculation and the ReCLAIM prediction are incomplete from the perspective of a typical screening assessment, and the predictions from these calculations need to be interpreted with caution. That said, the example provided demonstrates how the general assessment approach described in this publication can be applied, recognizing the need to consider all relevant radionuclides and pathways.

The two detailed software packages (DandD 2.1.0 and RESRAD) generated results that differed by a factor of two from each other and that were significantly higher than the hand calculation and the ReCLAIM prediction. The reasons for the difference between the two sets of predictions are not understood. However, considering the uncertainties inherent in this type of calculation, there is reasonable agreement between the two sets of predictions.

7.1.5. More realistic assessment

Consistent with the iterative nature of the general assessment approach presented in this publication, more realistic intermediate assessments were carried out, as described in Section 6.1.6. The basis for these assessments were the reported remedial actions that were implemented, as described in Section 6.1.8. This included the installation of a plastic liner and a clean soil cover on the stack.

Reliance on the integrity of the plastic liner was evaluated in the assessments by including or excluding the radon inhalation pathway. Both the DandD (version 2.1.0) and RESRAD codes assumed the necessary thickness of clean soil. The details and results of the calculations are provided in Section 6.1.8.

The MicroShield package was used to estimate the external dose rate as a function of cover thickness. The results are shown in Fig. 18 in Section 6.1.8. The MicroShield predictions were confirmed using RESRAD for a cover thickness of 2 metres.

Another calculation was performed using RESRAD, assuming the default values included in the code and considering all the exposure pathways (external exposure, inhalation, soil ingestion, vegetable ingestion, radon and drinking water) for a residential scenario without any remediation, and with the assumption that 50% of the food ingested by the representative person was produced on the site.

The HYDRUS-2D package was used to calculate the migration of water from the waste into the environment. The radium activity concentration in well water was estimated using both the HYDRUS−2D and RESRAD codes.

Although there were differences in the results of the calculations from the different modelling packages for individual pathways, and the assumptions were different in some cases, all the calculations led to the conclusion that without a retaining wall, a plastic liner and a soil cover in place, the site would not meet the screening criterion, whereas with a retaining wall, plastic liner and soil cover in place, the site might be suitable for recreational or industrial use.

7.1.6. Detailed assessment

Using the general assessment methodology (Section 4), a detailed assessment is needed in those cases where the predictions of the intermediate assessment exceed the screening criterion. In such cases, the detailed assessment focused on the effect of remedial actions that had already been implemented at the site. In particular, the assessment considered factors such as:

(1) The possible decrease in the integrity of the plastic liner with time;
(2) The effectiveness of the retaining wall over time in reducing leaching;
(3) The ingress of rainwater into the phosphogypsum stack.

Two model packages (MicroShield and RESRAD-OFFSITE) were used for this exercise. The calculations assumed residential scenarios both above and at the edge of the phosphogypsum stack.

The results from both models, as presented in Section 6.1.8, indicated that for both residential scenarios, the annual effective dose to the representative person would still exceed 1 mSv after several hundred years. The results also showed that if the land were to be released for other uses (e.g. recreational or industrial use), the reduction in the occupancy factor would significantly reduce the doses (Section 6.1.9).

The results of the calculations from both models also confirmed that with a retaining wall in place, the off-site doses are likely to remain low well beyond the projected institutional control period.

7.2. BELLEZANE SITE (FRANCE)

7.2.1. Description of the site

The Bellezane site is part of the Crouzille Mine Division and includes four tailings repositories (see Figs 40 and 41). It is situated in the catchment area of the Gartempe river and is located approximately 2 km southeast of Bessines-sur-Gartempe, a village near Limoges in the French department of ‘Haute-Vienne’ (Limousin region).

The property owner of the Bellezane site was AREVA (formerly COGEMA), the operating organization for the mine that carried out the remediation a few years ago. AREVA was responsible for the environmental monitoring on and in the vicinity of the site.

7.2.1.1. Nature and extent of mining works, historic landmarks

Mining

Mining work on the Bellezane site was initiated in March 1975 and ceased in January 1992 due to resource depletion. Ore extraction involved both open-pit mining (referred to as ‘MCO’) and underground mining (referred to as the ‘tailings management system’ or ‘TMS’).

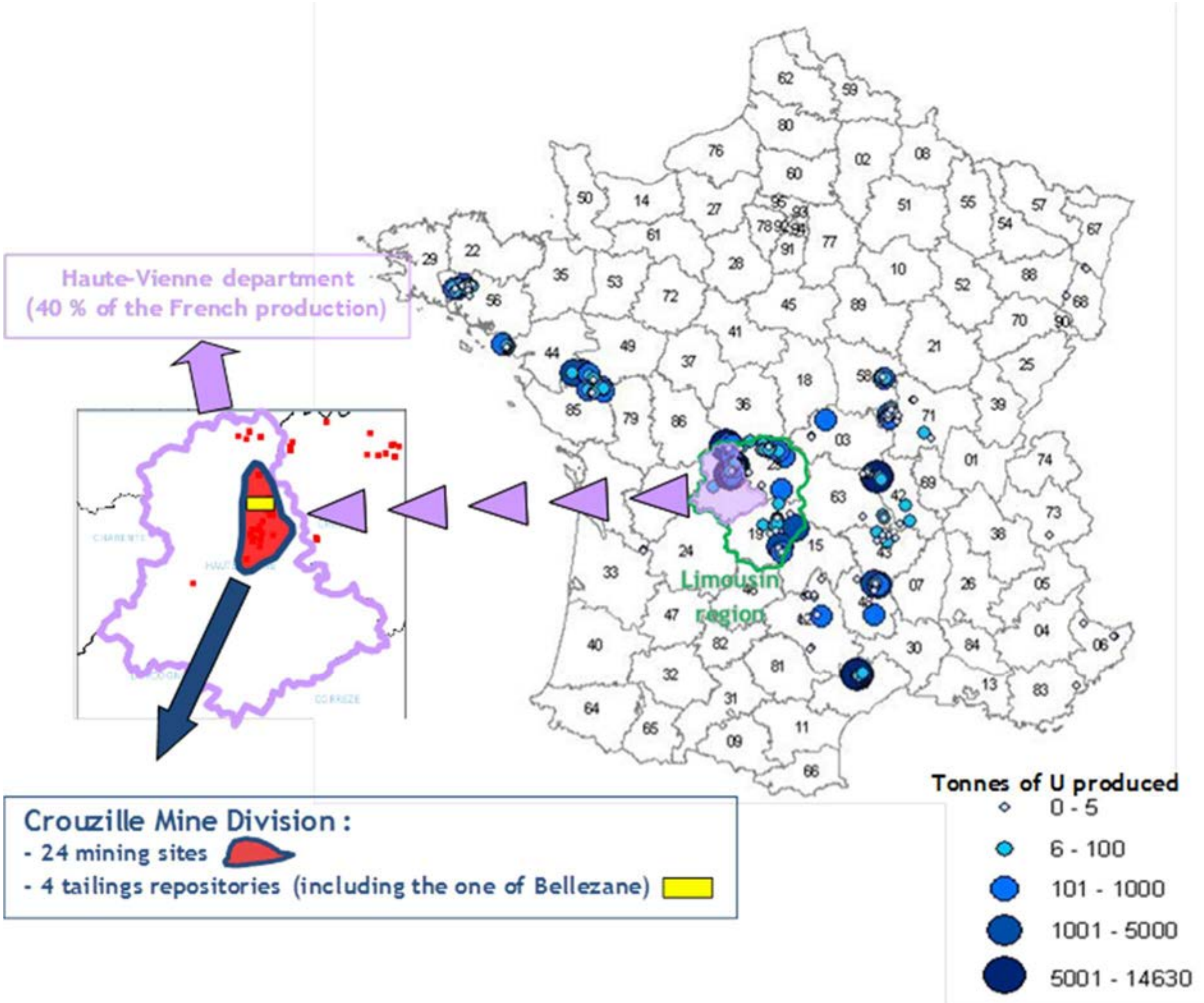


FIG. 40. Location of the Bellezane site in France.

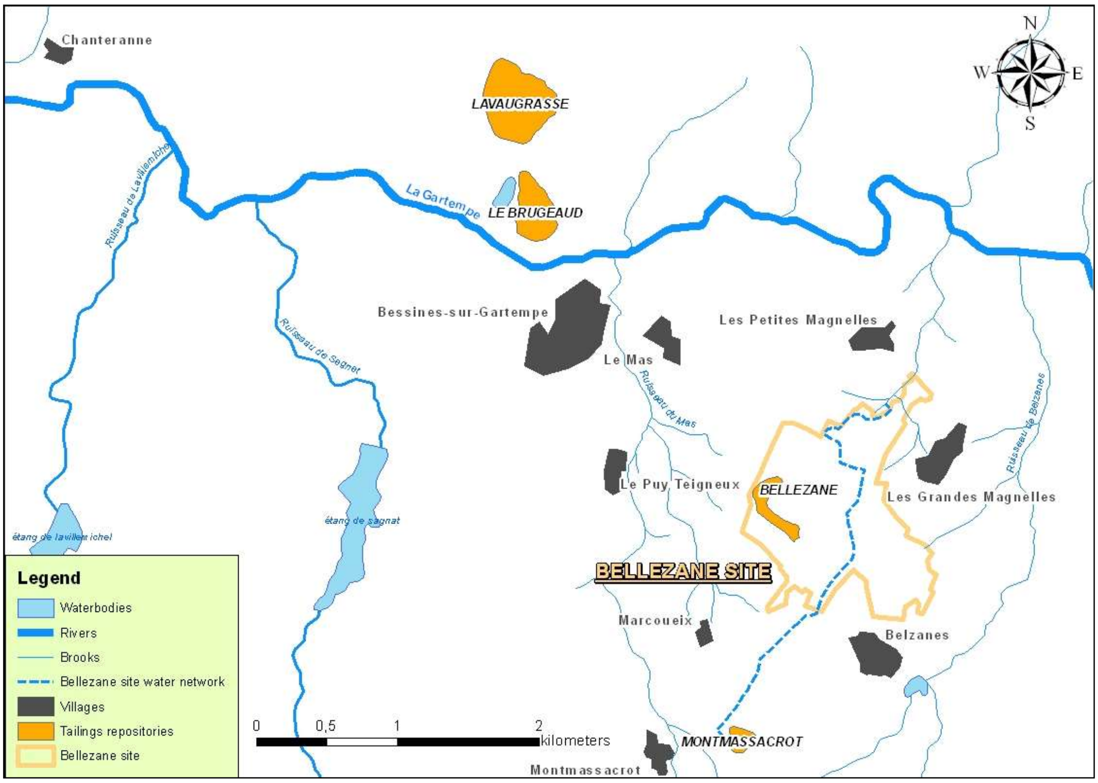


FIG. 41. Layout of the Bellezane site showing the four tailings repositories (Lavaugrasse, Le Brugeaud, Bellezane and Montmassacrot) within the Crouzille Mine Division.

Table 10 presents the amounts of ore mined in Bellezane, and the corresponding amounts of uranium produced by milling or heap leaching from open pit mining and underground mining.

Mining activities both in open pits and tailing management systems generated 18 378 775 and 2 194 687 tonnes of raw materials (ore and waste rock), respectively. Ore was milled in a nearby facility (by SIMO[44]).

TABLE 10. AMOUNTS OF ORE MINED IN BELLEZANE AND CORRESPONDING AMOUNTS OF URANIUM PRODUCED

Origin	Ore mined (t)	Uranium produced (kg)
TMS	1 237 286 109 177	2 617 324 46 814
MCO	799 061 687 569	1 167 290 221 777
Total	2 833 093	4 053 205

[44] Subsidiary of COGEMA at the time of site operation; this subsidiary has not existed since the closure of all the uranium mines in France. COGEMA was succeeded by AREVA in 2001 and later, by ORANO in 2018.

Tailings repository

The tailings generated from the SIMO milling facility, located at Bessines-sur-Gartempe, were stored in the tailings repository of Bellezane. More than 97% of the tailings that were disposed of in Bellezane were produced by a dynamic leaching treatment process. These tailings contain the highest activity concentrations of ^{226}Ra, with a level of 32 000 Bq/kg dry mass on average, compared to 14 000 Bq/kg dry mass on average for heap leaching residues. This repository contains a total of 1 513 591 tonnes of fine particle-sized tailings and 42 029 tonnes of heap leaching residues.

Remediation

Remedial actions that were carried out at the site included:

- Backfilling the mining works connecting the bottom and top of the mine;
- Backfilling the pits;
- Covering the tailings that had been disposed of in two of the pits with 250 000 m^3 of waste rock.

Management of water flows was achieved by:

- Establishment of adequate drainage on site;
- Operation of a water treatment plant with settling basins.

The treatment products that are currently used are:

- Barium chloride to precipitate radium with sulphate ions;
- Lye to adjust pH and aluminium sulphate to precipitate iron and for uranium fixation on hydroxides;
- Flocculant (aluminium polyhydroxychlorosulphate) to increase settling processes.

The water treatment facility consists of three serially connected settling basins.

Establishment of a disposal cell for sludge and sediment

By a prefectural decree dated August 2006, disposal of sludge generated from the dredging of the water treatment facility and of sediments from the dredging of various ponds located in the surroundings was allowed in a dedicated cell that was established in the southern part of the tailings repository. The cell capacity was set to hold 42 000 m^3 of dry matter. Flow of drainage water is piped through two different drains toward a tunnel.

7.2.1.2. Considerations in the design of the tailings repository

Considerations during the design phase of the tailings repository were presented in a licence application by AREVA (COGEMA at that time) in 1987. The management of potential impacts resulting from the repository is based on the collection and treatment of water likely to be in contact with tailings, and on the installation of a cover over the tailings.

Amounts of tailings and their characteristics

Tailings produced by dynamic treatment in the SIMO milling facility

Ore (uraninite, pitchblende) from the mines in the Limousin region was milled in the SIMO facility. Treatment in the milling facility was carried out for ore containing more than 0.6% of uranium. For less rich ore, the treatment involving in situ leaching.

Considerations relating to treatment and production of tailings

The ore was crushed and pulverized into a pulp to a size of smaller than 450 μm. This pulp was then leached using sulphuric acid at 65°C with sodium chlorate, $NaClO_3$, as an oxidizer (to facilitate a reaction to produce uranium in a more soluble state). The filtrate was fed into the process for uranium treatment, whereas tailings were washed and sent to the retention basin.

Dissolved uranium was extracted from solution with a solvent, followed by ammonium sulphate; the resultant solution was neutralized using ammonia. The final concentrated solution was ammonium diuranate, $(NH_4)_2U_2O_7$, which contained 75% of the source uranium. Treatment discharges were neutralized and precipitated with lime and calcium carbonate (limestone); tailings produced after neutralization were consolidated with those produced after acid attack, and sludge from water that was treated prior to discharge.

Some of the sandy fraction of the tailings produced were used as backfill for tunnels during mining. Tailings that had not been separated using cyclones were disposed of, after spin-drying, in an open pit mine or a remediated basin. Tailings originating from the SIMO milling facility were distributed over four other repositories.

Characteristics of tailings

Considering the efficiency of the entire mining and milling process, tailings produced by dynamic treatment contain only 5% of their initial uranium content. For heap leaching, the mining efficiency is lower (50–80%) and the residues contain a larger fraction of the initial ore uranium content.

Considering differences between initial content in ore milled by the SIMO facility by dynamic treatment or heap leaching, and the efficiencies of these two types of treatments, residual uranium concentrations range from 50–300 ppm in both cases.

Radiological characteristics of the tailings can be represented by data generated from a number of core samples taken in 1993–1994 from the Le Brugeaud repository. Gamma spectrometric analysis of 39 strata from one core sample (50 m deep) provided the mean values shown in Table 11.

Based on data from particle-size analysis, it appears that the fine particle-sized fraction contains most of the radionuclide content (measurements on ^{226}Ra). Radium activity in the coarse particle-sized sand fraction (>150 μm) is estimated to be 2500 Bq/kg dry mass. This is consistent with the results presented in Table 12.

TABLE 11. LE BRUGEAUD TAILINGS − MEAN VALUES MEASURED ON 39 STRATA (Bq/kg dry mass) AND RANGE OF VARIATION

U-238	U-234*	Th-230	Ra-226	Pb-210
1 188 (448–1961)	860 (375–1 100)	13 329 (1 547–30 380)	12 703 (1 500–28 140)	14 182 (1 505–29 830)

*U-234: measurements on 11 strata.

TABLE 12. PARTICLE-SIZE ANALYSIS OF SIX TAILINGS SAMPLES COVERING A WIDE RANGE OF ^{226}Ra ACTIVITY CONCENTRATIONS

Ra-226 (Bq/kg dry mass) in different strata	Particle size <200 µm (%)	Particle size >200 µm (%)
28 140	97.3	2.7
23 940	97.4	2.6
20 410	94.1	5.9
5 661	73.4	24.7
4 850	67.5	32.5
4 317	38.6	61.1

These results are consistent with other studies (e.g., see Refs [147−148]). For example:

- The particle-size fraction of smaller than 150 µm is made up of clayey minerals, sulphates (essentially, gypsum and barite), and iron and aluminium hydroxides, and contains more than 80% of the residual radioactivity;
- The sandy fraction (150–450 µm) has a chemical composition that is similar to the host rock, with a significantly lower activity than the fine particle-size fraction (<150 µm).

A chemical analysis has been performed on five samples from the core samples taken in 1993–1994. Through this analysis, the compounds presented in Table 13 were detected. These data confirm that the chemical composition of the tailings include chemical precipitates related to treatment, whereas their overall composition is mainly similar to that of the ore and the host rock. Table 14 presents the chemical composition of the tailings.

From a mineralogical point of view, tailings from acid treatment that was carried out in the SIMO milling facility are made up of:

- More than 90% mineral solid phases originating from the granitic ore, which was chemically resistant to treatment. Specifically, they are composed of quartz, feldspar, mica, sulphide and clay from chemical weathering;
- Between 1.5 and 8% of newly formed minerals arising from the acid treatment of the ore and from the treatment of effluents before discharge (e.g. gypsum, barite, iron sulphates, iron and aluminium hydroxides). This indicates the linkage between the tailings and the original minerals, which contain 30–90% of the activity of the residual uranium.

TABLE 13. CHEMICAL COMPOSITION OF TAILINGS MEASURED IN SAMPLES TAKEN FROM LE BRUGEAUD REPOSITORY

Anions	**Dry matter content (% or ppm)**	**Trace elements**	**Dry matter content (ppm)**
Sulphates	3.24–11.71%	Pb, Bi, Ba, Zn, Mn	100–500
Phosphates	0.304–0.707%	Cu, Ni, Sn, V, W, Y	10 –40
Carbonates	0.11–0.85%	Cd, Co, Ge, Mo, Sb, Se	< 10
Nitrates	5–88 ppm		

TABLE 14. THE CHEMICAL COMPOSITION OF THE TAILINGS

Oxides and elements	**Dry matter mass (% or ppm)**
$SiO_2+Al_2O_3$	65–90%
$CaO+Na_2O+K_2O+Fe_2O_3+P_2O_5+MgO$	10–35%
Uranium	70–280 ppm
Thorium	about 20 ppm
Radium	0.0006–0.003 ppm
Pb, Zn, Cu, Ni, Co, Ag, Bi, ...	Trace elements

Leaching tests on tailings

Tests were performed on the tailings core samples taken from the Le Brugeaud repository, which provide information on the chemical characteristics of leachates. These tests were carried out using distilled water that was saturated with carbon dioxide as a solvent, and air to simulate rainwater. The liquid-to-solid ratio was equal to 10. The observations made from these tests are as follows:

- pH of leachates fell between 6.7 and 7.3, whereas at the end of belt filter presses in the SIMO milling facility, the pH was more acidic with a value of approximately 5.5;
- Approximately 1% of the ^{226}Ra was leached after 3 successive contacts, and the proportions of barium and uranium were lower than 1%;
- Calcium sulphate and magnesium sulphate were present in large amounts. Calcium sulphate was leached from the sixth extract (of 10 successive leachings), and radium and barium then decayed within solutions;
- Sodium and chlorides were present in lower proportions and were leached from the second extract;
- Heavy metals were not detected, indicating limited extraction.

Other studies have reported the predominance of Ca, Mg and SO_4 in the water flows of Bessines-sur-Gartempe and have concluded that calcium and magnesium concentrations are strongly correlated to those of sulphate. Fluid composition is primarily controlled by the nature of newly formed phases and not by the main mineral phases. Gypsum controls Mg, Ca and SO_4 behaviour, whereas silicates control Si, Al and K behaviour. A study on another repository, located in the Vendée region, has revealed that 0.3–0.6% of ^{226}Ra and 34–47% of gypsum can be extracted using distilled water.

Successive extractions performed using different reagents enable the different phases containing ^{226}Ra to be traced as follows:

- Total dissolution of gypsum results in the release of 10–20% of ^{226}Ra within tailings;
- Dissolution of amorphous phases of iron, manganese and aluminium, and pyrite results in releases of the largest amounts of radium and uranium;
- Dissolution of barite, iron and manganese oxides, and phosphate oxides results in additional releases of a large amount of ^{226}Ra;
- The silicate fraction, which represents 60–70% of the tailings mass, only contains 3–7% of radium and 7% of uranium.

These results confirm that the radioactivity of sands after cyclonic separation, mainly composed of the silicate fraction, consists of much lower contents of radionuclides than in the fine particle-sized tailings.

Under the current conditions of pH and redox potential of the tailings, gypsum controls the activity of ^{226}Ra in porewater. Some leaching tests were performed between 1994 and 1995 by COGEMA on sludge from the water treatment facility of the Bellezane site and the same conclusion was drawn for ^{226}Ra, again indicating that the amounts of ^{226}Ra that are being discharged with water are low.

Comparison of radiological composition of tailings and waste rock

An estimation of radium and uranium contents in different residues was made, based on past data, to compare the activity of tailings disposed of on site with the activity of waste rock in dumps. Relevant data are presented in Table 15.

TABLE 15. AMOUNT AND TOTAL ACTIVITY OF THE MATERIALS EXTRACTED

Amount of the materials extracted	Total activity
1 513 591 tonnes of fine particle-sized tailings from dynamic treatment in pits MCO 105 and MCO 68[a]-	Ra-226: 48.4 TBq (hypothesis of mean Ra-226 content equal to 32 000 Bq/kg dry mass) U-238: 2.4 TBq (hypothesis: U-238 in ore = Ra-226; mining efficiency 95%; the same calculation made with the results acquired from Le Brugeaud repository would give 1.8 TBq)
42 029 tonnes of heap leaching residues disposed of in pit MCO 105	Ra-226: 0.6 TBq (hypothesis of mean Ra-226 content equal to 14 000 Bq/kg dry mass) U-238: 0.2 TBq (hypothesis: U-238 in ore = Ra-226; mining efficiency 70%)
14 179 tonnes of hydraulic fill in pit MCO 122	Ra-226: 0.04 TBq (hypothesis of mean Ra-226 content equal to 2 500 Bq/kg dry mass) U-238: 0.002 TBq (hypothesis: U-238 in ore = Ra-226; mining efficiency 95%)
17 740 000 tonnes of waste rock extracted and used to fill some TMSs, MCOs and to build dumps (estimation based on a uranium content of 50 ppm, i.e. 617 Bq kg^{-1}, to be compared to the uranium content in host rock which is about 20 ppm on average)	Ra-226: 11 TBq U-238: 11 TBq

[a] This comprises the tailings repository of Bellezane, which consists of the two former open pit mines (MCO 68 and MCO 105), separated by a dike

7.2.2. Scenario descriptions

Two scenarios of exposure were considered as part of this model–model intercomparison:

- A scenario for the current situation at the site;
- A scenario for a possible future situation involving intrusion.

The two scenarios are described in the following Sections 7.2.2.1 and 7.2.2.2.

7.2.2.1. Current situation

The scenario for the current situation corresponds to the configuration of the site in 2011, where most transfer pathways are supposed to be under control, as verified through environmental monitoring. Access to the Bellezane site is restricted and there is a fence around the site. Consequently, the receptors of exposure considered are the inhabitants of the nearest village to the site, 'Les Grandes Magnelles'; the village centre is approximately 1.5 km from the outer boundary of the Bellezane repository, and the nearest resident is approximately 1 km away.

The representative person is an adult who lives and works in the village. A part of his/her diet consists of locally grown products (meat of cows pasturing on fields close to the tailings repository, and vegetables grown on fields close to the repository).

In summer, the locally grown vegetables are assumed to be irrigated with water taken from a well pumping directly from the groundwater and are assumed to be located in the field.

In addition to exposure via inhalation of radon and its short-lived progeny (released from contaminated water only), the following ingestion exposure pathways are considered:

- Plants;
- Meat;
- Milk;
- Aquatic foods;
- Drinking water;
- Soil.

The primary contamination route for receptors is waterborne since:

- The receptor does not live on or very close to the tailings repository;
- The tailings repository is mainly shaped like a valley with a northwest–southeast orientation, perpendicular to the axis of the 'tailings repository-receptor's home'.

Therefore, inhalation and external gamma pathways are not considered in this scenario. Short visits on site for leisure activities we're not considered for the current scenario, as such visiting is unrealistic in the case of Bellezane, due to the institutional control of the site (and especially, the presence of a fence). However, for other types of sites or in other contexts, short visits may be considered as a realistic hypothesis in a risk assessment. It has been verified that consideration of a short intrusion of half an hour per day on the Bellezane site led only to a trivial exposure.

7.2.2.2. *Intrusion (future situation)*

A conservative case was considered, involving the formulation of a plausible intrusion scenario in the future. It was assumed that, in the year 2112, the responsibility of the Bellezane site is transferred to the French nation. Then, due to budget restrictions, the monitoring programme is ceased in 2163. Consequently, in 2209, the existence of the tailings is forgotten.

In 2253, a family builds a house on site. Once settled, they work at home, and they have a garden where they grow some vegetables. These vegetables are irrigated in summer with water from a well pumping directly from the groundwater. All relevant exposure pathways are considered for the intrusion scenario, i.e. ingestion of:

- Plants;
- Meat;
- Milk;
- Aquatic foods;
- Drinking water;
- Soil;

as well as:

- External gamma exposure;
- Inhalation of contaminated dust;
- Inhalation of radon and its short-lived progeny.

The primary routes of contamination considered from the point of view of the receptor are both airborne and waterborne.

7.2.3. Available environmental measurement data

For the current scenario, the available measurement data are from the environmental monitoring implemented by AREVA on the Bellezane site and in its vicinity. Available data have been synthesized in a single file readable by a geographic information system software package, which has been used for this model–model intercomparison. These data are related to the following environmental compartments:

- Groundwater;
- Surface water;
- Air (gamma dose rate and possible alpha level);
- Fish;
- Sediment;
- Milk;
- Soil;
- Vegetables.

Figure 42 shows the spatial distribution of the measurement points that have been established for the environmental monitoring.

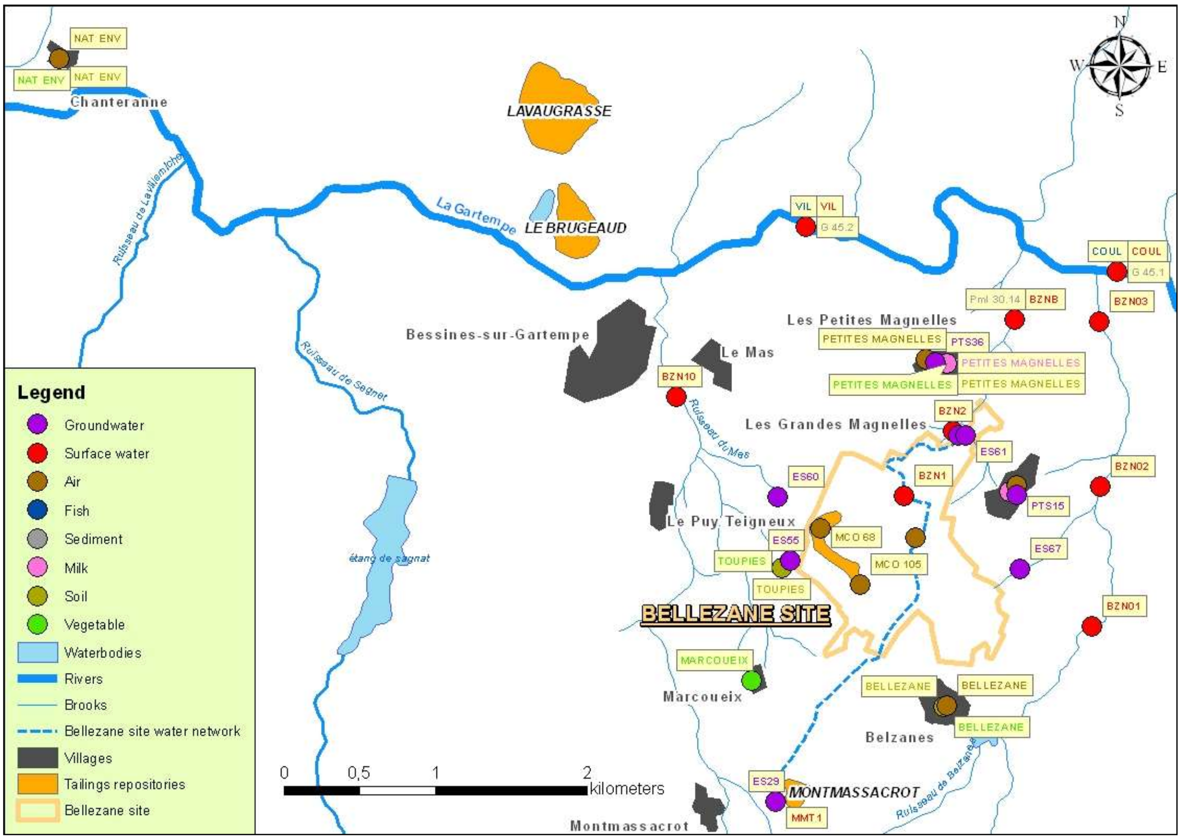


FIG. 42. Spatial distribution of the monitoring data available in the vicinity of the Bellezane site.

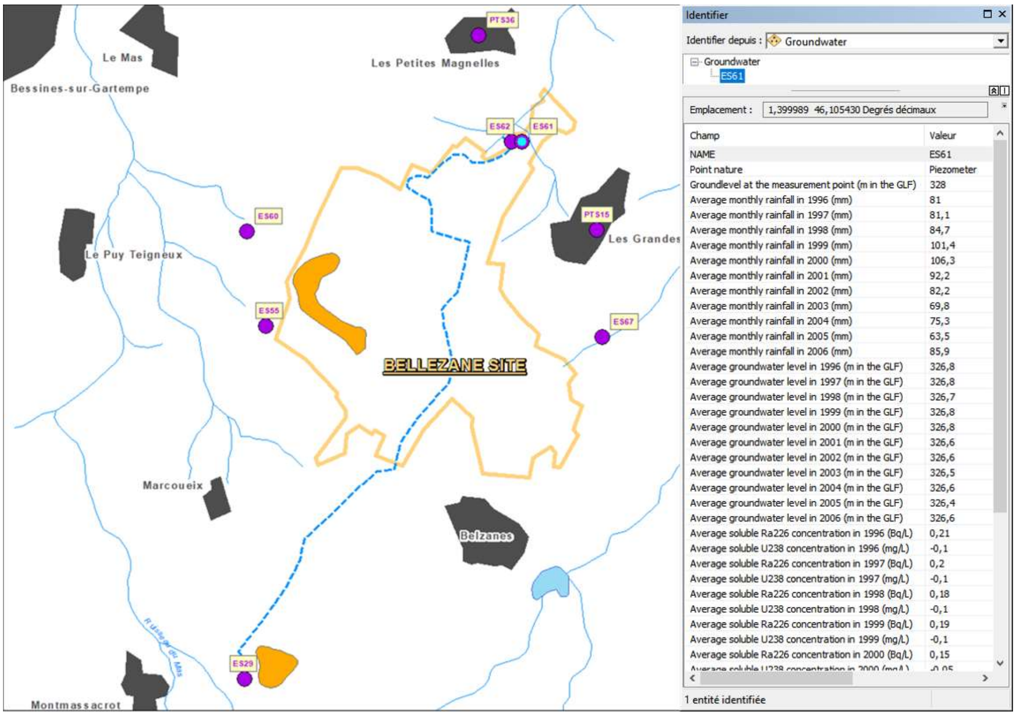


FIG. 43. Available measurement data for groundwater – Detail for the piezometer ES61.

No data were selected for animals (e.g. poultry, rabbit) because such data are scarce, and the monitoring results obtained all fall below the detection limit for the measurement devices used. Most data cover the period between 1994 and 2006 (remediation ended in 1996), and almost all the values provided for the model–model intercomparison represent an average over an entire year. However, many values fall under the detection limit, which has sometimes changed over time. This is the reason why few significant data are available for several monitoring points. Figure 43 (above) provides an example of available measurement data for groundwater.

7.2.4. Available site specific data

7.2.4.1. Source term

The source term considered is the tailings repository of Bellezane, which consists of the two former open pit mines (MCO 68 and MCO 105), separated by a dike.

Characteristics of MCO 68

Total surface area: ~2.2 ha.

Successive layers (from top to bottom) are as follows:

- Vegetation cover:
  - Thickness: 0.1–0.2 m [149];
  - No detectable radionuclides.
- Waste rock cover (compacted waste rock):
  - Thickness: 2 m;
  - Dry bulk density: ~1700 kg/m^3;
  - Radionuclide inventory: 40 ppm uranium [150] (1 ppm uranium = 12 Bq ^{238}U/kg dry mass), supposedly in equilibrium with its progeny;
  - Permeability: 2.8 × 10^{-7} m/s [151].
- Layer of tailings from dynamic treatment:
  - Thickness: 50–60 m;
  - Radionuclide inventory: 1.6 Bq ^{238}U/g dry mass – 32 Bq ^{226}Ra g dry mass (on average);
  - Permeability: between 10^{-5} and 10^{-8} m/s [151].

Characteristics of MCO 105

The characteristics of MCO 105 are as follows:

Total surface area: ~2.8 ha.

Successive layers (from top to bottom):

- Vegetation cover:
  - Thickness: 0.1–0.2 m [149];
  - No detectable radionuclides.
- Waste rock cover (compacted waste rock):
  - Thickness: 2–12 m, depending on the location;
  - Density: ~1700 kg/m^3;
  - Volume: 250 000 m^3 [149];

- Radionuclide inventory: 50 ppm uranium [150] (1 ppm uranium = 12 Bq ^{238}U/kg dry mass), assumed to be in equilibrium with its progeny;
- Permeability: 2.8×10^{-7} m/s [151].

— Layer of heap leaching residues:
  - Thickness: ~5 m;
  - Radionuclide inventory: 5 Bq ^{238}U/g dry mass – 14 Bq ^{226}Ra/g dry mass (on average) [149].

— Layer of tailings from dynamic treatment:
  - Thickness: 10–25 m;
  - Radionuclide inventory: 1.6 Bq ^{238}U/g dry mass – 32 Bq ^{226}Ra/g dry mass (on average) [149];
  - Permeability: 10^{-5} – 10^{-8} m/s [151];

— Concrete slab:
  - Thickness: 1 m.

— Waste rock and fluid concrete drainage layer:
  - Thickness: 5 m.

7.2.4.2. Groundwater and surface water

Groundwater

The following data are available:

— Catchment area of the Bellezane site: ~120 ha [149];
— Type of soil: quite sandy;
— Type of host rock: granitic;
— Description of unsaturated zone(s) under tailings: no unsaturated zone;
— Saturated zone: average piezometric level: +360 m NGF ('Nivellement Général de la France') [149];
— Main drainage and water circulation paths through the host rock [149];
— Direction of groundwater flow [151];
— Hydrogeological balance [151].

Surface water

The following data are available:

— Flow rate of the Les Petites Magnelles brook[45]: 60 m^3/h [149];
— Flow rate of the Gartempe river: 30 000 m^3/h [149];
— Runoff coefficient: 0.219 [151].

Figure 44 provides a synthesis some of the previous features presented.

[45] Water is discharged into this brook after treatment.

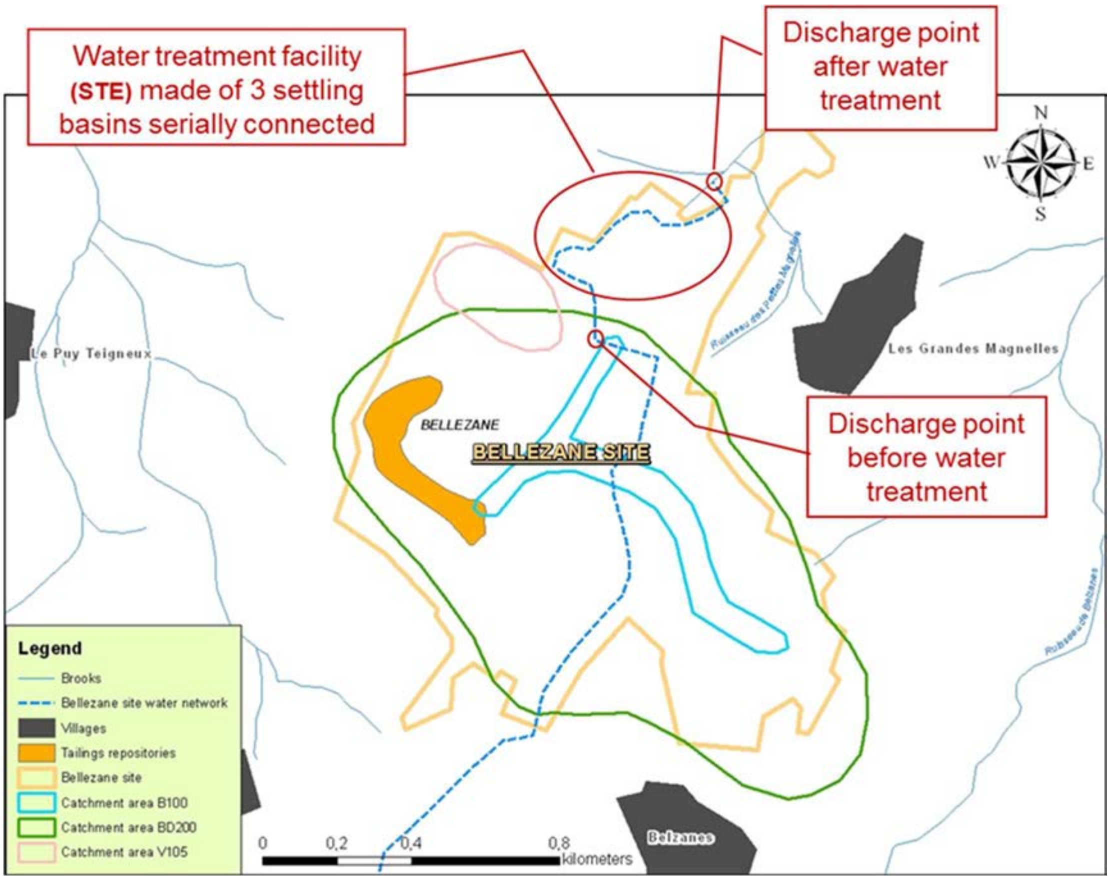


FIG. 44. Layout of the Bellezane site showing some key features.

7.2.4.3. Climate

Table 16 provides data relevant to the local climate. The average precipitation between 1993 and 2005 was 1050 mm/a [151].

TABLE 16. AVERAGE PRECIPITATION AND EXTREME TEMPERATURES OVER DIFFERENT PERIODS OF THE YEAR (DATA FROM A WEATHER STATION LOCATED IN BESSINES-SUR-GARTEMPE OPERATED BY MÉTÉO FRANCE)

Period of the year	Pluviometry (mm/d)	Lowest temperature (°C)	Highest temperature (°C)
September – November 2000	5.3	7.0	13.7
June – July 2007	3.3	12.4	23.1
April – June 2008	4.7	6.9	17.8

7.2.4.4. Diet of members of the public living in a rural area in France

Dietary data for an adult living in a rural area in France are presented in Table 17.

TABLE 17. DIET FOR AN ADULT LIVING IN A RURAL AREA IN FRANCE [152]

Foodstuff	Consumption for an adult (kg/a or L/a)	Fraction of consumed locally produced foodstuff
Leafy vegetables	28	0.707
Fruit vegetables	75	0.306
Root vegetables	12	0.675
Potatoes	44	0.767
Cereals	75	0.001
Beef meat	20	0.365
Mutton	4	0.435
Pork meat	27	0.292
Poultry meat	26	0.734
Eggs	10	0.631
Milk	88	0.301
Milk products	31	0.059
Fish	9	0.161

7.2.5. Radiological environmental impact assessment

The following modelling exercise is only a first attempt, based on two simple scenarios of exposure (corresponding to the current situation and a case study of a future intrusion situation). These scenarios are intended to provide a minimal representation of a REIA, as the Bellezane site is complex to model for most of the transfer pathways. Therefore, the results provided in this section are for illustration only and are not for use in any future management decisions.

Two modelling tools have been used to perform the REIA for this model–model intercomparison: RESRAD-OFFSITE and SATURN.

7.2.5.1. Modelling assumptions and hypotheses for RESRAD-OFFSITE

Small adjustments had to be made to relate (standardize) available data and the input dataset needed for RESRAD-OFFSITE.

Source term

To simplify the modelling process, a parallelepiped is considered for the source term (see Fig. 45). To conserve mass, the parallelepiped, representing the layer of tailings, is assumed to have:

— A density of 1200 kg/m^3;
— A total mass of 1 512 000 t [46];
— A surface area of 50 400 m^2.

The dimensions of the parallelepiped, representing the layer of tailings, are 420 m × 120 m × 25 m.

To conserve mass, the waste rock is assumed to have:

— A total volume of 295 000 m^3;
— A surface area of 50 400 m^2.

[46] Heap leaching residues are not included, considering their small contribution to the total mass.

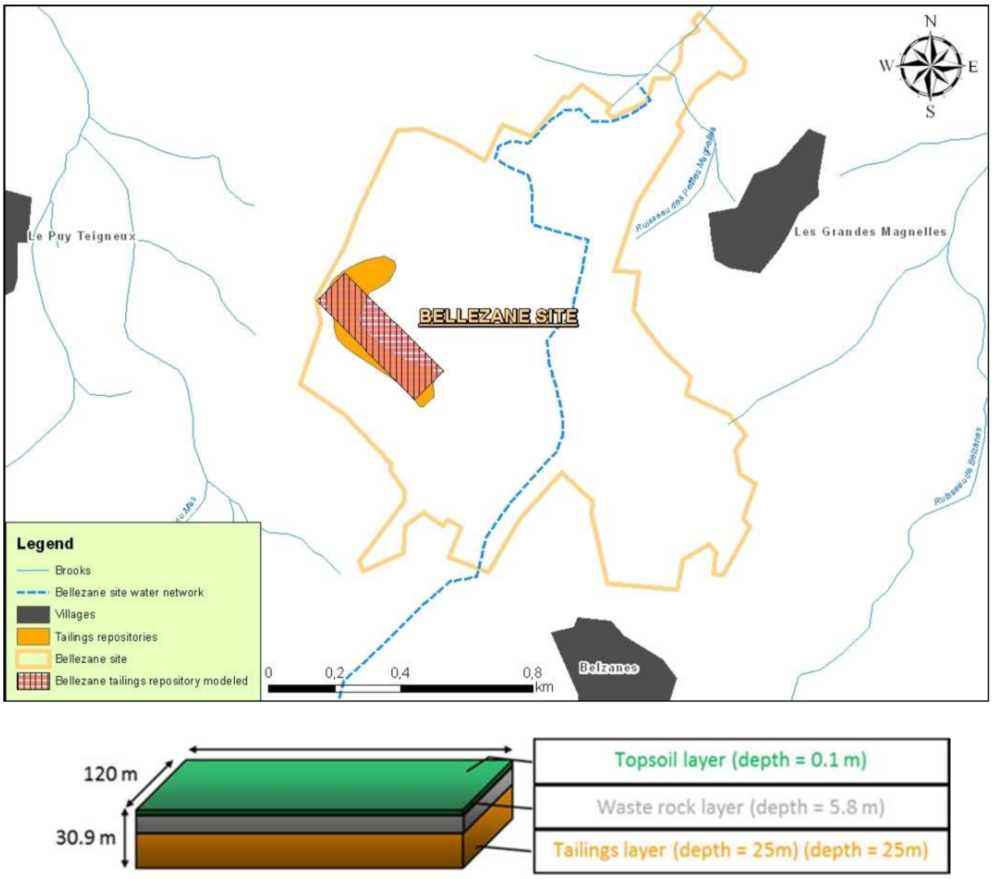


FIG. 45. Source term modelled.

The dimensions of this parallelepiped, representing the waste rock cover, are 420 m × 120 m × 5.8 m.

For the topsoil layer, the same area is considered, with a depth of 0.1 m. The resulting parallelepiped, made up of these three layers, is oriented northwest–southeast.

Dose coefficients

The dose coefficients for external exposure are taken from Ref. [128], whereas those for ingestion and inhalation originate from Ref. [153].

Solid–liquid distribution coefficient K_d

Table 18 presents the values that have been adopted for the solid–liquid distribution coefficient (K_d), sometimes called the sediment-to-water partition coefficient.

TABLE 18. SOLID–LIQUID DISTRIBUTION COEFFICIENTS (K_d) VALUES.

Radioactive element	K_d (m^3/kg dry mass)
Uranium	0.180 [74]
Thorium	2.6 [74]
Radium	0.500 [154]
Lead	0.140 [154]
Polonium	0.030 [154]

Diet

Considering the diet presented in Table 17 for people living in a rural area in France, and the need to relate the diet to the different kinds of foodstuffs used in RESRAD-OFFSITE, Table 19 presents the diet adopted for the calculations.

RESRAD-OFFSITE only considers beef and no other kinds of meat. As a result, it was decided to merge all the values for meat consumption into this one category, even if this might create a bias due to the differences between consumption rates and transfer factors from one animal type relative to another. In practice, the differences in consumption rates tend to compensate for the differences in transfer factors, as consumption rates increase with increasing body mass, whereas transfer factors decrease, although not always proportionately. The same approach for merging data has been done for the category 'Fruit, grain, non-leafy vegetables'.

TABLE 19. CONSUMPTION CONSIDERED FOR CALCULATIONS WITH RESRAD-OFFSITE

Foodstuff	Consumption considered for calculations with RESRAD-OFFSITE (kg/a or L/a)	Fraction of foodstuff from affected area
Fish	9	0.161
Crustaceans, mollusks	0	0
Fruit, grain, non-leafy vegetables	206*	0.315
Leafy vegetables	28	0.707
Meat	77**	0.468
Milk	88	0.301

* Includes the values from Table 17 above for fruit vegetables, root vegetables, potatoes and cereals.
** Includes the values from Table 17 above for beef meat, pork meat, mutton and poultry meat.

Other parameters

With the exception of occupancy factors, which were adapted to the respective scenarios of exposure, for other parameters, the default values used in RESRAD-OFFSITE were adopted.

7.2.5.2. RESRAD-OFFSITE results

Both the scenarios for the current situation and for a future intrusion were modelled using RESRAD-OFFSITE. Although the proposed timescale was 1000 years, the calculations were performed over a much longer time span, most likely beyond the scope of any regulatory control, to indicate the possible effects of radionuclide migration to surface water and groundwater.

In both scenarios, separate modelling was performed for the tailings layer and the waste rock layer. Although the degree of contamination of the waste rock layer is much smaller than the tailings layer, because the waste rock is at the surface, it is the main contributor to the external exposure and radon pathways.

Current scenario

The tailings repository was modelled as a contaminated area without considering the radioactive content of the waste rock layer[47].

An Excel spreadsheet with parameter values for the modelling exercise was distributed to participants. Figure 46 presents the main features provided in the input file.

The modelled groundwater flow is directed straight to a surface water body of limited extent (with a flow rate of 60 m^3/h), representing the monitoring point corresponding to the water discharge before treatment.

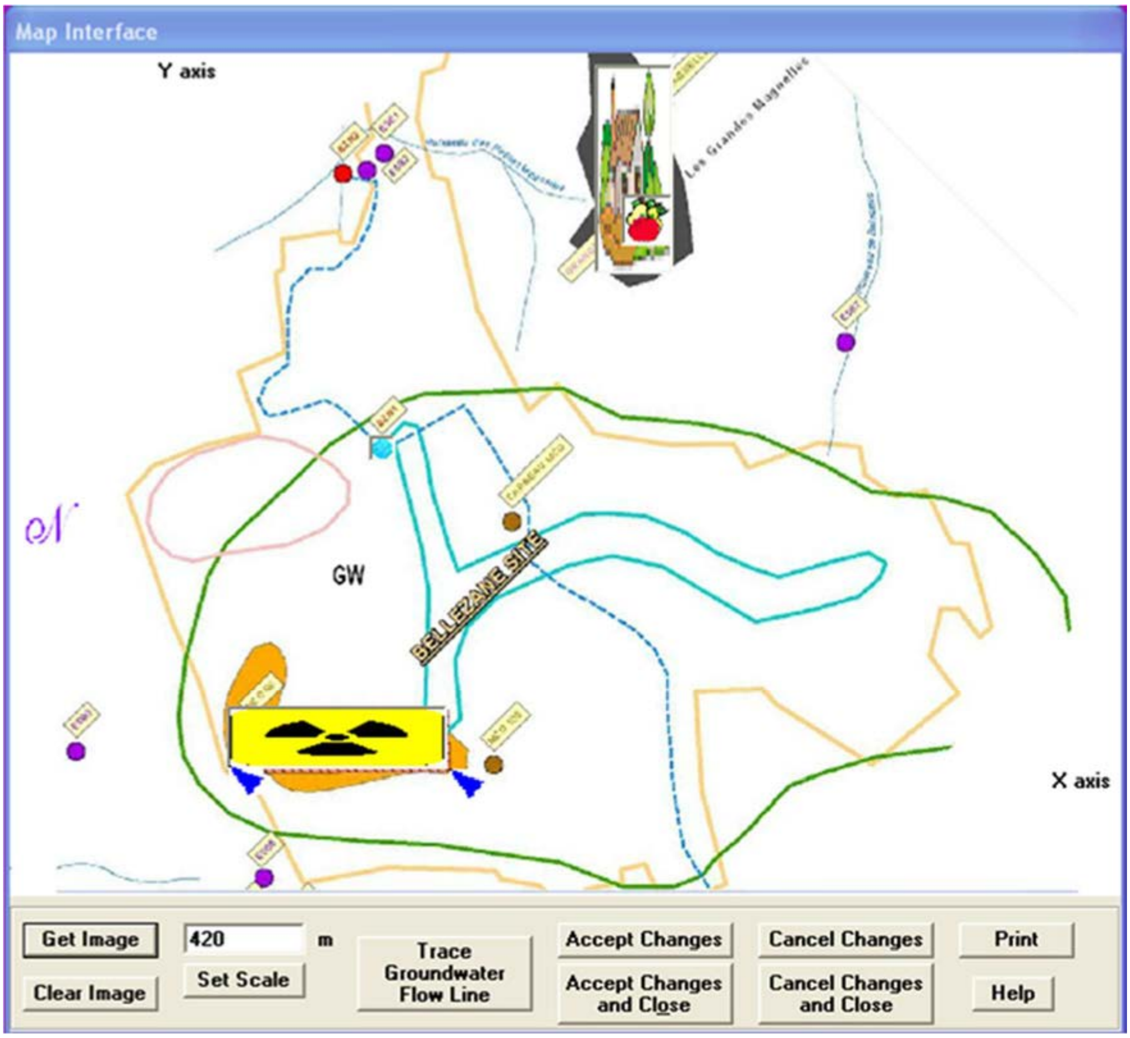


FIG. 46. RESRAD-OFFSITE map interface showing the main features considered and described for calculations related to the current scenario.

[47] In this case, the thickness of the clean cover is assumed to be equal to 5.9 m (topsoil layer + waste rock layer).

Calculated doses

Due to the long migration time of the radionuclides to surface water and to groundwater and because the exposure pathways considered for this scenario are only waterborne, there is no dose predicted for the first 1000 years. However, it is important to be note that these migration times may be over-estimated, as discussed further below.

If the timespan is extended to 100 000 years (the maximum value allowed in RESRAD-OFFSITE), the annual effective dose is trivial for most of the time and reaches 0.1 mSv only after approximately 80 000 years. Figure 47 provides an overview of the individual contributions of the exposure pathways to the total annual effective dose; a maximum annual effective dose of approximately 1.6 mSv is reached after 100 000 years, but is still increasing at that time.

The maximum annual dose that has been calculated is due to the progeny of ^{238}U and ^{234}U, and mainly due to ^{210}Po, as shown in Figs 48 and 49, respectively. The radionuclides with a shorter radioactive half-life that are present in the source term almost completely decay away before they can reach the surface via the groundwater.

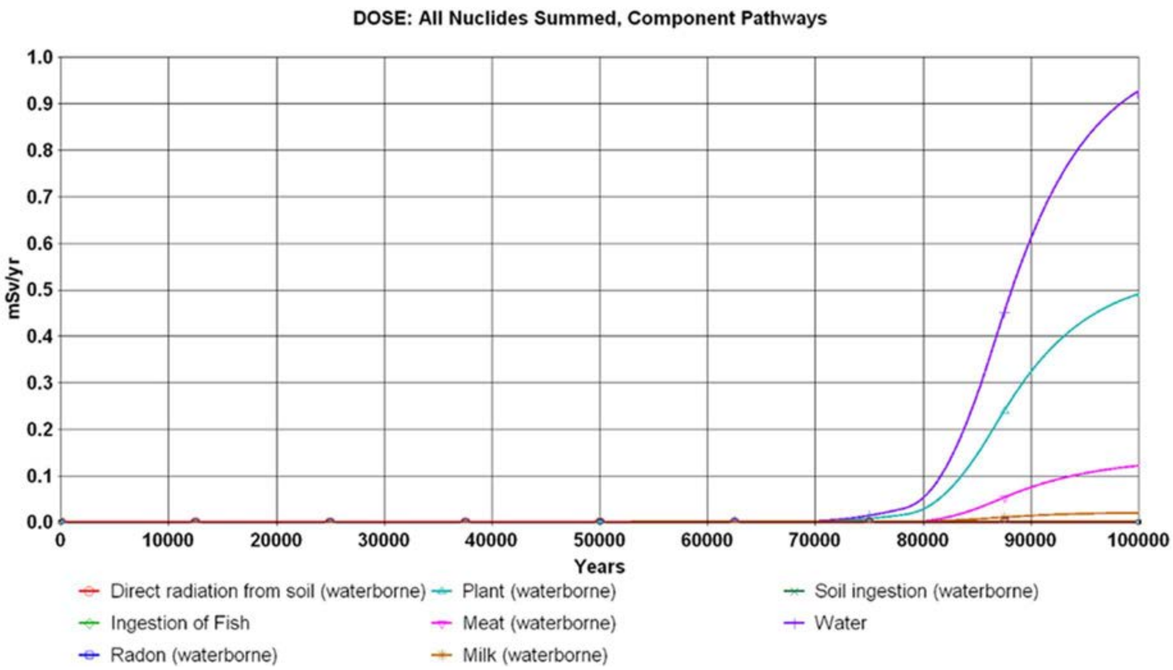


FIG. 47. Dose contribution of different exposure pathways over time for the current scenario (impact of tailings).

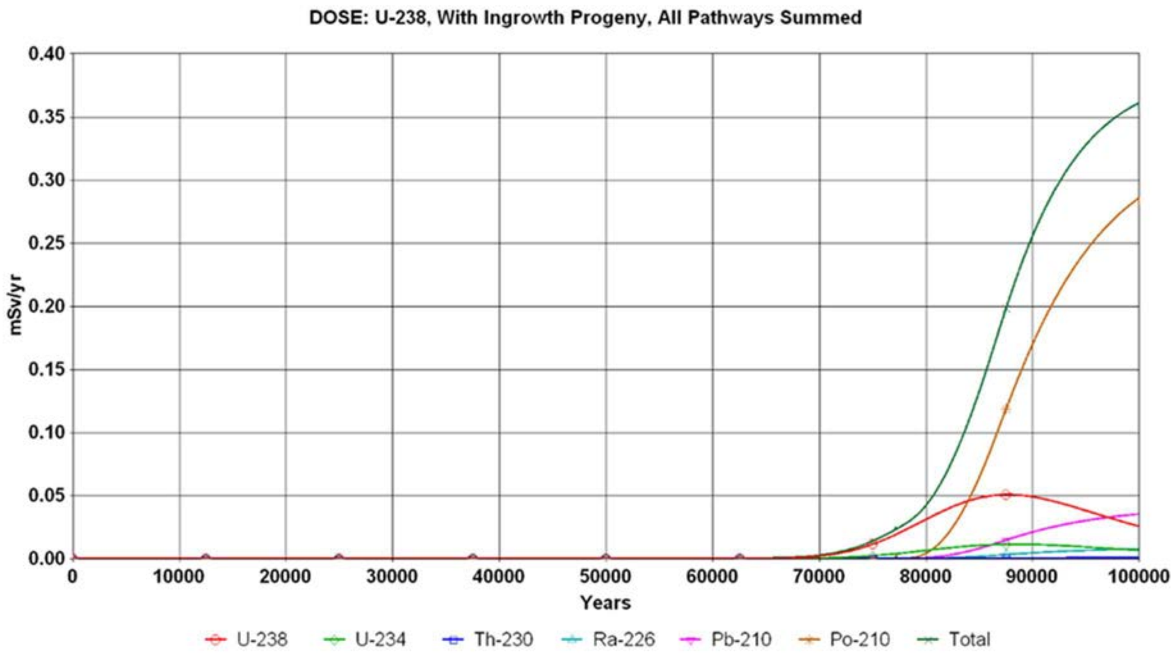


FIG. 48. Contributions to dose of ^{238}U and its ingrowing progeny in the current scenario (impact of tailings).

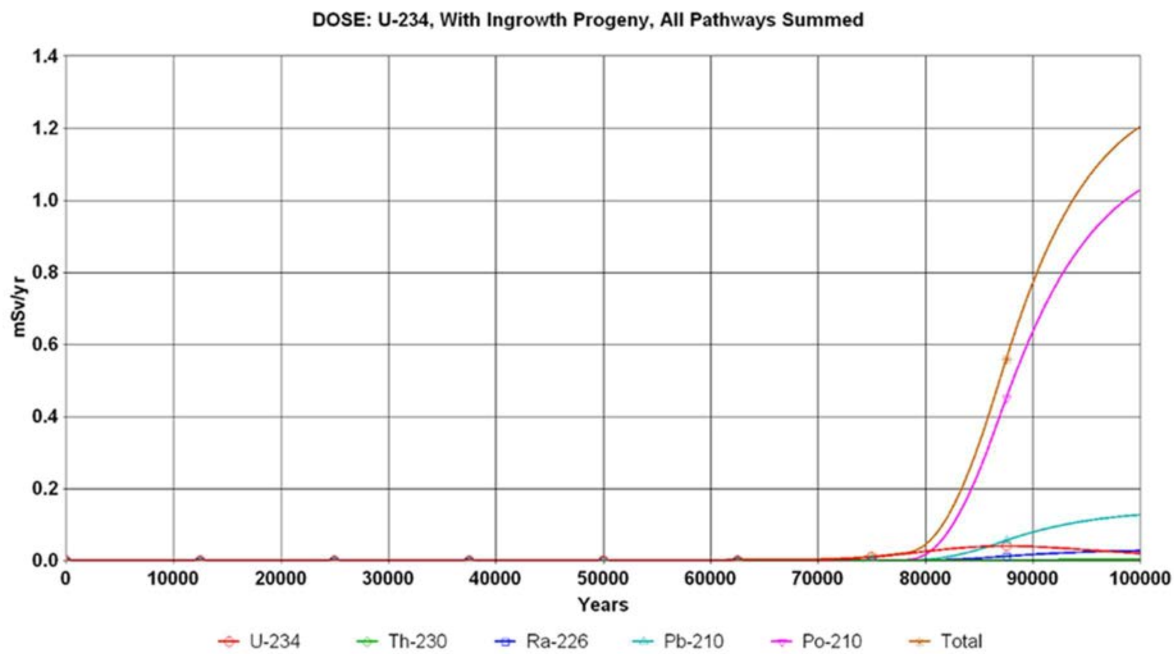


FIG. 49. Contribution to dose of ^{234}U and its ingrowing progeny in the current scenario (impact of tailings).

Radionuclide activity concentrations in well water

No radionuclide migration to well water is anticipated unless long timespans are considered.

The maximum activity concentrations of ^{238}U and ^{234}U in the well water is expected only after approximately 88 000 years; maximum activity concentrations of ^{230}Th and its progeny in well water is expected to occur after 100 000 years. Figure 50 depicts the anticipated evolution of the ^{210}Po activity concentration in well water over time.

Model outputs for surface water body

The model outputs indicate that there is no radionuclide migration towards the surface water body, except if very long timespans are considered (see Fig. 51). In 2007, the mean annual activity concentration of soluble ^{226}Ra that was measured in surface water at the discharge point before treatment reached approximately 0.5 Bq/L. It is therefore possible that the results provided by the model implemented in RESRAD-OFFSITE may underestimate the transfer of radionuclides through groundwater. The groundwater flow considered in this model is assumed to occur through a porous medium, whereas on the site under consideration, groundwater is mainly drained by some mining tunnels or through fractures in the granite. The transfer of radionuclides through groundwater is more accurately modelled by increasing the hydraulic conductivity of the saturated zone. If a value of 1 000 000 m/a is used, which is more representative of water transport through a cavity, instead of the default value of 100 m/a, the results presented in Fig. 52 are generated. However, a better approach would be to implement a discrete fracture network representation of the flow and transport system.

The water treatment process has not been modelled, although the process can result in a 10-fold decrease in activity concentration values in surface water.

Modelling the water treatment process is considered a difficult task, especially in terms of its spatial extent. In addition, it is difficult to calibrate a model with measurements in such a context where the average background level is quite high and difficult to discriminate.

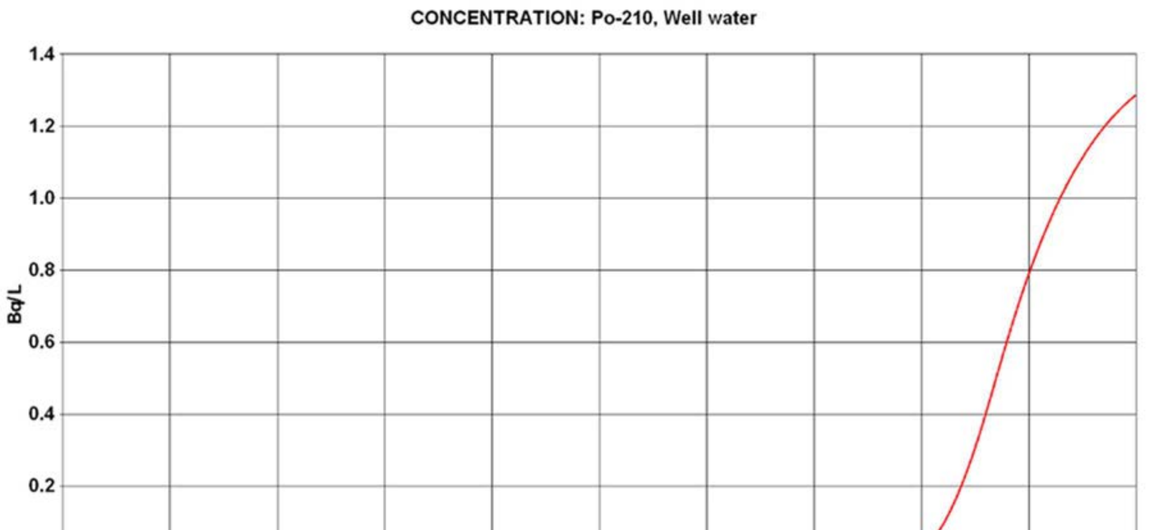


FIG. 50. Time evolution of ^{210}Po concentration in well water in the current scenario (impact of tailings).

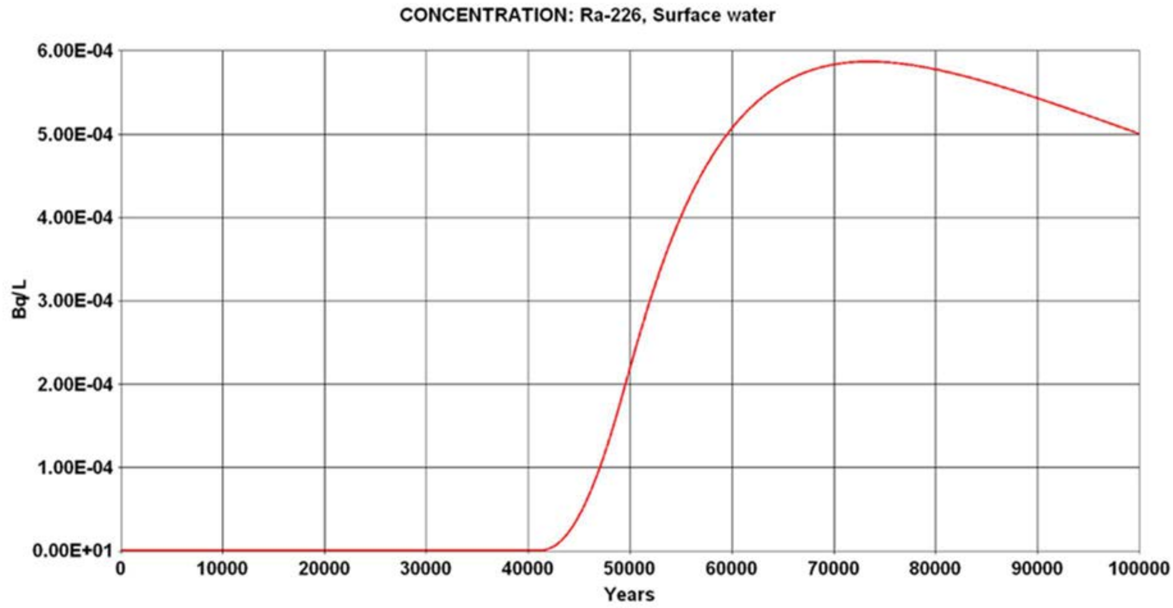


FIG. 51. Time evolution of ^{226}Ra concentration in the surface water body modelled for the current scenario (impact of tailings).

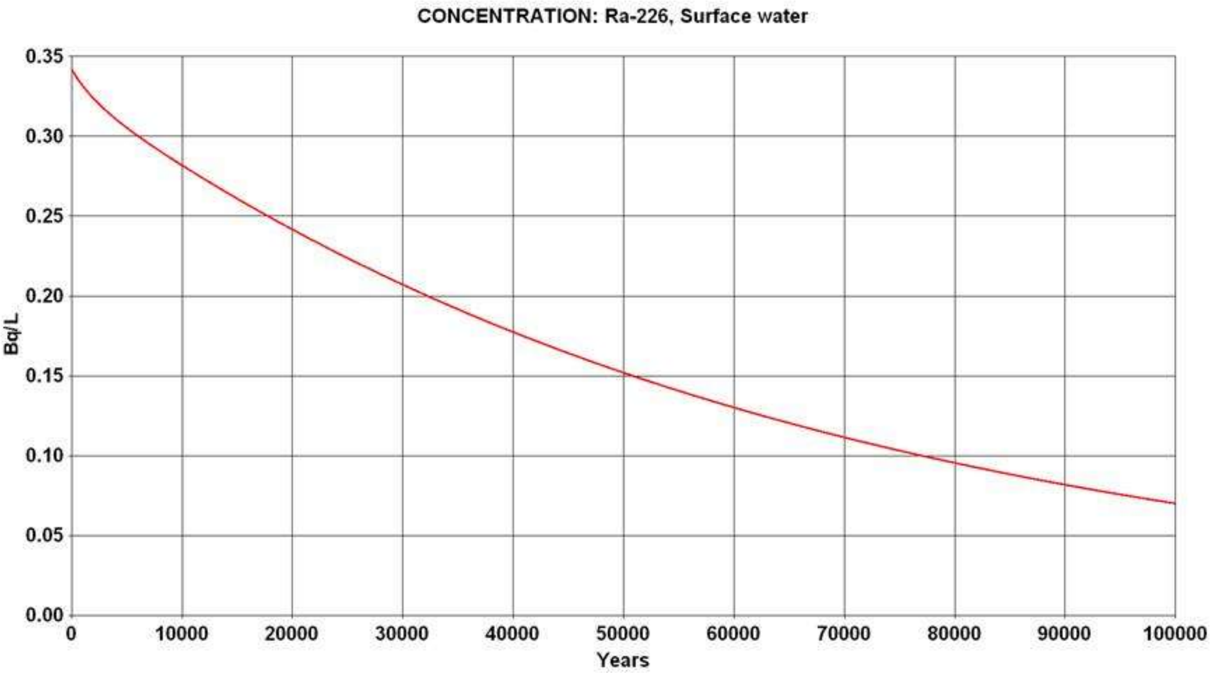


FIG. 52. Time evolution of ^{226}Ra concentration in the surface water body modelled for the current scenario (impact of tailings), with a hydraulic conductivity of the saturated zone equal to 10^6 m/a. The values obtained fit much better to measurement data since the appropriate order of magnitude is reached from the very beginning of the timespan studied.

Sensitivity analysis

Sensitivity analysis was performed on the following parameters:

— Hydraulic conductivity of the contaminated area and of the saturated zone;
— K_d for polonium in the saturated zone and in the contaminated area.

Changing the hydraulic conductivity of the contaminated area or the value of the K_d of polonium in the contaminated area has no influence on the results (it neither influences the value of the maximum dose rate nor the time of the maximum annual effective dose). The influence of the two other parameters (hydraulic conductivity of the saturated zone and K_d) on the maximum annual effective dose and the time when the maximum annual effective dose is reached is shown in Tables 20 and 21, respectively. It is emphasized that because of the long groundwater transport time estimated using the porous medium approach with reference parameter values, these sensitivity analyses may not be physically realistic.

TABLE 20. INFLUENCE OF HYDRAULIC CONDUCTIVITY OF THE SATURATED ZONE ON MAXIMUM ANNUAL EFFECTIVE DOSE

Hydraulic conductivity of the saturated zone (m/a)	Maximum annual effective dose (mSv)	Time of maximum annual effective dose (years)
100 (default)	1.598	99 885
50	0.0008	99 934
200	0.781	68 332

TABLE 21. INFLUENCE OF K_d FOR POLONIUM IN THE SATURATED ZONE ON MAXIMUM ANNUAL EFFECTIVE DOSE

K_d (dm^3/kg)	Maximum annual effective dose (mSv)	Time of maximum annual effective dose (years)
30 (base case)	1.598	99 885
3	12.38	100 000
300	0.429	99 885

Modelling the waste rock layer as a contaminated area

The waste rock layer[48] contains 0.5 Bq/g dry mass of ^{238}U in secular equilibrium with its progeny. In this part of the modelling exercise, the waste rock layer is defined as a contaminated area, and the tailings layer is considered as an unsaturated zone. These results are added to the ones compiled previously.

The maximum annual effective dose (0.2 mSv) is only reached after a long period of time (~97 000 years) (see Fig. 53). Again, this result depends on use of the reference parameterization for a continuous porous medium and may not be physically realistic.

[48] Here, the thickness of the clean cover is assumed to be 0.1 m.

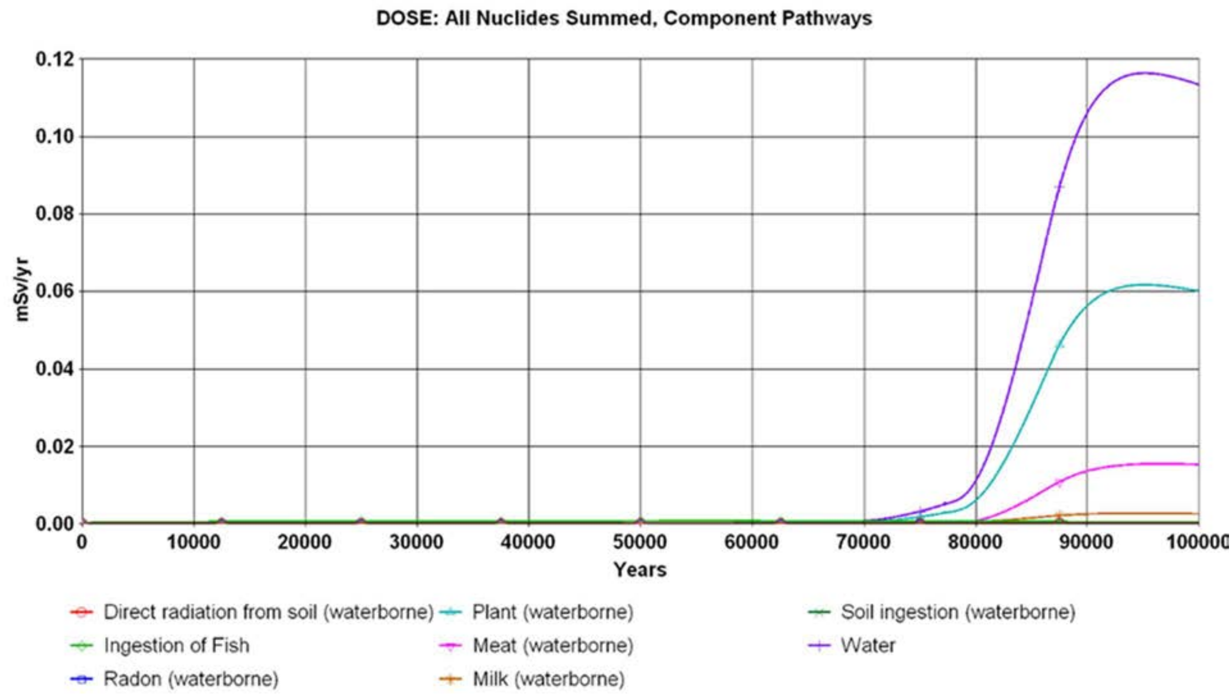


FIG. 53. Time evolution of the contribution of the different exposure pathways to the dose in the current scenario (impact of waste rock cover).

For the scenario for the current situation, the dose impact is trivial over a timeframe of 1000 years. A non-trivial, but still limited, annual effective dose only occurs if a long timeframe is considered; however, this timescale is determined by use of a simple continuous porous medium model that almost certainly underestimates the rates of groundwater flow and radionuclide transport due to site specific factors (e.g. the underestimation of hydraulic conductivities through fractures in the granitic bedrock and through mining tunnels).

Intrusion scenario

To model the intrusion scenario, the following parameters had to be changed compared with the scenario for the current situation:

— Site layout (location of the receptors);
— Location of well;
— Occupancy factors.

The fraction of the agricultural area directly overlying the primary contamination was assumed to be 1.0. Occupancy factors were set as follows:

— Fraction of time spent on primary contamination:
  - Indoors: 0.8;
  - Outdoors: 0.15.

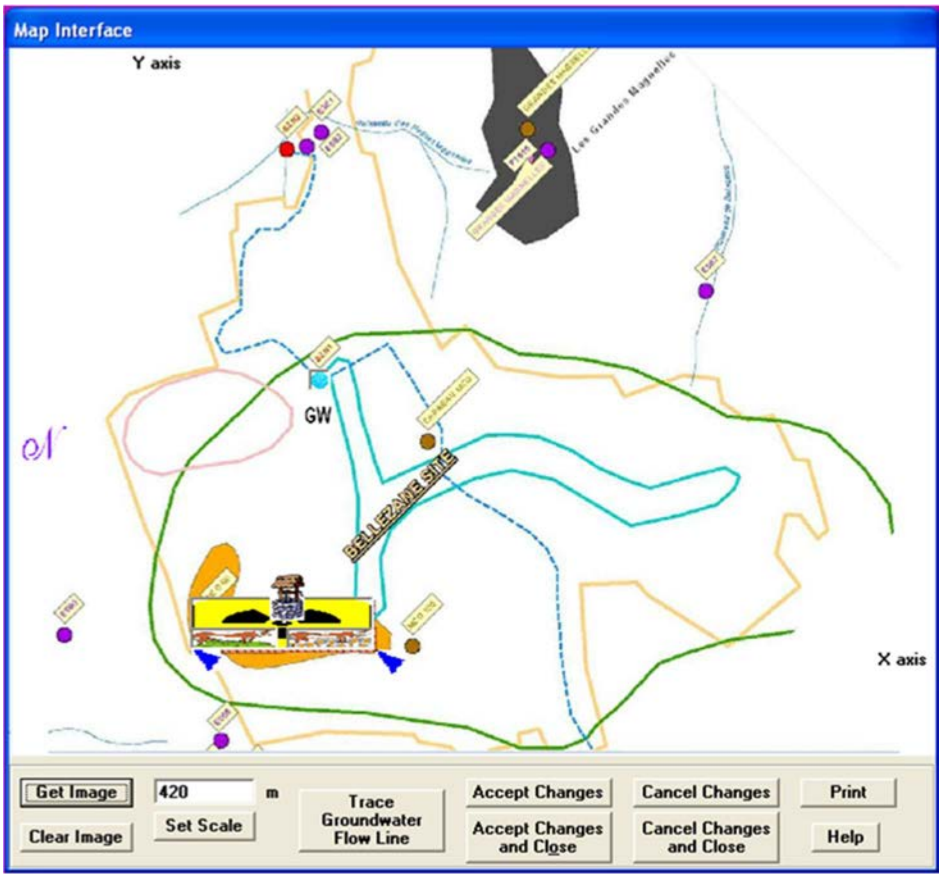


FIG. 54. RESRAD-OFFSITE map interface showing the major features considered and described for calculations related to the intrusion scenario.

— Fraction of time spent in farmed areas[49]:
- Fruit, grain, and non-leafy fields: 0.05;
- Leafy vegetable fields: 0.05;
- Pasture and silage fields: 0;
- Livestock grain fields: 0.

Figure 54 depicts the main features implemented in the input file.

Modelling of the tailings

Calculated doses

The water well is located directly on the contaminated area. Therefore, doses due to water-dependent pathways are significant compared with other pathways.

After 10 years, the annual effective dose is approximately 4.7 mSv (almost entirely due to radon arising from both water- and gas-mediated pathways). The maximum annual effective dose is reached after approximately 70 000 years, with a value of 152 mSv.

[49] Located fully inside the primary contamination for this scenario of exposure.

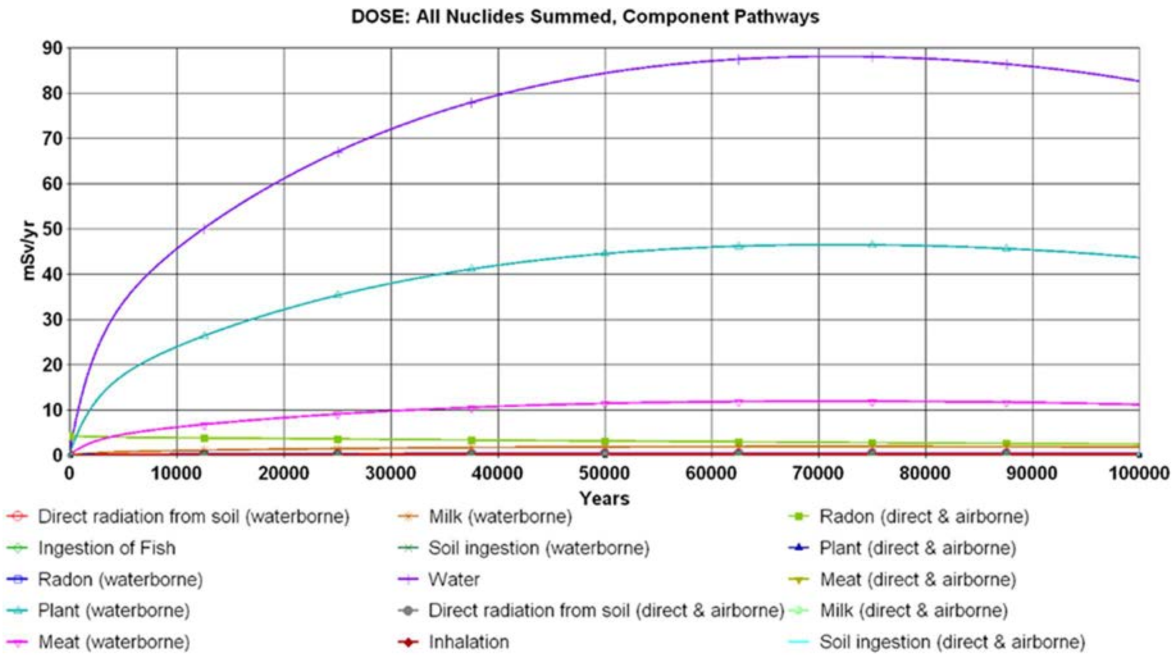


FIG. 55. Time evolution of the contribution of the different exposure pathways to the effective dose in the intrusion scenario (impact of tailings).

As shown in Fig. 55, this annual effective dose is almost entirely due to water-dependent pathways:

- Drinking water: ~90 mSv;
- Ingestion of plants: ~47 mSv;
- Ingestion of meat: ~11 mSv.

Sensitivity analysis

Sensitivity analysis has been performed for the same parameters as for the scenario for the current situation. Changing the hydraulic conductivity of the contaminated area or the value of the K_d of polonium in the contaminated area had no influence on the results. This is as expected because it is transport in the saturated zone that dominates. Changes in transport in the contaminated area would be trivial in comparison. The significant influence of the hydraulic conductivity and the K_d in the saturated zone on the maximum dose and the timing of this maximum being reached is shown in Tables 22 and 23, respectively.

TABLE 22. INFLUENCE OF THE HYDRAULIC CONDUCTIVITY OF THE SATURATED ZONE ON MAXIMUM ANNUAL EFFECTIVE DOSE

Hydraulic conductivity of the saturated zone (m/a)	**Maximum annual effective dose (mSv)**	**Time of maximum annual effective dose (years)**
100 (default)	152	71 018
50	40.5	69 211
200	526	46 254

TABLE 23. INFLUENCE OF THE K_d OF POLONIUM IN THE SATURATED ZONE ON MAXIMUM ANNUAL EFFECTIVE DOSE

K_d (m^3/kg dry mass)	Maximum annual effective dose (mSv)	Time of maximum annual effective dose (years)
0.030 (base case)	114	71 018
0.003	1212	71 409
0.300	36.38	69 455

Erosion of the cover

The current parameters (default values for the parameters affecting the erosion rate, with the exception of density) lead to an erosion rate for the cover of 10^{-5} m/a; the 5.9 m thick cover will, therefore, be maintained for the entire investigation period of 100 000 years. To simulate the erosion of the cover, a change of the cover management factor from its default value of 0.003 to a value of 0.1 leads to an erosion rate of 3.4×10^{-4} m/a for the cover and 4.8×10^{-4} m/a for the tailings. The cover will be eroded after 17 500 years, and the tailings, 52 000 years later. This will lead to a maximum annual effective dose of 657 mSv at t = 17 828 years. As shown in Fig. 56 ('component pathways'), this is mainly due to the large contribution of radon following the erosion of the cover. The effect of the complete erosion of the tailings, after 69 500 years, is also clearly visible in Fig. 56, with a large reduction in the annual effective dose.

Modelling of the waste rock layer

Calculated doses

Considering the radionuclide content of the waste rock layer, radon is the dominant exposure pathway over the entire timescale under consideration. The maximum annual effective dose occurs at early times, with a value of approximately 15.8 mSv, almost entirely due to radon, as shown in Fig. 57.

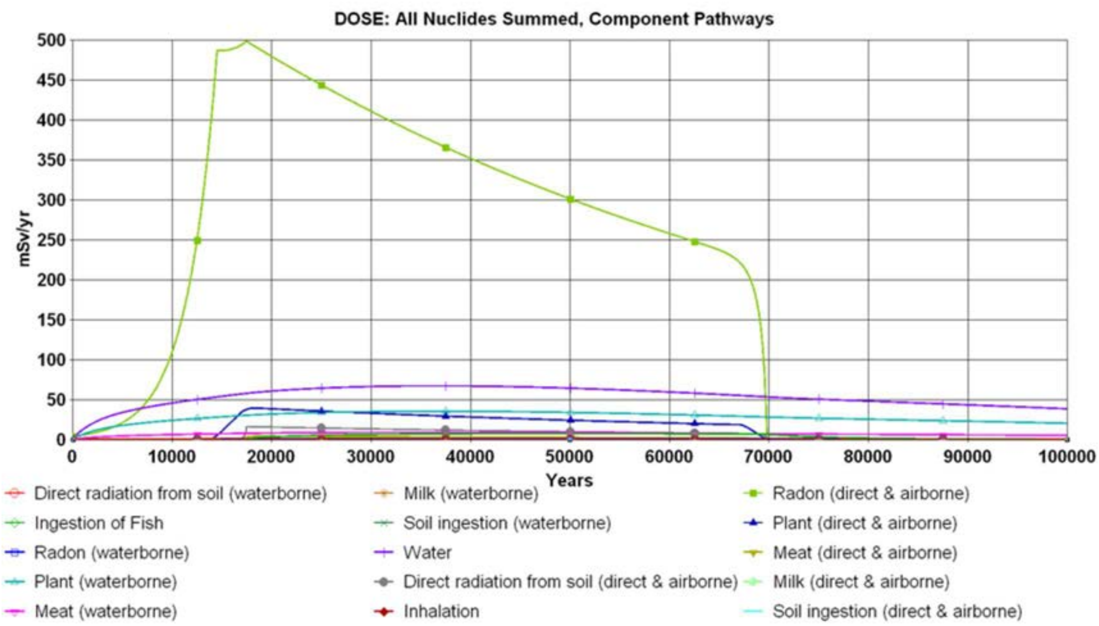


FIG. 56. Time evolution of the contribution of the different exposure pathways to the annual effective dose in the intrusion scenario (impact of tailings) – Cover eroded after 17 500 years.

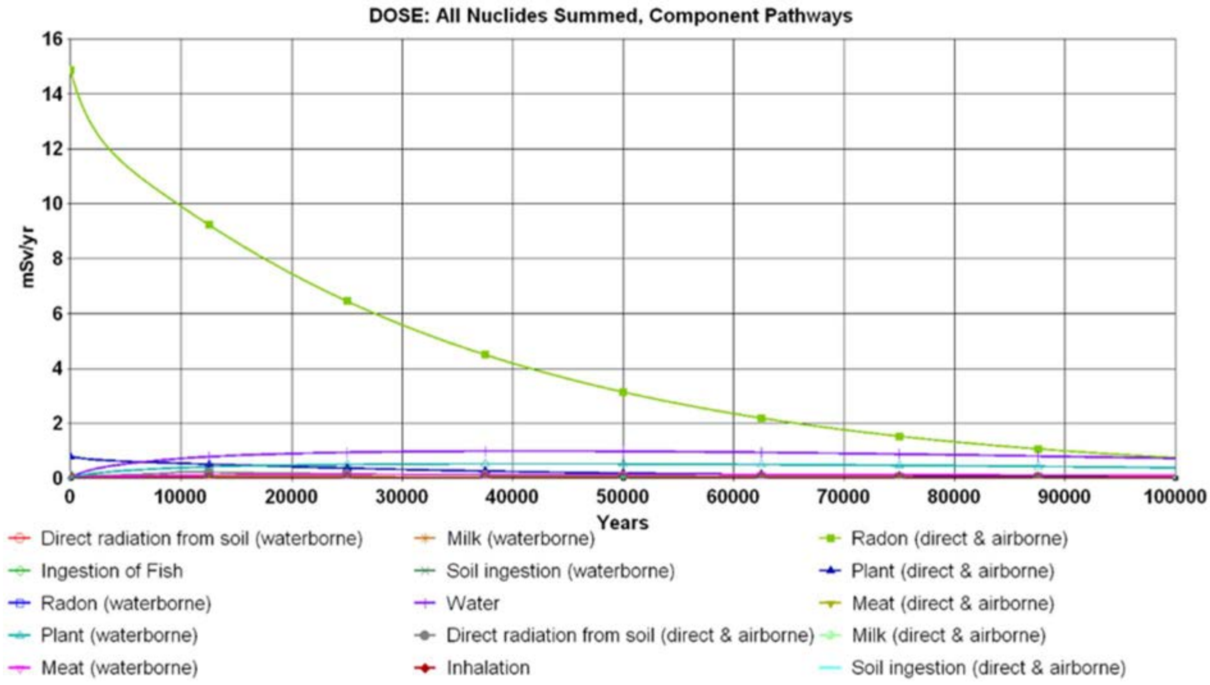


FIG. 57. Time evolution of the contribution of the different exposure pathways to the effective dose in the intrusion scenario (impact of waste rock cover).

Sensitivity analysis

As radon is the most significant exposure pathway, two radon-related parameters were selected for sensitivity analysis: the effective radon diffusion coefficient in the contaminated area (waste rock layer); and the total porosity of this contaminated area. The results are summarized in Tables 24 and 25, respectively.

TABLE 24. INFLUENCE OF THE EFFECTIVE RADON DIFFUSION COEFFICIENT IN THE CONTAMINATED ZONE

Rn-222 diffusion coefficient (m^2/s)	Maximum annual effective dose (mSv)	Time of maximum annual effective dose (years)
2×10^{-6} (default)	15.8	0
2×10^{-5}(default)	20.11	0
2×10^{-7}	10.73	0

TABLE 25. INFLUENCE OF TOTAL POROSITY OF THE CONTAMINATED AREA

Total porosity	Maximum annual effective dose (mSv)	Time of maximum annual effective dose (years)
0.4	15.8	0
0.2	24.75	0
0.8	9.71	0

Combination of the impact of tailings and waste rock layers

If the contributions to exposure from the waste rock cover and from the tailings are added, the curve shown in Fig. 58 is generated. The contribution of the waste rock cover is dominant for up to approximately 450 years.

The intrusion scenario may lead to a significant annual effective dose. For example, within a timeframe of 1000 years, both tailings and waste rock contribute to the dose. Over a time period of 450 years, the main contribution to the dose originates from the radon emanating from the waste rock layer, where the activity concentration of ^{226}Ra is of the same order of magnitude as that of phosphogypsum, for example. The tailings will affect the water-dependent pathways, which dominate for longer timescales.

7.2.5.3. SATURN

Dose calculations for the current and intrusion scenarios were also carried out using the SATURN radioecology modelling library, developed by Facilia AB using Ecolego. The SATURN library implements dose assessment models described in Ref. [155] and the source for the default values of model parameters are Refs [61, 155]. Additional monitoring data on the hydrogeological characteristics within the tailings-repository was extracted from Ref. [156].

Calculations were carried out separately for the two former open pit mines (MCO 68 and MCO 105), which contain tailings of the Bellezane repository.

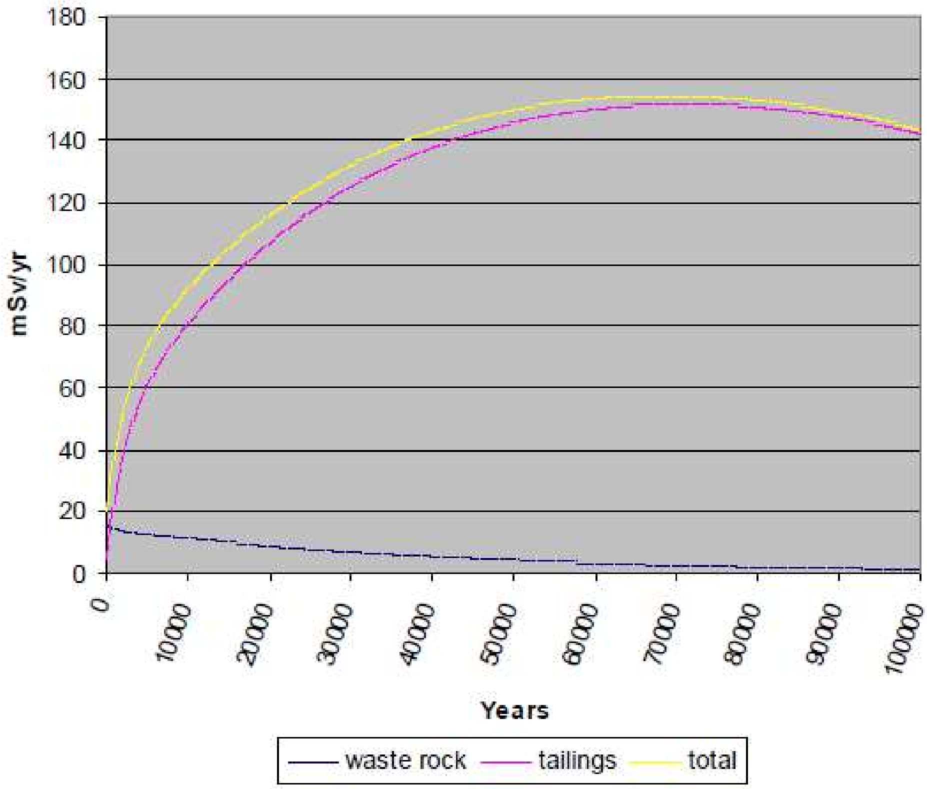


FIG. 58. Respective contributions to the total effective dose of the waste rock layer and of the tailings.

Estimation of doses for the current scenario

For the scenario relating to the current situation, the following calculations were performed using SATURN:

- Doses from consumption of crops irrigated with groundwater using measured levels of ^{226}Ra in groundwater;
- Doses by different transfer pathways resulting from radon releases from the tailings and external exposure from the tailings. The end points of the calculations were doses from staying in the village and from ingestion of crops, milk and meat produced on fields close to the tailings.

Transport of radionuclides via groundwater from the Bellezane tailings site was not modelled using SATURN for the current scenario. This is due to the complex hydrogeological setting and lack of detailed hydrogeological schematization. Nevertheless, an attempt was later made to estimate radionuclide migration in groundwater towards the well downgradient of the contaminated area, and to carry out calculations for comparison. Based on this assessment, it was concluded that groundwater transport is not a significant contributor to dose, except on extremely long timescales.

The distance considered from the contaminated area to the well was 1150 m, and the groundwater travel time (without retardation) was calculated as follows:

Flow velocity = Hydraulic conductivity × Gradient/Porosity
= 10 m/a travel time = Distance/Velocity = 115 years

The K_d for uranium, which originally considered as a default value was 0.05 m^3/kg; the corresponding retardation factor was, therefore, 376. This translates to an advective travel time of uranium to the well of 43 000 years. However, the time interval considered here for modelling is only 1000 years. As discussed above, it is debatable whether this travel time is realistic for channelled flow through fractured rock, which includes tunnels from former mining activities.

Doses from consumption of irrigated vegetables

Dose to adults from consumption of crops that are contaminated by irrigation with groundwater is included in the model outputs. Radionuclide activity concentrations in groundwater are taken from the monitoring data.

It was assumed that three types of crops comprise the diet of an adult (see Ref. [155]):

- ‘Leafy vegetables’ (consumption by an adult of 13 kg/a is assumed);
- Other ‘vegetables’ (40 kg/a);
- ‘Root vegetables’, such as potatoes (55 kg/a).

In addition, it was assumed that 25% of the crops consumed by the exposed adult originate from the contaminated area, and the remainder comes from other, uncontaminated areas. In the calculation of radium activity concentrations in crops, the soil-to-plant transfer factor values for radium, which have been published by the IAEA (see Ref. [74]), were used. These are reproduced in Table 26.

Two types of calculations were carried out: deterministic and probabilistic. The deterministic simulation used a best estimate value for the transfer factor value for radium (geometric mean from Table 17 of Ref. [74]). The calculations were done for the ^{226}Ra activity concentration in irrigation water of 0.45 Bq/L, which is the upper value of the range of activity concentrations in groundwater reported in the monitoring results. Results from the deterministic calculations of radium activity concentrations in different types of vegetables and resulting doses are presented in Table 27. The modelled time interval was assumed to be equal to 100 years. Activity concentrations of radium in vegetables and associated doses are expected to increase with time due to accumulation of radionuclides in soil (see Table 27).

In the probabilistic calculations, log-normal distributions were assumed for the transfer factor (truncated by the minimum and maximum values presented in Ref. [74]). Another model parameter that was treated as being uncertain was the fraction of the activity in irrigation water intercepted by the crops; this value was assumed to range from 0.2 to 0.4 (uniform distribution; a value of 0.3 was used in the deterministic simulation). Results from the probabilistic Monte Carlo simulation are presented in Table 28. The median value of the dose is close to 'best estimates' prediction from the deterministic simulations (for t = 50 years), with doses corresponding to the 5% and 95% percentiles differing by approximately a factor of 3, as shown in Table 28.

TABLE 26. ELEMENT SPECIFIC SOIL-TO-PLANT TRANSFER FACTORS (TF) [74]

Element	Plant group	TF geometric mean, kg dry mass/kg dry mass	Geometric standard deviation (GSD)	Min TF, kg dry mass/kg dry mass	Max TF, kg dry mass/kg dry mass
U	Leafy vegetables	2.0×10^{-2}	7.3	7.8×10^{-5}	8.8
Ra	Leafy vegetables	9.1×10^{-2}	6.7	1.8×10^{-3}	130
Th	Leafy vegetables	1.2×10^{-3}	6	9.4×10^{-5}	0.21
Po	Leafy vegetables	7.4×10^{-3}	6.9	2.5×10^{-4}	0.05
Pb	Leafy vegetables	8.0×10^{-3}	13	3.2×10^{-3}	25
U	Non-leafy vegetables	1.5×10^{-2}	4.2	5.2×10^{-4}	0.2
Ra	Non-leafy vegetables	1.7×10^{-2}	8.4	2.4×10^{-4}	6.3
Th	Non-leafy vegetables	7.8×10^{-4}	6.8	6.2×10^{-5}	1.6×10^{-2}
Po	Non-leafy vegetables	1.9×10^{-4}		1.6×10^{-5}	3.7×10^{-4}
Pb	Non-leafy vegetables	1.5×10^{-2}	26	1.5×10^{-3}	3.9
U	Roots	8.4×10^{-3}	6.2	4.9×10^{-4}	0.26
Ra	Roots	7.0×10^{-2}	9.2	2.0×10^{-3}	56
Th	Roots	8.0×10^{-4}	1.3	8.2×10^{-6}	0.09
Po	Roots	5.8×10^{-3}	4.3	2.4×10^{-4}	0.05
Pb	Roots	1.5×10^{-2}	1.6	2.4×10^{-4}	3.3

Note: For conversion from dry mass to fresh mass of vegetables, the following dry matter content values for plants were used: leafy vegetables – 10%; non-leafy vegetables – 7%; roots – 20% [74]. Where insufficient data were available, a geometric standard deviation was not calculated. Also, some of the geometric standard deviation values are large, reflecting sparse and/or variable data. It is debatable whether use of a log-normal distribution can be justified in these circumstances.

TABLE 27. RESULTS FROM THE DETERMINISTIC SIMULATION FOR THE CURRENT SCENARIO, ASSUMING THAT ^{226}Ra CONCENTRATION IN IRRIGATION WATER IS 0.45 Bq/L

Time, years	Ra-226 concentrations in crops, Bq/kg fresh mass			Annual effective dose to an adult, Sv
	Leafy vegetables	Vegetables	Roots	
0	0.70	0.47	0.47	1.6×10^{-5}
10	0.72	0.47	0.51	1.7×10^{-5}
20	0.75	0.47	0.54	1.8×10^{-5}
30	0.77	0.48	0.58	1.9×10^{-5}
40	0.80	0.48	0.62	2.1×10^{-5}
50	0.82	0.48	0.66	2.2×10^{-5}
60	0.85	0.49	0.70	2.3×10^{-5}
70	0.87	0.49	0.73	2.4×10^{-5}
80	0.90	0.49	0.77	2.5×10^{-5}
90	0.92	0.50	0.81	2.6×10^{-5}
100	0.95	0.50	0.85	2.7×10^{-5}

TABLE 28. RESULTS OF MONTE-CARLO SIMULATION FOR THE CURRENT SCENARIO

Parameter	Annual effective dose to an adult, Sv
Median dose	2.3×10^{-5}
5% percentile	1.9×10^{-5}
95% percentile	5.3×10^{-5}

Note: Annual effective dose at time t=50 years after starting irrigation.

Doses resulting from emanation of radon from the tailings and external exposure at the tailings

Radon emanating from the tailings layer through the cover layer will result in contamination of the air above the tailings and in surrounding areas. Radon decay products also appear in the atmosphere due to the radioactive decay of ^{222}Rn. In this calculation, the dose increment resulting from releases of ^{222}Rn originating in the tailing materials was considered. Contributions from the cover of the tailings were not considered.

Time-dependent radon fluxes at the top of the cover layer were calculated, and from this, the time evolution of ^{222}Rn, ^{210}Pb and ^{210}Po activity concentrations in the local atmosphere above the tailings and above the topsoil layer was predicted. These calculations were carried out using the 'Source' module of the SATURN library. in addition, the 'Atmospheric transport' module of the library was applied to calculate the ^{222}Rn activity concentrations in air at a village situated 1000 m from the source, as well as activity concentrations of ^{210}Pb and ^{210}Po in air above the topsoil layer resulting from radioactive decay of radon. These activity concentrations were used in the SATURN 'Building' and 'Land' modules to calculate doses from external exposure, inhalation, and soil ingestion. The calculations for this case were made to estimate the contribution to dose of the MCO 68 site only. In the calculation of external dose, the contribution from radionuclides in the tailings material was considered, accounting for attenuation by the tailings cover.

Using the SATURN 'Crop Land' model, activity concentrations in crops produced in a field located 400 m away from the tailings area were calculated. Activity concentrations in milk and meat from cows grazing in a field located 500 m from the repository were also calculated using the SATURN 'Pasture Land' module. Doses resulting from ingestion of contaminated crops, milk and meat were calculated.

Results of the calculations for the current scenario, at $t = 27$ years after closure of the tailings repository, are summarized in Table 29.

Based on the results, it was determined that the calculated values differed significantly from the monitoring measurements made at the site. One possible explanation for this difference is that measured values integrate the contributions of background radiation and other sources, whereas in this exercise, only the incremental dose from tailings has been evaluated, excluding potential contributions from radionuclides in the tailings cover or from other sources in the vicinity.

TABLE 29. RESULTS OF CALCULATIONS OF CONCENTRATIONS AND DOSE RATES BY THE ATMOSPHERIC PATHWAY FOR THE CURRENT SCENARIO (ELAPSED TIME = 27 YEARS)

Location	**Parameter, Units**	**Radionuclide**	**Calculated result**
Village	Concentrations in air outdoors, Bq/m^3	Rn-222	7.3×10^{-4}
		Pb-210	3.6×10^{-10}
		Po-210	1.0×10^{-14}
	Dose rate outdoors, Sv/h	Total	3.6×10^{-13}
	Concentrations in air indoors (above background), Bq/m^3	Rn-222	2.9×10^{-4}
		Pb-210	1.4×10^{-10}
		Po-210	4.1×10^{-15}
	Doses rate indoors, Sv/h	Total	1.4×10^{-13}
	Annual effective dose, Sv	External outdoors	2.1×10^{-5}
		External indoors	8.6×10^{-6}
		Inhalation outdoors	1.2×10^{-8}
		Inhalation indoors	4.9×10^{-9}
		Soil ingestion	1.6×10^{-8}
Crop land	Concentrations in air outdoors, Bq/m^3	Rn-222	3.6×10^{-3}
		Pb-210	1.8×10^{-9}
		Po-210	5.2×10^{-14}
	Dose rate outdoors, Sv/h	Total	1.8×10^{-12}
	Concentrations in crops*, Bq/kg fresh mass	Pb-210	9.8×10^{-8}
		Po-210	8.8×10^{-9}
	Annual effective dose, Sv	Ingestion of crops	1.4×10^{-12}
Pasture land	Concentrations in air outdoors, Bq/m^3	Rn-222	2.4×10^{-3}
		Pb-210	1.4×10^{-10}
		Po-210	4.1×10^{-15}
	Dose rate outdoors, Sv/h		2.0×10^{-12}
	Concentrations in milk, Bq/L	Pb-210	1.7×10^{-3}
		Po-210	1.3×10^{-4}
	Concentrations in meat, Bq/kg fresh mass	Pb-210	5.7×10^{-3}
		Po-210	2.0×10^{-3}
	Annual effective dose, Sv	Ingestion of milk	1.8×10^{-7}
		Ingestion of meat	2.2×10^{-7}
Overall	Total annual effective dose, Sv	External exposure	3.0×10^{-5}
		Inhalation	2.7×10^{-8}
		Food ingestion	4.0×10^{-7}
		Soil ingestion	1.6×10^{-8}
		All pathways	3.0×10^{-5}

*Conservative calculations of the activity concentrations for a generic crop were performed using the transfer factors given in Ref. [61].

Estimation of doses for the intrusion scenario

For this intrusion scenario, the following calculations were performed using SATURN:

- External dose and inhalation dose assuming occupancy on the tailings were calculated using measured data collected at the tailings. In these calculations, diffusion of ^{222}Rn from the tailings material through the basement of the house was not taken into account;
- The contribution from the tailings to the external dose and inhalation dose assuming occupancy on the tailings, as well as doses from ingestion of crops cultivated on the tailings were calculated for MCO 68. In these calculations, diffusion of ^{222}Rn from the tailings material through the basement of the house was taken into account;
- Doses from ingestion of crops irrigated with water from a well built directly on the tailings.

Doses from occupancy on the tailings calculated from data measurements

For this calculation, it was assumed that an adult spends 75% of the time indoors and 25% of the time outdoors at the tailings. It was also assumed that the house is built with uncontaminated construction materials, and provides a shielding factor of 0.3 for external exposure (which is the default parameter of SATURN) [155]. It was assumed that ^{222}Rn activity concentrations in indoor air are the same as outdoor activity concentrations (due to ventilation air exchange, for example, through doors and windows), and that the equilibrium factor for ^{222}Rn progeny is 0.4 (by comparison, outdoors on the tailings dump was assumed to be 0.2) [155]. In these calculations, the possible diffusion of ^{222}Rn from the tailings area through the basement of the house was not considered. Data on radionuclide levels measured at the site were used in the calculations. The results of the dose calculations are given in Table 30.

Dose contribution from tailings due to occupancy on the tailings and ingestion of crops cultivated on the tailings area

In this calculation, the contribution of tailings to external doses rates and ^{222}Rn activity concentrations in indoor and outdoor air were calculated using the ‘Source’ module of the SATURN library. The contribution from radionuclides in the tailings cover were disregarded, whereas the attenuation of external irradiation by the cover was included. In calculating the indoor ^{222}Rn activity concentration, diffusion of ^{222}Rn from the tailings through the tailings cover and the house basement were considered. The ventilation rate in the house was assumed to be 0.5 exchanges of air per hour. Calculations were also done for the case in which a vegetable garden has been established directly on the tailings. In this case, the contamination of vegetables comes solely from ^{222}Rn emanation from the tailings. Irrigation of vegetables is not considered. The results from these calculations are presented in Table 31.

TABLE 30. DOSES TO AN ADULT FROM OCCUPANCY OF THE TAILINGS CALCULATED FROM MEASURED DATA FOR THE INTRUSION SCENARIO

Parameter	**Units**	**MCO 68**	**MCO 105**
Dose rate	Sv/h	2.2×10^{-7}	2.9×10^{-7}
Annual effective dose from external exposure	Sv	4.6×10^{-4}	7.2×10^{-4}
Rn-222 concentration in air	Bq/m^3	135	470
Annual effective dose from Rn-222 inhalation	Sv	2.4×10^{-3}	8.7×10^{-3}
Total annual effective dose	Sv	2.8×10^{-3}	9.5×10^{-3}

TABLE 31. CONTRIBUTION FROM THE TAILING MATERIALS TO RADIONUCLIDE CONCENTRATIONS IN CROPS AND DOSES BY DIFFERENT PATHWAYS CALCULATED FOR THE INTRUSION SCENARIO USING THE SATURN MODULES

Location	Parameter, Units	Radionuclides/Exposure pathways	Calculated result
Tailings site	Concentrations in crops*, Bq/kg fresh mass	Pb-210	8.0×10^{-4}
		Po-210	7.9×10^{-5}
	Annual effective doses, Sv	Crops ingestion	1.2×10^{-8}
		Soil ingestion	3.1×10^{-7}
		External exposure indoors	2.6×10^{-7}
		External exposure outdoors	8.7×10^{-8}
		Inhalation indoors	3.6×10^{-5}
		Inhalation outdoors	4.3×10^{-7}

*Conservative calculations of activity concentrations for a generic crop were carried out using the transfer factors given in Ref. [61].
Note: It was assumed that the elapsed time from the closure of the tailings was 130 years.

Exposures from irrigation water from a well drilled directly on the tailings

In this calculation, it is assumed that a family builds a water well directly on the tailings and uses the well water directly for irrigation. Doses from ingestion of irrigated crops are calculated.

Radionuclide activity concentrations in groundwater were estimated using measured data for the activity concentrations of radionuclides in the tailings. The radionuclide partitioning between porewater and the tailings matrix was estimated in the model using generic K_d values, as described in Ref. [156]. In this model, it is assumed that the entire radionuclide inventory in the tailings layer is in a mobile (exchangeable) form. It is assumed that the activity of ^{234}U in tailings is in equilibrium with ^{238}U, and that activities of ^{226}Ra, ^{210}Po and ^{210}Pb in the tailings are in equilibrium with ^{230}Th. In the absence of site specific data, it is assumed that the bulk density of tailings is 2 kg dry mass/dm^3, and the porosity is 0.3.

The key parameter influencing calculated radionuclide activity concentrations in groundwater is the solid–liquid distribution coefficient (K_d), indicating sorption. No site specific K_d data are available for the Bellezane tailings site. For the element specific K_d values of tailings material, the values for mineral soils from Ref. [74] were used and are presented in Table 32. The lithological composition of soil was accounted for wherever possible. The tailings represent low-permeability, finely dispersed material. Therefore, the most appropriate K_d values are those corresponding to loam and clay material. Such data are available for Ra and Pb in Ref. [74].

TABLE 32. ELEMENT SPECIFIC K_d VALUES FOR MINERAL SOILS [74]

Element	Type of soil	Geometric mean K_d (L/kg dry mass)	Geometric standard deviation	Min K_d (L/kg dry mass)	Max K_d (L/kg dry mass)
U	Mineral	180	13	0.7	67000
Ra	Clay	38000	12	700	950000
Th	Mineral	2600	10	35	250000
Pb	Loam+clay	13000	3.6	3600	130000
Po	Mineral	190	5.1	12	7000

Note: The highest K_d values for Ra may reflect co-precipitation, rather than reversible sorption. Therefore, the geometric mean and geometric standard deviation for Ra listed may be over-estimated.

Two types of modelling analysis were carried out to predict radionuclide activity concentrations in crops:

(1) Deterministic simulations using best estimate values for the K_d, soil-plant transfer factors and other model parameters. The geometric mean of K_d and transfer factor values reported in Ref. [74] were used as best estimate values for this modelling case.
(2) Probabilistic simulation, accounting for uncertainty in K_d, transfer factor values and some other parameters. In particular, the stochastic case assumed that K_d and transfer factor values are log-normally distributed, based on statistical parameters reported in Ref. [74]. The element- specific K_d values used in the simulations are reproduced in Table 32.

In the probabilistic simulations, it was assumed that K_d values for Ra, Th and Pb are correlated random variables, with a correlation coefficient of $R = 0.5$ (i.e. the K_d values of these radionuclides tend to jointly increase or decrease). This is because the sorption behaviour of these radionuclides is governed by the ion-exchange process, and therefore, K_d is typically correlated with the cation-exchange capacity of soil. Uranium and polonium have more complex sorption behaviour, and no assumptions were made regarding the correlation between the K_d of these radionuclides with other K_d values. The K_d and transfer factor values were treated as uncorrelated. However, in practice, they are likely to be anti-correlated because radionuclides bound to soil solids are less likely to be available for plant uptake than radionuclides present in soil solution.

The fraction of the activity in irrigation water after interception by the crops was assumed to range from 0.2–0.4 (uniform distribution; in the deterministic simulation, a value of 0.3 was assumed).

Deterministic results

Estimated radionuclide activity concentrations in groundwater are provided in Table 33. These estimates are in good agreement with the monitoring data (Table 34) for the Bellezane site in 2008 [157].

Three types of crops were assumed to comprise the diet of an exposed adult: 'leafy vegetables', 'root vegetables' and other 'vegetables' [61]. Results of calculations of doses from consumption of vegetables irrigated with contaminated groundwater at MCO 68 and MCO 105 are presented in Table 35. The time over which modelling was performed was $t = 100$ years. Doses consider contributions from all radionuclides of the ^{238}U decay series.

TABLE 33. ESTIMATED RADIONUCLIDE ACTIVITY CONCENTRATIONS (Bq/L) IN WATER FROM A WELL CONSTRUCTED DIRECTLY ON THE TAILINGS

Radionuclide	MCO 68	MCO 105
U-238	8.9	27.8
U-234	8.9	27.8
Ra-226	0.84	0.37
Th-230	12.3	5.4
Pb-210	2.5	1.1
Po-210	168	73.6

TABLE 34. RADIONUCLIDE ACTIVITY CONCENTRATIONS IN POREWATER OF WATER-SATURATED TAILINGS, BASED ON MONITORING DATA FROM 2008 [157]

Radionuclide	Piezometer ES 90 (MCO 68)	Piezometers ES 85, 89 (MCO 105)
Uranium total (soluble)		
µg/L	4670–5900	2380–4460
Bq/L	56.4–70.8	28.8–54
Ra-226 (Bq/L)		
Soluble	0.72–0.84	0.36–0.95
Insoluble	15–39	8–21

TABLE 35. RESULTS OF DETERMINISTIC CALCULATIONS OF ANNUAL EFFECTIVE DOSES FROM INGESTION OF IRRIGATED CROPS FOR THE INTRUSION SCENARIO

Time, years	MCO 68 Annual effective dose for an adult (Sv)	MCO 105 Annual effective dose for an adult (Sv)
0	1.32×10^{-3}	6.41×10^{-4}
10	1.32×10^{-3}	6.43×10^{-4}
20	1.32×10^{-3}	6.44×10^{-4}
30	1.32×10^{-3}	6.45×10^{-4}
40	1.32×10^{-3}	6.45×10^{-4}
50	1.32×10^{-3}	6.46×10^{-4}
60	1.32×10^{-3}	6.47×10^{-4}
70	1.32×10^{-3}	6.48×10^{-4}
80	1.32×10^{-3}	6.49×10^{-4}
90	1.32×10^{-3}	6.50×10^{-4}
100	1.33×10^{-3}	$6.51\mathrm{E}\times10^{-4}$

Probabilistic results

Results of probabilistic simulations for MCO 68 and MCO 105 are presented in Table 36. The predicted confidence intervals for doses due to consumption of contaminated crops range over almost two orders of magnitude. An important point is that the 95th percentiles are significantly higher than the best estimates from the deterministic simulation. That said, the median values of doses are comparable to the best estimates from the deterministic simulation. An example frequency histogram of dose values for the intrusion scenario (for MCO 68 only) is shown in Fig. 59.

TABLE 36. RESULTS OF PROBABILISTIC MONTE-CARLO SIMULATION FOR THE INTRUSION SCENARIO

Parameter [a]	MCO 68	MCO 105
Median annual effective dose (Sv)	2.0×10^{-3}	1.2×10^{-3}
5% percentile (Sv)	3.1×10^{-4}	2.0×10^{-4}
95% percentile (Sv)	1.1×10^{-2}	7.4×10^{-3}

[a] Median values and confidence intervals of doses to an adult relating to consumption of crops irrigated using contaminated groundwater (for t = 50 years from the start of irrigation).

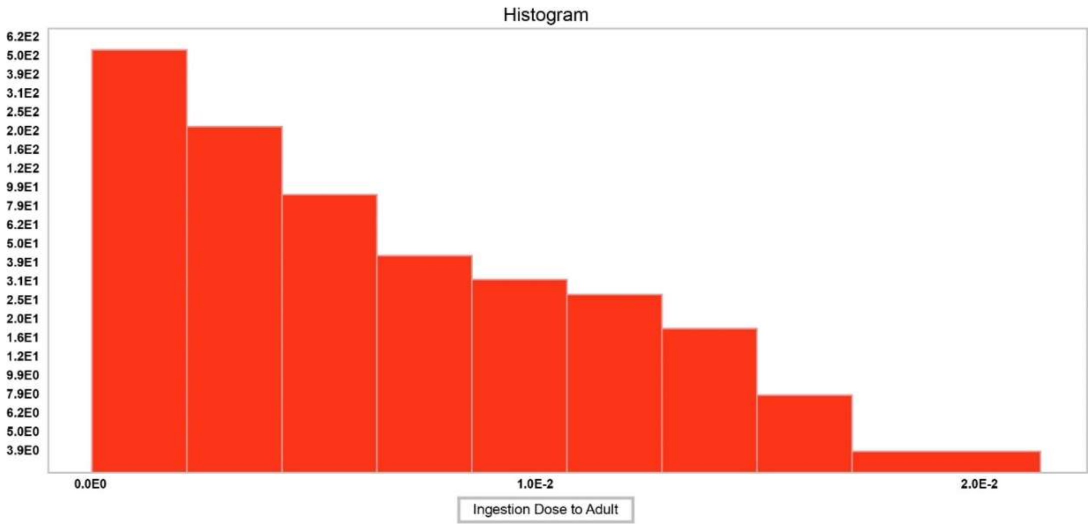


FIG. 59. Frequency histogram for doses to an adult due to ingestion of irrigated crops for the intrusion scenario. Values for MCO 68 (t=50 years).

Sensitivity analysis

A sensitivity analysis was carried out based on the outcomes of the probabilistic simulation for the intrusion scenario, and results are presented in a Tornado chart[50] (see Fig. 60). The sensitivity analysis indicated that the most sensitive model parameters are the K_d values for the tailings material, which govern radionuclide activity concentrations in irrigation well water. The K_d value of polonium has the highest contribution to the uncertainties of the predicted doses. In the case that was analysed, polonium has relatively low K_d values, and consequently, high polonium activity concentrations in the irrigation water have been estimated. Another important model parameter is the fraction of the activity in irrigation water intercepted by the crops. This parameter has a strong influence on the predicted polonium activity concentrations in irrigated vegetables.

7.2.6. Summary of the Bellezane modelling exercise

For the current scenario, conclusions, which are qualitatively similar, have been drawn, based on both models: a trivial annual effective dose is anticipated for a timeframe of up to 1000 years. For the intrusion scenario, a meaningful comparison between results generated using RESRAD-OFFSITE and the SATURN is difficult due to the different modelling assumptions and approaches between the two models. For example, in the RESRAD-OFFSITE calculations, Pits 68 and 105 containing tailings from the Bellezane repository were replaced by a single average value for the source term, whereas in the calculations that were performed using SATURN, the contributions from each pit were calculated separately.

The SATURN calculations focused on a few specific exposure pathways, neglecting, for example, the drinking water pathway, whereas the RESRAD-OFFSITE calculations included all exposure pathways. The latter also included the contribution of the radionuclide content of the waste rock cover, which is predicted to dominate the total dose in the intrusion scenario for the first 500 years.

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[50] A Tornado chart enables comparison of the influence of the input data on the results. For each input parameter, the length of bar indicates the magnitude of the change. The most influential input data are displayed on the top of the chart, followed in decreasing order by those which are less and less influential; this is why this kind of chart is often shaped like a tornado.

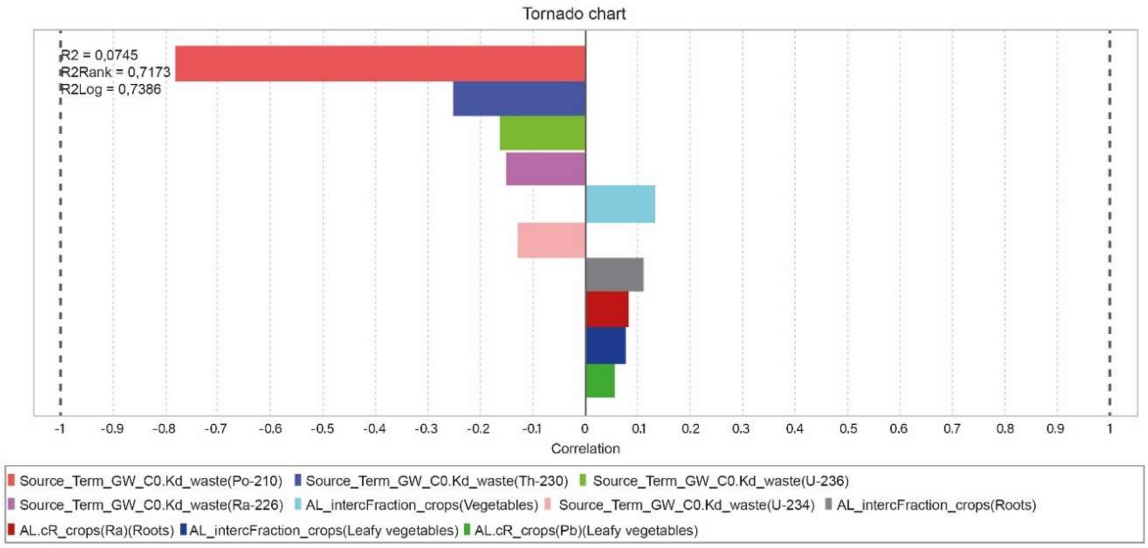

FIG. 60. Results of the sensitivity analysis for the intrusion scenario (only for MCO 105). Partial rank correlation coefficient values for model parameters are shown.

Despite these considerations, the quantitative results for the intrusion scenario for the dose from ingestion of plants was estimated to be similar for both models. At *t* = 100 years, the annual effective doses from this exposure pathway were estimated to be 1.18 mSv, based on the RESRAD-OFFSITE calculations, and 1.33 mSv, based on the SATURN calculations for Pit 68 and 0.65 mSv for Pit 105.

Results of the SATURN calculations included a probabilistic simulation, which is a key element in a decision making process, given the large number of uncertainties for the various parameters. Such a probabilistic simulation was not performed using RESRAD-OFFSITE.

The Bellezane exercise indicates how two independent groups of modellers can make different assumptions about the same complex reality. A more meaningful comparison of the two models used, SATURN and RESRAD, would need greater harmonization between assumptions and parameter values assumed by modellers, for example in terms of the timeframe, exposure pathways, layout of the site, and transfer parameters. It was noted that predictions over such a large timescale are purely academic; the hypotheses of the scenario might no longer be valid in the distant future due to societal and landscape changes that are likely to occur over the timeframe considered.

8. SUMMARY

Given the many sites affected by past activities existing around the world, a variety of tools and approaches are needed when performing REIAs for protection of people and the environment. The challenges faced by assessors and decision makers are diverse, as are the national legal and regulatory requirements in this field.

A broad range of freely available and commercially available models (27 in total) was considered by EMRAS II WG2 and are described in this publication. These models vary from simple screening tools to more sophisticated modelling codes for which specific training and expertise is needed in support of their use. Despite this broad range of available models, the user still needs to be aware of the limitations of the model they choose to apply. None of the models mentioned in this publication has all the applicabilities needed to represent the historic NORM and historic nuclear sites presented within this publication. Therefore, the assessor needs to be clear in terms of which outputs are necessary to generate through modelling in support of the selection of appropriate models for use. The assessments presented in this publication demonstrate the practical use of some of the models in the evaluation of actual sites.

The general assessment methodology (Section 4) presented in this publication is consistent with recommendations provided in GSG-15 [1], while providing more detailed information on assessment tools and their application to evaluate impacts and risks in support of remediation planning and implementation. The methodology is applicable to sites contaminated due to past activities (historic sites), accidental releases of radioactive material or routine operations. The general assessment methodology proved useful for several different sites with varying characteristics and challenges. This implies that the methodology is generally applicable for sites affected by past activities around the world. A graded approach (from screening to detailed assessment) is useful in optimizing the assessment process, for example from the perspective of regulatory compliance, radiological protection, and cost. The graded approach described is a method of optimizing an assessment process to support remediation, so that the regulatory oversight and effort is commensurate with risk which allows the assessors and decision maker(s) to stop at the point where it becomes clear that further work will not change the outcome of the assessment, for example.

The model–model intercomparison of the Gela phosphogypsum site in Italy involved assessments of different uses of the site in the future, based on the existing information. For screening purposes, a rural residential scenario was considered, where all the food ingested by the representative person was assumed to be grown directly on the waste stack. No remedial options were considered, therefore, representing a worst-case scenario. The models generated differing results for the screening assessment calculations, ranging from approximately the highly conservative screening criterion of 0.3 mSv that had been adopted for the site, for an exposed individual at the site, to up to 30 times this value. The model that predicted the lowest doses, however, did not take into consideration all the relevant pathways, which illustrates the importance of knowing the possibilities and limitations of each model. Fewer models were available for the intermediate and detailed assessments. The basis for these assessments was the reported remediation that was carried out, which included the installation of a plastic liner and several metres of clean soil covering the stack. Several options were considered, with evaluations that included determinations of the effect of the plastic liner on radon emanation and the depth of clean soil needed for the radiological protection of individuals. The assessment results were comparable between the models used: with the retaining wall, plastic liner and soil cover in place, the site might be suitable for recreational or industrial use, but not for residential use in the future.

A model–model intercomparison was also performed for the Bellezane site in France where two former open uranium pit mines (MCO 68 and MCO 105), which are separated by a dike, have been remediated. The site has restricted access and is fenced. Two scenarios of exposure were considered. The 'current' scenario involves exposure of nearby villagers whose diet comes partially from local produce and where groundwater is used to irrigate agricultural fields where the local crops are grown. In the intrusion scenario, it is assumed that in 250 years from now, there is no longer institutional control of the site and the knowledge about the former tailings is forgotten. A family builds a house on the site, which now looks like a green field. Once settled, they work at home, and they have a garden in which they grow some vegetables. These vegetables are irrigated in summer with water from a well pumping directly from the groundwater. Two modelling codes were used: RESRAD-OFFSITE and SATURN.

For the 'current' scenario, conclusions were drawn that were qualitatively similar for both models: a trivial annual effective dose for a timeframe of 1000 years. For the intrusion scenario, however, different modelling assumptions and approaches were used as input into RESRAD-OFFSITE and SATURN. For example, in the RESRAD-OFFSITE calculations, the two pits were represented by an average single source term, whereas using SATURN, the contributions of each pit to radionuclide releases were calculated separately. The SATURN calculations focused on a few specific exposure pathways, whereas RESRAD-OFFSITE considered all relevant pathways. Despite these and other differing assumptions, the quantitative results for the intrusion scenario regarding the annual effective dose from ingestion of plants were similar for both models. At $t = 100$ years, the annual effective dose of 1.18 mSv was estimated in the RESRAD-OFFSITE calculations, whereas the SATURN calculations produced estimates of 1.33 mSv and 0.65 mSv, respectively, for the two pits. Due to different assumptions and approaches, the exercise indicated how two independent groups of modellers could make different assumptions about the same complex reality.

9. FURTHER WORK

The work of IAEA international model validation programmes, such as EMRAS, and discussions with Member States relating to the need for international guidance on how to address existing exposure situations have highlighted the importance of assessing potential impacts of facilities and activities as part of responsible planning to reduce the possibility of creating future existing exposure situations (e.g. to avoid future legacy sites).

To accomplish this, it is necessary to:

- Develop abilities to undertake probabilistic calculations;
- Develop technically robust methods of uncertainty estimation for model predictions and estimates;
- Investigate how to harmonize approaches and models for hazardous substances and radioactive material (using a risk based approach, as opposed to a dose based approach);
- Run models for different remedial actions for actual sites;
- Foster cross-fertilization between groups through an exchange of knowledge and experience, e.g. between experts, institutions, and countries.

A comprehensive survey of internationally accepted practices for assessing and remediating (where appropriate) contaminated sites would be useful for both assessors and regulatory authorities and would be useful in terms of developing a harmonized international approach for the management of such sites.

IAEA's international model validation and data compilation programmes facilitate such outcomes.

APPENDIX I.
SITES TO ILLUSTRATE TYPICAL ISSUES AND FOR POSSIBLE MODELLING EXERCISES

Several historic NORM (e.g. uranium) and historic nuclear sites that illustrate the complex range of issues that arise when dealing with such sites, and which could be used as actual scenarios for a modelling exercise, were presented and discussed in the EMRAS II NORM and Legacy Sites Working Group [58]. These sites cover a number of different countries on different continents, which illustrates the ubiquity of the issue. These sites may be sub-divided into three main categories:

- Historic nuclear sites, e.g. former nuclear submarine maintenance sites;
- Historic uranium mining and milling sites;
- Historic NORM sites (other than those related to uranium mining and milling), e.g. phosphogypsum stacks.

Table 1 in Section 2 provides an overview of the sites that have been considered. Further details for each of the sites are provided in Appendix II (historic nuclear sites), Appendix III (historic uranium milling and mining sites) and Appendix IV (historic NORM sites). For most of the sites, data may be structured as follows:

(1) **Description of the source terms:** Volumes of contaminated materials, site area, relevant radionuclides, ranges of activity concentrations, and other information, as relevant;
(2) **Description of the parameters related to environmental transport of radionuclides:** Hydrogeology, climate, transfer factors, concentration ratios, and other parameters, as relevant;
(3) **Monitoring data:** Dose rates on site, radionuclide activity concentrations in groundwaters and surface waters and/or in percolate, radon emanation measurements; activity concentrations in soils, plants and animals, and other monitoring data, as relevant.
(4) **Remediation:** For sites which are already being remediated, the actions detailed in the approved remediation plan and descriptions of the remedial actions and other protective actions to evaluate their effectiveness.

Historic sites may or may not be in operation any longer, which means that the modelling process may not only be aimed at remediation of existing contamination, but also at prevention of future contamination. Moreover, for NORM facilities in operation, the REIA does not only focus on the possible impacts of the site on the public and the environment, but also on the operational radiation protection of workers on the site.

For some of the sites considered by EMRAS II WG2, a dose assessment had already been performed based on well-defined scenarios of exposure. These scenarios are not limited to the assessment of the current radiological impact, but may address future impacts, taking into account the physical evolution of the site (e.g. migration of the nuclides, ingrowth of some nuclides due to radioactive decay) and the evolution in the use of the site (like construction of dwellings on site, a typical intrusion scenario).

For several of these sites, an intrusion scenario is not only a theoretical possibility but an actual occurrence.

APPENDIX II.
HISTORIC NUCLEAR SITES IN THE RUSSIAN FEDERATION

The sites described in this Appendix are historic nuclear sites in the Russian Federation that were affected by past activities, as follows:

- Lepse nuclear vessel;
- Andreeva Bay.

A general description of the site, the source term, site specific data in support of modelling and information relating to the REIA are provided for each. In the case of Andreeva Bay, information on progress made on remediation is also provided.

II.1. THE TECHNICAL SUPPORT VESSEL LEPSE

II.1.1. General description

The Lepse, originally built as a cargo carrier more than 70 years ago, was rebuilt as a service ship in 1958, known as a 'floating technical base'. The vessel was used for refueling the first nuclear-powered icebreaker. It was also used for the next generation of icebreakers, known as Arktika and Sibir. It was taken out of service in 1988 and was officially declared as a 'laid-up' vessel in 1990. At this point, the storage compartments still contained SNF and radioactive waste in solid and liquid forms.

The decommissioning of Lepse was identified as an important task and work was initiated in 1989 by the Murmansk Shipping Company, but was then terminated in 1994 when funds ran out. Work continued, partially through international support[51] and in June 2021 the final stage of securing the nuclear material from the Lepse was completed[52].

II.1.2. Source term

Figures 61 and 62 depict the vessel 'Lepse', and the waste storage locations on the vessel. The vessel Lepse was located in Kola Bay near Murmansk, Russian Federation. The vessel contained two storage tanks for SNF from the icebreakers, with 639 fuel assemblies on board, many of which had damaged fuel elements. Liquid and solid radioactive waste were stored on board.

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[51] http://ndep.org/projects/decommissioning-of-the-lepse-ftb/
[52] https://www.highnorthnews.com/en/after-27-years-lepse-no-longer-poses-nuclear-threat-arctic

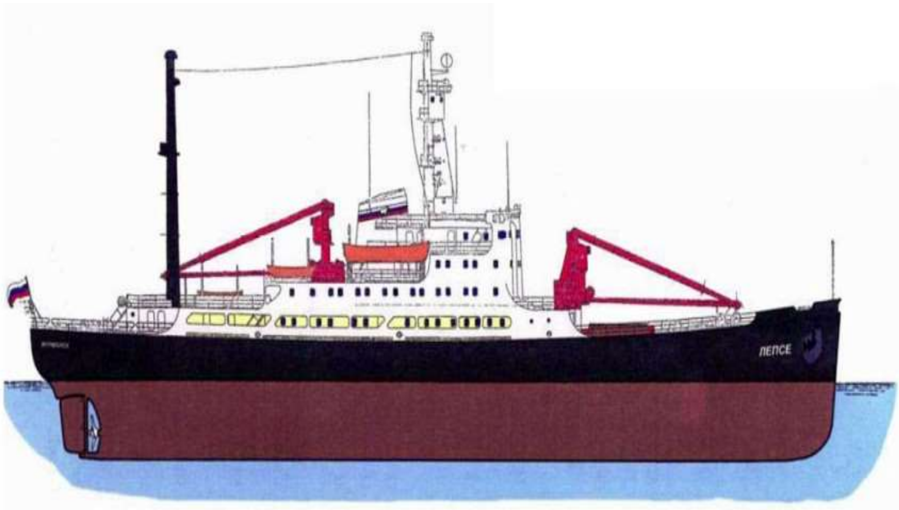

FIG. 61. The vessel 'Lepse'.

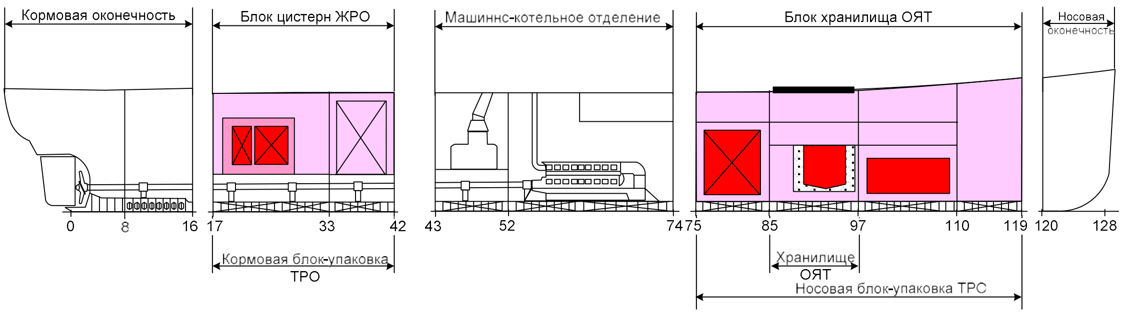

FIG. 62. The spent nuclear fuel (in red uncrossed) and solid and liquid radioactive waste (crossed) storages.

II.1.3. Radiological environmental impact assessment

The activity of the SNF in the storage hold was approximately 2.5×10^4 TBq, which was considered a substantial amount. The poor condition of a significant fraction of the fuel elements suggested that early action to avoid the consequences of an accident, including the possibility of criticality or further degradation of materials, had not been effective, which made decommissioning even more difficult. At the same time, the risks associated with decommissioning activities had been considered to avoid increased risks, at least in the short term, during hazardous recovery operations. At the time of decommissioning, although the radioactive waste inventory was much smaller than that of the SNF, contamination in various areas hampered decommissioning work.

The abnormal circumstances and condition of the vessel infrastructure and the condition of the waste stores and the radioactive material led to specific regulatory requirements [158]. Development of these requirements involved close cooperation between the operating organizations and the regulatory body, together with international inputs [159]. The guidance that was developed provided a framework for continued regulatory supervision and included [160] a quality management system and requirements for the:

- Composition of a set and contents of documents substantiating nuclear and radiation safety;
- Quality assurance programme of activities for unloading SNF assemblies;
- Nuclear and radiation safety analysis report.

Assessments, based on these requirements, included those for worker protection, analysis of potential for and consequences of accidents, and transport safety analyses.

II.1.4. Safety challenges and lessons learned

An initial major issue with planning the decommissioning of the vessel Lepse was the lack of information about the condition of the stored SNF. It was clear that early engineering work to investigate the circumstances needed to proceed very carefully and slowly, to provide at least the minimum information necessary for developing an adequate decommissioning plan.

Particularly important in this case was whether SNF assemblies could be removed from current locations without risk of breaking. Waste characterization was necessary so that appropriate handling operations and treatment processes could be developed, as well as waste acceptance criteria for long-term storage and eventual disposal.

Some of the damage to fuel might have occurred prior to its storage in the Lepse. Actions taken in the management of wastes following accidents needed to be appropriately recorded.

The radiation conditions in potentially relevant work areas were challenging, both in terms of high radiation dose rates and high contamination levels. Evaluation of remedial options to mitigate these conditions involved assessment not only of the measures taken to ensure worker protection, but also in terms of secondary waste production and management.

Abnormal working conditions, common for sites affected by past activities (existing exposure situations), necessitate the judicious application of radiation protection planning, as well as occupational health and safety in a broader sense. Delaying, or taking no action might avoid limited but controlled radiological impacts, such as from the release of secondary wastes or the relatively high exposure of workers to radiation over a limited period of time. However, such delay or inaction might result in the continued existence of a major hazard. In this case, there was risk of major release of radioactive material to the environment. The same risk balance could relate to taking action to make an unstable dump of NORM waste or mine tailings physically safe, for example. The solution to such problems is not likely to be easy, but an integrated consideration of the impacts of radiation, other pollutants, and physical hazards is typically needed in complex situations such as this.

II.2. SITE FOR TEMPORARY STORAGE OF SPENT NUCLEAR FUEL AND RADIOACTIVE WASTE, ANDREEVA BAY

II.2.1. General description

From the 1960s, submarines were maintained at Gremikha village on the coast of the Barents Sea and in Andreeva Bay, Kola Peninsula (northwest Russia). Activities included the receipt and storage of radioactive waste and SNF. No waste has been received by the technical bases since 1985 and they have been recategorized as sites of temporary storage.

Characteristics of the site at Andreeva Bay are as follows:

- Unsatisfactory condition of facilities, which hampers the safe management of SNF and radioactive waste;

- Dispersion of radioactive contamination from the temporary storage site to the marine environment;
- Lack of regulatory requirements and guidance for the abnormal radiation conditions;
- Lack of relevant standards for radioactive waste management.

Management challenges were increased as a result of the following factors:

- Damage to the SNF and the engineered barriers of the storage facilities, which led to radioactive contamination of the environment and risk of further releases;
- Gaps in regulations on procedures concerning radioactive waste and SNF management, including insufficient definition of requirements for remediation;
- Public concern that significant environmental impacts may occur in the Kola Peninsula, the European part of the Russian Federation, and other parts of northern Europe.

The Russian strategy to address the situation draws upon a broad range of industrial projects which receive support from donor organizations and technical institutions, coordinated through the IAEA's Contact Expert Group. The Norwegian Radiation Protection Authority's bilateral regulatory cooperation programme was designed to support Russian regulatory authorities to ensure that investments made to manage these historic nuclear sites were spent within the context of an effective regulatory regime.

II.2.2. Source term and site specific data

The temporary storage site in Andreeva Bay consists of the following main facilities [161]:

- Fixed-site technological berth;
- Three partly underground 1000 m^3 blocks of dry storage, re-equipped for SNF storage;
- Service site for the SNF storage;
- Basin-type SNF storage (Building 5 to be decommissioned following removal of SNF);
- Liquid radioactive waste storage;
- Intended water purification building;
- Facility for post-treatment storage of high-level concentrates of liquid radioactive waste;
- Numerous constructions and sites for solid radioactive waste storage.

Andreeva Bay was divided into four separate radiation protection areas for radiation protection of workers (see Fig. 63; see also Ref. [162]):

- **Controlled Access Area (CAA):** Radioactive waste and SNF were stored in facilities in this area and hazardous operations involving radiation exposure were carried out here [163]. These facilities were the main objects of remediation. Decontamination signs demarcated this area and specific procedures were established for work in this area. Personal protection equipment was required in the CAA for radiation protection of personnel designated as 'Group A' workers (annual dose limit = 20 mSv).
- **Uncontrolled (Free Access) Area (UA):** Facilities were located in this area, which support implementation of work in the controlled access area. No hazardous operations involving radiation exposure were performed within this area. 'Group B' workers (annual dose limit = 5 mSv) mainly worked here.

— **Health Protection Zone (HPZ):** This area was used for administrative and technical services. The border of this area was delimited by physical protection of the engineered area.
— **Supervision Area (SA):** This area covered a radius of approximately 10 km and served as an area in which the impact of the facility on the public and the environment were monitored (annual dose limit = 1 mSv).

The total radioactive inventory amounted to about 10^5 TBq, primarily in SNF, but also significant amounts of solid and liquid radioactive waste, as well as contaminated soil and sub-soil [161].

A range of environmental sampling was undertaken to support the planning of remediation and long-term management of the site. Figure 64 shows the ^{90}Sr and ^{137}Cs contamination ranges in designated areas, as reported in Ref. [164], in which a description is given of the site specific regulatory controls and requirements that covered:

— Optimization of radiation protection of workers;
— Site monitoring and control criteria;
— Management of very low level waste;
— Emergency preparedness and response.

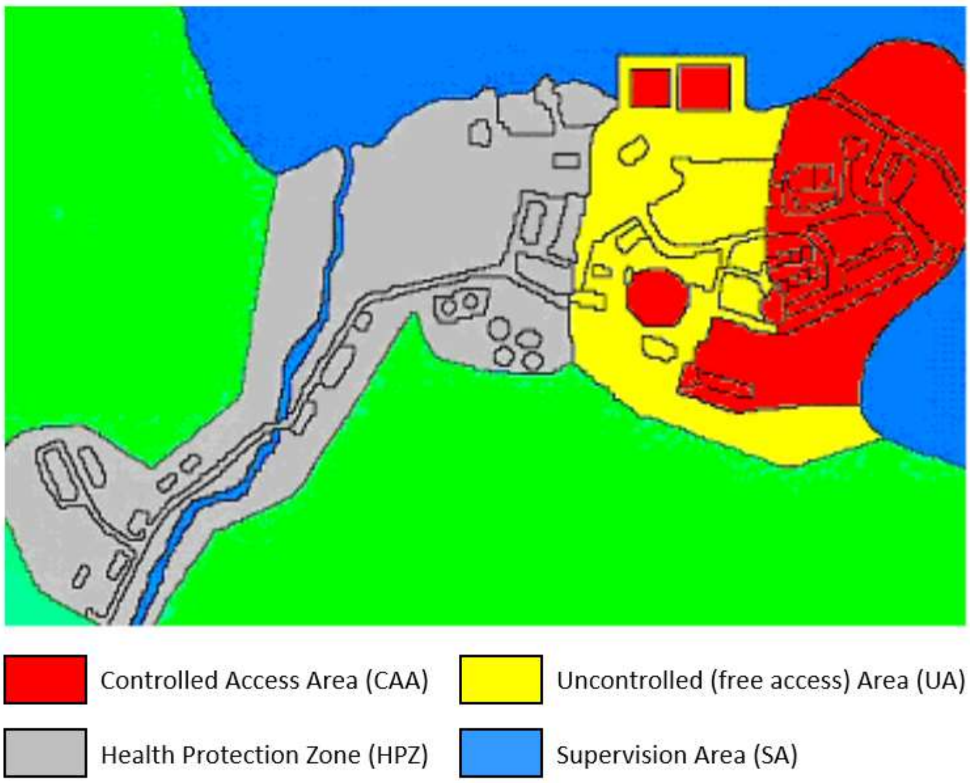

FIG. 63. Area categorization at Andreeva Bay. Reproduced with permission by the IOP from Ref. [161].

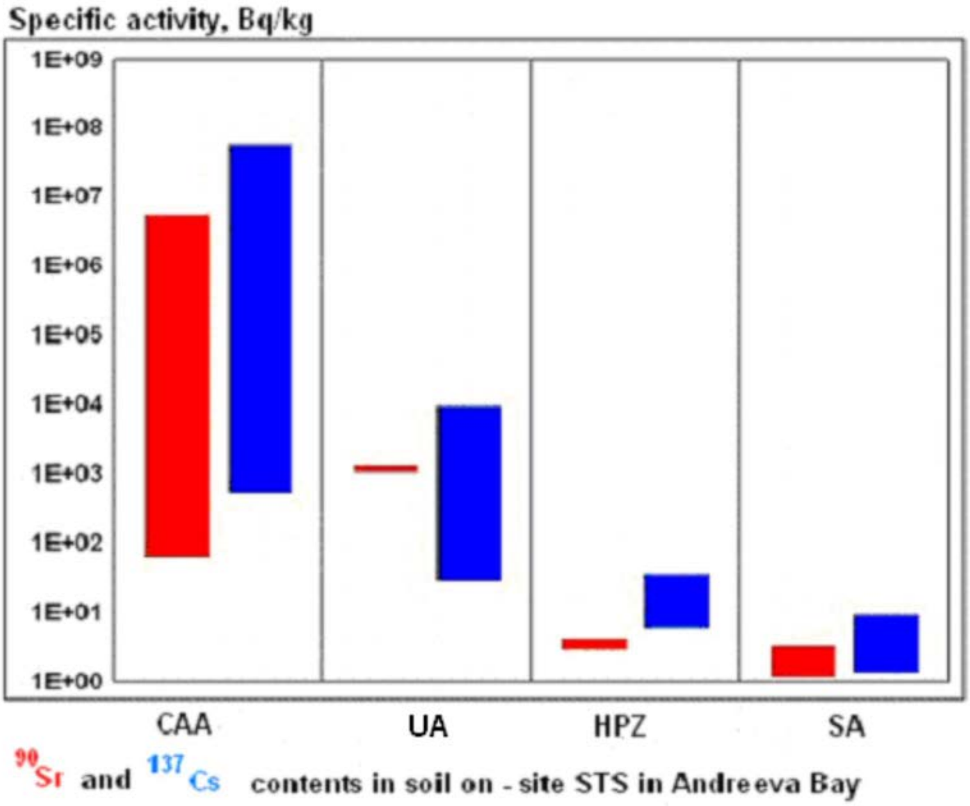

FIG. 64. ^{90}Sr and ^{137}Cs contents in soil within categorized areas at the Andreeva Bay temporary storage site [164]. Reproduced with permission by the IOP from Ref. [161].

II.2.3. Radiological environmental impact assessment

Prognostic REIA was important in remediation planning. It relies on sufficient understanding of the current situation with respect to existing facilities, radioactive sources and the nature of the environment. The assessment of future conditions, for example, to estimate dose rates at workplaces once a number of sources have been removed, supports the identification of appropriate remedial actions, corresponding management actions, and effective regulatory supervision of the site and its associated facilities.

Two assessment tools were developed, one for monitoring of radiation conditions and individual exposure (DOSEMAP) and another for radioecological assessment (DATAMAP). Both tools are supported by geographic information systems and are described in detail in Ref. [165].

Figure 65 provides an example of output from DOSEMAP, showing how to represent the industrial site and isolines of ambient dose rate. The example serves only to illustrate potential graphic representation of measured data, which can be generated from results of relevant measurements and put into the assessment tool database. Such techniques can be used to assess the current situation and the effect of removing specific source terms, and hence supports optimization of protection and safety of workers.

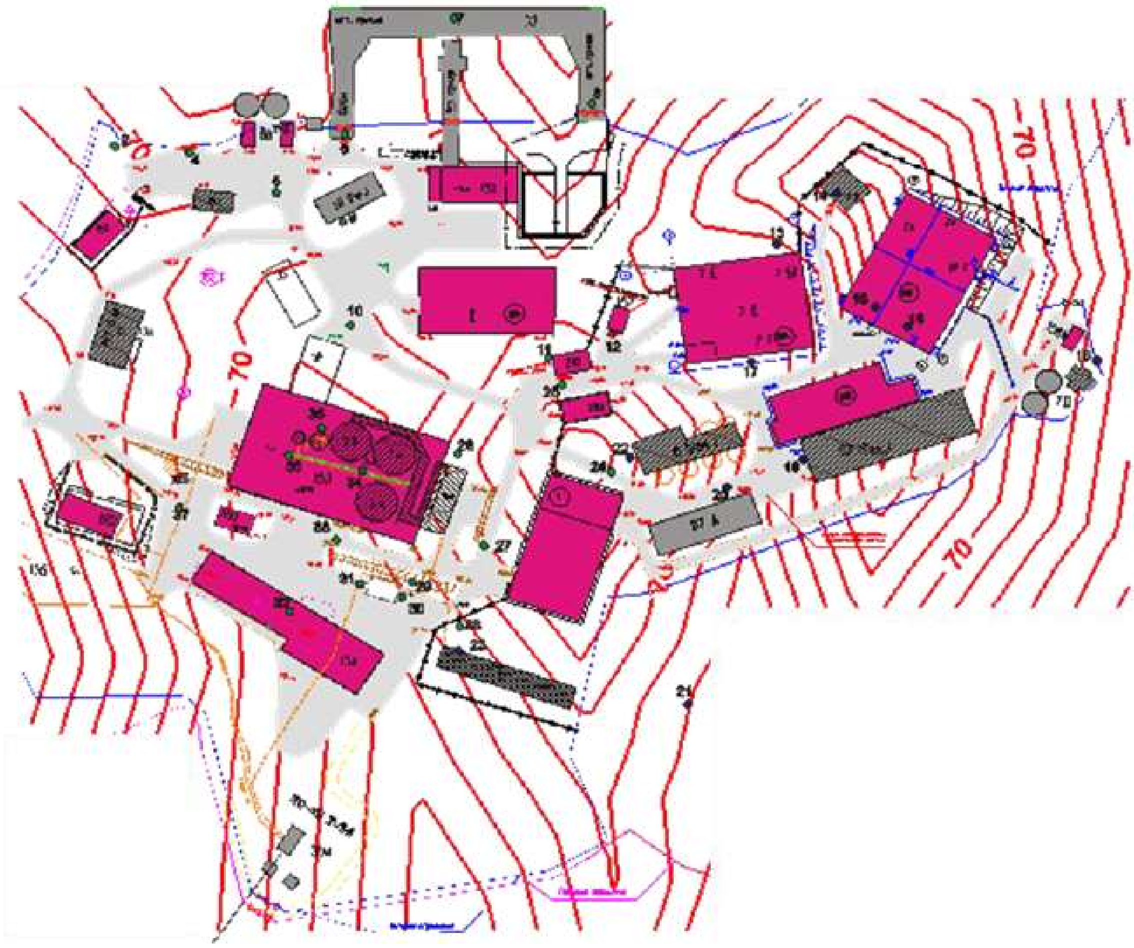

FIG. 65. Andreeva Bay facilities with dose curves as presented in DOSEMAP.

The DOSEMAP assessment tool focuses on environmental contamination management. It relies on measurements of environmental radioactivity in soils, in sub-soils, and in groundwater, making use of interpolation techniques and provides the following:

— 2D and 3D presentations of contamination;
— Identification of areas where available data are insufficient for analysis of the radioecological situation;
— Application of models for projection of future contamination levels, accounting for radioactive decay, migration of contaminants and accumulation of radionuclides.

II.2.4. Progress with remediation

Since 2002, there has been substantial construction and reconstruction of infrastructure. Table 37 summarizes the changes in the radiation situation in some of the more highly contaminated areas.

Remedial actions are still ongoing, resulting in lower dose rate working areas. Overall progress is described in Ref. [165], along with the need for continuing actions and supervision from a regulatory perspective.

TABLE 37. CHANGES IN DOSE RATE WITH TIME

Location	Dose rate, μSv/h		Action taken
	2002	2009	
Area near new pier	0.15–450	0.15–0.35	Old pier dismantled
Around building 50	0.3–1.5	0.25–0.57	Removal of scrap metal landfill
Various damaged buildings	0.58–2.7	0.38–1.1	Sand backfilling and asphalt covering
Motor transport decontamination area	2.5–30.7	0.57–0.7	Paving of the site

APPENDIX III.
HISTORIC URANIUM MINING AND MILLING SITES

Depending on the national regulations, uranium mining and milling sites can be defined either as nuclear sites or as NORM sites. NORM sites can include those relating to nuclear fuel cycle facilities, such as uranium mines and mills (covered in this Appendix) or other types of NORM sites (see Appendix IV).

The uranium mining and milling sites described in this Appendix are the following:

- Los Gigantes site in Argentina;
- Poços de Caldas site in Brazil;
- Buhovo site in Bulgaria;
- Žirovski vrh area in Slovenia.

A general description of the site, the source term, site specific data in support of modelling and information relating to the REIA are provided for each site.

III.1. MINERAL TAILINGS OF A FORMER URANIUM MANUFACTURING MINING FACILITY: LOS GIGANTES – ARGENTINA

III.1.1. General description

The Los Gigantes Manufacturing Mining Facilities are located in Sierra Grande, Province of Córdoba, Argentina, 30 km away from the city of Villa Carlos Paz. The complex is part of the provincial Water Reserve of Achala. In 1970, prospecting studies were developed, assigning the rights for the exploitation of uranium to a private company. These studies were then carried out from 1979 to 1990 on the site.

Through the industrial activities at the Los Gigantes site, 2 500 000 tons of ore (0.15‰ U_3O_8; 0.239‰ U_3O_8 mean ore, 0.123‰ U_3O_8 for the marginal ore) were processed and 206 tons of concentrated U was produced, generating 1 000 000 tons of sterile quarry, 600 000 tons of marginal mineral, 2 400 000 tons of ore tailings, 100 000 m^3 of precipitation mud, and 100 000 m^3 of liquid effluents.

Hot spots up to ~1.2 MBq/kg of ^{226}Ra are present in the mineral tailings and a remediation project is currently being carried out to remediate (isolate) them.

III.1.2. Source term

Layout of the site

The mineral tailings are located over an 800 m section of the northern riverbed of Arroyo de la Mina. Due to rainfall effects and springs associated with subsurface waters, the tailings lixiviate and release acidic solutions ($3.5 < pH < 6$), with high contents of dissolved salts ($0.23 <$ U (μg/L) < 500; $0.0148 <$ ^{226}Ra (Bq/L) < 1.48; $2.1 <$ SO_4^{2-} (mg/L) < 2800) to surface water courses. These lixiviates represent the major contribution of contaminants to watercourses in the area, resulting in decreasing water quality with respect to the protection of aquatic life (see Fig. 66).

The tailings piles have a slope (talus angle) of 35°, with a mean altitude of 70 m and are subject to erosion by rain.

Some characteristics of the tailings are as follows:

- Area: ~10 ha;
- Mass of waste: 2 400 000 t;
- Mean Optimum Humidity[53] = 12%;
- Mean Maximum dry density = 1.8 ton/m^3.

The mean width of the waste layer is approximately 130 m.

Radiological data

The dose rate results for waste areas fall in the range of 0.43 to 0.63 μSv/h; those of nearby areas, range from 0.19 to 0.27 μSv/h and those of background areas range from 0.19 to 0.24 μSv/h.

There are also measurements of radon concentrations in air and gamma radiation available over the mineral tailings. Mean values are presented in Table 38.

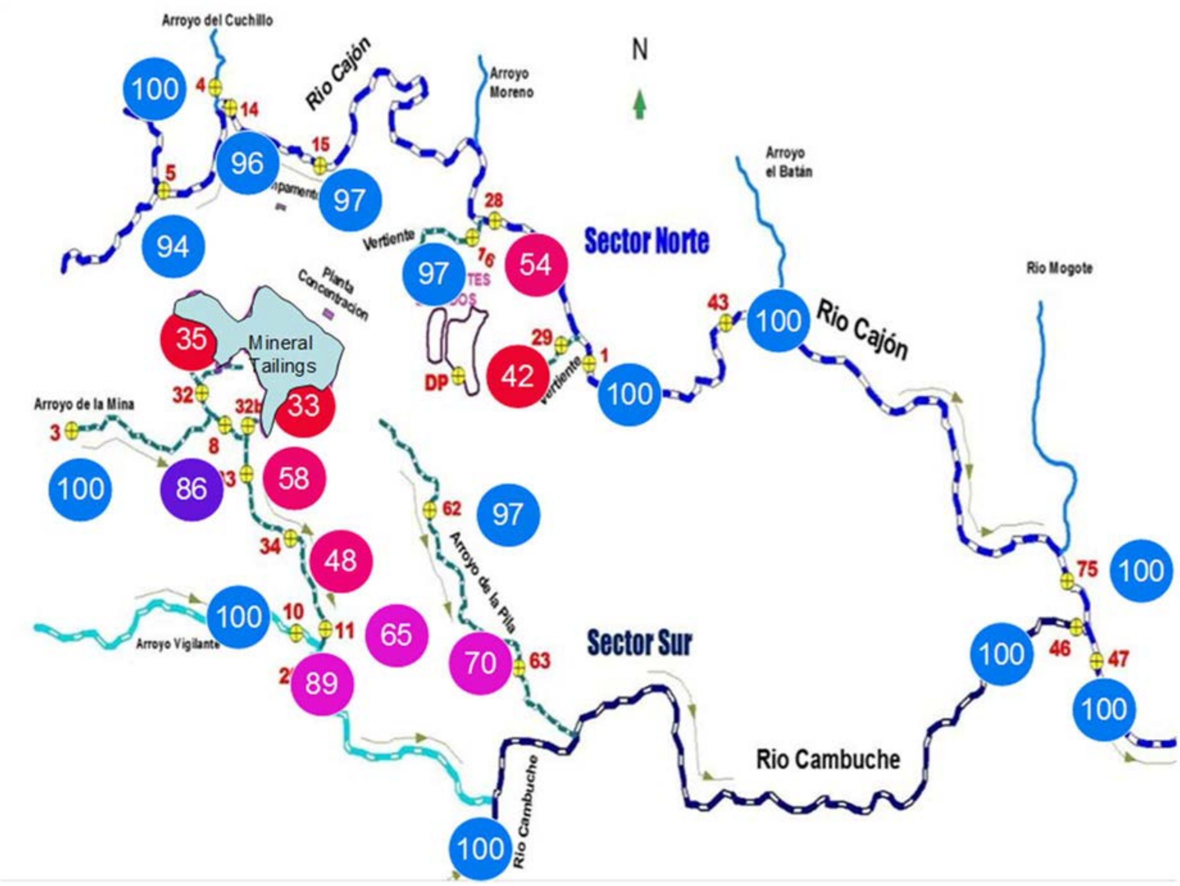

FIG. 66. Indicators of water quality around the site.

[53] "Optimum humidity is achieved when the inspired gas is at body core temperature and 100% relative humidity" (see Ref. [166]).

TABLE 38. MEAN VALUES OF RADIOLOGICAL PARAMETERS OVER THE TAILINGS

Parameter	Unit	Value
Rate of emission of radon over the tailings	Bq $m^{-2}s^{-1}$	0.33
Dose rate due to external irradiation over the tailings	µSv/h	0.35–0.45
Concentration of radon over the tailings	Bq/m^3	337–896

III.1.3. Site specific data

Geology

The geological environment consists of partially peneplanized areas of plutonites, limited by high grade metamorphic rocks. They are a significant part of the Achala batolite. A detailed description of the geology of the area can be found in Ref. [167].

Hydrogeology

The hydrological conditions of the area are limited by the characteristics of the geology previously described. A massive crystalline basement does not allow significant accumulations of meteoric water, which are controlled by a drainage system and a positive relief that favours quick percolation of the water.

The framework for this environment is given by the fracture zones, which are the only percolation and recharge lines for further drainage that supply water to the mountain creeks; there is a dry pluvial regime in winter and spring, and rainy in summer and autumn.

Surface waters

There are two springs associated with the fractured zones: one is in the middle of the quarry, over the west slope, and is permanent; the second is in the southern end of the quarry and is seasonal. Both springs join to form the Arroyo de la Mina, which is an affluent of Arroyo Vigilante; and this is an affluent of the Río Cambuche.

Table 39 presents the contribution of three effluent streams (Monitoring Stations 32, 32b and 33) to the water quality of Arroyo de la Mina (see also Fig. 66).

Climate

Data on climate, covering temperatures and precipitation in the area are provided in Refs [167−168].

TABLE 39. CONTRIBUTION OF THE EFFLUENT STREAMS

Ion	Stream Effluents (32)	Stream Effluents (32b)	Stream Effluents (33)	Others	Total
U	35.66%	62.79%	1.45%	0.10%	10%
Ra	36.84%	54.09%	8.95%	0.12%	100%
SO_4^{2-}	30.18%	58.20%	5.14%	6.48%	100%
TDS	33.31%	58.93%	5.61%	0.29%	100%
Acidity	20.41%	73.28%	6.29%	2.15%	100%
Flow	40%	25%	10%	25%	100%

Flora and fauna

Data on the flora and fauna of the area are available in Ref. [169].

Demography

The facilities of the complex are in the Department of Punilla, whose capital city is Cosquín (2592 km^2 and a population of 121 215 inhabitants – 46.8 inhabitants/km^2).

Transfer factors to agricultural products and diet of the representative person[54]

Soil-plant concentration factors have not been measured around the site.

III.1.4. Radiological environmental impact assessment

An environmental impact assessment is presently being performed on the impact of the mineral tailings on the water quality of the nearby mountain creeks, using the Canadian Water Quality Index) (CWQI)[55] and the United States Environmental Protection Agency Water Quality Analysis Simulation Program (USEPA WASP)[56].

In addition, a REIA is presently being performed by PRAMU[57].

III.2. POÇOS DE CALDAS SITE

The Poços de Caldas site (Brazil) consists of an open mine pit, tailings dam, waste rock dumps with an acid rock drainage problem, and a uranium mine and mill. The relevant aspects of these are described in the following Sections III.2.1–III.2.4 of this Appendix.

III.2.1. General description

The Poços de Caldas uranium mining and milling facilities began their commercial operation in 1982, with a target of producing 500 tons U_3O_8/year. After fifteen years of activity, the facilities were closed. During the operation, over 120 million tons of rock was removed from the open pit mine, 10 million tons of which were used as building material (e.g. roads, ponds). The remaining material is stored in piles on the site [170−171]. The main Waste Rock Piles (WRPs) are Waste Rock Pile 4 (WRP-4) and Waste Rock Pile 8 (WRP-8), which together contain approximately 60% of the total amount of rock removed during the ore extraction. On the site, a tailings dam was built, containing around 2×10^6 tons of residues from the milling process and wastes of different origins, among them calcium sulphate produced by the neutralization of the acid drainage and pyrolusite used to promote the uranium oxidation.

The Poços de Caldas region has radioactive anomalies and is characterized by bauxite associated with fluorite, pyrite, and manganese oxides. The removal of a huge amount of rock, coupled with the occurrence of pyrite and the relatively high rainfall rates (1800 mm/year), generates acid drainage that can mobilize metals and radionuclides from the rocks and tailings. The site has three sources where acid drainage occurs: the mine pit; the waste rock piles; and

[54] In the original study, the 'critical group' was considered, which has been superseded by the 'representative person' (see GSR Part 3 [3]).
[55] See https://www.gov.nl.ca/ecc/waterres/quality/background/cwqi/
[56] https://www.epa.gov/ceam/water-quality-analysis-simulation-program-wasp
[57] Proyecto de Restitucion Ambiental de la Mineria del Uranio - Uranium Mining Environmental Restoration Project.

the tailings dam. Without chemical treatment, the continuous release of this acidic solution will continue to result in increased metal and radionuclide concentrations in nearby surface waters.

A chemical treatment facility is still in operation at the site to decrease the concentrations of metals, fluorite and radionuclides in the acidic solution. Before release into the rivers of the region, the drainage from the waste rock piles, tailings dam and mine pit are treated by the addition of calcium hydroxide, $Ca(OH)_2$, to increase the pH of the the acidic drainage and make it alkaline, with a pH of 10–11. Following the settlement of solids, the liquid overflow from the settling dams is released into the Antas river and Soberbo creek, the latter of which runs to the Verde river.

III.2.2. Source term

Layout of the site

The Poços de Caldas plateau, where the uranium mine and mill site is located, is situated in the Minas Gerais State (geographic coordinates: 21º47’27”S and 46º47’56”W) in the southeast region of Brazil, 20 km from Poços de Caldas city (151 000 inhabitants). The site covers an area of 15 km^2 within which the mine occupies an area of 2.0 km^2 (Fig. 67). Two major water courses receive the releases from the site: the Antas river, which flows towards Pocos de Caldas city, and the Verde river, which flows towards the city of Caldas.

The region is characterized by well-developed agricultural activities in which the water of the rivers is intensively used for crop irrigation and cattle watering purposes.

Tables 40 and 41 provide information on the characteristics of the waste rock piles and the tailings dam [171].

TABLE 40. VOLUME, MASS AND AREAS OF THE WASTE ROCK PILES AT THE POÇOS DE CALDAS SITE

Waste rock pile	**Volume (10^6 m^3)**	**Mass (10^6 t)**	**Area (10^6 m^2)**
WRP-1	4.4	10.6	2.5
WRP-3	9.8	23.5	2.0
WRP-4	12.4	29.8	5.7
WRP-7	2.4	5.8	5.3
WRP-8	14.8	35.5	6.4
Total	**43.8**	**105.2**	**21.9**

TABLE 41. GENERAL CHARACTERISTICS OF THE TAILINGS DAM [172]

Parameter	**Value**
Total drainage area	0.86 km^2
Average waste discharge	0.15 m^3/s
Maximum volume	2.39 million m^3
Area	1.8 km^2
Average outflow rate	0.05 m^3/s

FIG. 67. Location of the site.

Radiological data

(a) Waste rock piles

The studies and characterization of the waste rock piles were focused on WRP-4 because its drainage is collected directly in a pond, whereas for WRP-8, only a part of the drainage is collected and the other part is released to the environment [171−172].

Table 42 presents the pH and radionuclide activity concentrations of the drainage from WRP-4 and Table 43 presents the radionuclide activity concentrations in a rock sample from WRP-4.

(b) Tailings dam

Metals and radionuclides are inhomogeneously distributed in the solid material of the dam. This heterogeneity in the tailings was caused by the different kinds of wastes and materials that were disposed of in the dam and by various mechanisms, among them the settling process, precipitation, preferential adsorption and leaching.

The mobilization of radionuclides in the tailings dam caused by the acid leaching of the radionuclides from wastes is described in Ref. [170]. The presence of acidic solution in the dam is an outcome of the residual pyrite oxidation. An estimation of the total amount of radionuclides in the dam is provided in Table 44, in addition to the fraction that is potentially mobile and the radionuclide activity concentrations in the dam water.

III.2.3. Site specific data

Geology

The Poços de Caldas Alkaline Massif is located in the central western part of the Cabo Frio Magmatic Lineament and is the largest alkaline complex in Brazil, with a diameter of 30 km and covering an area of 800 km^2. It consists of alkaline volcanic and plutonic rocks (mostly phonolites and nepheline syenites) with high background levels of U, Th and rare earth elements. The main morphology of the alkaline massif is a semi-circular plateau, with an average altitude of 1300 m, increasing to an elevation that is 400 m above the surrounding flatlands comprising the Poços de Caldas Plateau, with elevations of up to 1500–1600 m at its borders. The Poços de Caldas Plateau has been eroding at a rate of approximately 12 m per million years, on average, over the last 50 million years and is considered a stable feature. Local hydrothermal fluid-rock interactions have led to pyritisation, strong potassium enrichment and several important radioactive anomalies [173].

Climate

The tropical climate of the region has a marked rainy season from November to April and is dry the rest of the year. The annual average temperature is 19°C, with seasonal variations in the range of 1–36 °C. The maximum total annual rainfall is 1800 mm.

Surface water

The Antas and Verde hydrographic basins cross the site. The Antas basin drains approximately 70% of the plateau and the Verde basin 20%, which together drain the site. Hence, the Antas basin is the most important of the region, with a drainage area of 432 km^2. The Antas River has an average flux of 1.86 m^3/s, varying between 0.0025 m^3/s and 7.65 m^3/s. It is considered the ‘strategic’ stock of water for the region.

TABLE 42. ^{226}Ra AND ^{238}U CONCENTRATIONS AND pH VALUES IN THE DRAINAGE FROM WRP-4

Parameter	**Average**	**Minimum value**	**Maximum value**
pH	3.3	2.9	3.7
Ra-226 (Bq/L)	0.3	0.1	0.6
U-238 (Bq/L)	175	71	315

TABLE 43. RADIONUCLIDE CONCENTRATIONS IN A ROCK SAMPLE FROM WRP-4

Radionuclide	Concentration (Bq/kg dry mass)
U-238	4203
U-234	4090
Th-230	4066
Ra-226	5845
Pb-210	5120
Po-210	6600

TABLE 44. INVENTORY OF RADIONUCLIDES AT THE TAILINGS DAM

Element	Total inventory (Bq)	Mobile fraction (%)	Dam water (Bq/L)
U-238	4×10^{12}	7	0.22
Th-232	3×10^{11}	21	<0.01
Ra-226	5×10^{12}	59	0.03
Ra-238	3×10^{12}	16	0.38
Pb-210	7×10^{12}	65	<0.02

Site data

Several studies were performed in the Poços de Caldas region in order to determine the transfer factors from soils to plants and ultimately, to food [174−175], to study the transport of radionuclides in groundwater and surface water [176–178], and to assess the public dose due to the mining and milling operation [179−180]. Additionally, the owner of the site and the regulatory body have carried out pre-operational, operational, and post-operational environmental monitoring programmes. Through these studies, the levels of ^{226}Ra, ^{228}Ra, ^{238}U and ^{210}Pb in the effluents, surface waters, milk and vegetables are routinely monitored.

III.2.4. Radiological environmental impact assessment

Extensive studies of the impact of the mining site and enhanced metal concentrations in nearby surface waters are described in Ref. [171]. Furthermore, the main contaminants of concern in the Antas river were manganese, fluorite and uranium, which had concentrations that exceeded the water quality criteria established by Brazilian standards [181].

Tailings dam

A dose assessment of the impact of the tailings dam has been performed [170] using the model equations reported in Safety Reports Series No. 57[58]. The assessment included a worst-case scenario in which the control procedures would be interrupted and the acidic solutions released to downgradient water bodies without any kind of treatment. The values of estimated annual effective dose rate in a conservative approach were 8.5 mSv for children (0–1 year) and 8.1 mSv for adults.

A simulation using RESRAD-ONSITE showed that if houses are constructed directly on the tailings in a future scenario of unrestricted use of the site, high doses from direct gamma

[58] INTERNATIONAL ATOMIC ENERGY AGENCY, Generic Models for Use in Assessing the Impact of Discharges of Radioactive Substances to the Environment, Safety Reports Series No. 57, IAEA, Vienna (1982). Superseded by Ref. [61].

exposure and radon might result [171]. In this case, the calculated annual effective dose was 40 mSv for radon and 8 mSv for external exposure. In addition, the contamination of the groundwater was evaluated using the GWSCREEN code[59], based on conservative assumptions. Through this evaluation, it was determined that there would be a long contaminant transit time and that the contaminants might take thousands of years to reach the aquifer.

Waste rock piles

An annual committed effective dose of 0.5 mSv has been estimated due to the release of ^{226}Ra and ^{238}U with the acid drainage from the waste rock piles [176]. After the chemical treatment of the acidic solutions, the dose is expected to decrease to 0.05 mSv. The impact of releasing the acidic solution from the WRP-4 into rivers was assessed [170]. A screening model was used, which assumed that the representative person was located close to the zone of mixing of the river water and the acid drainage. The only exposure pathway that was considered was direct ingestion of water. In this case, the annual effective dose due to uranium, the main contributor to dose, would be 0.21 mSv. The dose was also evaluated for different remedial options.

III.3. BUHOVO URANIUM MINING AND MILLING SITE (BULGARIA)

III.3.1. Site description and history

Uranium mineralization in Bulgaria was first discovered in 1920. The first exploration activities were undertaken in 1935 at the Buhovo ore deposit, 25 km from Sofia. On the basis of technical research and economic calcuations, major exploration activities were undertaken in 1938–1939, in cooperation with German specialists [182].

During 1946–1947, Soviet geologists performed investigations of the Buhovo ore deposit, and a joint Soviet–Bulgarian enterprise was created in spring 1946. In 1956, the activities of this enterprise ceased and the ‘Rare Metals’ Bureau with the Council of Ministers of Bulgaria was established.

A number of exploration methods were applied. These included geological, geophysical, technological and a combination of methods. Aerial gamma spectrometry and hydro-radiochemistry were also used, in addition to other methods.

As a result of these explorations, 39 ore deposits were found and developed in Bulgaria. The main ore deposits for underground extraction are: Buhovo near Sofia; Eleshnitsa in northwest Bulgaria; Sliven in Central Bulgaria; and Smolyan, Dospat and Selishte in the Rhodope Mountains.

Two hydrometallurgical plants were built to process uranium ore and to produce uranium concentrate (U_3O_8) in Buhovo and Eleshnitsa. This became the centre of the South-Central uranium mining region.

In 1992, all uranium mines were closed and uranium milling in Bulgaria was shut down by decree of the Government.

[59] A semi-analytical model designed for assessment of the groundwater pathway from surface or buried contamination.

III.3.2. Remediation activities

The activities foreseen in the decree of the Government regarding protection from the consequences of the extraction and processing of uranium raw materials in the Republic of Bulgaria are:

- Technical liquidation;
- Technical and biological remediation;
- Purification of uranium contaminated mine waters;
- Environmental monitoring in the regions affected by the uranium extraction activities.

Remediation activities carried out in Bulgaria started with technical liquidation. For aboveground facilities, this included:

- Dismantling of the mining and milling facilities;
- Demolition of the buildings and structures;
- Decontamination of the technical equipment for reuse;
- Deposition of the radioactive wastes in the tailing ponds.

Underground facilities were secured with two concrete walls. Technical liquidation of the uranium mining facilities was completed. The second stage is 'technical remediation'[60] and included:

- Stabilization and reshaping of the surface of the waste rock piles and backfilling the tailing ponds and open pits;
- Building diversion channels for removing surface water for protection of the waste rock piles against erosion;
- Covering the surface with a non-radioactive layer to reduce gamma dose rates.

Technical remediation was completed for approximately 60% of the sites. For some of the sites, remediation activities included purification of contaminated water.

After the cessation of activities for uranium extraction and processing, the pumping stations for circulating water in the mines were dismantled; this created conditions for which water contaminated with radionuclides could flow out of the adits. As a result, mining waters containing radionuclides were released to the surface allowing for the contamination of surface waters in the area of the closed facilities.

In the areas where these processes are particularly intensive, treatment facilities for sorption purification of uranium-contaminated mine waters have been set up.

III.3.3. Radiological data

The Buhovo uranium mining and milling site is situated 15 km northeast of Sofia, the capital of Bulgaria. The Buhovo mining region is the main area for the uranium industry in Bulgaria

[60] This stage was originally referred to as 'technical rehabilitation', but has been changed to 'technical remediation', for consistency with the IAEA safety standards (see [6] and [1]) and to avoid implying that is feasible to rehabilitate the site to its original state.

and covers the area around Buhovo, Seslavtsi, Kremikovtzi, Yana village and Gorni Bogrov. Various contaminated ‘objects’ are present on the site, which can be divided by type of contamination (i.e. those resulting in contamination of water or soil).

Objects with contaminated water

Closed facilities with potential water contamination include Seslats and Kremikovtzi. The concentration of uranium and gross alpha levels in water samples at the point of discharge for Kremikovtzi and the Kremikovska river in the village are provided in Table 45.

Water passes through many tunnels and underground facilities. The activity concentrations of radionuclides in these tunnels are unknown. Water is used for watering of animals and irrigation in support of agricultural production (Fig. 68).

Water from two sections (Chora and Iskra) of the Buhovo mining region is purified through a sorption column. Water is directed through underground channels to the lowest point where the column is installed. Radionuclide activity concentrations in water collected before and after the treatment plant are monitored. Bottom sediments are also monitored in the area of the outflow of effluent water.

The hydrology of the area has not been reported, but the direction and level of groundwater may have been determined for environmental impact assessments.

TABLE 45. CONCENTRATION OF URANIUM AND ALPHA ACTIVITY CONCENTRATION IN WATER SAMPLES

Sampling place	**$U_{natural}$ (mg/L)**	**Gross α (Bq/L)**
Point of discharge	1.71	5.9
The Kremikovska river in the village	1.00	3.2
Uncertainties (2δ)	*20%*	*10%*

FIG. 68. Contaminated water used by grazing animals in the area.

Objects with contaminated soil

In conducting the mining of uranium ore, numerous waste rock piles were brought to the surface; most of them are in Buhovo and Seslavtsi. Dumps are remediated only in the horizontal part. Remediation is completed and adopted by a State Commission. Closing the horizontal part of the dumps in Seslavtsi has stopped the infiltration of surface water into the waste tip; however, this water then flows on the surface and contaminates the steep side slopes. Through this process, contaminated material migrates towards the village. The activity concentration of ^{226}Ra is approximately 1100 Bq/kg dry mass.

Dumps located in Buhovo are not subject to erosion by surface water because they are older. They are, however, used by motorcyclists for extreme sports, which is detrimental to their integrity and likely contributes to the spread of radionuclides. The activity concentration of ^{226}Ra in soil in the dumps is approximately 700 Bq/kg dry mass.

The lack of environmental protection measures during the initial operation of the uranium sites increased the likelihood of contamination from past activities. The most significant source of contamination is likely from the hydrometallurgical processing plant in Buhovo. The plant operated for almost 5 years without provision for the storage of tailings and radionuclides released to the environment could, therefore, migrate to the villages of Yana and Gorni Bogrov. The contaminated areas are now parks in the village of Yana and farmlands for agricultural production. The activity concentration of ^{226}Ra in soil in the contaminated areas in Yana and Gorni Bogrov is approximately 2200 Bq/kg dry mass.

An area of particularly high uranium contamination in Bulgaria is the milling plant in Buhovo. It is now a private enterprise and decommissioning is being carried out. A project to remediate this site has also been initiated. The maximum concentration of ^{226}Ra is 19000 Bq/kg dry mass and for ^{238}U is 530 000 Bq/kg dry mass.

The Buhovo uranium milling site has two tailings ponds, one is dry and the other is filled with water. The total surface area is 900 acres. The mass of waste deposited is 8320 tons and the total radionuclide inventory is 1.8×10^{15} Bq.

III.4. ŽIROVSKI VRH: WASTE PILES OF A FORMER URANIUM MINING AND MILLING FACILITY (SLOVENIA)

III.4.1. General description

Uranium ore was discovered in the Žirovski vrh area in 1960. After that, different research activities were conducted to determine the amount of uranium ore and the feasibility of commercial exploitation. It was assessed that within the area of Žirovski vrh there are approximately 12 million tons of uranium ore that could be used to produce about 16 000 tons of U_3O_8. In 1976, the company Rudnik urana Žirovski vrh was established, which initiated industrial excavations of uranium ore in 1982. Two years later, industrial production of U_3O_8 concentrate (yellowcake) started and lasted until July 1990 when all mining and milling activities were temporarily ceased by a decree of the Slovenian Government. Mining of uranium was ceased because the price of uranium on the world market decreased after the end of the Cold War and because of rising environmental protection and anti-nuclear movements after the Chernobyl accident. In 1992, the act of permanent cessation of exploitation of uranium ore and prevention of consequences of mining at the Rudnik urana Žirovski vrh uranium mine was adopted, and the uranium mine was permanently closed. After that, activities for permanent remediation of the area affected by uranium mining were finished in 2010. Approximately 0.6 million tons of uranium ore along with more than 2.5 million tons of other waste rock and low-grade uranium ore were excavated. From the uranium ore, 450 tons of U_3O_8 in the form of yellowcake were produced [183].

After the closing of the mine, all mining and milling facilities were dismantled, and the area was decontaminated. Within the area of the former facility for production of yellowcake, there is now an industrial area. All wastes were deposited into two piles, located close to the uranium mine (Jazbec and Boršt waste piles).

III.4.2. Source term

Layout of the site

Figure 69 presents the site layout. The Jazbec waste pile is in a small valley, whereas the Boršt waste pile is located on the site of a former swamp. Both waste piles were constructed above the temperature inversion point, which prevents radon retention in the Brebovščica and Todraščica valleys under meteorological conditions when a temperature inversion occurs. The Jazbec waste pile contains waste rock, low-grade uranium ore, red mud (remaining from the purification of U_3O_8 concentrate) and all contaminated material remaining after the dismantling of the facilities. Uranium mill tailings (UMT) have been deposited on the Boršt waste pile. Both waste piles have had remedial actions carried out on them. This includes the placement of an approximately 2 m thick cover on the wastes..

Some relevant attributes of the Jazbec waste pile are as follows:

- Area: ~4 ha;
- Amount of waste rocks and low-grade uranium ore: 2.5 Mt;
- Amount of red mud: 0.05 Mt.

Some relevant attributes of the Boršt waste pile are as follows:

- Area: ~3 ha;
- Amount of uranium mill tailings waste: 0.6 Mt.

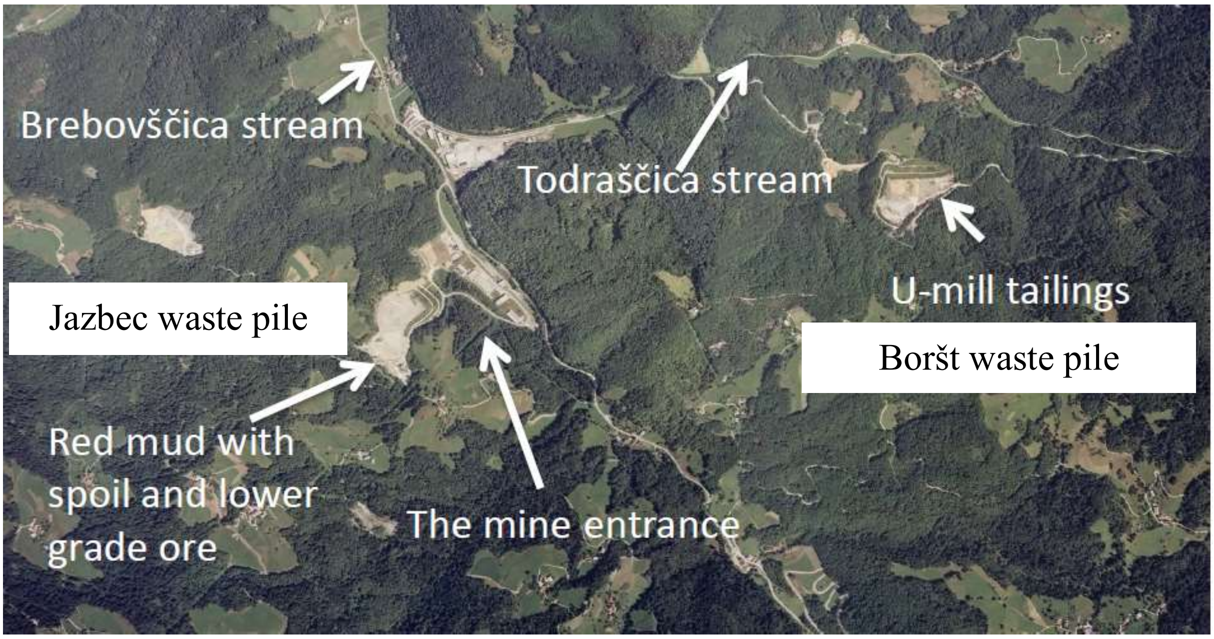

FIG. 69. Layout of the site (RUŽV – Rudnik urana Žirovski vrh).

Radiological data

The activity concentrations and inventories of natural radionuclides in the processed wastes deposited on the Jazbec and Boršt waste piles are presented in Tables 46 and 47. It is evident that in the Jazbec pile, red mud has elevated activity concentrations of ^{230}Th, which represents approximately 58% of the ^{230}Th originally present in the uranium ore. Nevertheless, the amount of red mud is small compared with other waste rock deposited in the Jazbec pile. The uranium mill tailings that have been deposited in the Boršt waste pile contain almost all (~95%) of the original ^{226}Ra inventory, in addition to approximately 42% and 11% of the ^{230}Th and ^{238}U inventories, respectively, compared with what was originally contained in the uranium ore. In addition, the uranium mill tailings contain 7610 ± 495 Bq/kg of ^{210}Pb and ^{210}Po, which represents approximately 87% of what was originally contained in the uranium ore.

TABLE 46. ACTIVITY CONCENTRATIONS OF NATURAL RADIONUCLIDES IN WASTES DEPOSITED AT THE JAZBEC AND BORŠT WASTE PILES [184]

Sample	a_{U-238} **(Bq/kg dry mass)**	a_{Th-230} **(Bq/kg dry mass)**	a_{Ra-226} **(Bq/kg dry mass)**
Uranium ore*	8780 ± 700	8780 ± 700	8780 ± 700
Jazbec site, red mud	495 ± 75	65 100 ± 9800	190 ± 20
Jazbec site, waste rocks*	750 ± 100	750 ± 100	750 ± 100
Boršt site, uranium mill tailings	995 ± 80	3930 ± 580	8630 ± 340

*Based on data for average U_3O_8 content of 840 mg/kg dry mass in uranium ore and 70 mg/kg dry mass in waste rocks.

TABLE 47. INVENTORIES OF NATURAL RADIONUCLIDES IN PROCESSED WASTES DEPOSITED AT THE JAZBEC AND BORŠT WASTE PILES [184]

Sample	A_{U-238} **(TBq)**	A_{Th-230} **(TBq)**	A_{Ra-226} **(TBq)**
Uranium ore	5.37	5.37	5.37
Jazbec site, red mud	0.02	3.1	0.01
Boršt site, uranium mill tailings	0.58	2.33	5.12

Average dose rates in 2010 at the Jazbec waste pile surface were approximately 100 nSv/h, and on the top of the Boršt waste pile, approximately 120 nSv/h. Radon activity concentrations in 2010 for the Jazbec waste pile ranged from 8.4–120 Bq/m^3, and for the Boršt waste pile, from 6.9–77 Bq/m^3 [185].

III.4.3. Site specific data

Surface waters and hydrogeology

A description of surface waters and the hydrogeology can be found in Ref. [183]. Figure 70 shows surface waters at the area of the former uranium mine Žirovski vrh. Seepage waters from the Jazbec waste pile are released into the Brebovščica stream, and from the Boršt waste pile, into the Todraščica stream. The Todraščica stream flows into the Brebovščica stream and the Brebovščica eventually flows into the Selška Sora river.

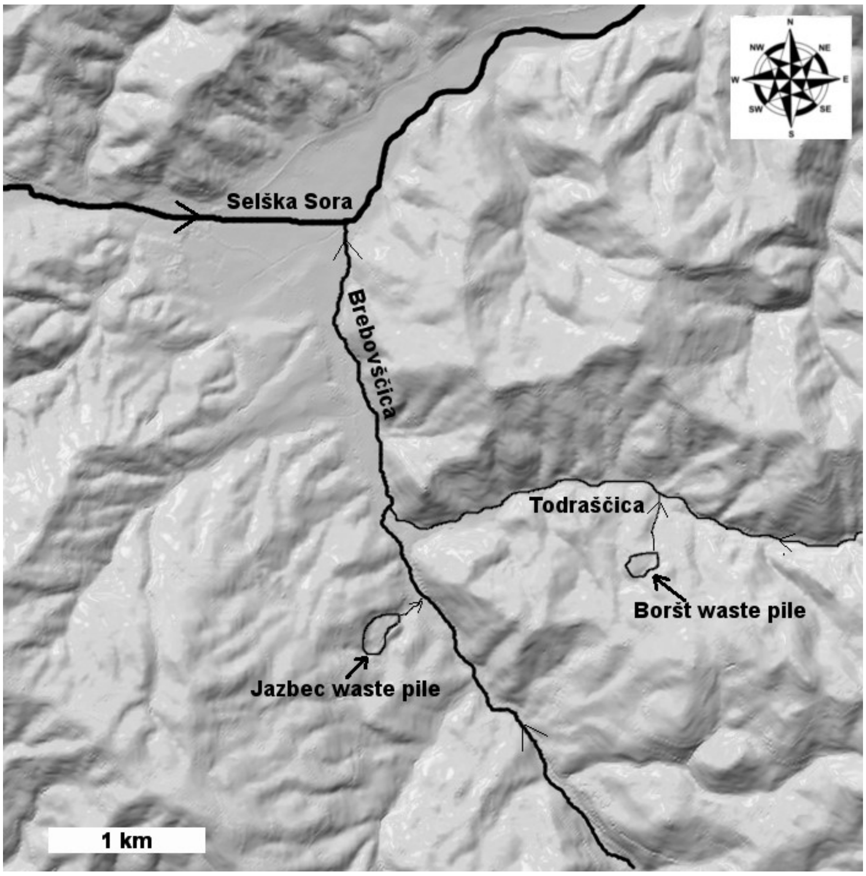

FIG. 70. Surface waters in the area of the former uranium mine Žirovski vrh.

Climate

Both waste piles are in a subalpine region with relatively high rainfall. A meteorological station in the vicinity of the Boršt waste pile provides data for this area.

Radioecological studies

Since 1968, routine radiological measurements of the area of the former uranium mine Žirovski vrh have been carried out. During the last five years, research has been focused on the migration and mobility of natural radionuclides within the Boršt waste pile, transfer of those radionuclides to plants, milk and fish, and assessments of doses to plants [186–192]. The results of the investigations show that, at the Boršt waste pile, the most mobile radionuclide is ^{238}U, followed by ^{226}Ra and ^{210}Pb, whereas ^{230}Th and ^{210}Po are extremely immobile. Activity concentrations of natural radionuclides in milk and fish are low and mostly comparable with the reference location. Transfer of natural radionuclides to plants is highest for ^{226}Ra, followed by ^{238}U and ^{230}Th. Dose assessment to plants using the ERICA Tool [132−133] showed that the dose rates are lower than the derived consideration reference levels (DCRLs)[61]

[61] Referred to as 'action values' in the original documentation for this case study.

III.4.4. Radiological environmental impact assessment

The additional dose to the people living nearby is assessed annually, based on available radionuclide and radiation monitoring data, which comprises all possible transfer pathways (e.g. air, aerosols, water, food). In the year 2010, the additional contribution to the annual effective dose of the nearby population due to the former uranium mine Žirovski vrh was approximately 0.118 mSv for an adult inhabitant.

APPENDIX IV.
HISTORIC NORM SITES

The sites described in this Appendix are historic NORM sites other than uranium mining and milling sites, as follows:

- Tessenderlo site in Belgium (phosphate);
- Botuxim site in Brazil (monazite);
- Inner Mongolia BaoTou Iron and Steel Plant (iron, rare earth metals);
- Upper Silesian Basin in Poland (coal);
- Compostilla II site in Spain (coal).

A general description of the site, the source term, site specific data in support of modelling and information relating to the REIA are provided for each site.

IV.1. TESSENDERLO'S CAF2 SLUDGE DUMP SITES (TESSENDERLO – BELGIUM)

IV.1.1. General description

Tessenderlo is in northeast Belgium. The Tessenderlo Chemie chemical company processes Moroccan phosphate ore for cattle feed production. The company uses the hydrochloric acid process, creating solid waste consisting predominantly of CaF_2 sludges. Several dumpsites (sludge basins) have been exploited by the company. One of them (the Veldhoven sludge basin) consists of three separate disposal sites adjacent to each other (i.e. S1, S2 and S3). S1 is no longer in operation, and S2 and S3 are still operating. The S1 site was used as a case study in the European Commission report "Investigation of a possible basis for a common approach with regard to the restoration of areas affected by lasting radiation exposure as a result of past or old practice or work activity" [144].

A few other dumpsites that are no longer in use are scattered around the site. Moreover, because of relatively high radium concentrations in past liquid discharges, the banks of the discharge streams, Laak and Winterbeek, have also been contaminated.

IV.1.2. Source term

Site layout

The Veldhoven dumpsite is located just North of a canal (i.e. the 'Albert canal') and comprises three distinct mining areas, i.e. S1, S2, S3 (Table 48). There is another dumpsite on the opposite side of the canal, on the factory premises [193]. Two other disused dumpsites are located 1–2 km SW of the Veldhoven site.

TABLE 48. OVERVIEW OF THE DUMPSITES

Dump site	Area (ha)	Volume of residues (tons - dry)	Years of exploitation
S1	25	900 000	1963–1986
S2	4	50 000	(buffer dump) ~1980–present day
S3	26	900 000	1987–present day
Sludge basin on factory premises	5.6	150 000	1931–1968
Sludge basin Kepkensberg	19.7	~550 000	~1946–1979
Landfill Spoorwegstraat	2.4	630 000	1942–1983 + 1989–1996

The total capacity of the S3 dump is approximately 2 400 000 m^3. The density of the sludge is approximately 1.6 ton/m^3.

Radiological data

Until 1990, the average ^{226}Ra activity concentration in the CaF_2 sludges was approximately 3.5 Bq/g. Since 1990, the ^{226}Ra activity concentration has significantly increased, reaching a level of up to approximately 11 Bq/g[62]. The ^{226}Ra activity concentration for the S3 sludge basin is, therefore, significantly higher than that of the other sites (abandoned prior to 1990). The surface dose rates for the site have reached a maximum value of just over 2.5 µSv/h.

The contamination on the banks of the discharge streams, Laak and Winterbeek, is patchy and covers a wide area due to the dredging of contaminated sediments, and has reached approximately a few Bq/g of ^{226}Ra. Figure 71 shows dose rate measurements on parts of the banks of the Winterbeek stream.

The distribution of contamination around the Laak and Winterbeek streams has been determined based on several dose rate measurement campaigns. In addition to ^{226}Ra contamination, heavy metals (such as arsenic and cadmium) are also present. Dose rate measurements and cadmium contamination show some degree of correlation, which allows use of the dose rate as an indicator for identifying areas contaminated with heavy metals.

[62] In order to decrease the radium concentration in wastewater, barium was added to the process to co-precipitate radium. This process was very efficient in reducing the ^{226} Ra activity concentration of wastewaters (it decreased from 20-25 Bq/L down to <0.2 Bq/L); however, this led to an increase in the ^{226} Ra concentration in solid waste.

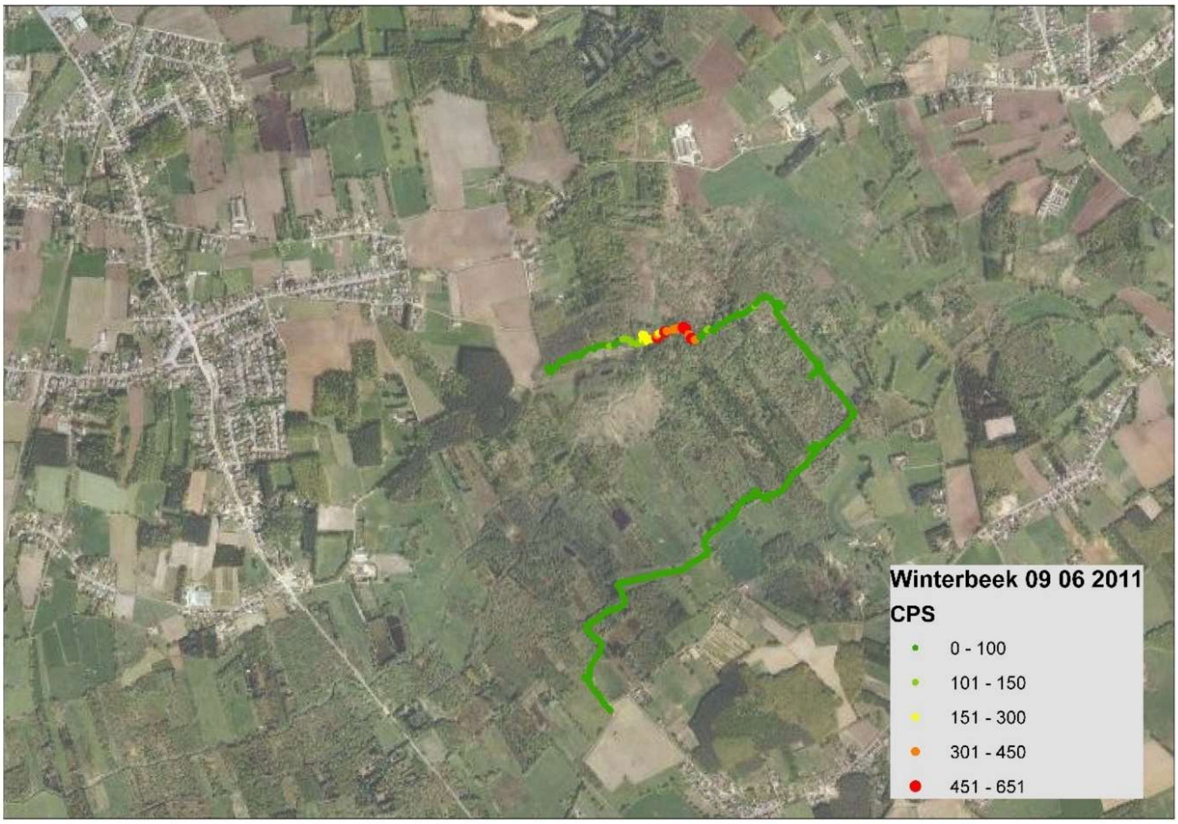

FIG. 71. Gamma flux measurements on the banks of the Winterbeek stream. These may reach 15 times background values.

A radon monitoring program, consisting of 14 measurement points, has been established on and around basins S1–S3. Table 49 shows a summary of the results for the period of 2008–2010.

TABLE 49. RADON MEASUREMENTS (Bq/m^3) ON AND AROUND THE S1–S3 SLUDGE BASINS

Measurement site	2008	2009	2010
On areas S1 and S2	160	80	65
On area S3	14	20	20
Environment	11	10	10
Sludge basin on factory premises	53	60	30

Note: Normal background values for outdoor radon concentrations in the area are approximately 10 Bq/m^3.

IV.1.3. Site specific data

Surface waters and hydrogeology

In addition to the Albert canal, there are two streams flowing next to the dump site (~50 m away from the site): ‘Bosloop’ to the northeast, which follows the edge of S2 and S3; and ‘Grote Beek’ to the southeast of S3. According to piezometric measurements, groundwater is flowing in the direction of the ‘Grote Beek’ river. The flow rate of ‘Grote Beek’ is 2000 m^3/hour. The nature of the soil under the dumpsite is sandy.

Groundwater on the site has the following characteristics:

- Cross-sectional area: 2600 m^2;
- Flow rate: 10 m/year.

Distribution coefficients

There are no site specific distribution coefficients available in Ref. [144]. Therefore, K_d values were taken from Technical Reports Series 364, Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments [194].

Transfer factors

The transfer factors from soil to pasture grass and agricultural crops, from plants to animals and from surface water to aquatic foodstuffs used in Ref. [144] are not site specific. Therefore, these values have also been taken from Ref. [194].

IV.1.4. Radiological environmental impact assessment

The S1 dumpsite was used as a test case for the generic assessment methodology described in Ref. [144]. A generic assessment model has been developed for the assessment and restoration of contaminated sites (AMCARE: Assessment Model for a Common Approach to REstoration of contaminated sites) [144]. Moreover, Ref. [144] also describes all parameter values used in the AMCARE model (e.g. radionuclide specific distribution coefficients, transfer factors to animal and aquatic foodstuffs, and transfer factors to pasture grass and agricultural crops, characteristics of the representative person).

Two scenarios of exposure have been considered [144]:

(1) A normal evolution case, with farmers living and working close to the site (‘status quo’ regarding the site and local population characteristics);
(2) An intrusion scenario, which assumes that the representative person inhabits houses built directly on the contaminated site.

The results of the dose assessment for these two scenarios are described in detail in Ref. [144].

Radon inhalation is by far the main exposure pathway for the two scenarios. For the normal evolution scenario, it leads to an annual effective dose of 0.5 mSv. For the intrusion scenario, it leads to an annual effective dose of 357 mSv.

It is noted that in another study of the radiological impact of the S1 sludge basin [195], the contribution of radon is much smaller, with an annual effective dose for the intrusion scenario estimated to be 38 mSv in this latter study (i.e. approximately an order of magnitude lower in the latter study). Radiological impact studies have also been performed for other former dumpsites [196].

IV.2. BOTUXIM STORAGE SITE FOR RESIDUES FROM THE MONAZITE INDUSTRIAL PROCESS (BOTUXIM – BRAZIL)

IV.2.1. General description

Low-level radioactive wastes containing long lived radionuclides were generated from 43 years of monazite processing in Brazil. The material generated is referred to as ‘mesothorium cake’

(Ra and Pb isotopes) and 'Cake II' (a mixture of 0.9% of U_3O_8 and 22% of ThO_2). Around 3500 tons of Cake II has been stored since 1975 at Botuxim site, located in a rural area in the Itu District, 80 km far from São Paulo city [197]. Approximately 3500 tons of Cake II have been stored since 1975 at the Botuxim site, located in a rural area in the Itu District, 80 km from São Paulo city [197]. At the site, Cake II is stored in seven rectangular pools (silos) that were loaded between 1975 and 1980.

Soil contamination was found in the controlled area associated with the silos. The activity concentrations in the surface soil samples reached values of 70 Bq/g for ^{228}Ra [198]. The activity ratios of ^{228}Ra/^{226}Ra were consistent with the ratios expected to be present in Cake II, leading to the assumption that the contamination was a consequence of material overflow which may have occurred during the filling of the silos.

Since the 1980s, high radium concentrations have been detected in a groundwater well downstream from the storage site. The highest radium activity concentrations occur during the peak rainfall period from April to May. Some characterization studies were conducted to identify the origin of the abnormal levels of radium measured in this groundwater well. One possibility that was considered was that the contamination might be associated with soil contamination or with a spill of the Cake II through a crack in the concrete of the silos, with subsequent leaching of the waste residues in the soil, ultimately, resulting in the contamination of the well water. However, measured data indicated that the activity ratio of ^{228}Ra/^{226}Ra found in the groundwater itself does not fall within the range to be expected for contamination due to the Cake II disposed of at the site [199]. In addition, a survey revealed high natural radioactivity levels in soil and the mainly granitic rocks in the Botuxim region and indicated the possibility that the high levels of radium in the well water might be due to leaching from such rocks [200]. The origin of the contamination has not yet been clearly identified, and additional research would be needed to confirm the above assumptions.

IV.2.2. Source term

Layout of the site

The control area for the storage of the silos is located on the Botuxim site. A map of the site is provided in Fig. 72. The residue is stored in seven pools, which are three metres deep and are surrounded by 30 cm thick concrete walls and floors. Each pool extends 0.5 m above the soil surface and 2.5 m underground. They are capped with concrete (Fig. 73). A small creek, the Monjolinho creek, runs through the site and flows into the public water supply of Itu city (150 000 inhabitants).

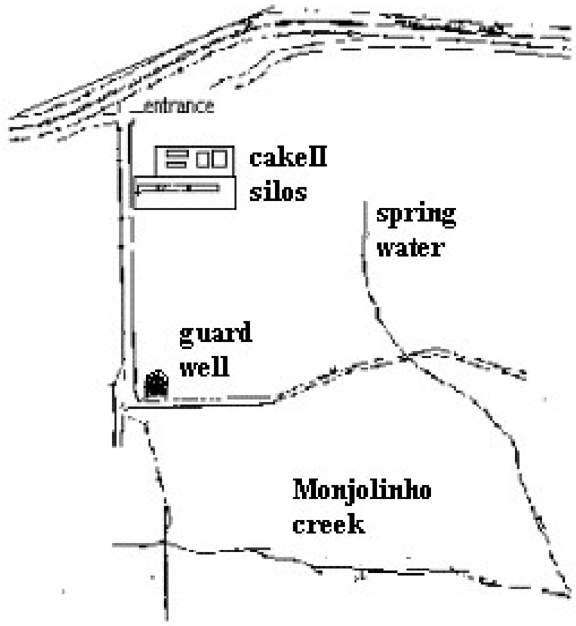

FIG. 72. Map of the Botuxim site.

FIG. 73. Controlled silos area at the Botuxim site.

Radiological data

The stored residues consist of approximately 22% of thorium oxide and 0.9% of uranium oxide, as part of the associated decay chains. The total specific activity is approximately 1820 Bq/g.

The radionuclide activity concentrations in soil are highly heterogeneous. The main radionuclide in soil is ^{228}Ra, with activity concentrations varying from 0.034 Bq/g dry mass to 70 Bq/g dry mass, followed by ^{226}Ra and then ^{238}U, with activity concentrations between 0.02 Bq/g dry mass and 0.8 Bq/g dry mass and from 0.02 to 13 Bq/g dry mass, respectively. Gamma radiation surveys have been carried out, which revealed contaminated areas surrounding the concrete storage silos (Fig. 74).

Additionally, an area with elevated activity concentrations of radionuclides was found outside of the fenced controlled silos area.

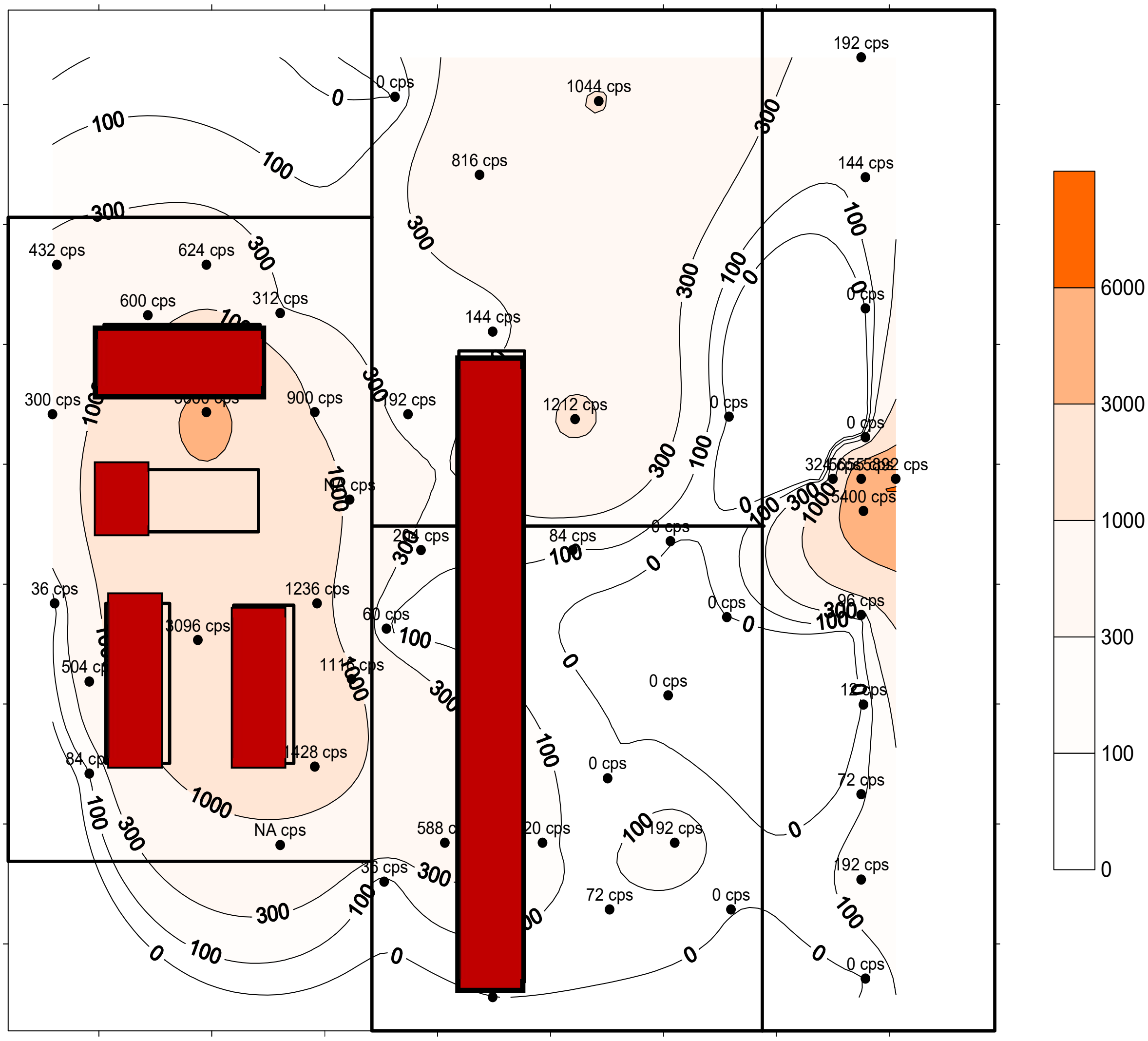

FIG. 74. Distribution of contamination (in counts per second, cps) in the surface soil of the silos area.

Activity concentrations in well water measured as part of a monitoring programme of the Institute of Radiation Protection and Dosimetry (IRD) showed values of 0.10 (0.023–0.820) Bq/L for ^{226}Ra and 0.14 (0.023–0.740) Bq/L for ^{228}Ra, for the guard well sampling station, while the geometric mean radium activity concentrations in surrounding wells showed values of around 0.08 Bq/L for ^{226}Ra and 0.01 Bq/L for ^{228}Ra. Values of 0.02 and 0.16 Bq/L were measured in an upstream spring for ^{226}Ra and ^{228}Ra, respectively.

Total alpha activity concentrations in water samples from sampling stations are presented in Table 50. It can be verified that the values measured for the guard well station are higher than those measured in Monjolinho creek, and in the water supply of Itu city.

TABLE 50. ALPHA ACTIVITY CONCENTRATIONS IN ENVIRONMENTAL SITES AND IN THE GUARD WELL (Bq/L)[a]

Locality	N	Geometric mean	Minimum	Maximum
Guard well	77	2.97×10^{-1} (3.90)	1.00×10^{-2}	4.00×10^{0}
Itu city water supply treatment station	6	1.03×10^{-1} (4.07)	1.00×10^{-2}	5.00×10^{1}
Monjolinho creek	8	1.04×10^{-1} (2.93)	2.00×10^{-2}	3.30×10^{-1}

[a] Data supplied by the CETESB (Companhia Ambiental do Estado de São Paulo).

IV.2.3. Site specific data

Geology

The Itú intrusive suite defined in Ref. [201] is located between the cities of Itu, Itupeva and Indaiatuba, in the State of São Paulo, Brazil, 60 km from São Paulo City. It covers an area of about 400 km^2 and contains at least four granitic bodies, separated by basement exposures, faults and masses of undifferentiated granites. The composition of the bodies is mainly Fe–hastingsite biotite granites and biotite granites.

The Th/U activity ratio exceeds the average of approximately 1.2, with a value of approximately 2.5 for the granite from Itú. The high values for the Th/U activity ratio indicates granite leaching [200].

Climate

The region where the site is located experiences a tropical type of climate, with temperatures ranging from 16–22°C and an annual average rainfall of 1200 mm. The dry season is between July and December and the rainy season is January to June, with most rainfall occurring in April to May.

The water table of the aquifer lies at a depth of approximately 16 m beneath ground level.

IV.2.4. Radiological environmental impact assessment

The REIA for the Botuxim repository has faced some challenges through the years. For example, it is uncertain whether the source of contamination found within a groundwater well (see Section IV.2.2) originates from the disposal site or whether the high measured radium concentration may be from high natural background radiation. Further field studies to determine the origin of the high radium level found at that groundwater well are needed.

IV.3. BAIYUN OBO, CHINA: MINING AND PROCESSING FROM A DEPOSIT OF IRON AND RARE EARTH ORES

IV.3.1. General description

The Inner Mongolia BaoTou Iron and Steel Plant (BTISP) in Baiyun Obo, China, was founded in 1954 and consists of the following entities:

Baiyun Obo:

- Mining and crushing;
- Milling since 2008.

Baotou:

- Milling;
- Refining in the Iron and Steel Plant;
- Processing in the Rare Earth plant since 1974.

The BTISP processes 1.2×10^7 tons of ore per year from the Baiyun Obo mine, of which there are 9×10^6 tons of iron and steel per year and more than 7×10^3 tons per year of thorium-rich oxide equivalent of rare earth ore produced.

Routine radiation monitoring is required by law and is carried out by the Radiation Environmental Monitoring Station of Autonomous Region of Inner Mongolia (REMSARIM). In addition, an assessment of radioactive contamination originating from mining of the Baiyun Obo mine has been conducted.

The background radiation level is 85 nGy/h in Baiyun Obo. Higher radiation levels were primarily detected in the mining area and its surroundings, including on the mining sites, in some office buildings and on the dump sites. Typical dose rates lie in the range of 200–800 nGy/h, covering approximately 55 km^2, with values from 600–2000 nGy/h at mining sites, and from 400–800 nGy/h at the dump sites.

Elevated radiation levels in the areas of the new rare earth plant and the tailings pond, sintering plant, iron plant, as well as in other facilities and areas, with values that lie mainly in the range of 300–500 nGy/h. The upper 10 cm layer of soil contains most of the contamination; the activity concentration of thorium is ca. 80–120 Bq/kg dry mass of soil.

The background dose rate in Baotou, China is 65 nGy/h. Higher dose rates are detectable in an area of approximately 7 km^2, located mostly in the BTISP, including in the sorting plant, rare earth plant, steel refining plant, and the NORM residue storage areas. The typical dose rates here lie between 500–1000 nGy/h, with a maximum value of 1518 nGy/h. The radiation level in the central area of rare earth plants ranges from 200–600 nGy/h. Several hot spots can be found in some rare earth plants in the city. The dose rates in the plant and its surrounding environment range from background to 143 nGy/h. The dose rates in the tailings pond without a water cover is in the range of 650–1200 nGy/h. The dose rates from soil contaminated by dust from the tailings pond lie in the range of 85–150 nGy/h, with an activity concentration that falls in the range of 80–200 Bq/kg dry mass and a value exceeding 400 Bq/kg dry mass near the tailings pond.

IV.3.2. Source term

Mining sites in Baiyun Obo (see Fig. 75)

Some key attributes of the mining sites in Baiyun Obo are as follows:

- Open pit mines: 1520 × 1080 m^2 for Main Mine, 1400 × 1020 m^2 for East Mine, and 4600 m in length by 1000–1200 m in width for West Mine.
- Waste rock dumps: Approximately 10 × 10^6 tons of waste rocks are produced annually. The total amount of material within the waste rock dumps is approximately 560 × 10^6 tons.

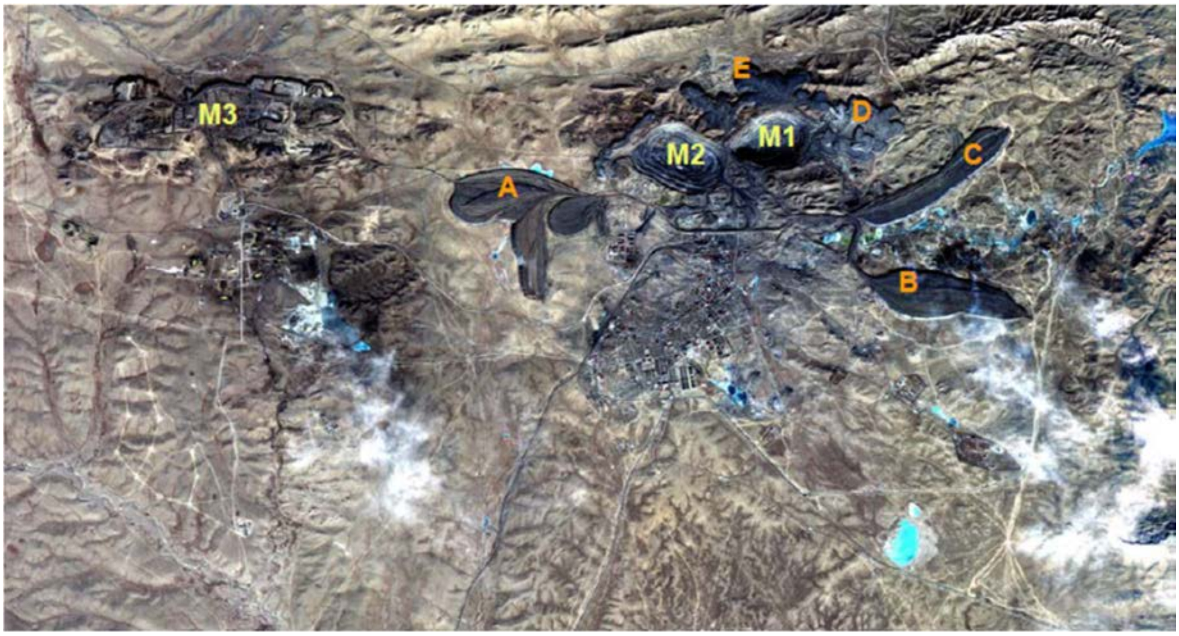

FIG. 75. Baiyun Obo mining sites. M1, M2 and M3 are east mine, main mine and west mine, respectively. A, B, C, D and E are dumping sites.

The BTISP and Baotou city (see Fig. 76)

Some key attributes of the BTISP and Baotou city are as follows:

- **Tailings pond:** An area of 11 km^2, with approximately 149 × 10^6 tons of tailings (based on an estimate made in 2006). Approximately 6.55 × 10^6 tons of tailings, consisting of 0.048% Th, are produced annually.
- **Ferrous slag dump:** Covers an area of 1 km^2. Approximately 3.55 × 10^6 tons of ferrous slag are produced annually.
- **Others:** 60 000 tons per year of rare earth slag are disposed of in the Baotou Radioactive Waste Storage Facility. The liquid effluent is discharged into the tailings pond. Treated exhaust gas that has been treated by gas cleaning is discharged to the environment from stacks.

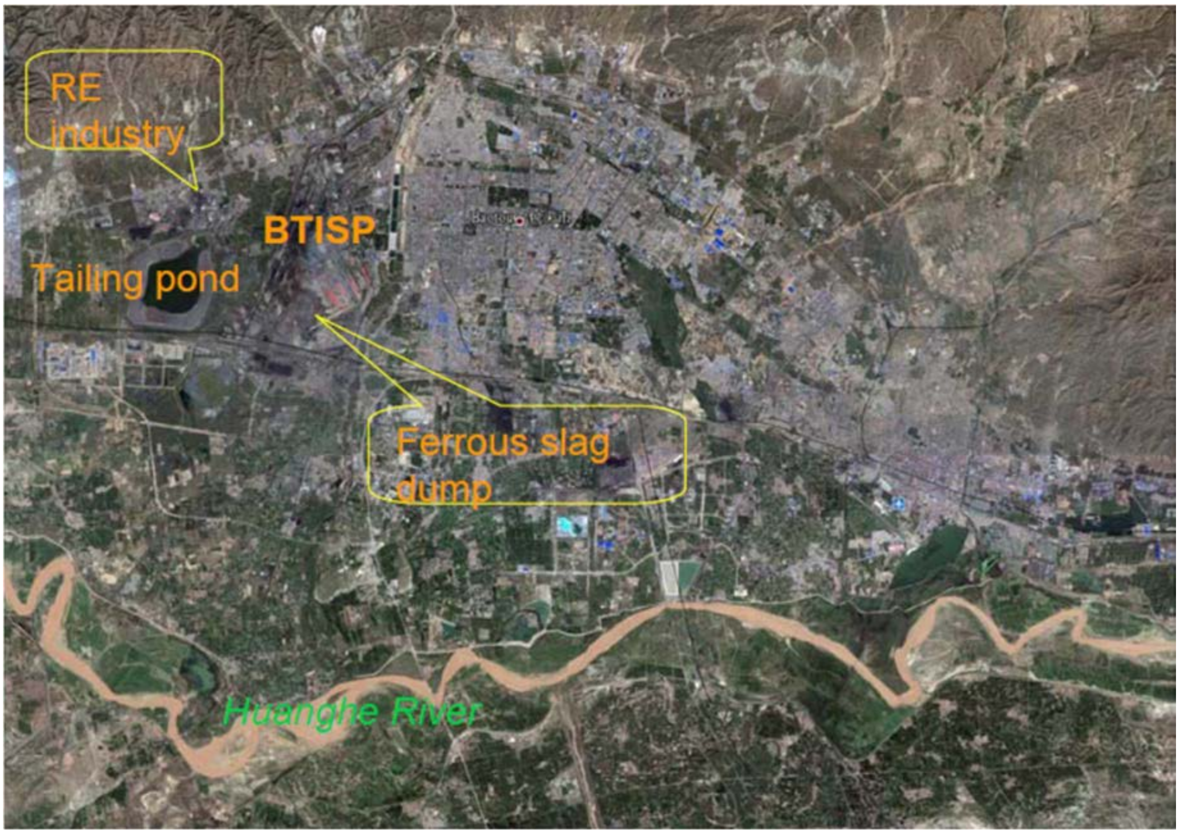

FIG. 76. The BTISP and Baotou city.

IV.3.3. Site specific data

Monitoring data available for modelling includes:

- **Regional radiological data:** Activity concentrations of natural nuclides from aerial survey and ground measurements in the Baotou and Baiyun Obo areas, as shown in Figs 77 and 78, respectively:
- **Activity concentrations of natural radionuclides** in samples of soils, water, gases and other materials;
- **Indoor radiation data** (radon and radionuclide activity concentrations in building materials);
- **Regular monitoring data:** Collected from the Radiation Environmental Monitoring Station of Autonomous Region of Inner Mongolia (REMSARIM);
- **Other data:** Meteorology, geology and hydrology.

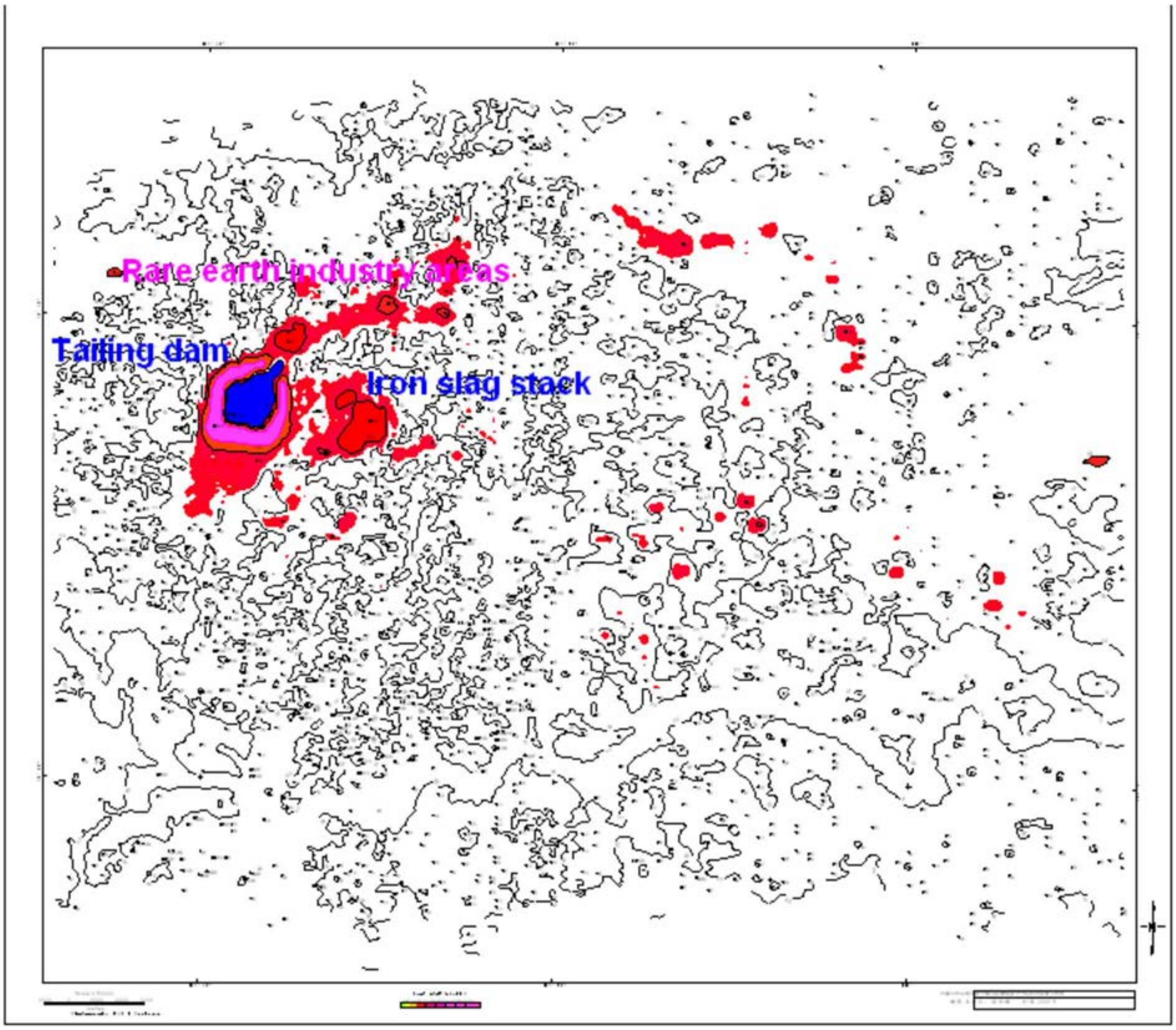

FIG. 77. Radiation levels in the Baotou area.

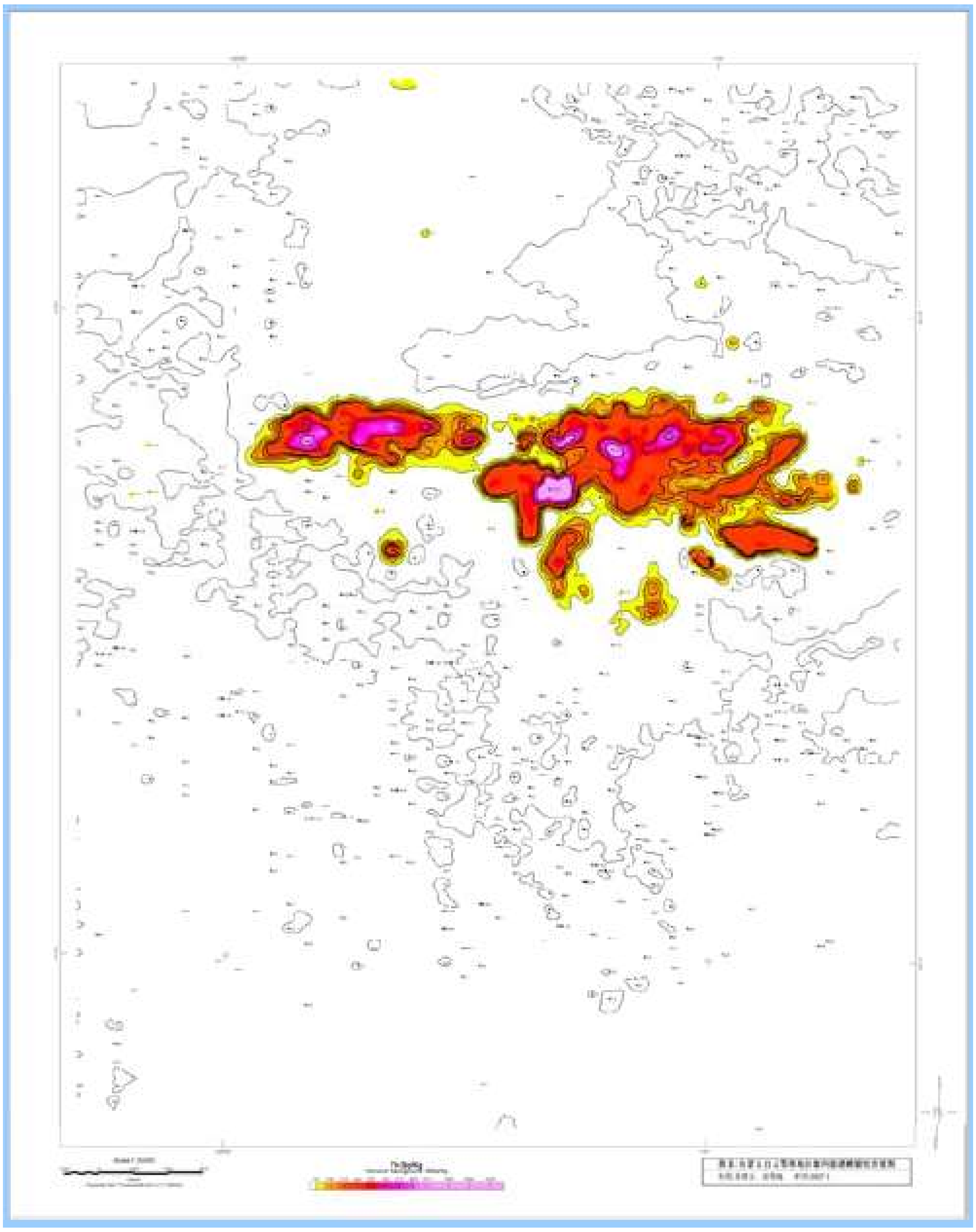

FIG. 78. Radiation levels in Baiyun Obo area.

Tables 51 and 52 provide detailed information on monitored constituents and the frequency of monitoring for historic NORM sites and for environmental quality.

TABLE 51. MONITORED CONSTITUENTS AND FREQUENCY OF MONITORING OF HISTORIC NORM SITES

Discharge	**Locations**	**Sampling points**	**Frequency**	**Monitored constituents**
External radiation	Industrial park, mining sites and waste rock dumps, tailing pond	69	Once a year	Gamma dose rate
Aerosol	Iron and steel, and rare earth industrial park	2	Once a year	Rn progeny
Well water	Iron and steel, and rare earth industrial park, and villages around the park	5	Once a year	Gross α Gross β U Th ^{226}Ra
Wastewater	In tailing pond, leaking from tailing pond, and discharging from plants	4	Once a year	Gross α Gross β
Solid waste	Ores, raw materials (minerals), waste rocks, tailings, ferrous slag and rare earth slag.	8	Once a year	^{238}U ^{232}Th ^{226}Ra ^{40}K Gross α Gross β

TABLE 52. MONITORED CONSTITUENTS AND MONITORING FREQUENCY FOR ENVIRONMENTAL QUALITY

Discharge	**Locations**	**Sampling points**	**Frequency**	**Monitored constituents**
External radiation	Around the building of the REMSARIM	1	Continuous sampling	Gamma dose rate
	In public areas	33	Twice a year	Gamma dose rate
	Around the building of the REMSARIM and Radioactive Waste Storage Facility	2	Twice a year	Gamma dose rate
Aerosol	Around the building of the REMSARIM and on the Tuanjie Square	2	Twice a year	Rn progeny
	Around the building of the REMSARIM	1	Once a year	Gross α, gross β
Deposition	In public areas	7	Four a year	Gross α Gross β
Surface water	Huanghe river, Nanhaizi lake and Kundu reservoir	7	Twice a year	Gross α Gross β U Th ^{226}Ra
Drinking water	Waterworks in Baotou city	3	Twice a year	Gross α Gross β U Th ^{226}Ra
Well water	Around Radioactive Waste Storage Facility	3	Twice a year	Gross α Gross β U Th
Wastewater	3 streams, those are Sidaoshahe, Kunhe,and Xihe.	3	Once a year	Th ^{226}Ra
Soils	Around Radioactive Waste Storage Facility and on the Tuanjie Square	5	Once a year	^{238}U ^{232}Th ^{226}Ra ^{40}K Gross α Gross β

IV.3.4. Radiological environmental impact assessment

In the plants, the residual external exposure to workers falls in the range 0.29–0.41 mSv, while for workers in mining areas, it is in the range of 0.24–1.0 mSv. If the inhalation of thorium containing aerosols and dust is considered, the additional annual effective dose likely exceeds 1.0 mSv for many workers.

The radiation level is higher in the buildings that have been constructed using slag materials, but additional external exposure to members of the public is not significant.

Additional information on REIAs carried out is available in Refs [202–205].

IV.4. POLAND: DESCRIPTION OF SETTLING PONDS FROM THE COAL MINING INDUSTRY

IV.4.1. Upper Silesian Coal Basin, South of Poland (approximately 50 km from Katowice)

General description

The Silesian Upland, approximately 50 km from Katowice in southern Poland, contains a coal-mining region. Activities associated with the coal mining include the use of settling ponds. Information regarding one of these ponds, which was used for sedimentation of suspended solids from mine waters since 1977, is provided here. In the 22 years up until 2011, ca. 70 million m^3 of saline waters was discharged into the pond. The content of the suspension ranged from 0.3 up to 2.4 kg/m^3. The area of the pond is approximately 36 ha (0.6 km^2). The main purpose of this pond was the retention of saline water prior to discharge to the river to protect freshwater from a high content of salt. Clarification from suspended matter was also carried out.

Radium-bearing waters have been pumped into the reservoir [206]: ‘Type A’ waters contain significant amounts of barium and traces of sulphate ions; and ‘Type B’ waters contain no barium, but radium and sulphate ions are present. In 1998, ca. 5600 m^3/day of these types of water were released into the pond. Radium and barium spontaneously co-precipitated as insoluble deposits of $BaSO_4 + RaSO_4$, which settled to the bottom of the pond. These types of ‘scales’ are characterized by highly enhanced concentrations of radium isotopes.

Polish Atomic Energy Agency inspectors measured gamma dose rates around the settling pond and conducted monitoring at 37 locations around the reservoir. A significant increase in dose rates was reported, with values of up to 42 μGy/h near the point of inflow of waters into the settling pond. The dose rate decreased with increasing distance from the discharge point, down to a value of 2 μGy/h at 15 metres from the end of the discharge pipe. This area is secured against entry of non-authorized persons. Lower gamma dose rates were measured in the remaining area, with values not exceeding a level of 1.2 μGy/h. By comparison, the national background gamma dose rates lie in the range 0.02–0.09 μGy/h [207−208].

Analyses from 1999 showed that the average activity concentrations for ^{226}Ra and ^{228}Ra were 2.27 and 2.37 kBq/m^3, respectively, in saline waters discharged into the pond. Approximately 9.5 GBq of each radium isotope are released into the pond per year. As a result of the high radium deposition rate, a detailed investigation of the bottom sediments was undertaken. The measurements were also evaluated to assess the radiation risk. The pond and the surrounding area are shown in Fig. 79.

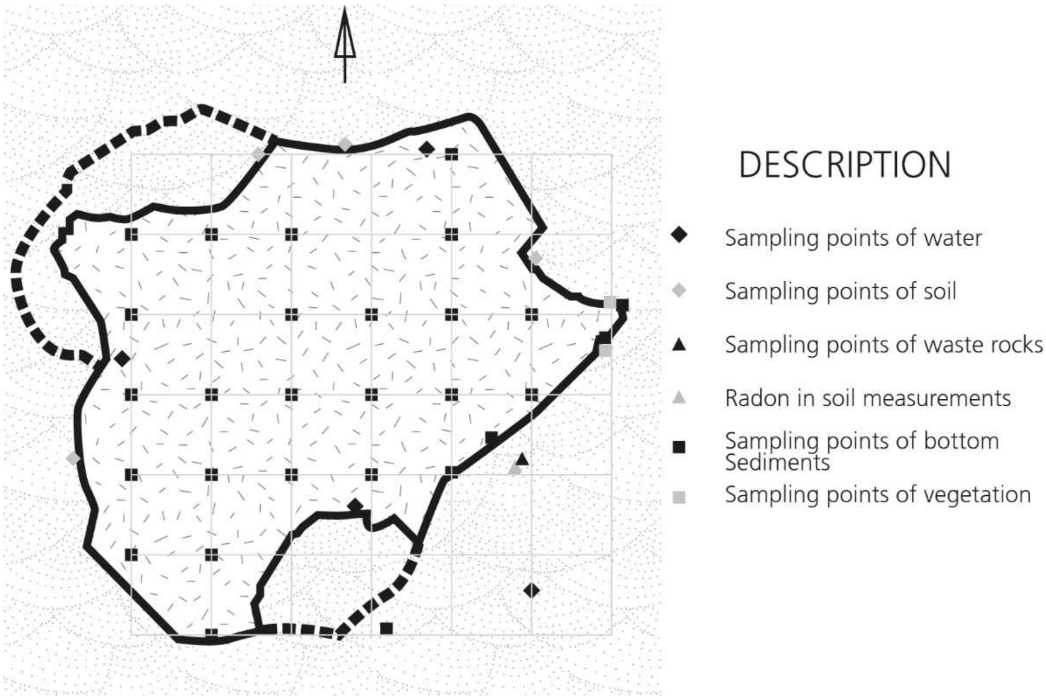

FIG. 79. Sketch plan of the reservoir (with sampling points).

Results of measurements

At each of the 28 sampling points in the reservoir, placed on a regular grid, water and sediment samples were collected and the following measurements were taken:

- Water depth above the sediments;
- Bathymetric shape of the reservoir bottom (by bathymetric scanning);
- Thickness of the sediments;
- Gamma dose rate above the water surface.

Gamma spectrometry (natural and artificial radionuclides) was performed on bottom sediments that had been collected at each sampling point; however, because the sediments were soft and water laden, it was not possible to collect samples of the different layers that represent different ages of sediments.

Results of radium measurements in water and sediment samples

Table 53 summarizes the radium activity concentrations in sediments from gamma spectrometry, which show that there is a large range in values (Fig. 80). Results of radium analysis of water samples are provided in Table 54, and Fig. 81 depicts the spatial distribution of total radium ($^{228}Ra+^{226}Ra$) activity concentrations in water when the settling pond was in use.

In the centre of the reservoir (distant from the point of inflow), both radium isotopes are more homogeneously distributed spatially, as there is little exchange of waters between the centre and the rest of the pond. In general, at locations where sediments had a higher radium activity concentration, the corresponding levels in water were lower than average.

TABLE 53. RESULTS OF RADIUM MEASUREMENTS IN BOTTOM SEDIMENTS FROM THE POND (n=28)

Bottom sediment statistics	**Activity concentration** (Bq/kg dry mass)	
	^{226}**Ra**	^{228}**Ra**
Average	5105	1407
Median	1191	593
Maximum	49 151	6388
Minimum	67	62

TABLE 54. RESULTS OF RADIUM MEASUREMENTS IN WATER SAMPLES FROM THE POND (n=28)

Water statistics	**Activity concentration (Bq/m^3)**	
	^{226}**Ra**	^{228}**Ra**
Average	3176	2931
Median	3609	3220
Maximum	3996	3740
Minimum	337	520

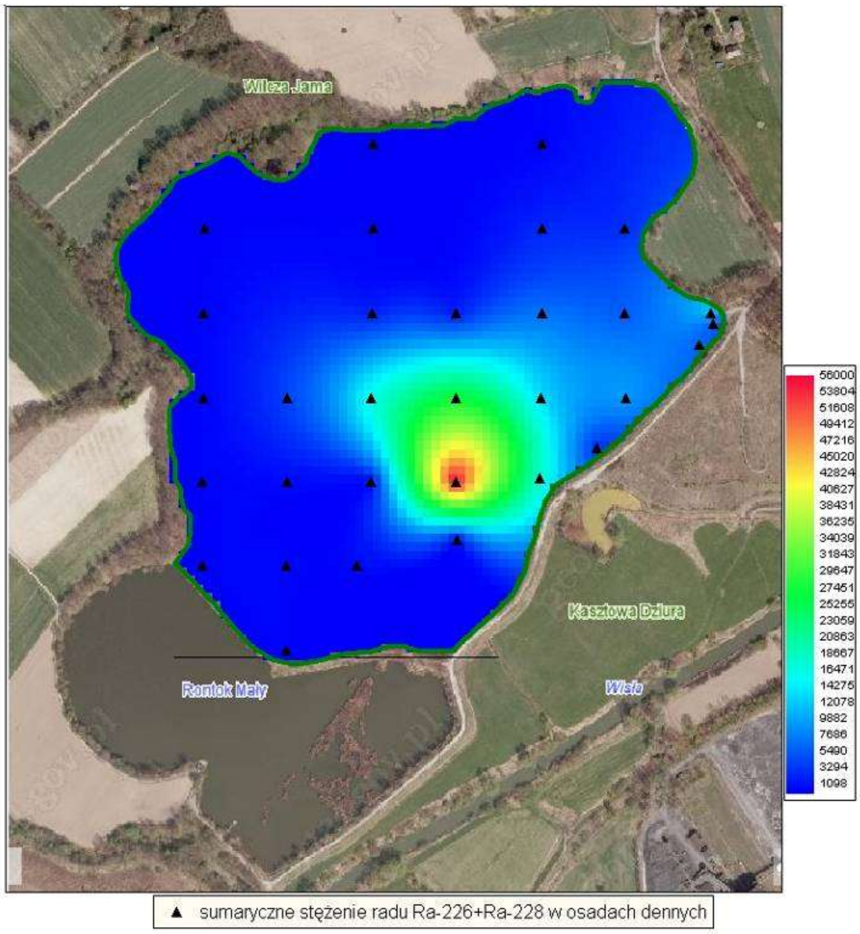

FIG. 80. Distribution of ^{228}Ra+^{226}Ra radionuclides in bottom sediments (Bq/kg dry mass).

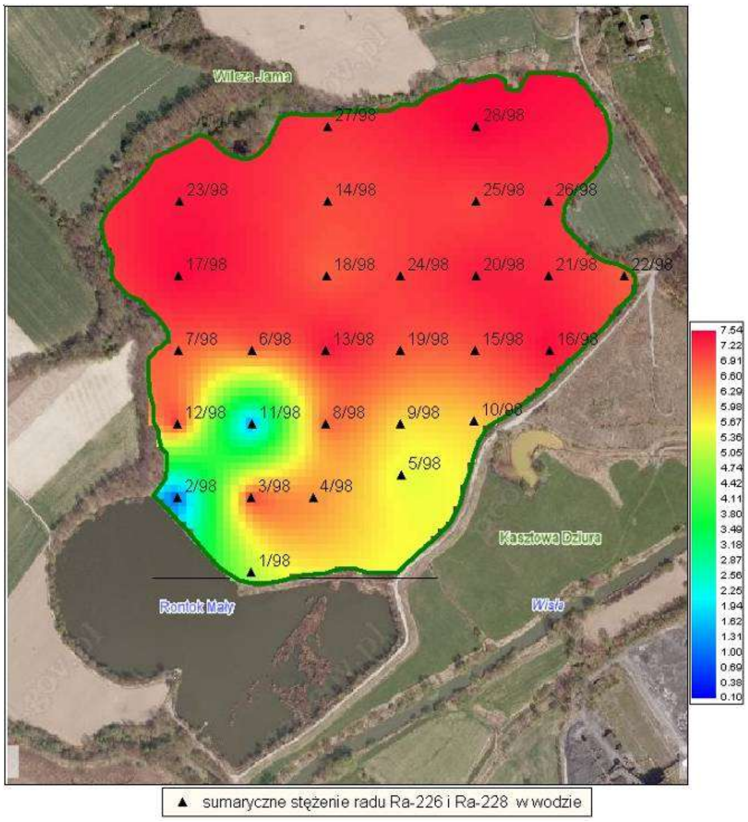

FIG. 81. Distribution of ^{228}Ra+^{226}Ra in water (Bq/L) (when settling pond was in use).

Additionally, to assess the extent of any contamination on the pond banks, soil samples have been collected, along with samples of waste rock used for bank construction. Gamma spectrometry measurements are shown in Table 55. The radium activity concentration in waste rock at one site was clearly elevated, likely due to contamination with mine sediments. The ^{137}Cs activity concentrations in soil samples are elevated due to fallout from the Chernobyl accident.

TABLE 55. ACTIVITY CONCENTRATIONS OF SELECTED RADIONUCLIDES IN WASTE ROCKS AND SOIL SAMPLES FROM THE BANKS OF THE SETTLING POND

Sampling type	**Activity concentration ranges (Bq/kg dry mass)**			
	^{226}Ra	**^{228}Ra**	**^{40}K**	**^{137}Cs**
Waste rocks	30–3508	40–2482	414–645	<2–79
Soil	29–74	33–77	377–811	77–238

Activity concentration of radon in soil gas

The contamination of the banks of a pond, due to sediment and waste rock with elevated radium concentrations, would be expected to lead to an increase of radon in soil. Therefore, measurements were taken to determine the radon concentration in soil gas along the pond bank. Soil gas that was collected from a depth of 1 metre, was transferred into Lucas cells and measured. Sampling locations are shown in Fig. 81. The maximum value of radon concentration in soil gas was 12.7 kBq/m^3. However, at many sampling locations, the activity concentration of radon was very low, likely as a result of the high water table in the ground around the pond.

Analysis of investigation results

According to UNSCEAR [9], the normal range of activity concentrations for radium isotopes in the Earth's crust is 17–60 Bq/kg dry mass and 11–64 Bq/kg dry mass, for ^{226}Ra and ^{228}Ra, respectively, whereas the range for ^{40}K is 140–850 Bq/kg dry mass. Table 56 presents values measured for select radionuclides in the current study relative to average values for the natural environment.

Activity concentrations of radium in sediments significantly exceeds the average value of the Earth's crust. This is due to the influence of radium-bearing mine waters. Barium and radium co-precipitate as insoluble sulphates, which settle out in the bottom of the pond, together with non-radioactive suspended matter which is released into the pond with mine waters. It is possible to estimate the rate of co-precipitation based on the activity concentrations of radium isotopes in the sediment. The wide range of radium activity concentrations that has been found in the sediments and their distribution indicate that the process of barium and radium co-precipitation and sedimentation principally takes place near the point of discharge of saline waters into the pond. However, large areas of the pond have very low sediment contamination, even where the activity concentration of radium in the water is elevated.

High radium content in waste rock only occurs near the point of inflow, perhaps as a result of dredging of bottom sediments, which are then placed on the banks of the pond. No contamination of soils on the adjacent land has been found. In addition, no radioactive contamination has been detected in the swamps that are used to supply therapeutic mud. Finally, contamination is limited to the bottom of the reservoir, except for a region around the outlet of the discharge pipe, carrying saline waters from the mine to the settling pond,

Cesium-137, an artificial radionuclide with a radioactive half-life of approximately 30 years, is present in elevated concentrations because of fallout from the Chernobyl accident in 1986. It is not present in deeper sediments, or in freshly excavated waste rocks. However, soil and sediment samples from areas with low sedimentation rates have enhanced levels of ^{137}Cs. The activity concentration of radium in these same sediments is low, indicating that they were transported to the pond from local fields. This is supported by the similar ratio of activities of radium and ^{137}Cs that have been measured in samples of therapeutic mud.

TABLE 56. SUMMARIZED RESULTS OF THE INVESTIGATION OF NATURAL RADIONUCLIDES IN SOLID SAMPLES IN THE AREA OF THE SETTLING POND

Radionuclide and range of concentrations in the Earth's crust (Bq/kg dry mass)	**Activity concentration (Bq/kg dry mass)**		
	Bottom sediments	**Waste rocks**	**Soils and therapeutic mud**
^{226}Ra (17–60)	67–49151	30–3508	22–74
^{228}Ra (11–64)	62–6388	40–2482	17–47
^{224}Ra (11–64)	65–8990	38–1791	17–44
^{137}Cs	<2–1014	<2–79	77–238
^{40}K (140–850)	155–848	414–664	102–811

Radium balance in settling pond sediments

Activity concentrations of radium in sediments are heterogeneous, and there is a considerable local enhancement of radionuclides above average values at many locations. This will, therefore, need careful consideration during remediation planning. The radium activity in bottom sediments is shown on Table 57.

Sediment thickness and water depth measurements were recorded during sampling, and a computer-generated bathymetric chart was then produced, and sediment thickness over the entire pond area was calculated. The maximum thickness of this layer is approximately 1.2 m and is located in the southeastern part of the reservoir, close to the discharge point of mine waters. Sediment thicknesses exceeding 0.4 m are found only in one small part of the pond covering approximately 2.5 ha (0.025 km^2). For the remaining 33 ha (0.33 km^2), the sediment thickness falls in the range of 0.1–0.4 m, occurring in the northern and western parts of the pond where a minimum thickness of precipitates can be found. It seems likely that inflows of fresh water are low in this part of the reservoir, resulting in a very slow rate of precipitation of radium and barium from the water column. Moreover, coal was absent here, and the radium content in sediments was also low (< 500 Bq/kg dry mass).

The bathymetric map indicates subsidence in the northern and eastern areas of the pond due to underground mining. Water depths are very shallow in the remaining area, so that sediments occur just below the water surface or sometimes even above the water, particularly in the area of the pond outflow to the Vistula River.

The maximum activity concentration of radium in sediments (approximately 55 kBq ^{226}Ra+^{228}Ra/kg dry mass sediments) is in the southern area of the pond, midway between the mine water inflow and the pond outflow to the Vistula River (see Fig. 82). Near the inflow from the pipeline, the average value is lower (10 kBq ^{226}Ra+^{228}Ra/kg dry mass sediments), likely due to faster deposition of non-radioactive suspended material and much slower settlement of fine radium and barium sulphate crystals.

A significant proportion of sediments (35–40%), mainly in the north and west areas of the pond, has average radium activity concentrations of less than 350 Bq/kg dry mass for ^{226}Ra and 230 Bq/kg dry mass for ^{228}Ra. These were proposed as maximum permissible activity concentrations for radium in such deposits and waste rocks, above which the material would be considered waste that would need to be safely stored and managed [209].

The total activity of radium isotopes that was deposited in the pond is approximately 315 GBq, most of which (at least 240 GBq) is ^{226}Ra, with a half-life of 1620 years. In addition, 74 GBq of the total activity is from ^{228}Ra, with a half-life of only 5.8 years. Therefore, enhanced radioactivity from mining activities will remain for centuries after the end of mining, and without proper remediation, might affect people in the vicinity of the pond now and in the future. Table 58 indicates that the radium activity concentration in the surface water of the pond is negligible likely because the amount of water present in the settling pond lowers the activity concentration values.

For ^{228}Ra, correction due to decay is a significant factor and it is assumed that the activity concentrations of ^{228}Th and ^{224}Ra in sediments will becomes equal to 1.4 times the activity concentration of ^{228}Ra. Activity concentrations of ^{210}Pb and ^{210}Po can be calculated by assuming that just following sedimentation, their activity concentrations in sediments were close to zero.

TABLE 57. AMOUNT AND TOTAL ACTIVITY OF SETTLED BOTTOM SEDIMENTS IN THE POND

Area of the reservoir (m^2)	Deposits volume (m^3)	Water volume (m^3)	Total activity of ^{226}Ra (Bq)	Total activity of ^{228}Ra (Bq)	Total activity of ^{226}Ra+^{228}Ra (Bq)
360 000	113 107	262 084	2.40×10^{11}	7.4×10^{10}	3.14×10^{11}

TABLE 58. RADIUM ACTIVITY CONCENTRATIONS IN WATER FROM THE SETTLING POND

Year	Ra-226 (Bq/L)	Ra-228 (Bq/L)
2005	0.101	0.04
2006	0.272	0.15
2007	0.213	0.03
2008	0.152	0.05
2009	0.106	0.04
2010	0.065	0.02

IV.4.2. Bojszowy Reservoir, Upper Silesian Coal Basin, South of Poland (approximately 50 km from Katowice)

The Bojszowy Reservoir is situated approximately 50 km from Katowice in the south of Poland within the Upper Silesian Coal Basin. Saline, radium-bearing waters from two mines are released into the Bojszowy Reservoir. The pond waters are released into the small river Gostynka, a tributary of the Vistula. Information regarding contamination has been published previously [210], with much of this published information being used as the basis of this section. The inflow from the 'Czeczott' Coal Mine to the pond is approximately 15 000 m^3/day, whereas from the 'Piast' mine, the discharge is approximately 20 000 m^3/day. In both cases, the type of water is the same (type B – no barium, only radium and sulphate ions). The activity ratio of ^{226}Ra:^{228}Ra is 1:2. Due to the absence of barium, radium does not precipitate in the pond. Nevertheless, as a result of sorption, sediments have enhanced activity concentrations of radium of up to several hundred Bq/kg dry mass. From 1980 until 1999, approximately 227 million m^3 of water were discharged into the reservoir.

Bojszowy Reservoir – Radium-bearing waters

Waters released into the Bojszowy Reservoir are of type B, with higher concentrations of ^{228}Ra than ^{226}Ra. The average radium activity concentrations in discharges from the Piast Mine were approximately 4.1 kBq/m^3 for ^{226}Ra and approximately 7.2 kBq/m^3 for ^{228}Ra. Corresponding values for inflows from the Czeczott Mine were lower, with values of 3.2 kBq/m^3 for ^{226}Ra and 4.9 kBq/m^3 for ^{228}Ra. These are average values over 2 years.

The annual total activity of Radium (^{226}Ra + ^{228}Ra) within water flowing to the Bojszowy settling pond has been calculated to be approximately 124 GBq, comprising 55% of the total radium carried in waters from all the coal mines of Poland [211]. The average radium activity concentrations in waters discharged to the pond were calculated to be 3.6 kBq/m^3 for ^{226}Ra and 6.2 kBq/m^3 for ^{228}Ra.

TABLE 59. RADIUM ACTIVITY CONCENTRATIONS IN WATER SAMPLES FROM BOJSZOWY RESERVOIR (n=42)

	Radium activity concentration (kBq/m^3)	
	^{226}Ra	^{228}Ra
Average	3.45	6.95
Median	3.34	6.76
Maximum	5.21	8.32
Minimum	2.12	4.67

The reservoir water and bottom sediments were sampled in 1996. The grid spacing was 50 m × 50 m. The activity concentration of radium from 42 water samples is presented in Table 59.

The distribution of radium in water in the Bojszowy reservoir is more uniform than in the settling pond described in the previous section. The main reasons seem to be a less variable radium activity concentration in water flowing into the pond and the type of the water from which radium is removed only by sorption, as there is no carrier for co-precipitation of radium.

By comparing the results of radium analysis of inflows and outflows from the Bojszowy reservoir to the Gostynka river, it has been calculated that only 2.9% of the ^{226}Ra and 3.3% of the ^{228}Ra activities remain in the pond and are sorbed onto bottom sediments. The average radium activity concentration in the outflow from the settling pond has been calculated to be similar to that of the inflows, with values of 3.5 kBq/m^3 for ^{226}Ra and 6.0 kBq/m^3 for ^{228}Ra. This means that only small amounts of radium are deposited within the pond. Moreover, the total activity concentration of radium in water released to the Gostynka stream is approximately 10 kBq/m^3, which is more than 10 times greater than the permitted level for wastewater. A significant improvement of the situation was achieved in 1998, due to the start-up of an underground treatment plant for mine waters in the Piast Colliery.

More than 95% of the radium is discharged with saline waters into the Gostynka river. Upstream of the point of discharge, activity concentrations of radium isotopes are low (<0.1 kBq/m^3); such a value is typical for Polish groundwater and river waters [212]. Downstream of the point of discharge, the radium activity increases rapidly. Typically, the activity concentration of radium does not exceed 0.7 kBq/m^3, during spring and winter when water levels are higher. However, during summer, the activity concentration of ^{226}Ra in Gostynka river varies within the range of 0.5–0.7 kBq/m^3, whereas for ^{228}Ra the concentration is higher, falling within the range of approximately 1.0–1.3 kBq/m^3. The total activity concentration of radium in the Gostynka river can reach 1.5–2.0 kBq/m^3 [213]. Some radium (several relative percent) is adsorbed onto sediments, whereas most of it is transported into the Vistula River. In this large river, due to dilution and further adsorption to particles, the radium activity concentration in water decreases [212−213].

Bottom sediments

Sediment samples were taken at the same sampling locations as water samples. Drilling of boreholes was conducted into the bottom of the settling pond, and sediment cores were collected and subsequently analysed by gamma spectrometry. Activity concentrations of ^{226}Ra in sediments from the Bojszowy Reservoir fell within a range of 95–950 Bq/kg dry mass, with activity concentrations of ^{228}Ra ranging between 124 and 1705 Bq/kg dry mass [213]. In almost all cases, these sediments characteristically showed activity concentrations of ^{228}Ra and ^{224}Ra that were close to equilibrium, and frequently the activity concentration of ^{226}Ra was only slightly less than that of ^{228}Ra, which implies that the sediments are at least a few years old and that adsorption is slow. Recent deposits are found in only a few locations far from the banks. Radium activity concentrations in bottom sediments are shown in Table 60.

Based on these measurements, the balance of radium in bottom deposits in the settling pond are shown in Tables 61 and 62. It was assumed that the distribution of radium isotopes in the bottom sediments was uniform, and therefore, the average activity concentrations of both radium isotopes was used for calculations. Based on these calculations, over 19 years of operation of the Bojszowy Reservoir, the total ^{226}Ra activity accumulated in sediments is approximately 66 GBq, and that of ^{228}Ra is 100 GBq. The rates of deposition are approximately 3.5 GBq/year for ^{226}Ra and 5.8 GBq/year for ^{228}Ra, which represents only 7% of the annual aquatic discharge of radium from the mine into the settling pond. Previous calculations estimated lower rates of deposition (~4% per year), but these past assessments were not very accurate because they did not take into account the large uncertainties in the parameters measured.

The above results can be used to describe the radium behaviour in both settling ponds and rivers and can be used to correlate them with the chemical composition of radium-bearing waters.

A natural succession of flora was observed after the settling pond dried. Between 2000 and 2007, approximately 80% of the total settling pond surface became overgrown by a variety of plant species. In 2008, a survey was conducted, which revealed more than 40 plant species. Of these, 12 species were dominant. The sediment-to-plant transfer factor was measured for three species, and the relationship was assessed between radium activity concentration in sediments and the transfer factor. Dose rates and radium activity concentration of up to 3 μSv/h and 7 kBq/kg dry mass, respectively, were measured in hot spots.

TABLE 60. RADIUM ACTIVITY CONCENTRATIONS IN BOTTOM SEDIMENTS FROM THE BOJSZOWY RESERVOIR

	Radium concentration	
	^{226}Ra (Bq/kg dry mass)	**^{228}Ra (Bq/kg dry mass)**
Average	414	627
Median	406	628
Maximum	950	1705
Minimum	95	124

TABLE 61. ASSESSMENT OF THE AMOUNTS OF DEPOSITS IN BOJSZOWY RESERVOIR AND THE TOTAL RADIUM ACTIVITY IN DEPOSITS

Area of the pond (m^2)	**Volume of deposits (m^3)**	**Amount of water (m^3)**	**Total activity of ^{226}Ra (Bq)**	**Total activity of ^{228}Ra (Bq)**	**Amount of radium in the pond ^{226}Ra+^{228}Ra (Bq)**
160 000	240 000	262 084	6.6×10^{10}	1.00×10^{11}	1.66×10^{11}

TABLE 62. RADIUM BALANCE IN BOJSZOWY RESERVOIR

Settling pond	Area (m^2)	Volume of deposits (m^3)	Total amount of radium discharged into pond (GBq)	Amount of radium deposited in the pond (GBq)	Deposition rate (%)
Bojszowy	160 000	240 000	2356	166	7

The process of remediation of contaminated land in the area of the Bojsziwy Reservoir has been completed. Sediments were left in situ and covered with inert material; initially, a 1 m thick layer of waste rock from the coal mine was used, followed by a thin sand layer, which was put on top. The upper level of the settling pond area is now 2–3 m above the level of neighbouring land but is below the banks (see Fig. 82).

FIG. 82. Photographs of the settling pond area.

IV.4.3. Chwałowice stream

Chwałowice Stream is a small river in the vicinity of a coal mine, which received releases of mine waters over many years. In the past, it was a part of the mine dewatering system. The nearest coal mine is one of the oldest coal mines in Poland (more than one hundred years old) and is still in operation. Currently, the water from this coal mine is discharged to the mine water collector and then discharged directly to a large river (the Oder River).

The discharge of water with a high suspended matter concentration will result in an increase in the thickness of bottom sediments. The bed of the Chwałowice Stream has been dredged many times and the sediments placed near the stream banks. There are no data available regarding the radium activity concentrations of removed sediments. Results of recent screening measurements on land along the Chwałowice Stream indicated an increment in the dose rate of up to 6 μSv/h (see Fig. 83). Activity concentrations of ^{226}Ra in soil samples were as high as 7 kBq/kg dry mass, but activity concentrations of ^{228}Ra are quite low, with values of up to only 75 Bq/kg dry mass. As the activity concentrations of both radium isotopes in mine water are comparable and the half-life of ^{228}Ra is 5.75 years, the contamination had to have occurred at least several years ago. The existing data from 1982 and 1986 indicates that soil contamination reached a level of 53 kBq/kg dry mass for ^{226}Ra in some hot spots on the nearest arable land.

The extent of contaminated land has not been fully assessed, however, along the stream banks, it likely covers a few hundred metres or more. The area currently neighbouring the banks of the stream is overgrown by different species of herbaceous plants. In the immediate vicinity, arable land and inhabited buildings also exist.

FIG. 83. Dose rate measurements along the Chwalowice Stream. The red dots are sampling locations.

IV.5. THE COAL-FIRED POWER PLANTS OF COMPOSTILLA II (LEON – SPAIN)

IV.5.1. General description

One of the largest coal power plants in Spain, Compostilla II, was located in Ponferrada (Leon province, Spain) (see Fig. 84). This facility belongs to ENDESA, a Spanish electrical utility company, and has five production groups (I–V), commissioned from 1961 to 1985, that used local and imported coal. Group I was stopped in 2002, and in 2009, a decision was taken to gradually substitute this group and groups II and III with gas-combustion plants.
In December 2018, ENDESA applied to the Spanish authorities to close the remaining three coal-fired groups. The Compostilla II thermal power plant was permanently closed on 30 June 2020. In 2021, the decommissioning and dismantling activities started.

FIG. 84. View of the Compostilla II coal-fired power plant, showing the four stacks, the two cooling towers and the two main groups.

The Compostilla II thermal power plant was included within a national project that analysed the main NORM industries in Spain. The analysis included this plant and three other Spanish coal-fired power plants with different characteristics, fuels and geographic situations. The project was carried out by the Unit of Radiological Protection of the Public and the Environment from CIEMAT, together with the laboratory of environmental radioactivity (LARUEX24) from the University of Extremadura.

A complete radiological characterization of the power plant and dose assessments for workers and members of the public were performed for Compostilla II. This included the management of residues and maintenance operations within the boilers. Part of the fly and bottom ashes have been dumped to a surface store on the site of the plant, which are uncovered and near a small river. The main data used for the dose assessment, including meteorological data and the radioactive characterization of materials, are provided in Sections IV.5.3 and IV.5.4, respectively.

Layout of the site

Compostilla II is sited in the northwest of Spain at a latitude of 42.56 N and a longitude of 6.58 W (see Fig. 85). The site is situated at an average altitude of 512 m above sea level in a continental Mediterranean climate, with relatively humid and temperate weather. The annual average temperature is 13ºC, with an annual average precipitation of 640 mm and an annual average relative humidity of 70%.

FIG. 85. Location of the Compostilla II plant in Spain.

The plant is close to the reservoir Bárcena and a small river used for irrigation of vegetables grown on the banks.

IV.5.2. Source term

Within the plant, there were two storage sites for coal and one storage site for residues. The residues that have not been recycled, including the fly and bottom ashes from the five groups, have been stored without cover.

In the beginning of its operation, the plant used low quality coal from the local mines of el Bierzo. Later, it used a mixture of local and imported coal, mainly from South Africa and Ukraine. The transport of coal to the plant was carried out by trucks and train.

A schematic illustration of a coal-fired power plant is provided in Fig. 86.

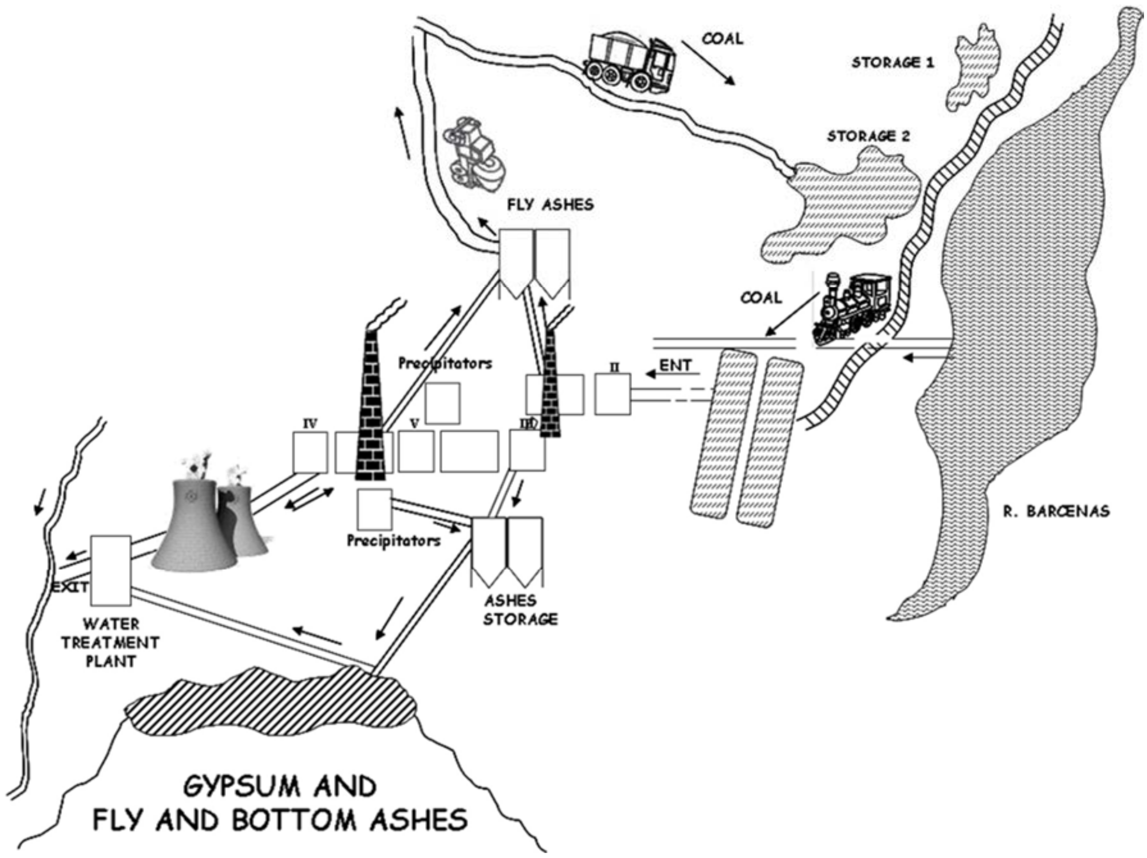

FIG. 86. Schematic illustration of a coal-fired power plant.

The total electrical output of the Compostilla II thermal power plant in 2015 was 1200 MW. A summary of power by group is provided in Table 63.

TABLE 63. SUMMARY OF DATA OF COMPOSTILLA II

Group	Power (MW)
I	Out of service since 2008
II	147.9 (Out of service since 2016)
III	337.2 (Out of service since 2020)
IV	358.6 (Out of service since 2020)
V	355.9 (Out of service since 2020)

A large proportion of the generated fly ash has been recycled and used in building materials. When this was not the case, the fly ash was treated as conventional material and dumped in an uncovered storage inside the plant, together with bottom ashes and gypsum created during the desulphuration process that took place in specific filters which removed much of the sulphur in the flue gases.

The main filters used in the plant were electroprecipitators that remove more than 99.9% of the fly ash flowing in the flue gases, and desulphurators that convert limestone into gypsum in a wet process.

Water used for the treatment of solid residues was processed within a treatment plant, together with rainwater and service water, to treat it before discharge to the nearby river.

Flue gases and some solid particles not captured by filtration were released to the atmosphere through two stacks with heights of 270 m and 290 m, respectively. The approximate diameter of each stack at the top was 8 m. The flow rate of the flue gases emitted by the first stack was 2×10^6 m^3/h and 3×10^6 m^3/h for the second one. The average temperature of the gases when discharged was 120ºC.

The total coal used was typically approximately 3.2×10^6 t/a. The corresponding quantities of fly and bottom ash generated were 866 231 t/a and 96 248 t/a, respectively. The estimated quantity of suspended particles emitted through the stacks was 4.331×10^5 kg/a.

The quantity of material in uncovered storage was approximately 20 million tons.

IV.5.3. Site specific data

Meteorological data

Hourly meteorological data for two consecutive years, measured from a local meteorological tower, are available. They include precipitation rate, solar irradiation, barometric pressure, wind velocity, wind direction and temperature at three different heights (10 m, 25 m and 60 m). Two-year averaged wind rose data are provided in Fig. 87.

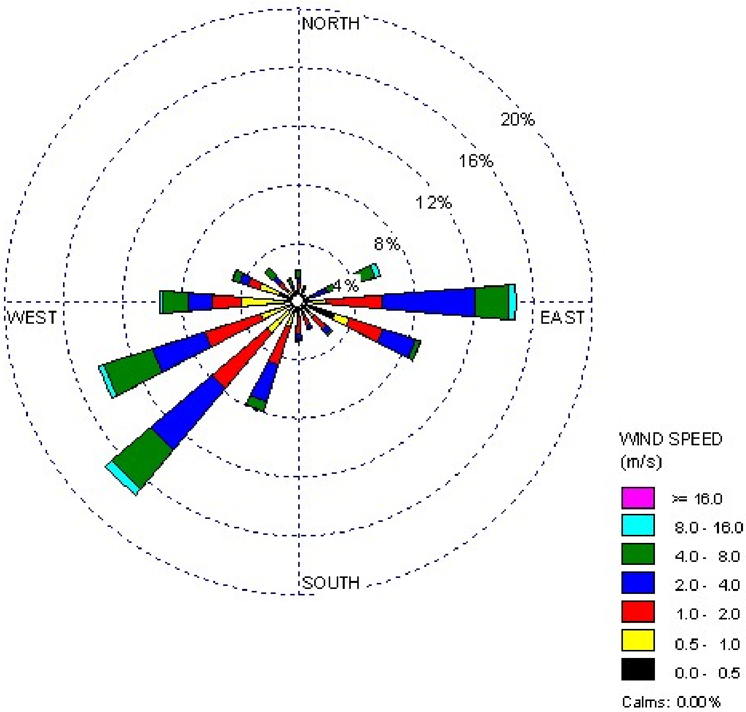

FIG. 87. Two-year averaged wind rose (data from the meteorological tower at 60 m).

On average, the data indicates a Pasquill stability category D, an average pressure of 944 mbar, and a relative humidity of 65%. The average temperature at 10 m, 25 m and 60 m, is 10.0ºC, 10.9ºC and 11.9ºC, respectively.

Additional data on the Compostilla II site has been previously published (see Refs [214−215]).

IV.5.4. Radiological environmental impact assessment

Description of the representative person

For the REIA performed for the plant, five groups of workers were selected (Table 64). For the public, the three closest villages were considered. Data for the main population groups within the vicinity of the plant are presented in Table 65. Water from the local small river may be used as drinking water for consumption by humans or livestock, and for irrigation of crops that subsequently may be consumed by the local population.

TABLE 64. WORKERS GROUPS CONSIDERED IN THE EVALUATION

Workers groups
Drivers of bulldozers in the coal piles
Maintenance operators for the plant
Drivers of the trucks carrying fly ash
Workers at the waste storage
Boiler maintenance workers

TABLE 65. MAIN POPULATION IN THE VICINITY OF THE PLANT

Name	Distance (km)	Direction	Population
Ponferrada	7.6	SSE	67 969
Cubillos del Sil	1.4	NNE	1656
Congosto	3.9	ENE	1747
Toreno	10.9	NNE	3806

Radiological characterization

All the materials used and produced in the coal-fired power plant were radiologically characterized. The range of activity concentrations measured in coal and other subproducts are provided in Table 66. The equivalent ambient dose rate (also known as H*(10) results) for areas of the power plant are presented in Fig. 88.

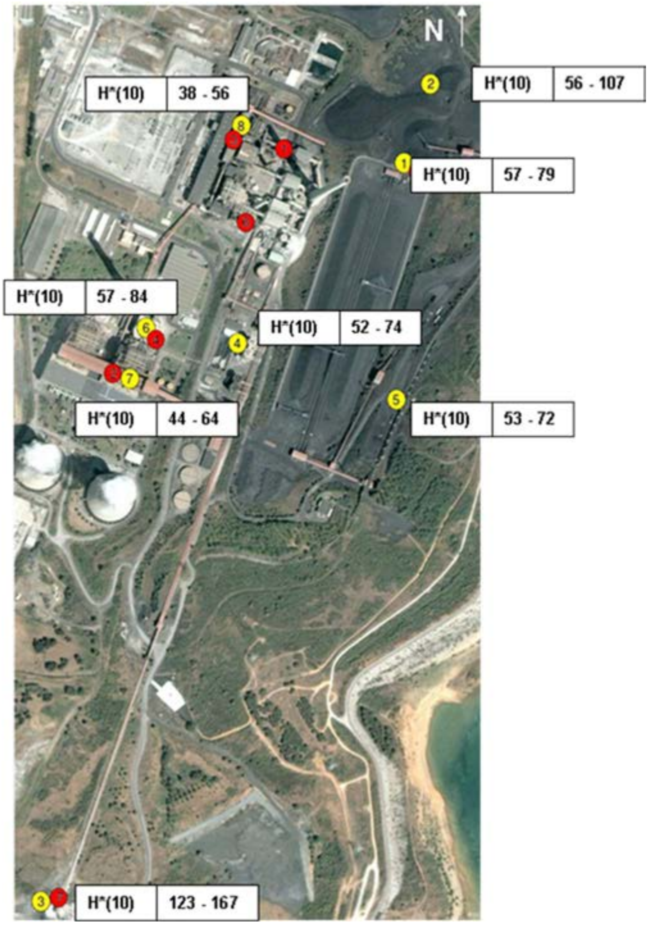

FIG. 88. Ranges of the measurements of H(10) (nSv/h) in the installation.*

TABLE 66. RANGE OF ACTIVITY CONCENTRATIONS MEASURED IN COAL AND OTHER MATERIALS

	Activity concentration (Bq/kg dry mass)							
Sample	**Po-210**	**Th-232**	**Th-230**	**Th-228**	**Ra-226**	**U-238**	**U-234**	**K-40**
Coal	79–181	28–46	36–42	44–52	10–37	36–62	34–52	113–358
Fly Ash	300–590	81–119	87–148	108–181	83–147	71–112	74–119	1030–1196
Bottom Ash	6–113	81–190	83–227	93–250	76–158	76–114	67–120	950–1140
Limestone	82–455	< 2	–	–	11–19	–	–	< 8
Gypsum	79–271	< 5	–	–	5–15	–	–	< 5

To assess the annual effective dose to boiler maintenance workers, several measurements were performed. Figure 89 identifies where the measurements were taken, and which types of measurements were made. The H*(10) results obtained from the measurements outside the boiler during the first phase of this study are provided in Table 67. The results for in situ surface contamination inside the boiler and H*(10), measured in the second phase of this study, are presented in Table 68. The results from alpha and gamma spectrometry carried out in the laboratory in the third and last phase are presented in Table 69. In this case, measurements correspond to the activity on the surface of the tubes, which were chemically treated to extract possible lead or polonium deposits.

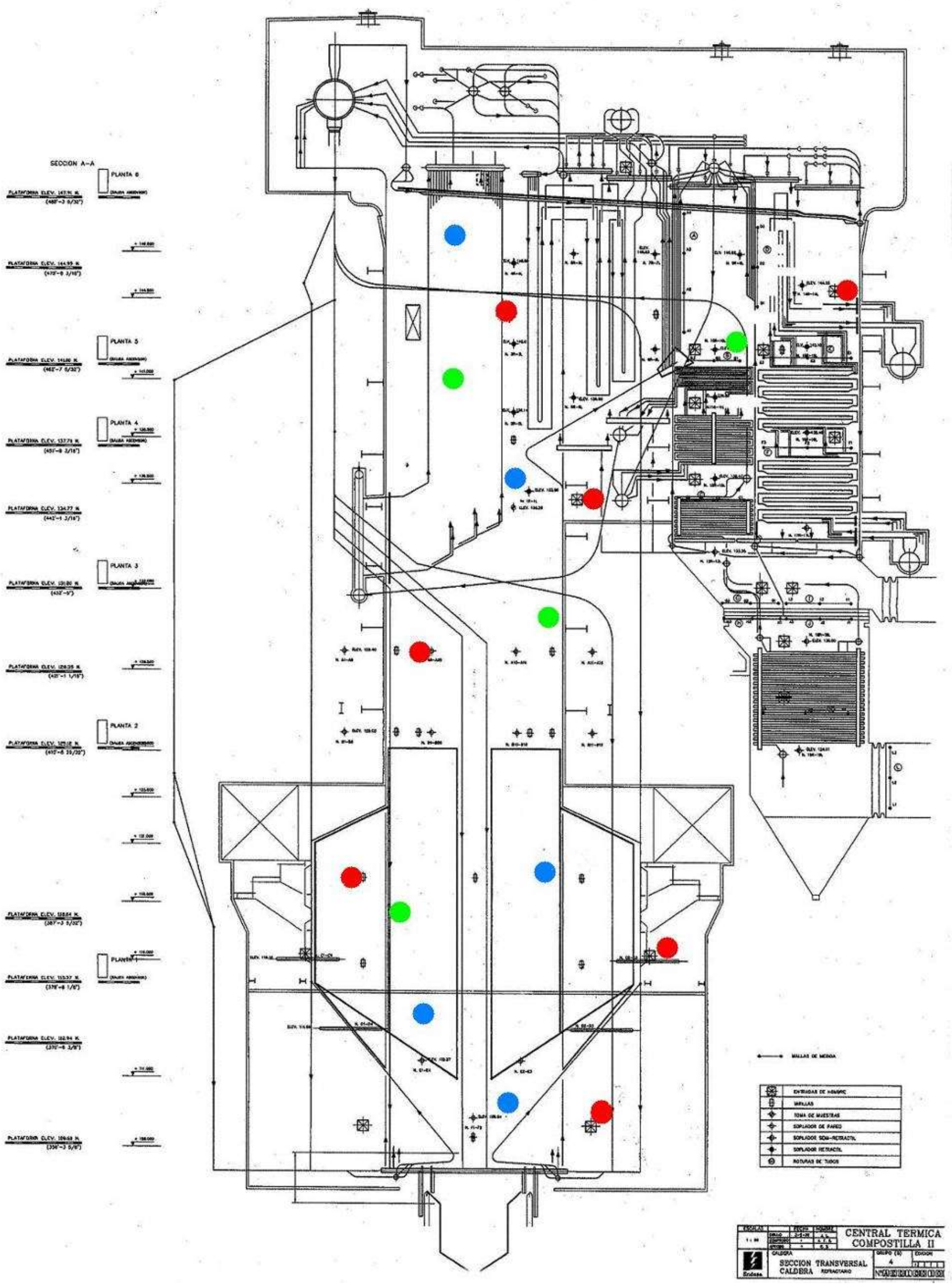

FIG. 89. Diagram showing the location of measurements taken from the boiler. Marked in red are the points where H(10) was measured from the external parts of the boiler; in blue the points where H*(10) and β contamination were measured inside the boiler; and in green the points where tubes were sampled for laboratory measurements.*

TABLE 67. H*(10) MEASURED AT SEVERAL LOCATIONS FROM THE OUTER PART OF THE BOILER DURING MAINTENANCE WORK (see Fig. 91 for locations)

Point of identification	$H^*(10)$ (µSv/h)
Background 1. Level 33 m	0.010
Background 2. Level 33 m	0.011
Background 3. Level 0.	0.028
Point 1. Level 33 m	0.087
Point 2. Level 33 m	0.050
Point 3. Level 30 m	0.054
Point 4. Level 24 m	0.042
Point 5. Level 20 m	0.053
Point 5. Level 10 m	0.105
Point 6. Level 7 m	0.094
Point 7. Level 0 m	0.065

TABLE 68. H*(10) AND BETA COUNTING MEASUREMENTS INSIDE THE BOILER (see Fig. 91 for locations)

Point of identification	$H^*(10)$ (µSv/h)	β counting(cps)
Point 8. Level 37 m	0.026	19–20
Point 9. Level 37 m	0.015– 0.021	20
Point 10. Level 28 m	0.053–0.060	40–47
Point 11. Level 28 m	0.014–0.016	–
Point 12. Level 10 m	0.020	–
Point 13. Level 10 m	0.020–0.024	–
Point 14. Level 10 m	0.050–0.060	40–120
Point 15. Level 0 m	0.036–0.040	85–87
Point 16. Level 0 m	0.1	37

TABLE 69. ^{210}Po SURFACE ACTIVITIES MEASURED FROM SELECTED TUBES

Sample description	Po-210 (mBq/cm^2)
Boiler's water walls	3.6 ± 0.6
On radiant	25 ± 6
On radiant	18 ± 4
On primary	29 ± 6
On the end	24 ± 6
Reheating	7.0 ± 1.1

Finally, to assess the annual effective dose to the public, several measurements were taken. The range of activity concentrations measured in water from the small river, upstream and downstream of the power plant, are presented in Table 70. As there are big differences in the pluviometry (measurement of rainfall) for the dry and wet seasons, values are presented for both cases. Several samples were taken from vegetables grown next to the river and activity concentrations are provided in Table 71.

Other measurements were taken from areas surrounding the power plant. However, all the measured values are of the same order of magnitude and may not be represent radionuclides released from the plant. As an example, ^{210}Po measurements are provided in Fig. 90.

TABLE 70. ACTIVITY CONCENTRATIONS IN WATER FROM THE RIVER (Bq/m^3)

Sample	U-234	U-238	Ra-226	Po-210
Dry season upstream	5.0 ± 0.8	3.5 ± 0.6	4.1 ± 0.9	1.5 ± 0.3
Dry season downstream	5.3 ± 1.1	5.1 ± 1.1	2.7 ± 0.5	2.5 ± 0.6
Wet season upstream	28.4 ± 1.8	21.9 ± 1.4	3.3 ± 0.9	1.3 ± 0.4
Wet season downstream	9.3 ± 0.7	7.3 ± 0.9	4.6 ± 1.4	7.5 ± 1.9

TABLE 71. ACTIVITY CONCENTRATIONS IN VEGETABLES (Bq/kg fresh mass)

Sample	Activity concentration (Bq/kg fresh mass)		
	K-40	Ra-226	Th-232
Cauliflower	175 ± 4	0.5 ± 0.2	0.9 ± 0.6
Cabbage	189 ± 4	0.4 ± 0.2	< 1
Apple	470 ± 18	< 9	< 5

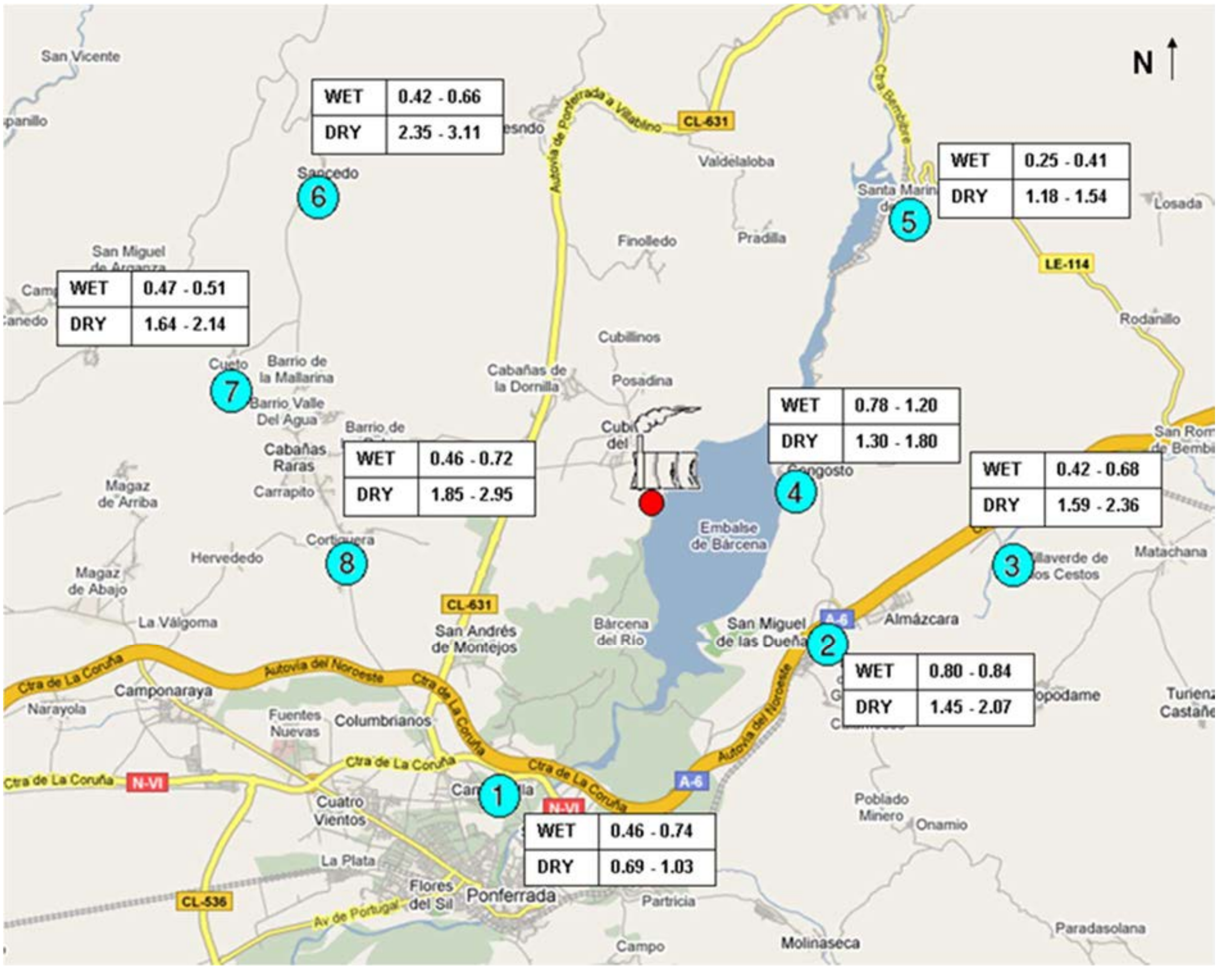

FIG. 90. ^{210}Po measurements in air (mBq/m^3).

APPENDIX V.
AVAILABLE MODELS FOR RISK AND ENVIRONMENTAL IMPACT ASSESSMENTS

A range of models has been developed over the years to perform risk and environmental impact assessments, for example, relating to radioactive waste or radioactive releases in planned and emergency exposure situations. The models range from simple screening tools to complex modelling codes embedded in general purpose simulation packages with various applications. Some models are available free of charge, while others are only commercially available. This Appendix describes models used by participants for the modelling work performed as part of EMRAS II WG2 (Table 72), in addition to some other available models and screening tools (Table 73). An overview of alternative modelling approaches and models that implement them is provided in Section 5.

V.1. MODELS USED FOR WORK IN EMRAS II WG2

The models used by participants as part of WG2 work are listed in Table 72, together with a brief description of each model. More detailed descriptions are provided in the following Sections V.1.1−V.1.10 of this Appendix.

V.1.1. CROM

CROM[63] is a generic environmental modelling code, which has been developed by CIEMAT[64] with the Polytechnic University of Madrid, based on the approach described in Ref. [61], with some variations from Ref. [216]. The code allows automated calculation of radionuclide activity concentrations in different environmental compartments and their impact on radionuclide transfer in food chains. Estimated activity concentrations are also used in the calculation of annual effective dose to humans, which allows the assessor to focus on analysis and interpretation of model results. A user's manual has been published to facilitate the application of the models that may be implemented within the CROM code [96−97]. Quality control of the code was undertaken by CIEMAT and RPD-HPA[65] [98] in support of its distribution by the IAEA as a reference tool applying its models. Training has been provided through IAEA's international model validation programmes (e.g. MODARIA II). The current version of the code is CROM 8.

CROM applies generic models for diffusion and dilution, which have adequate flexibility for use in more realistic calculations applying site specific parameters. The source term (types and quantities of radionuclides to be released) and the mode and characteristics of release are specified as model input parameters to estimate the radionuclide activity concentrations in environmental media at up to five receptor locations. Figure 91 presents the main screen with the dispersion modules of the CROM model.

CROM uses a Gaussian plume model for air dispersion, which assesses annually averaged radionuclide activity concentrations in air. The CROM model has been validated for distances of less than 20 km from the point of discharge. The code can be applied to calculate the rate of deposition at various locations in an area of interest, assuming 30 years of continuous discharge

[63] CROM is available for free from: ftp://ftp.ciemat.es/pub/CROM (see also https://radioecology-exchange.org/content/crom)
[64] Centro de Investigaciones Energéticas, Medioambientales y Tecnológicas.
[65] Radiological Protection Division of the Health Protection Agency, United Kingdom (Health Protection Agency was renamed Public Health England in 2013).

and neutral atmospheric stability conditions (based on Safety Reports Series No. 19[66], which was applicable to planned exposure situations). The model includes the effects of buildings near the discharge and the effect of ground roughness in the wind profile. The geometric mean wind speed and the wind direction at the height of the point of discharge are the basic meteorological variables needed for each individual calculation of radionuclide activity concentration in air. It is possible for assessors to use other diffusion factors, different atmospheric stability classes, and to introduce alternative effective heights of discharge in the code, but the code cannot be used to calculate them.

Dispersion in rivers, small lakes, large lakes, estuaries and along the coast of oceans and seas can be estimated using the surface water models in CROM. These models have been developed using analytical solutions to dispersion or advection-diffusion equations that describe radionuclide transport in surface waters under steady state, uniform flow conditions. Introduction of radionuclides into surface water from routine (planned) atmospheric discharge is considered for small lakes. Default values have been provided in the models within the CROM code for a large number of parameters and these can be used in the absence of site specific (local) information or data. The models can also be applied to large lakes and sewerage systems (see Footnote 67).

It is possible to input radionuclide data from both the atmosphere and the hydrosphere into the terrestrial food chain models, which also account for buildup of radionuclides on surface soil over 30 years. Radioactive decay and radionuclide ingrowth are accounted for in estimating retention of radionuclides on vegetation surfaces and on soil, and in assessing losses due to radioactive decay, which might occur between the time of harvest and consumption of vegetables. Types of foodstuffs that are considered in CROM are milk, meat and vegetables. Radionuclide uptake and retention by aquatic biota is estimated using element specific concentration ratios (also called ‘bioaccumulation factors’) for selected radionuclides, which assume steady state activity concentrations for a given radionuclide between aquatic biota and surface water. Types of aquatic foodstuffs that are considered in CROM are freshwater fish, marine fish and marine shellfish (also called ‘seafood’).

The maximum annual effective dose for combined internal and external exposure is estimated using estimated radionuclide activity concentrations in air, soil, surface water, sediments, and foodstuffs, along with annual intake rates, occupancy factors and the appropriate dose coefficients, assuming 30 years of discharge; the maximum annual effective dose is determined by summing annual effective dose values (internal and external) for all radionuclides under consideration. Dose coefficients for external exposure were calculated using the coefficients and equations provided in Ref. [128], which has now been superseded, and dose coefficients for internal exposure have been taken from Ref. [3]. External gamma dose rates from cloud immersion, deposition on soil, water submersion and deposition on sediment are accounted for in the model for gamma and beta exposure. For the six age categories[67] recommended by the ICRP, annual effective doses from external exposure and intake of radionuclides are calculated [217]. The code contains a database with default data for 152 radionuclides and provides eight examples of model application based on several examples that were presented in Safety Reports Series No. 19 (see Footnote 66).

[66] INTERNATIONAL ATOMIC ENERGY AGENCY, Generic Models for Use in Assessing the Impact of Discharges of Radioactive Substances to the Environment, Safety Reports Series No. 19, IAEA, Vienna (2001). Superseded by Ref. [61].

[67] ICRP 101a [217] recommends that annual dose to the representative person be defined by three age classes (1 year old infant, 10 year old child, adult). ICRP Committee 2 also provided dose coefficients for six representative age classes for use in circumstances where other age classes need to be considered: 3 month old infants, 1 year old, 5 year old, 10 year old and 15 year old children, and adults (>17 years of age).

TABLE 72. SUMMARY OF THE MODELS USED IN EMRAS II WORK BY WG2

Model	Mechanism	Scope	Radioactive Medium	Type	Availability	Website Link
CROM	Gaussian plume and box models in steady state	Screening to detailed	Gaseous and liquid releases	Steady state	Free	ftp://ftp.ciemat.es/pub/CROM
DandD	Analytical	Screening	Building and surface soil	Steady state	Via the RAMP program	https://ramp.nrc-gateway.gov/DandD
DOSDIM+HYDRUS	Analytical	Detailed	Surface, near surface	Time dependent	DosDiM is an in-house SCK·CEN model. HYDRUS is free	http://www.pc-progress.com/en/Default.aspx?hydrus-1d
PC-CREAM 08	Gaussian plume module, other modules	Detailed	Gaseous and liquid release	Steady state	Commercially from Public Health England - Radiation Services	https://www.phe-protectionservices.org.uk/pccream
ReCLAIM		Screening	Contaminated Land		Free	https://www.nnl.co.uk/2014/05/reclaim-v3-0-3-now-released/
RESRAD-OFFSITE	Analytical	Screening to detailed	Surface, near surface	Time dependent	Freely available upon request	http://resrad.evs.anl.gov/codes/resrad-offsite/
SATURN	Analytical	Detailed	Building, surface and near surface	Time dependent	Commercially	http://saturn.facilia.se/saturn/show/HomePage
ERICA	Semi-empirical	Screening to detailed	Dose rates to non-human biota	Steady state	Free	https://wiki.ceh.ac.uk/display/rpemain/ERICA+Tool
MicroShield	Analytical	Detailed	Photon/gamma shielding and dose assessment	Time dependent	Commercially	https://radiationsoftware.com/microshield
AMCARE	Gaussian plume and box models	Screening	Soil, groundwater, surface water and air	Steady state	In-house SCK·CEN model	

TABLE 73. SUMMARY OF THE MODELS IDENTIFIED AS POTENTIALLY USEFUL IN EMRAS II WORK BY WG2

Model	Mechanism	Scope	Radioactive medium	Type	Availability	Link to the website
AMBER	Analytical	Screening to detailed	All environmental media	Time dependent	Commercially	https://www.quintessa.org/software/AMBER
CARAIBE	Analytical	Detailed	Radon in building and surface soil	Time dependent	CARAIBE is an in-house IRSN model	
CHAIN	Analytical	Detailed	Solute transport in surface and near surface soils	Time dependent	Freely available upon request	
CHAIN 2D	Analytical	Detailed	Solute transport in surface and near surface soils	Time dependent	Freely available upon request	https://www.ars.usda.gov/pacific-west-area/riverside-ca/agricultural-water-efficiency-and-salinity-research-unit/docs/model/chain-2d-model/
CITRON	Gaussian plume, point source	Detailed	Atmospheric dispersion of radon and progeny	Time dependent	CITRON is an in-house IRSN model	
ECOLEGO	Analytical	Screening to detailed	Surface, near surface	Time dependent	Commercially except ECOLEGO Player Free	http://ecolego.facilia.se/ecolego/show/Downloads
FECUTZ	Analytical	Detailed	Solute transport in surface and near surface soils	Time dependent	Reports freely available	http://nepis.epa.gov/Exe/ZyNET.exe/900G0V00.TXT?ZyActionD=ZyDocument&Client=EPA&Index=2000+Thru+2005&File=D%3A%5CZYFILES%5CINDEX+DATA%5C00THRU05%5CTXT%5C00000011%5C900G0V00.TXT&User=anonymous&Password=anonymous&ImageQuality=r85g16%2Fr85g16%2Fx150y150g16%2Fi500&Display=hpfrw&Back=ZyActionS&MaximumPages=5&Query=fname%3D%22900G0V00.TXT%22
FRAME	Analytical	Risk and dose assessment	Air, water and human impacts	Steady state	Freely available upon request	http://mepas.pnnl.gov/FramesV2/download.stm
MILDOS AREA	Gaussian plume and analytical	Detailed	Uranium mining dose assessments from buildings, surface and near surface	Time dependent	Freely available upon request	http://mildos.evs.anl.gov/

TABLE 73. (cont.)

Model	Mechanism	Scope	Radioactive medium	Type	Availability	Link to the website
MODELMAKER	Analytical	Screening to detailed	All environmental media	Time dependent	Commercially	https://www.apbenson.com/about-modelmaker-4
MULTIMED Daughter Process (MULTIMED DP)	Analytical	Detailed	Multimedia waste disposal unit	Time dependent	Freely available upon request	https://www.epa.gov/ceam/multimed
RESRAD BUILD	Analytical	Screening and detailed	Building, surface, near surface	Time dependent	Freely available upon request	http://resrad.evs.anl.gov/codes/resrad-build/
ROOM	Analytical	Screening	Indoor gamma dose from building materials	Time dependent	In-house STUK model	
Preliminary Remediation Goals for Radionuclide Contaminants at Superfund Sites+Dose Compliance Concentrations (PRG+DCC)	Analytical	Screening	Setting up of cleanup goals for contaminated soil, water, and air	Steady state	Free internet based calculator	http://epa-prgs.ornl.gov/radionuclides/ http://epa-dccs.ornl.gov/
Preliminary Remediation Goals for Radionuclides on Outdoor Surfaces at Superfund Sites + Building Dose Compliance Concentrations for Radionuclides (BPRG+BDCC)	Analytical	Screening	Setting up of cleanup goals for contaminated buildings	Steady state	Free internet based calculator	http://epa-bprg.ornl.gov/ http://epa-bdcc.ornl.gov/
Preliminary Remediation Goals for Radionuclides on Outdoor Surfaces at Superfund Sites + Dose Compliance Concentrations for Radionuclides on Outdoor Surfaces (SPRG+SDCC)	Analytical	Screening	Setting up of cleanup goals for contaminated surfaces	Steady state	Free internet based calculator	http://epa-sprg.ornl.gov/ http://epa-sdcc.ornl.gov/
Radioactively contaminated land exposure assessment (RCLEA) tool	Gaussian plume and box models in steady state	Screening	Dose assessment from land use, buildings	Steady state	Free	https://www.gov.uk/government/publications/rclea-software-application

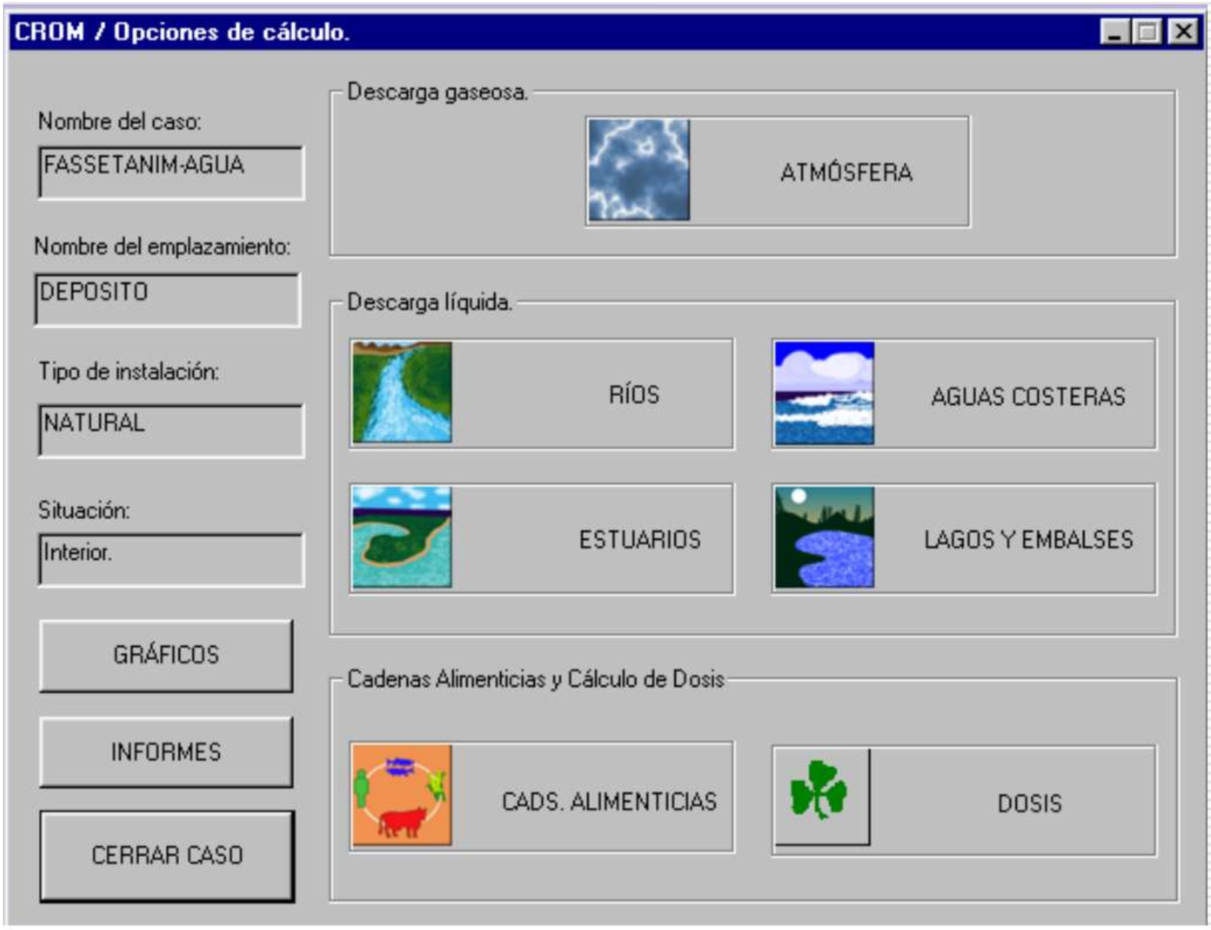

FIG. 91. Main screen of CROM, showing the main dispersion modules.

V.1.2. DandD

Decontamination and Decommissioning (DandD, version 2.1.0)[68] was developed by Sandia National Laboratories [72, 100−101, 106]. The code may be used for probabilistic dose assessments that are based on probability distribution functions for the parameters used in the models, scenarios and exposure pathways presented in the DandD tool. It is also possible to conduct sensitivity analyses using DandD to determine which parameters have the greatest influence on the dose distribution. Such information can be beneficial in support of decision making, for example, to determine the need for gathering site specific parameter values.

The code can be used to evaluate the impact of initial contamination in a building in the context of the building occupancy scenario. It can also be used to evaluate the impact of initial contamination in soil in the context of the residential scenario.

[68] DandD is available at: https://ramp.nrc-gateway.gov/DandD

The building occupancy scenario relates levels of surface contamination and volume in buildings to the calculated total effective dose equivalent. For this scenario, exposure pathways include external exposure, and internal exposure relating to inhalation and secondary ingestion.

A more generalized residential scenario, which is more complex than the other scenarios, has been designed for sites with contaminated soils. A generic water-use model can be used for the evaluation of the annual total effective dose equivalent due to exposure from ingestion of drinking water (well water) and from contaminated soils via multiple pathways. A simple 3-box model describing water-use considers radioactive decay, ingrowth of progeny, and environmental transport of radionuclides. The three boxes (also called 'layers') in this water-use model are surface soil, unsaturated soil and the aquifer. A conservative analysis can be conducted through generic treatment of potentially complex groundwater systems, with outputs that are limited to suggestions of when it is warranted to collect additional site specific data and to undertake more sophisticated modelling. The residential scenario considers external exposure, inhalation of dust and ingestion of fish, drinking water, soil, land-based foods (fruit, grain, leafy vegetables, other vegetables, milk, beef, poultry, eggs) and food grown using irrigation water as exposure pathways. The types of animal feeds considered are stored hay, stored grain and forage.

For each scenario and exposure pathway, distributions of input parameters were developed in a consistent manner to conduct screening dose assessments. This approach increased the likelihood of overestimating rather than underestimating potential exposure and corresponding dose. The DandD code provides a simple and straightforward option to modify selection of the scenario, source profile, exposure pathways and many of the parameters in the model to account for site specific conditions.

The code is user friendly with easy access to the User's Guide, in which a number of useful example applications are described. However, buried contaminated material or the radon exhalation pathway are not considered in the simulation. It is assumed that the thickness of the superficial contaminated soil is 15 cm.

V.1.3. DOSDIM + HYDRUS

DOSDIM (Dose Distribution Model) is a biosphere compartmental model, which is partly dynamic, depending on the exposure pathways and the time frame considered. An additional module has been added to the model to calculate radon activity concentrations in air from large areal sources. The equations used in the Immission Frequency Distribution Model (IFDM) serve as a basis for the calculations in this module; IFDM is a multi-source Gaussian dispersion model for modelling of ground-level pollutant concentrations that have been released from point and area sources [102]. DOSDIM has been used in several international biosphere model verification studies.

The HYDRUS−1D program, in combination with HYDRUS−2D, has been applied to model transport of radionuclides in the variably saturated medium underneath waste. Both of these models can be applied to calculate transport of water and solutes in the unsaturated and saturated zones. Richards equation, which may be used to describe variably saturated water flow and convection-dispersion for solute and heat transport, can be solved numerically using the HYDRUS computer code. Movement of water and solutes in fully saturated, partially saturated or unsaturated porous media can be modelled using the program, and finite element numerical methods may be used to solve flow and transport equations [103−104]. The one-dimensional version of the program (HYDRUS−1D) was used in this publication to model radionuclide transport through the vadose zone (unsaturated) underneath the waste and into the aquifer (saturated zone). HYDRUS−2D was then used to calculate radionuclide activity concentrations at the location of the point of exposure in the aquifer (i.e. a well at the house) using the output values from HYDRUS−1D as inputs.

V.1.4. PC-CREAM 08

The 'Consequences of Releases to the Environment: Assessment Methodology' (PC-CREAM 08) is comprised of a suite of models and data for application in the assessment of radiological consequences to the population relating to continuous routine discharges of radionuclides to the environment (atmosphere and aquatic) [99].

PC-CREAM 08 is an updated version of PC-CREAM 98. Updated features include:

- A comprehensive list of radionuclides based on Ref. [218];
- Inclusion of radioactive progeny;
- Inclusion of recent reviews of models and input parameters;
- Modelling of discharges to river environments, as well as to atmospheric and marine environments;
- Calculation of activity concentrations in environmental media.

Some parts of this model, originally developed for the European Union, have been used elsewhere. Results are reported in terms of individual or collective doses. The model can also be used to perform prospective assessments as input to authorizations for radioactive discharges from nuclear facilities and to decisions on radioactive waste management. PC-CREAM 08 cannot be used to model short-duration releases unless these occur with a frequency that can then be approximated to a continuous, uniform release.

PC-CREAM 08 is divided into mathematical 'models' and an assessment program called 'ASSESSOR', the latter of which serves in central dose assessment within the programs. Environmental transfer of radionuclides and corresponding activity concentrations in various environmental media can be calculated using a series of models within the 'models' of PC-CREAM 08, which are:

- PLUME model;
- RESUS model;
- GRANIS model;
- FARMLAND model;
- DORIS model;
- River models.

Outputs from the models can then serve as inputs to the dose assessment part of PC-CREAM 08 (i.e. the ASSESSOR).

V.1.4.1. PLUME

PLUME is a Gaussian plume model [90, 99], which is applied to calculate atmospheric dispersion within PC-CREAM 08, and represents:

- Meteorological conditions during a radiological release;
- Characteristics of the radionuclides being released;
- Roughness of the land surface.

Radionuclide activity concentrations in air, external gamma dose rates from radionuclides in the cloud ('cloud gamma'), and rates of deposition at a range of distances downwind of the point of release can be calculated using the PLUME model. Model outputs are subsequently inputted into ASSESSOR, which combines these outputs with actual radionuclide release rates and site specific meteorological data. Doses from various exposure pathways arising from atmospheric discharge of radionuclides are estimated based on deposition rates from the PLUME model, along with results from the other models (i.e. RESUS, GRANIS and FARMLAND).

V.1.4.2. RESUS

Radionuclide activity concentrations in air as a result of resuspension of previously deposited radionuclides can be estimated using the RESUS model. This model applies an equation from Ref. [99] that is independent of the radionuclide under consideration, with the exception of differences due to radioactive decay, which are considered. The activity concentrations generated using RESUS then serve as input to ASSESSOR for the calculation of doses from inhalation of resuspended material.

V.1.4.3. GRANIS

The 'Gamma Radiation Above Nuclides In Soil' (GRANIS) model[69] may be used to calculate external gamma exposure from radionuclides deposited on the ground [99]. Radionuclide transfer through soil can be modelled using GRANIS, along with soil shielding properties when estimating doses at 1 m above the soil surface. Doses are calculated using user-defined deposition rates. GRANIS is the only PC-CREAM 08 model that considers some organ doses, in addition to the effective dose. Effective doses are then used as input to ASSESSOR for the subsequent calculation of external exposures at a 1 m height above the ground surface.

V.1.4.4. FARMLAND

Radionuclide transfer into terrestrial foodstuffs from radionuclides deposited onto the ground can be predicted using the FARMLAND suite of models [99]. Food categories that are considered the most important in the human diet are considered in FARMLAND; these are fruit, green vegetables, root vegetables, grain, sheep meat, sheep liver, cow's milk, cow milk products, cow meat, and cow liver. Radionuclide activity concentrations can be calculated for each type of food using user-defined deposition rates. Activity concentrations are then used as input to ASSESSOR for the calculation of ingestion doses.

[69] See https://www.gov.uk/government/publications/granis-assessment-model-for-external-photon-irradiation

V.1.4.5. DORIS

DORIS is the marine dispersion model within PC-CREAM 08 [99], which is closely based on the MARINA II model [219]. Radionuclide activity concentrations can be predicted in sea water, sediments, and marine biota using the DORIS model, based on user-defined discharge rates. The activity concentrations generated using DORIS can then serve as input to ASSESSOR for the calculation of doses from external exposure to beach sediments, ingestion of marine foods and inhalation of sea spray.

V.1.4.6. River models

Radionuclide dispersion following release to rivers can be calculated using two models in PC-CREAM 08, a screening model and a dynamic model. The screening model is a simple dilution model, with an assumption of instantaneous equilibrium between surface water and river sediments. This model is intended for use as a screening tool and may be used in three modes, depending on the amount of available input data. The dynamic model is a time-dependent compartment model, which can be applied to conduct more detailed assessments and which needs a larger amount of input data to run.

V.1.5. ReCLAIM

The 'Review of Contaminant Levels for the Assessment calculation of de minimis Inventory Model' (ReCLAIM)[70] is a tool that consists of an electronic spreadsheet, which can be used to conduct simple generic or site specific assessments of radioactively contaminated land. The tool has been designed, principally, for assessment of nuclear licensed sites, but can potentially be applied more widely [105].

ReCLAIM has been specifically developed to evaluate future land use scenarios for a range of realistic exposure pathways associated with contaminated lands. The tool is designed to be of use for scenarios where the change in land use, for example, from industrial to residential, will be an issue.

The radionuclides considered in ReCLAIM v3.0 include those that are associated with the nuclear power production and the nuclear fuel cycle and have medium to long physical half-lives. The tool also includes a select list of naturally occurring radionuclides.

V.1.5.1. Exposure pathways

The following exposure pathways are considered within ReCLAIM v3.0:

- External irradiation due to exposure to contaminated ground when indoors or outdoors;
- External irradiation due to immersion in contaminated water;
- External exposure due to dermal contact with contaminated ground (with the exception of open wounds);
- Internal exposure from inhalation of contaminated dust (indoors and outdoors);
- Internal exposure due to ingestion of contaminated soil and dust;

[70] ReCLAIM was developed by the National Nuclear Laboratory (NNL) through support of the Nuclear Decommissioning Authority (NDA) in the United Kingdom.

- Internal exposure due to ingestion of contaminated plant-based foodstuffs (green vegetables, root vegetables, cereals, fruit);
- Internal exposure due to ingestion of contaminated animal-based foodstuffs (cow's milk, beef, cattle offal);
- Internal exposure due to ingestion of freshwater fish;
- Internal exposure due to ingestion of contaminated water.

Four geometries are available in the model for to estimate external irradiation from contaminated ground, which relate to:

- Surface contamination (Bq/cm^2) with 'zero' thickness;
- Shallow contamination (Bq/g dry mass) located at a depth of 5 cm from the surface;
- Deep contamination (Bq/g dry mass) that extends to an infinite depth from the surface;
- Buried contamination (Bq/g dry mass) located underneath either a clean soil cover with a thickness of either 0.1 m or 0.5 m.

V.1.5.2. Scenarios

The following scenarios can be accommodated within ReCLAIM v3.0:

- Agricultural (agricultural worker and family);
- Recreational (swimmer in a lake, general park user, fisherperson and family, park worker);
- Construction (construction worker who is working outdoors and disturbing the soil);
- School (teacher, school children, caretaker);
- Office (industrial office worker);
- Householders (family);
- Drinking water;
- Covered area (e.g. a car park).

In addition, the following land use scenarios can also be accommodated:

- Residential (with or without home-grown produce);
- Allotment;
- Commercial or industrial.

V.1.6. RESRAD

V.1.6.1. RESRAD (onsite)

Argonne National Laboratory developed the RESRAD (RESidual RADioactive) computer code for the US Department of Energy to calculate site specific guidelines for residual radioactive material, radiation doses, and excess lifetime cancer risk to an on-site receptor receiving chronic exposure. The RESRAD code is the first in a series known as the RESRAD Family of Codes. The original RESRAD code has been renamed to the 'RESRAD (onsite) code' to distinguish it from other codes within the RESRAD Family of Codes.

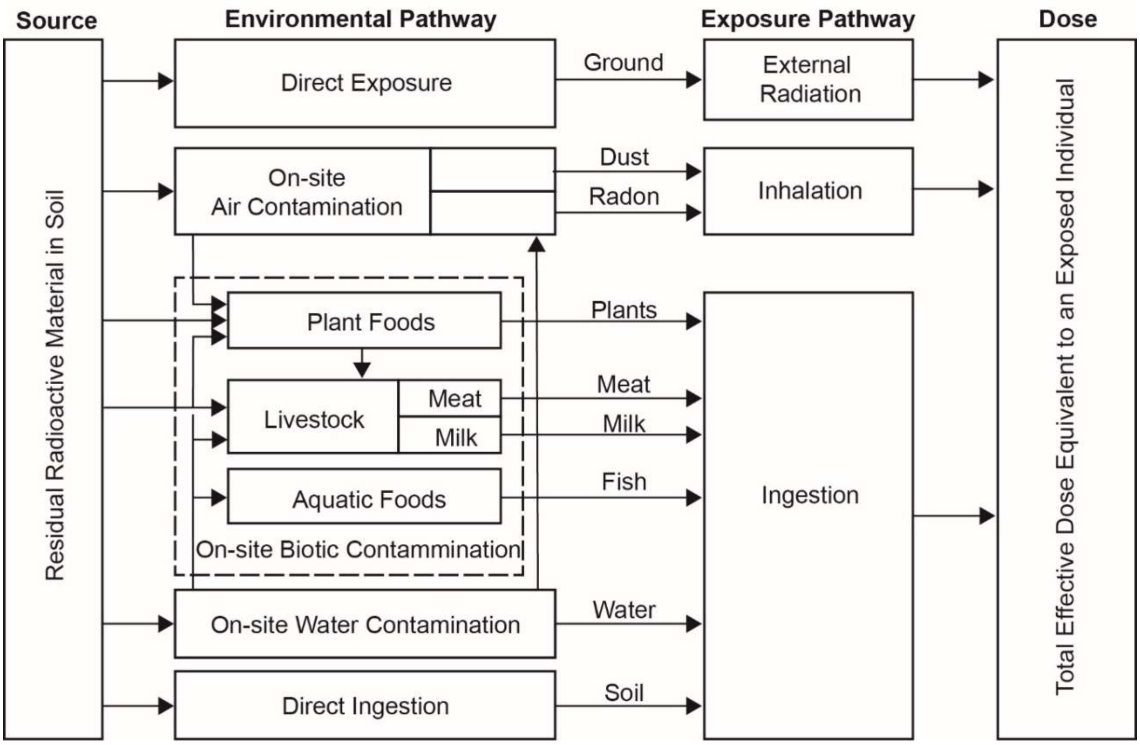

FIG. 92. A diagrammatic summary of the pathways considered in RESRAD (onsite).

The suite of models within RESRAD can be used for on-site screening or detailed assessment, assessment of near-surface burial of solid waste or landfill, estimation of doses to non-human species, assessment of impacts of buildings, chemical impacts, and other aspects.

The RESRAD (onsite) code considers the following major exposure pathways (see Fig. 92):

— External radiation exposure from direct exposure to contaminated soil;
— Internal exposure due to inhalation of:
  - Airborne radionuclides,
  - Radon progeny,
— Internal exposure due to ingestion of:
  - Drinking water from a contaminated well or pond,
  - Crops grown in contaminated soils and irrigated using contaminated water,
  - Milk and meat from livestock that have been fed with contaminated water and fodder,
  - Fish from a contaminated water body,
  - Contaminated soil.

RESRAD is a multifunctional and includes the following applications:

— Deriving soil cleanup criteria for remediation that are compliant with regulatory requirements, such as DOE Order 5400.5;
— Calculating potential radiation exposure, annual dose and lifetime cancer risks to workers or members of the public, due to residual radioactive material in soil;

- Estimating future radionuclide activity concentrations in environmental media (air, surface water, and groundwater) due to contamination in soil;
- Analyses in support of ensuring exposure is as low as reasonably achievable (ALARA) or cost benefit analyses in support of the decision making process relating to decontamination, decommissioning and remediation;
- Prioritizing oversight, effort and budget for collection of data on hydrogeological and soil properties that affect the distribution of radioactive material in the environment and consequently, decisions on management of radioactive material, including waste.

The RESRAD model can be applied to address situations involving buried waste, as well as landfill (uncovered waste) [107−108]. The model has limited applicability related to source region geometry and has not been designed to predict impacts off-site (see RESRAD-OFFSITE description in Section V.1.6.2 below). However, the RESRAD model can be applied to make conservative estimates of impacts off-site, despite this limitation. The model can handle a broad range of radionuclides and users can change the cut-off in half-life when setting short-lived progeny in equilibrium with the parent radionuclide.

V.1.6.2. RESRAD-OFFSITE

Radiological consequences to receptors that are located on-site or outside of the area of primary contamination can be estimated using RESRAD-OFFSITE[71], which is an extension of the RESRAD (onsite) code. RESRAD-OFFSITE can be applied to estimate radiation dose and excess lifetime cancer risk. Soil cleanup guidelines can be derived using activity concentrations of radionuclides in the environment that have been calculated using RESRAD-OFFSITE [109−110]. Figure 93 depicts the features of RESRAD-OFFSITE.

V.1.6.3. Exposure locations considered

Initial contamination in soil can be assessed using RESRAD-OFFSITE. The contaminated soil can have a clean cover on top and up to five partially saturated layers below it. Radiation exposure of an individual spending time on-site directly above the primary contamination and away from the primary contamination (off-site) can be modelled using the code.

It is possible that an individual will spend time in buildings located on-site or off-site, and the same individual might consume animal- and plant-based foods grown on the site or harvested from off-site agricultural fields contaminated by material from the primary contamination. Drinking water and water used by the individual might be drawn from a surface water body or a well located on-site or off-site. The individual might also gather aquatic food for consumption from a contaminated surface water body.

[71] DOE Office of Health, Safety and Security, and the Office of Environmental Management sponsored code development, with support from US Nuclear Regulatory Commission. Argonne National Laboratory (ANL) developed the code, and code and version control are maintained by US DOE through ANL.

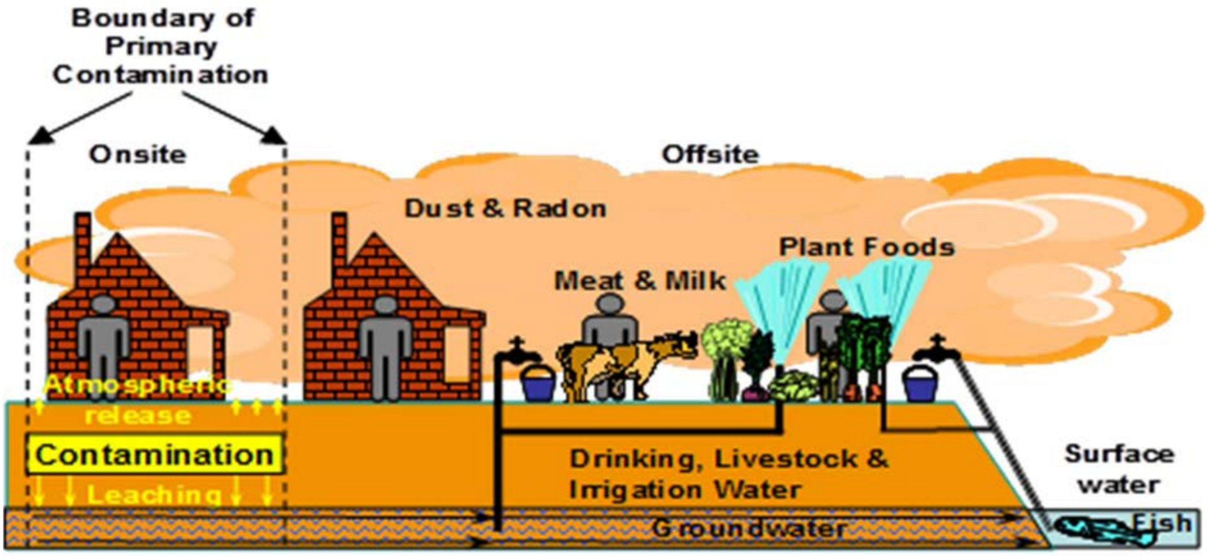

FIG. 93. The RESRAD-OFFSITE conceptual site model.

V.1.6.4. Pathways

RESRAD-OFFSITE considers eight exposure pathways, as follows:

— Direct exposure from soil contamination;
— Internal exposure due to inhalation of radon and particulates;
— Internal exposure due to ingestion of:
  • Plant-based foods,
  • Meat,
  • Milk,
  • Aquatic foods,
  • Drinking water,
  • Incidental ingestion of soil.

V.1.6.5. Scenarios of exposure considered

Through selection of different exposure pathways, various exposure scenarios may be simulated using RESRAD-OFFSITE. Examples of possible scenarios that can be modelled might include:

— Urban resident scenario;
— Rural resident farmer scenario;
— Industrial worker scenario;
— Recreational scenarios.

V.1.6.6. *Main features of the RESRAD-OFFSITE package*

The main features of the RESRAD-OFFSITE[72] package are as follows:

- User friendly with help function and User's Guide;
- Interface for mapping of primary contamination and specified off-site areas for the region of interest;
- Inclusion of all exposure models that have been provided in RESRAD (onsite), with minor changes, to facilitate model extension between on-site and off-site locations and consideration of off-site exposure;
- Possibility to specify various exposure scenarios through activation or suppression of exposure pathways and modification of model parameters;
- Possibility to calculate radionuclide activity concentrations in environmental media, doses and risks progressively over time through application of numerical methods;
- Generation of text output reports after each model run, which include lists of all input parameters, along with maximum dose and minimum soil quality guidelines;
- Compatibility of ICRP's database of radionuclides [218], which are used as default parameters in the model.

V.1.6.7. *RESRAD-OFFSITE data and uncertainty analysis input screen*

The RESRAD-OFFSITE data and uncertainty analysis input screen provides the following:

- Text reports for each user-specified time point, which provide lists of doses, soil quality guidelines, and health risks by radionuclide and exposure pathway;
- Temporal plots of activity concentration, dose, soil quality guidelines, risk and dose-to-source ratio;
- Sensitivity and probabilistic analyses (including graphic representations of results) to evaluate the influence of input parameters on model outputs;
- A database containing age specific dose factors from Refs [128, 153, 220], cancer risk coefficients for environmental exposure to radionuclides from Ref. [129], and Health Effects Assessment Summary Tables with morbidity or mortality slope factors from Refs [221−222]. It is possible for users to select dose and risk factors from the tool or to establish their own dose and risk library;
- An option to input the following types of temporal data:
  - (i) Activity concentrations of radionuclides in the primary contaminated area and in the mixing layer;
  - (ii) Fluxes of radionuclides to groundwater, surface runoff, and the atmosphere;
  - (iii) Dimensions of primary contamination, the cover and the mixing layer;
  - (iv) The eroded soil mass.

Figures 94 to 96 provide graphic depictions of RESRAD-OFFSITE screens and options for the display of some analysis results.

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[72] RESRAD-OFFSITE is available free of charge and can be downloaded from the RESRAD website (http://resrad.evs.anl.gov/codes/resrad-offsite/)

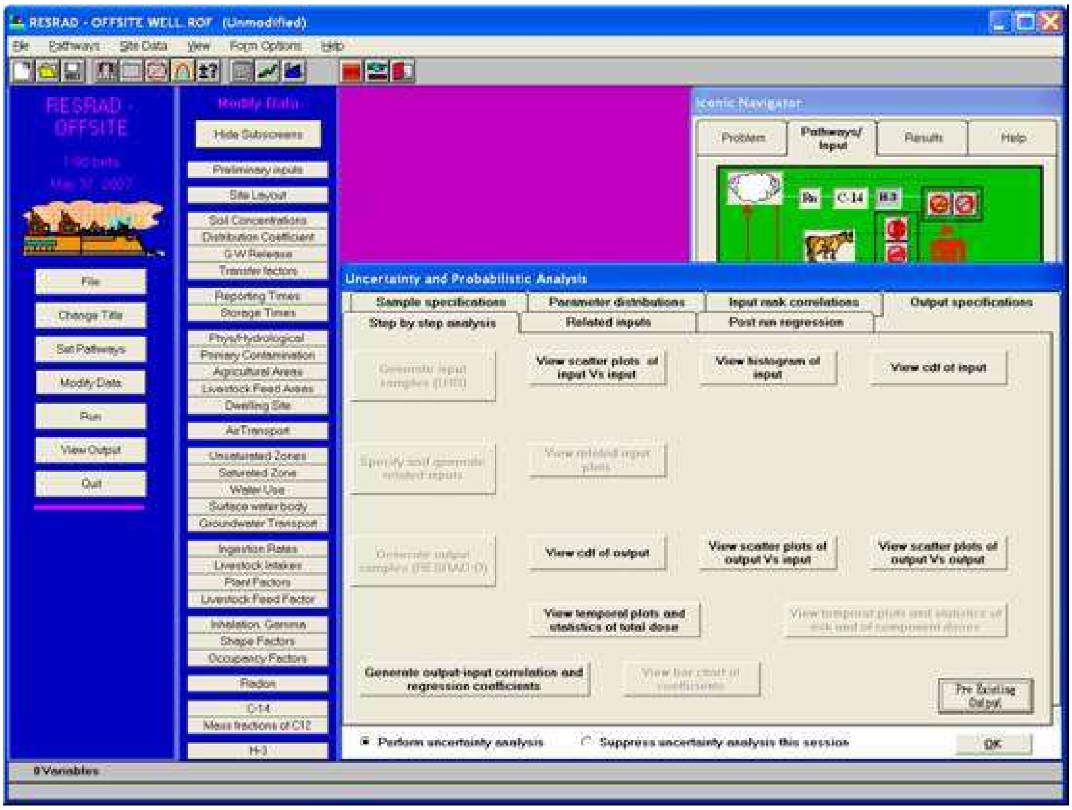

FIG. 94. The RESRAD-OFFSITE data and uncertainty analysis input screen.

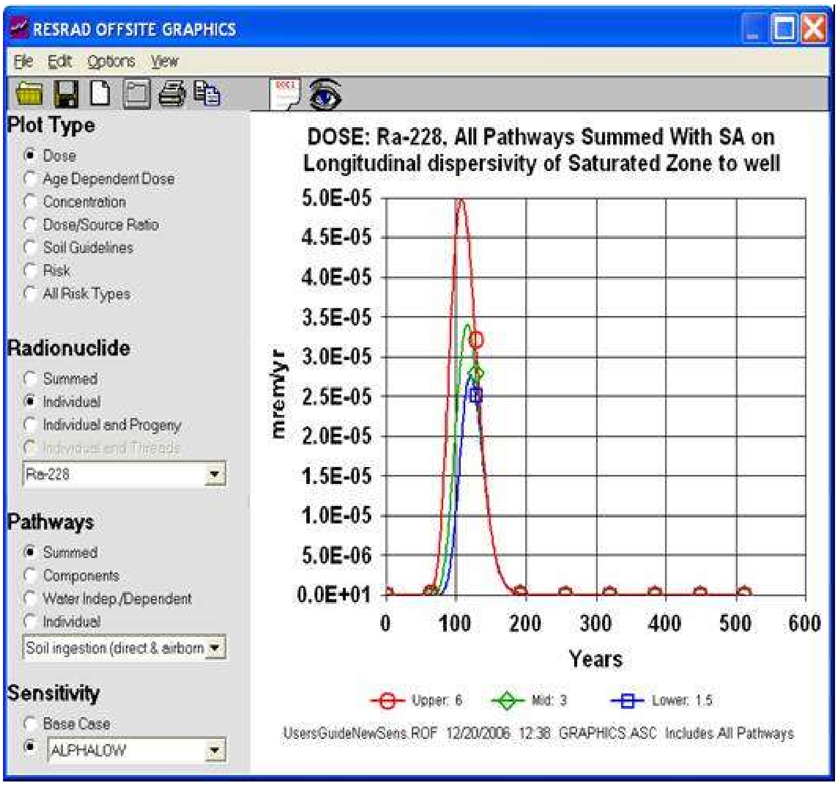

FIG. 95. Typical RESRAD-OFFSITE sensitivity analysis result screen.

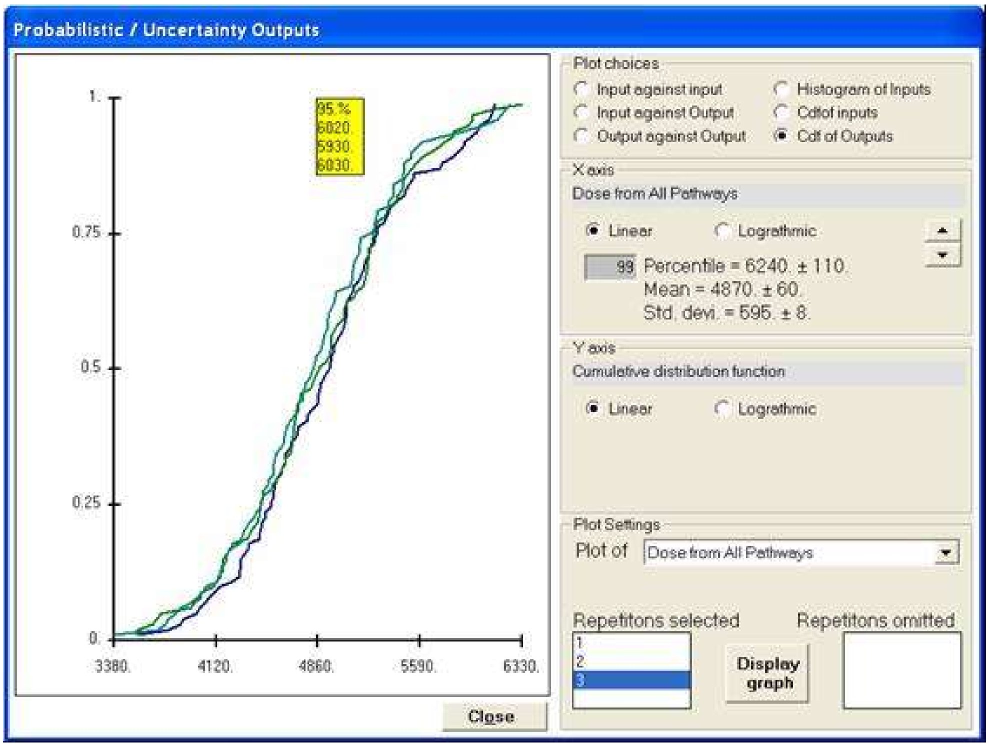

FIG. 96. RESRAD-OFFSITE probabilistic analysis result.

V.1.6.8. Types of releases from primary contamination

The types of radioactive releases that are considered by the code and that can lead to off-site contamination are (see Fig. 97):

(1) A model of the rate controlled release to estimate the amount of contaminants removed through leaching by water flowing through the area of primary contamination (see Fig. 98);
(2) An equilibrium model for dust release for atmospheric releases;
(3) A model of the erosion of material by surface runoff and subsequent release to surface water.

Releases to the atmosphere and through runoff are considered once the surface soil has become contaminated. Radionuclide accumulation at off-site locations is modelled considering atmospheric deposition and irrigation.

V.1.6.9. Groundwater transport in RESRAD-OFFSITE

In addition to dispersion and convection, the groundwater transport model within RESRAD-OFFSITE accounts for decay of parent radionuclides, ingrowth of progeny, and their respective retardation as a result of sorption/desorption in the solid phase. Differential equations, which are solved using numerical analysis methods, are used to characterize radionuclide behaviour. An option to further divide each saturated zone and unsaturated zone into thinner sub-layers is available to increase the precision of calculated results; application of this option also increases model calculation times.

V.1.6.10. *Atmospheric transport model in RESRAD-OFFSITE*

RESRAD-OFFSITE includes an atmospheric transport model with the following features:

- A Gaussian plume model, which assumes an area source release and allows the calculation of activity concentrations in air at off-site locations;
- A plume rise model for estimation of buoyancy induced plume rise;
- Possibility to use either standard Pasquill-Gifford dispersion coefficients or Briggs dispersion coefficients as input parameters;
- Consideration of wet and dry deposition of plume content for particulate radionuclides or vapour;
- Possibility for spatial integration over the area of interest through application of spacing grids to estimate the average radionuclide activity concentration in air.

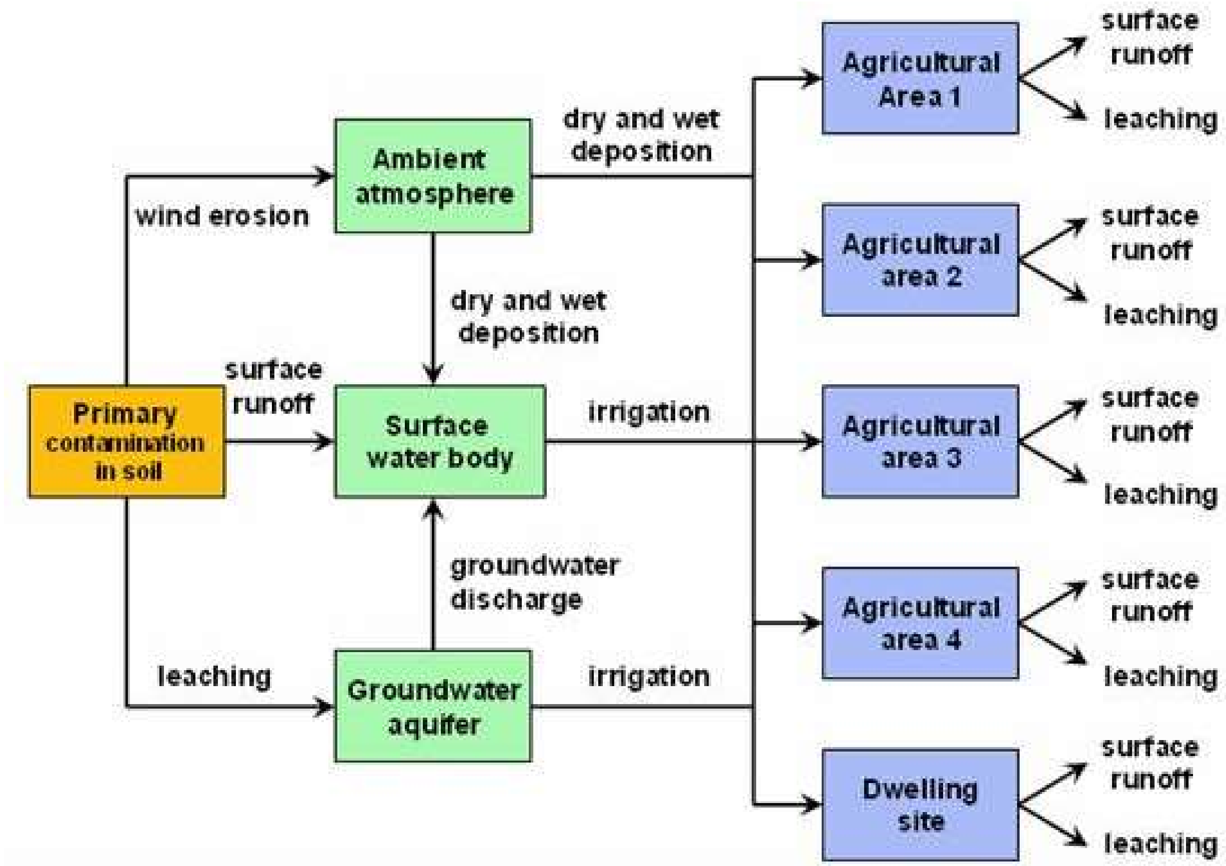

FIG. 97. Environmental transport processes considered in RESRAD-OFFSITE.

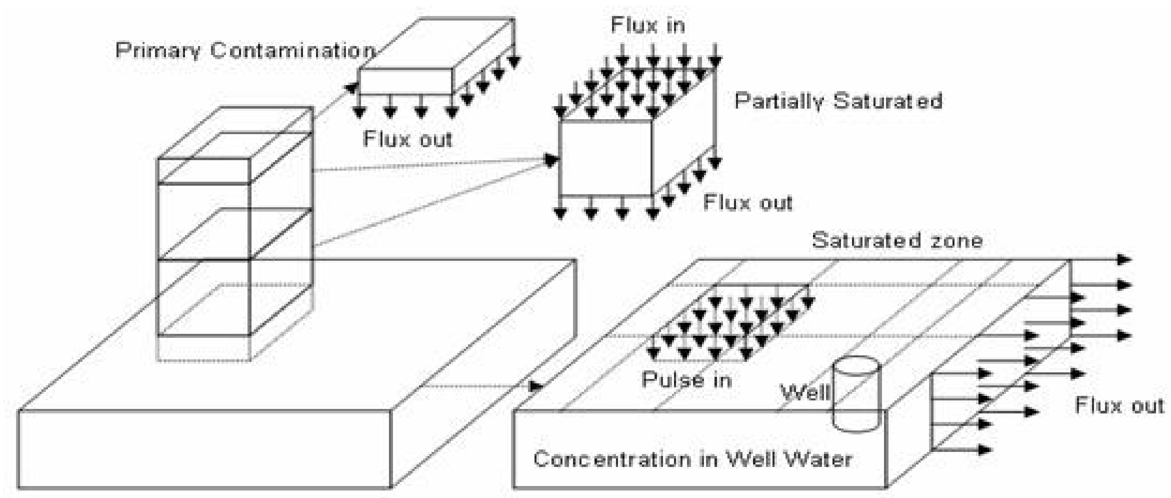

FIG. 98. Representation of the RESRAD-OFFSITE leaching and groundwater transport models.

V.1.7. SATURN

SATURN is a software package that can be used to implement mathematical models for the assessment of present and future radiological impacts from contaminated land. SATURN has been developed using the Ecolego software and can be run using the Ecolego Player, which is freely available.

SATURN consists of a library of modules that can be combined to assess radionuclide transport from contaminated land and human exposures by different pathways of exposure. An overview of the SATURN modules and their possible interactions is presented in Fig. 99. The modules are designed to allow for input data relating to several radionuclide release scenarios. Input data can be provided directly by the user, if available, for example, from monitoring programmes or calculations using external models. Alternatively, the module outputs can be used as inputs to other modules.

V.1.7.1. Source term modules

SATURN includes two source term modules, one for releases of radionuclides to the local atmosphere above a contaminated land via resuspension of particles or gas releases, and one for releases of radionuclides to the unsaturated zone below the contaminated land or directly to the aquifer. The source term models can address the presence of one or more uncontaminated cover layers above the contaminated materials. Gas releases into buildings constructed directly above the contaminated land are also included in the source term module.

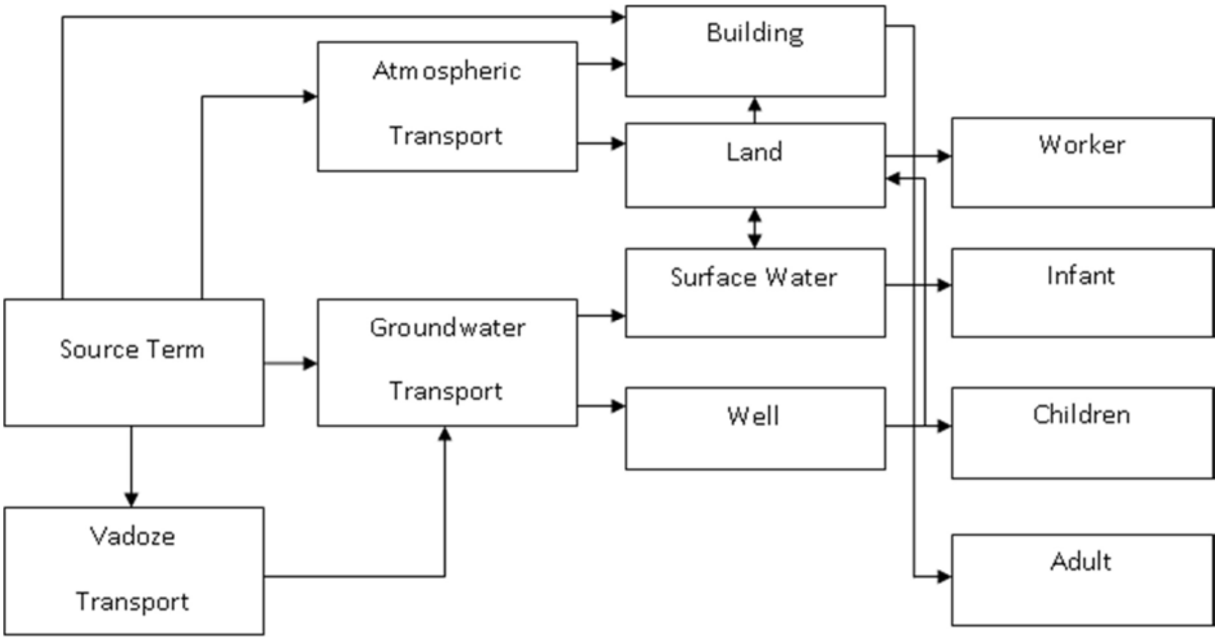

FIG. 99. Schematic representation of the SATURN modules and their interactions.

V.1.7.2. Transport modules

SATURN includes modules for the transport of radionuclides via groundwater and atmospheric pathways. A schematic representation of these transport processes can be seen in Figs 102 and 103.

Radionuclides leached from contaminated land migrate vertically in the unsaturated zone (vadose zone) towards the groundwater table. Upon entering the aquifer (saturated zone), the radionuclides are transported by groundwater flow towards a downstream water well, and finally, release into a surface water system. It is assumed that radionuclide migration occurs under steady state flow conditions. The geometry of the groundwater flowlines (flow tube) needs to be defined prior to radionuclide transport modelling. This can be done by means of groundwater modelling or from interpretation of hydraulic head measurements in monitoring wells. The transport process taken into account by the models include advection, hydrodynamic dispersion, radioactive decay, and radionuclide sorption by the soil matrix. For radionuclide sorption, instantaneous and reversible sorption is assumed, as described by a linear isotherm, also known as a K_d model (where K_d is described as sorption distribution coefficient). The schematization in Fig. 100 corresponds to the case where the groundwater system is deep, whereas the contaminated land has a relatively small lateral (horizontal) extent. The case in which the aquifer has a relatively small thickness, whereas the waste site has a large lateral extent, is shown in Fig. 101.

The atmospheric transport module implements the simple atmospheric dispersion screening models, as described in Safety Reports Series No. 19 (see Footnote 66). By applying this module several times within an assessment, it is possible to represent the effect of the distribution of radionuclides on a contaminated land, as well as the impact on receptors at different locations with respect to the source.

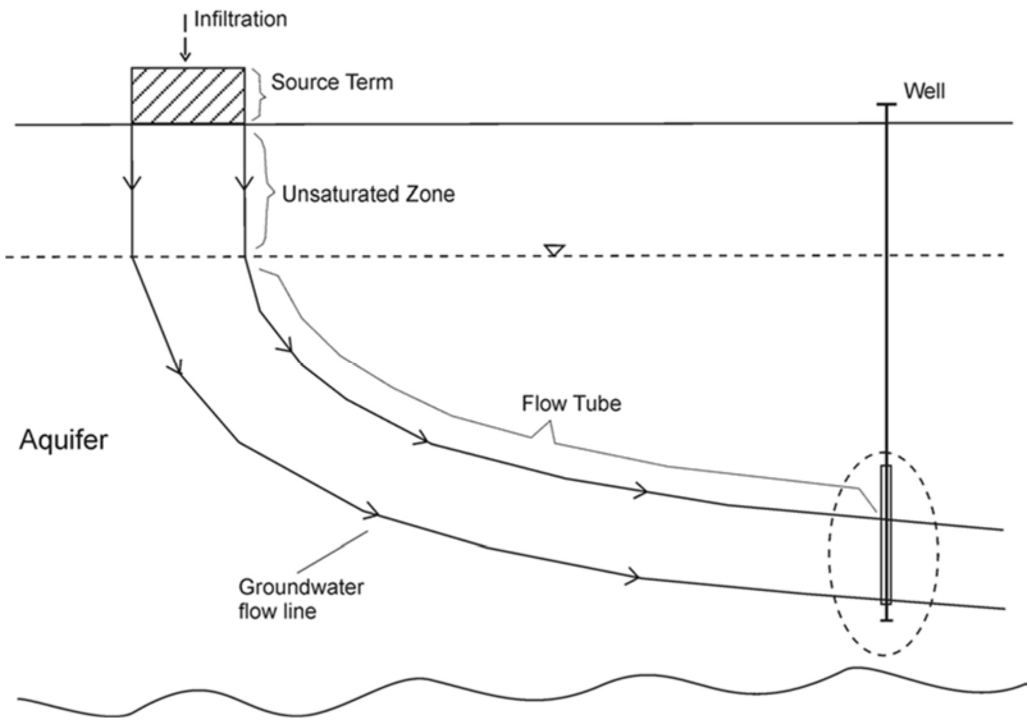

FIG. 100. Schematization of the SATURN Groundwater Transport module for a deep groundwater system (aquifer of large thickness) and a relatively small lateral extent of the contamination ('source term').

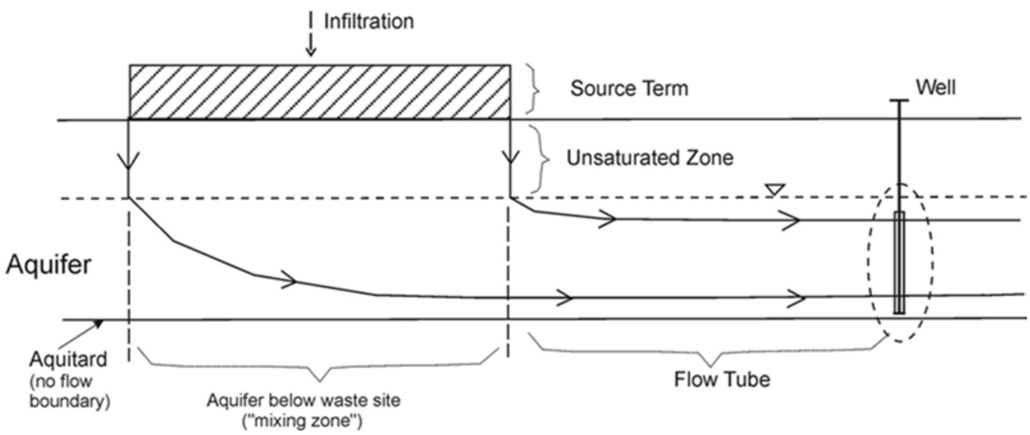

FIG. 101. Schematization of the SATURN Groundwater Transport module for a groundwater system (aquifer) of a relatively small thickness and large lateral extent of contaminated land ('source term').

V.1.7.3. Receptor environments

The receptor modules included in SATURN are:

— **Well** – can be used to calculate radionuclide activity concentrations in water from a well receiving radionuclide releases and to calculate the corresponding doses from drinking of the water. The activity concentrations in well water can be used as input to the land module when it is used for irrigation and/or animal watering.

- **Surface Water** – Application options within this module in SATURN include river, small lake, large lake and coastal area. The modules can calculate activity concentrations in surface water and different foods, and to calculate corresponding doses via ingestion of food and water. The models implemented are those described in Safety Reports Series No. 19 (see Footnote 66), with some modifications.
- **Land** – Several application options are included within this module in SATURN: crop land, pasture and land where people spend time but do not produce food. The modules can calculate activity concentrations in soil, air and different foods, and to calculate corresponding doses via inhalation, external irradiation and food ingestion. The models implemented are those described in Safety Reports Series No. 19 with some modifications.
- **Building** – can be used to calculate radionuclide activity concentrations in air inside a building and corresponding doses via inhalation and external irradiation.

In the receptor modules, conservative doses for standardized exposure conditions are calculated. These conservative dose estimates can be used when comparing exposure from different sources and corresponding risks arising from the contaminated land. In addition, doses from the actual use of the contaminated objects by different groups are also calculated. The groups considered are workers, adults (including lactating mothers), children and infants. These groups are defined in the corresponding exposure group modules. The main purpose of these modules is to integrate the exposure from different sources.

SATURN includes a parameter database, in which default parameter values are collated. Users can add their own parameter values to the database and share these with other users.

SATURN supports both deterministic and probabilistic simulations. Several sensitivity analysis methods, such as regression methods and variance decomposition methods, are available.

V.1.8. ERICA assessment tool

The ERICA assessment tool[73] package contains a suite of generic models from Safety Reports Series No. 19 (see Footnote 66). These models use a simplified uptake and dosimetry approach, with concentration ratios adopted to calculate radionuclide activity concentrations in non-human species (or ‘biota’) relative to environmental media. ERICA uses the concept of screening dose rate with the aim of protecting the function and sustainability of ecosystems by ensuring that dose rates to representative organisms are below a threshold screening value. The tool uses a tiered approach, ranging from simple to complex.

V.1.9. MicroShield

The MicroShield program[74] can be applied in photon and/or gamma ray shielding and dose assessment, for example in the design of shields, the estimation of source strengths based on radiation measurements, and the teaching of shielding principles. MicroShield is an interactive program that utilizes input error checking and provides integrated tools to graph results and create material and source files. It also affords access to material- and radionuclide specific data.

[73] The ERICA software is free, and its development is ongoing. It can be downloaded from http://www.erica-tool.com/
[74] MicroShield is available for purchase from Grove Software Inc.

The program includes tools that may be used to calculate source inference with decay, project exposure rate over time due to decay, and estimate decay heat. MicroShield has been used to evaluate direct exposure from contaminated equipment and materials in several industries, including the oil and gas industry.

V.1.10. AMCARE

The AMCARE model has been developed under the CARE (Common Approach for REstoration of contaminated sites) project of the European Commission [144]. AMCARE is a modular model for use in generic assessment of remedial options that might be applied to remediate sites contaminated as a result of various past activities and processes, and that have a range of different physical characteristics. In particular, AMCARE has been primarily developed as an assessment tool that provides a common approach for the relative ranking of possible remedial options; the tool is not applicable for the detailed assessment of a site that take account of site specific characteristics and exposure to 'local critical groups'[75].

As a result, in general, AMCARE is conservative for all radionuclide transport and exposure pathways, and the habits of exposed groups are assumed to be homogeneous across sites, as defined by European-wide generic habits surveys. In addition, in general, all foodstuffs are assumed to be grown at the location of inhabitation. That said, there is sufficient flexibility in the model for key site specific features to be considered (e.g. radionuclide inventory, type of waste, area and volume of waste disposal, proximity to nearest surface water). AMCARE has been designed to meet the following objectives:

(1) Prediction of doses to local critical groups for a given site over time, with or without implementation of remedial actions. Results are generally expressed for short term impacts relating to dose; however, it is also possible to derive maximum dose (and avoidance of maximum dose), over time periods of up to 10 000 years;
(2) Prediction of doses to on-site remediation workers and other workers (e.g. construction workers);
(3) Prediction of doses to a hypothetical group, inhabiting houses that have been constructed on a contaminated site in close contact with the main area where waste is located;
(4) Approximation of local collective dose over the period of 'institutional control' assumed to be 100 years, and over an intergenerational period of 500 years;
(5) Representation of inherent uncertainties in the applicability and accuracy of data that are being used to define key parameters as distributions of values and to evaluate dose impacts (expressed as 'best estimates' and distributions).

The modular construction of AMCARE includes multiple components, as follows:

- The GASAM[76] component to estimate atmospheric dispersion of radon emanating from the waste;
- The GWAM component to estimate migration of radionuclides contained in the wastes via groundwater;
- The DOSEAM[77] component to calculate the corresponding doses.

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[75] Superseded by the concept of the 'representative person' (see GSR Part 3 [3]).
[76] GASAM and GWAM are both FORTRAN-based modules.
[77] DOSEAM is a relatively simple spreadsheet model, which has been developed using MS Excel and applies steady state assumptions. It was developed as part of the CARE.

In addition, CRYSTAL BALL[78] is a separate (supporting) software package, which is applied to derive uncertainty distributions.

GASAM provides a Gaussian plume atmospheric dispersion model based on characteristics, which have been defined by the United Kingdom Working Group on Atmospheric Dispersion [90].

GWAM provides a one-dimensional groundwater pathway model to represent radionuclide migration through soil in the dissolved phase from a source to a location, which might then become a secondary source of exposure, over time.

Local contamination levels in environmental media (air, soil, groundwater, surface water) are estimated using outputs from both GASAM and GWAM; these contaminant levels then serve as inputs to DOSEAM for use in dose assessment. The characteristics of the waste can also serve as an input in defining local conditions within DOSEAM, as direct intrusion is considered as a potential exposure pathway within AMCARE.

V.2. OTHER MODELS IDENTIFIED AS POTENTIALLY USEFUL BY EMRAS II WG2

Table 73 provides a non-exhaustive list of other models that were identified by the participants as potentially useful in assessing risk and environmental impacts related to existing exposure situations, and in particular, sites affected by past activities.

V.2.1. AMBER

AMBER[79] is a tool that enables users to construct compartment models of any desired degree of complexity. he package can be used to take the model specified graphically by the user, to construct a set of simultaneous differential equations corresponding to that model and to provide several differential equation solvers to solve these equations under defined initial and boundary conditions.

V.2.2. CARAIBE

The CARAIBE computer code enables the user to estimate the average ^{222}Rn activity concentration inside buildings. CARAIBE considers different phenomena, such as ^{222}Rn generation in the underlying soil and bedrock, its transport up to the soil-to-building interface, its infiltration inside the structure, and air flows inside the building. These parameters can be determined from data gathered in the field (e.g. source term, building structure, weather conditions, soil permeability) or from scientific knowledge (e.g. ^{222}Rn diffusion coefficient, ^{222}Rn emanation coefficient).

CARAIBE combines a ^{222}Rn transport model (within the underlying soil and through the building levels) and an airflow model inside the building.

The ^{222}Rn transport model is one-dimensional and vertical, which enables the user to estimate the ^{222}Rn diffusive flux (Fick's law) and advective flux (Darcy's law) through several homogeneous layers representing the underlying soil, and through the building levels as well.

78 CRYSTAL BALL is a commercially available package for use in uncertainty and sensitivity analyses [114].
79 AMBER is only available commercially.

The airflow model can be used to determine the air inlet and outlet flows for the building, and the air flows between the building levels. These parameters are necessary for use in the model of radon transport inside the building.

The radon transport model is based on a partial differential equation with an analytic solution. This solution is used to determine the radon fluxes.

The airflow model is based on a non-linear equation system solved by the Newton-Raphson method, in association with a lower–upper matrix decomposition.

V.2.3. CHAIN

CHAIN is a simple model for use in simulation of radionuclide fate and transport in a uniform soil under unsaturated, steady state flow conditions [223]. The model soil system is assumed to be homogeneous, with constant moisture content and infiltration over time (assuming steady state flow). CHAIN applies analytical solutions to simultaneously solve one-dimensional advective-dispersive transport of contaminants for as many as four members of a first-order, sequential decay chain. Linear reversible isotherms are used to represent adsorbed contaminant concentrations. Either a constant concentration, a decaying source defined by an arbitrary general release mechanism, or the specific release mechanism defined by the Bateman equations can be used as the transport contaminant boundary condition [224]. Model outputs (activity concentration relative to distance (depth) at selected times; activity concentrations relative to time at selected depths) are generated using analytical solutions.

V.2.4. CHAIN 2D

Heterogeneously saturated flow, transport of contaminants, and heat transport can be simulated using the two-dimensional CHAIN 2D model [117]. In the model, water flow is represented by Richards equation for unsaturated and saturated flow and the effects of anisotropy and heterogeneity in non-uniform soils can be accounted for within the flow region. A prescribed head, flux or gradient boundaries, or free drainage, along with a simplified representation of nodal drains can be included to describe the flow model boundary conditions.

The model includes a sink term for uptake of water by plant roots.

The advective-dispersive equation, including processes of convection and conduction [223] is used to model heat and contaminant transport. Simulation of contaminant transport can be modified to describe linear steady state reactions between gaseous and liquid phases, and non-linear non-equilibrium reactions between liquid and solid phases. Simulation of contaminant transport also considers zero order production, along with two first order decay reactions, as follows:

(1) A reaction that is independent of other solutes;
(2) A reaction with solutes in a sequential chain of decay reactions.

Up to six species may be independently simulated using CHAIN 2D, or may be simulated in a unidirectional chain of decay. A specific distribution coefficient and a specific decay rate are needed for each radionuclide as inputs to the model simulation. It is possible to represent boundary conditions for contaminant transport as a constant flux or as a constant activity concentration. In addition, flow and transport may be estimated in a horizontal or vertical plane, or in an axisymmetric cylindrical coordinate system. The Galerkin finite element method is used for the solution of flow and transport equations.

In cases where the applicability of the assumption of vertical flow in the unsaturated zone is uncertain, the CHAIN 2D model may be applied to simulate radionuclide leaching from a disposal facility to groundwater. This might be particularly relevant in cases where there is a small leaching area, which can result in significant horizontal flow in highly stratified soils. CHAIN 2D can be applied to generate outputs of radionuclide activity concentration in soil water as a function of depth and time. In addition, the model can be applied to estimate cumulative solute flux across the base of the soil profile (i.e. at the water table).

V.2.5. CITRON

This computer code enables the user to calculate the activity concentrations of ^{222}Rn and its short half-lived progeny in open air following dispersion in the atmosphere from the ground. CITRON applies a Gaussian plume model with different representations of atmospheric stability (Briggs, Hosker, Pasquill) or a puff model (Doury). This model represents losses from the plume or puff by radioactive decay, and wet and dry deposition of the ^{222}Rn progeny. CITRON enables the user to model either point sources or area sources. Area sources can be represented either by a unique point source with a correction for the standard deviations used to represent dispersion, or with a regular meshwork consisting of point sources. Typical corrections for standard deviations or for other parameters of the model are available (e.g. to take account of the vertical wind profile).

V.2.6. ECOLEGO

Ecolego[80] is a simulation software tool, which can be applied to create dynamic models and to conduct deterministic and probabilistic simulations. The tool for risk assessments can also be used in the evaluation of complex dynamic systems that are evolving over time. Ecolego is primarily used to conduct risk assessments in the fields of radioecology, safety assessment for radioactive waste management, environmental physics and physiologically based pharmocokinetic modelling, but may also be applied in a variety of other technical disciplines. Specialized databases and other add-on tools have been developed for use in radioecology. This includes data for a large number of radionuclides and their decay products, which have been integrated into the software.

To improve the understandability of complex models with many interconnections, the models within Ecolego are represented by interaction matrices, as opposed to more traditional flow diagrams. These interaction matrices, together with hierarchical ‘containers’ (sub-systems) within the tool, facilitate the construction of large, complex models and their documentation.

Objects can be designated with units, images, comments and/or hyperlinks referring to other documents or to other Ecolego objects. Reports documenting and depicting complex models that have been developed using Ecolego, along with underlying assumptions, parameter values and model outputs, can be generated. Such reports can include a diverse range of information, such as interaction matrices, decay chains, equations, plots, tables, parameter values and descriptions, and can be saved in a variety of formats (e.g. pdf, HTML). Ecolego has been developed with no restrictions regarding the order of model creation to increase flexibility of use. For example, it is possible to use a parameter in equations before it has been defined. The

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[80] Available from http://ecolego.facilia.se/ecolego/show/HomePage

tool includes a real-time validation engine, which reports issues to the user, for example objects that have not yet been defined, objects without values or invalid equations.

The Ecolego model is typically applied as a compartmental model, as a solver of differential equations and an array of numerical solvers which the user can choose from. Some of the numerical solvers are optimized for numerically difficult models, whereas others may be applied for trivial models. An extensive list of probability density functions is available in Ecolego, which may be applied with Monte Carlo or Latin hypercube sampling and parameter correlation settings, to conduct advanced probabilistic analysis.

Ecolego has been used for assessment of doses from uranium tailings at sites in Ukraine, Tajikistan, and Uzbekistan. Ecolego was used for development of the SATURN software package, which was applied in the Bellazane case study.

Ecolego is commercially available, but also provides a freely available player with software features that have the same functionality as Ecolego, except for the flexibility to change the integral structure of the model. That said, it is possible for the user to assemble models by combining the components from a module library that has been created in Ecolego.

V.2.7. FECTUZ

FECTUZ is a one-dimensional model for use in prediction of the fate and transport of contaminants in the unsaturated zone [118−119], and is an extension of the VADOFT code [225]. Contaminant migration from a surface impoundment or landfill, through the unsaturated zone and to an unconfined aquifer with the water table at depth, can be simulated using FECTUZ. The Bateman's equation [119, 224] can be used to represent finite or infinite sources, taking account of decay. In addition, linear and non-linear adsorption and first order decay can be simulated using the model.

The applicability of FECTUZ is limited to simulation of hydrologically simple sites and either branched or unbranched chain decay for as many as seven radionuclide species (one parent with as many as six successive progeny) can be represented using the model. FECTUZ specifies a decay rate and distribution coefficient for each radionuclide.

Steady state flow is assumed within the unsaturated zone, consisting of one or more uniform layers of soil. Transport of contaminants in soil can be simulated in FECTUZ, applying an advective-dispersive equation, which applies three options for solution:

(1) Analytical, for single species, steady state decay with linear adsorption;
(2) Semi-analytical, for steady state and transient chain decay with linear sorption;
(3) Finite element, for chain decay with non-linear sorption.

Input of time varying precipitation data is possible, assuming steady state water flow during each precipitation event. Model outputs include radionuclide activity concentrations for specific locations and time points.

V.2.8. FRAMES

The 'Framework for Risk Analysis Multimedia Environmental Systems' (FRAMES [81], version 2.x) consists of several models covering environmental transport and risk assessment, in addition to user-developed databases, that function together in a flexible manner to accommodate user-designed exposure scenarios and exposure pathways. Data transfer between models is enabled through a common application programming interface. A tool is also available that applies data that have been generated using a variety of deterministic models for to conduct sensitivity and uncertainty analyses.

The user can select appropriate models and databases for each scenario. Once the model has been set up for a given scenario, it is possible for the user to redefine parameter values or the database being applied and rerun the model. FRAMES has an integrated operating package that can be used to run scenarios involving radionuclides, chemicals, atmospheric dispersion, and surface water and groundwater flow without specialized programming knowledge. As a result, it is possible for the user to focus on the environmental issue of interest, as opposed to program development.

Multiple medium specific models (e.g. air, water), along with the database of chemical properties and associated environmental parameters within the software tool[82], may be applied to conduct dose assessment and/or risk assessment for radiation and/or hazardous substances in the environment as a result of human activities.

The FRAMES model was evaluated by a participant in EMRAS II WG2, who noted that, although the model is extremely flexible and potentially useful in the modelling of historic NORM sites, its applicability might be limited due to the significant amount of effort that is needed to input large sets of analytical field data for sites that have been well characterized. The tutorial and supporting materials were found to be helpful, however, there was a lack of default selections for many of the data input screens, which might be challenging for inexperienced users.

V.2.9. GoldSim

GoldSim is a simulation software tool that applies Monte Carlo analysis in dynamic modelling of complex systems in science, engineering, and business. It is applied extensively in post-closure performance assessments of geological disposal of radioactive wastes. GoldSim can be applied in support of decision making and risk analysis through simulation of future performance. It is also possible to represent uncertainties and risks that are inherent to complex systems[83] using the tool.

GoldSim has a similar range of applications to AMBER (see Section V.2.1), however, the models are structured differently. In AMBER, the emphasis is on an overall model with sub-models, analogous to a main routine and subroutines in a high-level language program, such as FORTRAN. In contrast, GoldSim models are based on 'containers'. A container could hold, for example, all the material properties needed in a model, or all the input probability density

[81] FRAMES was developed by the Pacific Northwest National Laboratory (PNNL) under the sponsorship of the U.S. Environmental Protection Agency (EPA) Offices of Radiation and Indoor Air (ORIA) and Research and Development (ORD), U.S. Army Corps of Engineers (ACOE) Engineer Research and Development Center (ERDC), U.S. Nuclear Regulatory Commission (NRC), U.S. Department of Energy (DOE), and American Chemistry Council (ACC).
[82] The software, tutorial, and documentation can be found at http://mepas.pnnl.gov/earth/index.stm
[83] See https://www.goldsim.com/

functions being used. A container could be equivalent to a sub-model in AMBER, if it held the compartments between which transfers were calculated. GoldSim is also more focused on facilitating simulations of advective-dispersive transport than the other packages discussed in this publication.

V.2.10. MILDOS−AREA

MILDOS−AREA is a software tool[84] that is used by the US Nuclear Regulatory Commission (NRC) in the estimation of doses and radiation risks from authorized uranium mining activities. It has been updated to reflect changes in regulations, mining technologies relating to in situ leaching, graphical user interfaces, and technologies relating to internet software distribution. In situ leaching processes can be specified by users using an object-based geographic information system interface, incorporating updated dose assessment methodologies.

The MILDOS code was developed by Argonne National Laboratory based on the Uranium Dispersion and Dosimetry (UDAD) code [112, 123–124, 226−227]. Using a Gaussian plume dispersion model, the code can be used to estimates radionuclide activity concentrations in air and ground and the corresponding doses received by individuals within an 80 km radius of a source of radon and particulates. MILDOS includes models are for point sources (vents, stacks), as well as area sources (tailings areas, ore pads). Releases of particulates that are explicitly considered within the tool are limited to ^{238}U, ^{230}Th, ^{226}Ra and ^{210}Pb. Other radionuclides are accounted for implicitly by assuming secular equilibrium. For gaseous releases, consideration within the tool is limited to ^{222}Rn plus ingrowth of radioactive progeny. The transport model includes processes, such as plume depletion through deposition, resuspension of deposited radionuclides, radioactive decay, and ingrowth of radioactive progeny. Using MILDOS, it is possible to consider external exposure from groundshine and cloud immersion, internal exposure from ingestion of vegetables, milk and meat, and the inhalation pathway.

The MILDOS code does not consider releases to surface waters and groundwater. The main input parameters are radon release rate, meteorological conditions (as a site specific joint frequency distribution) and receptors located within 80 km from the source. The user provides parameter values that are used in the food pathway model.

It is possible to download the MILDOS code and documentation at no cost by accessing the Argonne National Laboratory MILDOS website.

V.2.11. MODELMAKER

ModelMaker4 defines model components and the relationships between them in a generalized way that allows almost any physically plausible system of ordinary differential equations (ODEs) to be represented. Therefore, by defining displacement and velocity as the two components and relating them through acceleration calculated from their instantaneous values, a damped, non-linear, harmonic oscillator can be simulated. Similarly, non-linear changes in the abundance of populations in a contaminated area, in response to changes in resources, variations in radiation dose rates, migration in and out of the area, and changes in the relative sizes of groups within the population with different radiosensitivities, can be readily simulated.

[84] See https://www.nrc.gov/reading-rm/doc-collections/nuregs/contract/cr7212/index.html

V.2.12. MULTIMED_DP

The MULTIMED_DP tool was initially intended for use in modelling the multimedia fate and transport of contaminants to simulate migration from a waste disposal unit (MULTIMED) via pathways in air, soil, groundwater and surface water [120–122]. Version 1.0 of the MULTIMED_DP model allows users to model transport in the unsaturated zone alone; however, the applicability of the model has been extended to also include the saturated zone through Monte Carlo simulation. The surface water module can be used to simulate contamination of a surface stream due to the complete interception of a plume at steady state in the saturated zone.

The semi-analytical solution within the unsaturated flow model can be used to simulate one-dimensional steady flow. The model includes an option that allows consideration of seasonal fluctuations in precipitation and evapotranspiration, while keeping the steady state assumption. The influence of advection, dispersion, volatilization, linear or non-linear sorption, hydrolysis and biodegradation are considered in the estimation of transport in the unsaturated zone. The model can be used to address steady state or time-variable infiltration and finite or infinite sources. Although one-dimensional uniform steady flow is assumed in the saturated zone, the saturated transport module considers three-dimensional dispersion, dilution due to recharge, linear adsorption and first-order decay. The extent of the disposal facility parallel to the flow direction, specified vertical dispersivity, saturation zone thickness, groundwater velocity and rate of infiltration are considered in estimating mixing in the underlying saturated zone.

The MULTIMED_DP model can handle three-generation chain decay, which can either consider one parent with one or two progeny or one parent with one immediate progeny and one or two second-generation progeny. An overall decay rate that combines different decay rates for material in various forms, individual decay rates, and distribution coefficients are needed for all members of the decay chain.

Application of MULTIMED_DP necessitates adequate understanding of the model and a large amount of input data.

V.2.13. RESRAD-BUILD

The RESRAD-BUILD model[85] may be applied in the analysis of radiation doses relating to remediation and occupancy of buildings contaminated with radioactive material. Model features include:

- Calculation of doses due to external exposure, inhalation (dust, radon), and ingestion (soil, dust);
- Possibility to model up to 10 sources and 10 receptors;
- Representation of different source geometries (point, line, plane, or volume);
- Simulation of building structure with up to three compartments;
- Representation of radioactive contamination in building materials or on surfaces;

[85] The RESRAD-BUILD code is part of the RESRAD family of codes, which now has its own website (http://resrad.evs.anl.gov/). The site has information on updates to the codes, and users can download the most recent release of the code(s) of interest. The website also contains information on upcoming training workshops and links to many documents relevant to application of the codes. Documentation for several of the RESRAD family of codes, including RESRAD-BUILD Version 3, can also be found on the RESRAD website. Further details are available in the User's manual for RESRAD-BUILD Version 3, which may be accessed at: https://resrad.evs.anl.gov/docs/ANL-EAD-03-1.pdf.

- Scenarios relating to occupancy of buildings (office worker, residential use) and building remediation (building renovation worker, decontamination worker).

Uncertainty and probabilistic analyses may be performed using the code [111−112], which incorporates default distributions of parameter values (based on average national data) for the selected parameters. Analytical results from RESRAD-BUILD may be generated in the form of text reports, graphic output and interactive output. It is possible to use the results of uncertainty analysis to determine the cost effectiveness of collecting additional information or data on variables and input parameters [113].

The time integrated dose over the duration of exposure can be calculated at user specified time points using the RESRAD-BUILD code. A quantity termed as the 'instantaneous dose at the specified time' can be calculated by setting the number of time integration points to one (see Ref. [110] for details). The RESRAD-BUILD database includes inhalation and ingestion dose coefficients [220], direct external exposure and air submersion dose coefficients [128], and radioactive half-lives [218].

Improvements of RESRAD-BUILD Version 3 made since its launch in 1994 include:

- Addition of a table for simpler input and review of shielding properties of sources/receptors;
- Improved 3-D display of source-receptor locations;
- Possibility for users to input radionuclide activity and the corresponding dose in SI units;
- Updates to the Help File and Table of Contents in the text report to improve use and navigability.

Model updates have not affected model outputs or predictions.

V.2.14. ROOM

Indoor gamma doses from building materials can be estimated by inputting geometrical and structural information on the room being modelled (e.g. wall dimensions), along with activity concentrations of natural radionuclides in building materials into the ROOM model. The details of the methodology that is applied in the ROOM model are described in Ref. [116].

V.3. SCREENING MODELS

Screening models available from the United Kingdom and the USA are described in the following Sections V.3.1−V.3.4 of this Appendix. US EPA recommends the use of screening models for risk assessments for its sites, rather than the use of dynamic transfer models.

V.3.1. Preliminary Remediation Goals (PRG) and Dose Compliance Concentrations (DCC) Calculators

The Preliminary Remediation Goals (PRG) and Dose Compliance Concentrations (DCC) Calculator is a screening model [86] that was developed by the US EPA to set preliminary remediation goals for remediation of a site for which more extensive evaluation and sampling have been undertaken through a field investigation.

[86] Internet based calculator is available at: http://epa-prgs.ornl.gov/radionuclides/. It is suggested that users begin the process by consulting the home page prior to any calculations being made.

Preliminary site specific remediation goals can be set for specific radionuclides using the PRG calculator, with input of responses to the following key questions about the site:

- Which are the important radionuclides that are present on-site?
- Which specific environmental media are contaminated?
- What are the assumptions regarding land use?
- What are the assumptions regarding with pathways of exposure of individuals?

Using the calculator, it is also possible to change the default exposure parameters when calculating preliminary site specific remediation goals. It is relatively straightforward to use the calculator, which provides default parameter values for use in cases where site specific values are not available, for example, if site specific characterization has not been undertaken. The calculator has a well-documented User's Guide and provides several risk assessment scenarios that may be using for evaluations. The PRG calculator comes with a free archived on-line training course linked to its on-line User's Guide. The PRG calculator is also consistent with US EPA's recommended model for risk assessment for chemicals in soil, water, and air – the regional screening level[87].

US EPA also developed a dose assessment calculator, called the DCC calculator[88], which is similar to the PRG calculator, for demonstrating compliance with dose-based regulations.

V.3.2. Building Preliminary Remediation Goals (BPRG) and Building Dose Compliance Concentrations (BDCC) Calculators

The Building Preliminary Remediation Goals (BPRG) calculator, which was developed by the Oak Ridge National Laboratory for the US EPA[89], provides an Excel-based screening model that has been designed to evaluate preliminary remediation goals for contaminated buildings. The tool can also be used for risk assessment and decision-aiding relating to such buildings. Contamination in the buildings considered can originate from radioactive construction materials or materials brought into or manufactured in the building.

Using the BPRG calculator, risks to indoor workers in an industrial building or to members of the public living in a contaminated house can be calculated. The tool can be used to evaluate of exposure due to direct external exposure (gamma), deposited dust (external exposure, ingestion) and ambient air (submersion, inhalation). It is possible for users to input site specific data or to use default values that are provided in the model, if specific parameter values are not available. Model outputs include preliminary estimates of both activity per unit mass and activity per unit area for the building under consideration, for example, in support of evaluation of remedial options or assessment of remediation effectiveness.

The BPRG calculator is consistent with US EPA's latest methodology for risk assessment for chemicals in buildings.

The US EPA also developed dose assessment calculator (i.e. the 'BDCC calculator')[90] similar to the BPRG calculator for demonstrating compliance with dose-based regulatory criteria for contaminated buildings.

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[87] The Regional Screening Level Calculator is available at: https://www.epa.gov/risk/regional-screening-levels-rsls-generic-tables
[88] Available at: http://epa-dccs.ornl.gov/
[89] The BPRG model is available at: http://epa-bprg.ornl.gov/
[90] The BDCC calculator is available at: https://epa-bdcc.ornl.gov/

V.3.3. Surface Preliminary Remediation Goals (SPRG) and Surface Dose Compliance Concentrations (SDCC) Calculators

The Surface Preliminary Remediation Goals (SPRG) calculator, which was developed by the Oak Ridge National Laboratory for the US EPA, is an Excel-based screening model calculator that has been designed to evaluate preliminary remediation goals for contaminated hard outdoor surfaces (e.g. streets, roads, pavements, sides of buildings). Contamination can be due to radioactive material used in construction materials. The tool allows users to evaluate risks to:

(1) Indoor workers in an industrial building;
(2) Outdoor workers at an industrial site;
(3) Members of the public in a contaminated house.

Sources of exposure considered are direct external exposure (gamma), deposited dust (external exposure, ingestion) and ambient air (submersion, inhalation). It is possible for users to input site specific data or to use default values that are provided in the model, if specific parameter values are not available. Model outputs include preliminary estimates both activity per unit mass or activity per unit area and for the building or surface of interest, for example, in support of evaluation of remedial options or assessment of remediation effectiveness. The SPRG calculator comes with free archived training courses linked to the on-line User's Guide[91].

US EPA also developed the Surface Dose Compliance Concentrations (SDCC) Calculator, which is a dose assessment calculator that is similar to the SPRG calculator for demonstrating compliance with dose based regulations for contaminated outdoor surfaces.

V.3.4. RCLEA

The 'Radioactively Contaminated Land Exposure Assessment Methodology' (RCLEA)[92] is applied in the United Kingdom Environment Agency to conduct exposure assessments for the management of sites under the extended Part 2A regime for managing contaminated land in the United Kingdom.

RCLEA is applicable in the calculation of potential doses for comparison with United Kingdom's regulatory criteria, and the calculation of 'guideline values' for radionuclide activity concentrations, in cases where reliable measurements are not yet available. Through initial generic calculations, users are able to select from the following four basic options for specified radionuclides (and activity concentrations, if available):

— Reference land uses: residential (without or with home grown vegetables), land allotments, commercial or industrial use;
— Building type: brick, timber-framed;
— Age of the exposed individual: adult, child, infant;
— Sex of the exposed individual: male, female.

[91] The SPRG model is available at: http://epa-sprg.ornl.gov/. The SDCC calculator is available at: http://epa-sdcc.ornl.gov/
[92] Information on RCLEA is available at: https://www.gov.uk/government/publications/rclea-software-application

REFERENCES

[1] INTERNATIONAL ATOMIC ENERGY AGENCY, Remediation Strategy and Process for Areas Affected by Past Activities or Events, IAEA Safety Standards Series No. GSG-15, IAEA, Vienna (2022).

[2] O'BRIEN, R., et al., Environmental modelling of NORM, Radioprotection **44**(5) (2009) 23−28.

[3] EUROPEAN COMMISSION, FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS, INTERNATIONAL ATOMIC ENERGY AGENCY, INTERNATIONAL LABOUR ORGANIZATION, OECD NUCLEAR ENERGY AGENCY, PAN AMERICAN HEALTH ORGANIZATION, UNITED NATIONS ENVIRONMENT PROGRAMME, WORLD HEALTH ORGANIZATION, Radiation Protection and Safety of Radiation Sources: International Basic Safety Standards, IAEA Safety Standards Series No. GSR Part 3, IAEA, Vienna (2014).

[4] INTERNATIONAL ATOMIC ENERGY AGENCY, Modelling the Transfer of Radionuclides from Naturally Occurring Radioactive Material (NORM): Report of the NORM Working Group of Theme 3, of the IAEA's Environmental Modelling for RAdiation Safety (EMRAS) Programme, IAEA-TECDOC-1678, IAEA, Vienna (2012).

[5] INTERNATIONAL ATOMIC ENERGY AGENCY, Governmental, Legal and Regulatory Framework for Safety, IAEA Safety Standards Series No. GSR Part 1 (Rev. 1), IAEA, Vienna (2016).

[6] INTERNATIONAL ATOMIC ENERGY AGENCY, IAEA Safety Glossary: Terminology Used in Nuclear Safety and Radiation Protection (2018 Edition), IAEA, Vienna (2019).

[7] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, The 2007 Recommendations of the International Commission on Radiological Protection, ICRP Publication 103 (2007).

[8] UNITED NATIONS SCIENTIFIC COMMITTEE ON THE EFFECTS OF ATOMIC RADIATION, Sources, effects and risks of ionizing radiation, Report to the General Assembly, with Annexes, UNSCEAR, United Nations, New York (1988).

[9] UNITED NATIONS SCIENTIFIC COMMITTEE ON THE EFFECTS OF ATOMIC RADIATION, Sources, effects and risks of ionizing radiation, Report to the General Assembly, with Annexes, UNSCEAR, United Nations, New York (2000).

[10] INTERNATIONAL ATOMIC ENERGY AGENCY, Management of Residues Containing Naturally Occurring Radioactive Material from Uranium Production and Other Activities, IAEA Safety Standards Series No. SSG-60, IAEA, Vienna (2021).

[11] INTERNATIONAL ATOMIC ENERGY AGENCY, Application of the Concepts of Exclusion, Exemption and Clearance, IAEA Safety Standards Series No. RS-G-1.7, IAEA, Vienna (2004).

[12] NATIONAL ACADEMY OF SCIENCES, Evaluation of Guidelines for Exposures to Technologically Enhanced Naturally Occurring Radioactive Materials, National Academy Press, Washington, DC (1999).

[13] INTERNATIONAL ATOMIC ENERGY AGENCY, Monitoring and Surveillance of Residues from the Mining and Milling of Uranium and Thorium, Safety Reports Series No. 27, IAEA, Vienna (2002).

[14] INTERNATIONAL ATOMIC ENERGY AGENCY, Radiation Protection and the Management of Radioactive Waste in the Oil and Gas Industry, Safety Reports Series No. 34, IAEA, Vienna (2003).

[15] INTERNATIONAL ATOMIC ENERGY AGENCY, Assessing the Need for Radiation Protection Measures in Work Involving Minerals and Raw Minerals, Safety Reports Series No. 49, IAEA, Vienna (2006).

[16] INTERNATIONAL ATOMIC ENERGY AGENCY, Radiation Protection and NORM Residue Management in the Zircon and Zirconia Industries, Safety Reports Series No. 51, IAEA, Vienna (2007).

[17] INTERNATIONAL ATOMIC ENERGY AGENCY, Radiation Protection and NORM Residue Management in the Production of Rare Earths from Thorium Containing Minerals, Safety Reports Series No. 68, IAEA, Vienna (2011).

[18] INTERNATIONAL ATOMIC ENERGY AGENCY, Radiation Protection and NORM Residue Management in the Titanium Dioxide and Related Industries, Safety Reports Series No. 76, IAEA, Vienna (2012).

[19] INTERNATIONAL ATOMIC ENERGY AGENCY, Radiation Protection and Management of NORM Residues in the Phosphate Industry, Safety Reports Series No. 78, IAEA, Vienna (2013).

[20] INTERNATIONAL ATOMIC ENERGY AGENCY, Management of NORM Residues, IAEA-TECDOC-1712, IAEA, Vienna (2013).

[21] INTERNATIONAL ATOMIC ENERGY AGENCY, Occupational Radiation Protection in the Uranium Mining and Processing Industry, Safety Reports Series No. 100, IAEA, Vienna (2020).

[22] INTERNATIONAL ATOMIC ENERGY AGENCY, Safety Assessment Methodologies for Near Surface Disposal Facilities: Results of a Coordinated Research Project, Volume 1: Review and enhancement of safety assessment approaches and tools, IAEA, Vienna (2004).

[23] INTERNATIONAL ATOMIC ENERGY AGENCY, Safety Assessment Methodologies for Near Surface Disposal Facilities: Results of a Coordinated Research Project, Volume 2: Test Cases, IAEA, Vienna (2004).

[24] OECD NUCLEAR ENERGY AGENCY, Features, Events and Processes (FEPs) for Geologic Disposal of Radioactive Waste: An International Database, Nuclear Energy Agency, Organisation for Economic Co-operation and Development, Paris (2000).

[25] INTERNATIONAL ATOMIC ENERGY AGENCY, Regulatory and management approaches for the control of environmental residues containing naturally occurring radioactive material (NORM), IAEA-TECDOC-1484, IAEA, Vienna (2006).

[26] FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS, INTERNATIONAL ATOMIC ENERGY AGENCY, INTERNATIONAL CIVIL AVIATION ORGANIZATION, INTERNATIONAL LABOUR OFFICE, INTERNATIONAL MARITIME ORGANIZATION, INTERPOL, OECD NUCLEAR ENERGY AGENCY, UNITED NATIONS OFFICE FOR THE COORDINATION OF HUMANITARIAN AFFAIRS, WORLD HEALTH ORGANIZATION, WORLD METEOROLOGICAL ORGANIZATION, Arrangements for the Termination of a Nuclear or Radiological Emergency, IAEA Standards Series No. GSG-11, Vienna (2018).

[27] ROUDAK, S.F., SNEVE, M.K., KISELEV, M., SHANDALA, N.K., Progress report on the regulatory cooperation program between the Norwegian Radiation Protection Authority and the Federal Medical Biological Agency of Russia. Final report of projects and other activities completed in 2008–2009 and plans for 2010–2011, Strålevern Rapport 2011:7, Norwegian Radiation Protection Authority, Østerås (2011).

[28] SWEDISH RADIATION PROTECTION INSTITUTE, An Overview of the BIOsphere MOdel Validation Study (BIOMOVS) Final Report, BIOMOVS, Technical Report No. 15, SRPI, Stockholm (1993).

[29] SWEDISH RADIATION PROTECTION INSTITUTE, An Overview of the BIOsphere MOdel Validation Study II (BIOMOVS II) Study and its Findings, BIOMOVS II, Technical Report No. 17, SRPI, Stockholm (1996).

[30] INTERNATIONAL ATOMIC ENERGY AGENCY, Testing of environmental transfer models using data from the remediation of a radium extraction site, Report of the Remediation Assessment Working Group of BIOMASS Theme 2, IAEA-BIOMASS-7, IAEA, Vienna (2004).

[31] FEDERAAL AGENTSCHAP VOOR NUCLEAIRE CONTROLE, Generic content of an orientation and descriptive study, FANC Technical Note No. 008-192-F (2009) (in French and Dutch).

[32] FEDERAAL AGENTSCHAP VOOR NUCLEAIRE CONTROLE, Intervention levels for lasting exposure situation, FANC Technical Note No. 009-050-F (2009) (in French and Dutch).

[33] FEDERAAL AGENTSCHAP VOOR NUCLEAIRE CONTROLE, FANC guidelines regarding assessment of dose for radioactively contaminated soil, FANC Technical Note No. KM_C458-20190705161647 (2019).

[34] INTERNATIONAL ATOMIC ENERGY AGENCY, UNITED NATIONS ENVIRONMENT PROGRAMME, Prospective Radiological Environmental Impact Assessment for Facilities and Activities, IAEA Safety Standards Series No. GSG-10, IAEA, Vienna (2018).

[35] INTERNATIONAL ATOMIC ENERGY AGENCY, The International Working Forum on the Regulatory Supervision of Legacy Sites – A Summary of Activities and Outcomes, IAEA-TECDOC-2016, IAEA, Vienna (2022).

[36] OECD NUCLEAR ENERGY AGENCY, Challenges in nuclear and radiological legacy management: Towards a common framework for the regulation of nuclear and radiological legacy sites and installations, Report of the Expert Group on Legacy Management, NEA Report No. 7419, Nuclear Energy Agency, Organisation for Economic Co-Operation and Development, Paris (2019).

[37] SNEVE, M.K., et al., Radiation Safety during Remediation of the SevRAO Facilities: 10 years of Regulatory Experience, J. Radiol. Prot. **35** (2015) 571–596.

[38] SNEVE, M.K., STRAND, P., Regulatory Supervision of Legacy Sites from Recognition to Resolution. Report of an international workshop, StrålevernRapport No. 2016:5, Østeras: Norwegian Radiation Protection Authority (2016).

[39] ALMESTAD, J., AMUNDSEN, I., SCHELLBERG, U., UHLENBRUCK, H., Activities within the Framework of the IAEA Contact Expert Group: Focus on input from Norway and Germany, StrålevernRapport No. 2017:14, Østerås: Statens strålevern (Eds Standring, W.J.F., Schüler, K.) (2017).

[40] SNEVE, M.K., POPIC, J.M., MIEGIEN-IWANIUK, K., Regulatory Supervision of Legacy Sites: The Process from Recognition to Resolution. Report of an international workshop, Lillehammer, Norway, 21–23 November 2017, StrålevernRapport No. 2018:4. Østeras: Norwegian Radiation Protection Authority (2018).

[41] MRDAKOVIC POPIC, J., SNEVE, M.K., VANDENHOVE, H., Radioecology as a Support to Regulatory Decision making on NORM and other Legacies, Related Waste Management and Disposal, Report of an International Workshop, StrålevernRapport No. 2018:2. Østerås: Statens strålevern (2018).

[42] NORWEGIAN RADIATION PROTECTION AUTHORITY, Study of Issues Affecting the Assessment of Impacts of Disposal of Radioactive and Hazardous Waste, NRPA Report No. 2018:6. Østerås: Statens strålevern (2018).

[43] SNEVE, M.K., Regulatory Framework of Decommissioning, Legacy Sites and Wastes from Recognition to Resolution: Building Optimization into the Process. Report of an international workshop, Tromsø, Norway, 29 October–1 November 2019. DSA Report No. 2020:05, Østerås: Norwegian Radiation and Nuclear Safety Authority (2020).

[44] TAZHIBAYEVA, I., et al., Management and regulatory supervision of legacy sites and radioactively contaminated lands in Republic of Kazakhstan, J. Radiol. Prot. **41** (2021) S454.

[45] NATIONAL RADIOLOGICAL PROTECTION BOARD, Methodology for estimating the doses to members of the public from the future use of land previously contaminated with radioactivity (W.B. Oatway and S.F. Mobbs), NRPB Report No. W36, National Radiological Protection Board, Chilton (2003).

[46] SHCHEBLANOV, V., SNEVE, M.K., BOBROV, A., Monitoring human factor risk characteristics at nuclear legacy sites in Northwest Russia in support of radiation safety regulation, J. Radiol. Prot. **32** (2012) 465–477.

[47] SMITH, J., et al., A framework for evaluating sustainable remediation options, and its use in a European regulatory context, Project: SuRF-UK, January 2010 (2010).

[48] EUROPEAN ATOMIC ENERGY COMMUNITY, FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS, INTERNATIONAL ATOMIC ENERGY AGENCY, INTERNATIONAL LABOUR ORGANIZATION, INTERNATIONAL MARITIME ORGANIZATION, OECD NUCLEAR ENERGY AGENCY, PAN AMERICAN HEALTH ORGANIZATION, UNITED NATIONS ENVIRONMENT PROGRAMME, WORLD HEALTH ORGANIZATION, Fundamental Safety Principles, IAEA Safety Standards Series No. SF-1, IAEA, Vienna (2006).

[49] YANKOVICH, T.L., et al., Practical Application of International Recommendations and Safety Standards in the Systematic Planning and Implementation of Remediation of Sites or Areas with Residual Radioactive Material, J. Radiol. Prot. **42** 020513 (2022).

[50] INTERNATIONAL ATOMIC ENERGY AGENCY, Guidelines for Remediation Strategies to Reduce the Radiological Consequences of Environmental Contamination, Technical Reports Series No. 475, IAEA, Vienna (2012).

[51] YANKOVICH, T., HACHKOWSKI, A., KLYASHTORIN, A., Options evaluation for remediation of the Gunnar Site using a decision-tree approach, Proceedings of the International Conference on Radioecology and Environmental Radioactivity, 7–12 September 2014, Barcelona, Paper No. ICRER-14-P-065, Spain (2014).

[52] YANKOVICH, T.L., et al., Balancing risk: Site remediation outside the environmental assessment. Waste Management, Decommissioning and Environmental Restoration for Canada’s Nuclear Activities, Canadian Nuclear Society, 11–14 September 2011, Toronto, Ontario (2011).

[53] YANKOVICH, T.L., HACHKOWSKI, A., KLYASHTORIN, A., Filtering of Remedial Options and Identification of a Preferred Option for the Gunnar Environmental Impact Statement (EIS), Saskatchewan Research Council Technical Report No. 12194-320-4B12, Saskatchewan, Canada (2012).

[54] INTERNATIONAL ATOMIC ENERGY AGENCY, INPRO Methodology for Sustainability Assessment of Nuclear Energy Systems: Environmental Impact of Stressors, INPRO Manual, IAEA Nuclear Energy Series No. NG-T-3.15, IAEA, Vienna (2016).

[55] INTERNATIONAL ATOMIC ENERGY AGENCY, Lessons Learned from Environmental Remediation Programmes, IAEA Nuclear Energy Series No. NW-T-3.6, IAEA, Vienna (2014).

[56] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Radiological Protection against Radon Exposure, ICRP Publication 126 (2014).

[57] INTERNATIONAL ATOMIC ENERGY AGENCY, Communication and Consultation with Interested Parties by the Regulatory Body, IAEA Safety Standards Series No. GSG-6, IAEA, Vienna (2018).

[58] PEPIN, S., et al., The IAEA Environmental Modelling for Radiation Safety programme (EMRAS II) – working group on “Reference approaches to modelling for management and remediation at NORM and legacy sites”, EU NORM 1st International Symposium, 5−8 June 2012, Tallinn, Estonia (2012).

[59] KONTIĆ, B., et al., Demonstrating the Use of a Framework for Risk-Informed Decisions with Stakeholder Engagement through Case Studies for NORM and Legacy Sites, J. Radiol. Prot. **42** 020504 (2022).

[60] PEPIN, S., et al., Intermodel comparison for the radiological assessment of Zapadnoe and Tessenderlo Case Studies with implications for selection of remediation strategy, JRP Special Issue on MODARIA II programme, J. Radiol. Prot. **42** 020510 (2022).

[61] INTERNATIONAL ATOMIC ENERGY AGENCY, Generic models for use in assessing the impact of discharges of radioactive substances to the environment, Safety Reports Series No. 19, IAEA, Vienna (2001).

[62] INTERNATIONAL ATOMIC ENERGY AGENCY, The Safety Case and Safety Assessment for the Predisposal Management of Radioactive Waste, IAEA Safety Standards Series No. GSG-3, IAEA, Vienna (2013).

[63] UNITED STATES DEPARTMENT OF DEFENCE, UNITED STATES DEPARTMENT OF ENERGY, UNITED STATES ENVIRONMENTAL PROTECTION AGENCY, UNITED STATES NUCLEAR REGULATORY COMMISSION, Multi-Agency Radiation Survey and Site Investigation Manual (MARSSIM) NUREG-1575, Rev. 1, EPA 402-R-97-016, Rev. 1, DOE/EH-0624, Rev. 1, August (2000).

[64] UNITED STATES NUCLEAR REGULATORY COMMISSION, A Proposed Nonparametric Statistical Methodology for the Design and Analysis of Final Status Decommissioning Survey, NUREG-1505, US NRC, Washington, DC (1997).

[65] HERROLD, J.F., BRIGHTWELL, M.S., Using Regulatory Guidance in Decommissioning Under an NRC Broad-Scope Byproduct Material License, Radiation Protection Management, **20** 3 (2003) 20–28.

[66] BURGESS, P.H., Handbook on measurement methods and strategies at very low levels and activities, European Commission Report EUR 17624 (1998).

[67] INTERNATIONAL ATOMIC ENERGY AGENCY, Leadership and Management for Safety, IAEA Safety Standards Series No. GSR Part 2, IAEA, Vienna (2016).

[68] INTERNATIONAL ATOMIC ENERGY AGENCY, Leadership, Management and Culture for Safety in Radioactive Waste Management, IAEA Safety Standards Series No. GSG-16, IAEA, Vienna (2022).

[69] INTERNATIONAL ATOMIC ENERGY AGENCY, Application of radiation protection principles to the cleanup of contaminated areas, Interim report for comment, IAEA-TECDOC-987, IAEA, Vienna (1997).

[70] INTERNATIONAL ATOMIC ENERGY AGENCY, Integrated Approach to Planning the Remediation of Sites Undergoing Decommissioning, IAEA Nuclear Energy Series No. NW-T-3.3, IAEA, Vienna (2009).

[71] BALONOV, M., et al., Optimisation of environmental remediation: how to select and use the reference levels. J Radiol Prot. **38**(2) (2018) 819–830.

[72] SANDIA NATIONAL LABORATORIES, Residual Radioactive Contamination from Decommissioning, Parameter Analysis, Draft Report for Comment, NUREG/CR-5512, Volume 3, SAND99-2148 prepared for the US Nuclear Regulatory Commission, Washington, DC (1999).

[73] INTERNATIONAL ATOMIC ENERGY AGENCY, Quantification of Radionuclide Transfer in Terrestrial and Freshwater Environments for Radiological Assessments, IAEA-TECDOC-1616, IAEA, Vienna (2009).

[74] INTERNATIONAL ATOMIC ENERGY AGENCY, Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Terrestrial and Freshwater Environments, Technical Reports Series No. 472, IAEA, Vienna (2010).

[75] PARKIN, G., et al., A physically based approach to modelling radionuclide transport in the biosphere, J. Radiol. Prot. **19** (1999) 319–332.

[76] INTERNATIONAL ATOMIC ENERGY AGENCY, Handbook of Parameter Values for the Prediction of Radionuclide Transfer to Wildlife, Technical Reports Series No. 479, IAEA, Vienna (2014).

[77] SAFEGROUNDS LEARNING NETWORK, Main SAFEGROUNDS guidance document: Good practice guidance for the management of contaminated land on nuclear and defence sites, Version 2, June 2009 (2009).

[78] GALLERAND, M.-O., Review of the radiological characteristics of the environment around a former uranium mine site in France: the case of the Saint Pierre site in the Cantal district of France, International Workshop on Ra-226 Environmental Behaviour, 4–5 May 2010, Châtenay-Malabry, France (2010).

[79] ZEEVAERT, TH., et al., Manual on Restoration Strategies for Radioactively Contaminated Sites. RESTRAT-TD 14, EC-RESTRAT project co-founded by the Nuclear Fission Safety Programme of the European Commission, F14P-CT95-0021a (1999).

[80] ZEEVAERT, TH., et al., Evaluation and ranking of restoration strategies for radioactively contaminated sites, J. Env. Radioact. **56** (2001) 33–50.

[81] NUCLEAR DECOMMISSIONING AUTHORITY, The NDA Value Framework, Version 1.2 (2016).

[82] UNITED STATES DEPARTMENT OF ENERGY, Guidebook to Decision-making Methods, Report No. WSRC-IM-2002-00002 (2001).

[83] AUSTRIAN FEDERAL ENVIRONMENT AGENCY (on behalf of CLARINET), Review of Decision Support Tools for Contaminated Land and their Use in Europe, A report from the Contaminated Land Rehabilitation Network for Environmental Technologies (2002).

[84] ULANOVSKY, A., et al., ReSCA: decision support tool for remediation planning after the Chernobyl accident, Radiat Environ Biophys. **50**(1) (2011) 67–83.

[85] KLOS, R., THORNE, M.C., Use of interaction matrices to formalise the development of conceptual models of contaminant transport in the biosphere and the translation of those conceptual models into mathematical models, J. Radiol. Prot. **40** (2020) 40–67.

[86] THORNE, M., Modelling radionuclide transport in the environment and calculating radiation doses, In: Poinssot, C and Geckeis, H (Eds.), Radionuclide Behaviour in the Natural Environment: Science, implications and lessons for the nuclear industry, Woodhead Publishing Limited, Sawston, Cambridge, UK (2012) 517–569.

[87] BEAR, J., Hydraulics of Groundwater, McGraw-Hill, New York (2007) 592 pp.

[88] HISCOCK, K.M., BENSE, V.F., Hydrogeology: Principles and Practice, Second Edition, Wiley Blackwell, Oxford, UK (2014) 552 pp.

[89] SHAO, Q., MATTHÄI, S.K., GROSS, L., Efficient modelling of solute transport in heterogeneous media with discrete event simulation, J. Comput. Phys. **384** (2019) 134–150.

[90] CLARKE, R.H., A model for short and medium range dispersion of radionuclides released to the atmosphere, National Radiological Protection Board Report, NRPB-R91 (1979).

[91] GALLACHER, D.J., et al., Dispersion of positron emitting radioactive gases in a complex urban building array: a comparison of dose modelling approaches, J. Radiol. Prot. **36** (2016) 746–784.

[92] GALLACHER, D.J., ROBINS, A.G., HAYDEN, P., Conversion of simulated radioactive pollutant gas concentrations for a complex building array into radiation dose, J. Radiol. Prot. **36** (2016) 785–818.

[93] JONES, A.R., THOMSON, D.J., HORT, M., DEVENISH, B., The U.K. Met Office's next-generation atmospheric dispersion model, NAME III, In Borrego C and Norman A-L (Eds.), Air Pollution Modeling and its Application XVII (Proceedings of the 27th NATO/CCMS International Technical Meeting on Air Pollution Modelling and its Application), Springer (2007) 580–589.

[94] ESTOURNEL, C., et al., Assessment of the amount of Cesium-137 released into the Pacific Ocean after the Fukushima accident and analysis of its dispersion in Japanese coastal waters, J. Geophys. Res.: Oceans, **117** (C11014) (2012).

[95] INTERNATIONAL ATOMIC ENERGY AGENCY, Sediment Distribution Coefficients and Concentration Factors for Biota in the Marine Environment, Technical Reports Series No. 422, IAEA, Vienna (2004).

[96] ROBLES, B., SUAÑEZ, A., CANCIO, D., Metodología de Evaluación del Impacto Radiológico a la Población con Aplicación de Nuevos Criterios de Protección Radiológica – iniciativa ATYCA, Editorial CIEMAT, Madrid (2000) (in Spanish).

[97] ROBLES, B., SUAÑEZ, A., MORA, J.C., CANCIO, D., Modelos implementados en el código CROM, Colección Documentos CIEMAT, Madrid (2007) (in Spanish).

[98] ROBERTS, D., SMITH, J.G., Evaluation of CROM – A software Application that Implements IAEA SR-19, v 2.0, Health Protection Agency, Radiation Protection Division, RPD-EA-11-2005 (2005).

[99] SMITH, J.G., SIMMONDS, J.R. (Eds), The methodology for assessing the radiological consequences of routine releases of radionuclides to the environment used in PC-CREAM 08 (2009).

[100] SANDIA NATIONAL LABORATORIES, Comparison of the Models and Assumptions used in the DandD 1.0, RESRAD 5.61, and RESRAD-Build 1.50 Computer Codes with Respect to the Residential Farmer and Industrial Occupant Scenarios Provided in NUREG/CR-5512, Draft Report for Comment, NUREG/CR-5512, Volume 4, SAND99-2147, prepared for the US Nuclear Regulatory Commission, Washington, DC (1999).

[101] SANDIA NATIONAL LABORATORIES, Residual Radioactive Contamination From Decommissioning, User's Manual, DandD Version 2.1, NUREG/CR-5512, Volume 2, SAND2001-0822P, prepared for the US Nuclear Regulatory Commission, Washington, DC (2001).

[102] KRETZSCHMAR, J.G., et al., IFDM: The Immission Frequency Distribution Model, SCK/CEN, Mol (1977).

[103] SIMUNEK, J., SEJNA, M, VAN GENUCHTEN, M.TH., The HYDRUS-2D software package for simulating two-dimensional movement of water, heat, and multiple solutes in variably saturated media, Version 2.0, IGWMC-TPS-53, International Groundwater Modelling Center, Colorado School of Mines, Golden, CO (1999) 251 pp.

[104] SIMUNEK, J., VAN GENUCHTEN, M.TH., SEJNA, M., The HYDRUS-1D Software Package for simulating the one-dimensional movement of water, heat and multiple solutes in variably-saturated media, Version 3.0, HYDRUS Software Series 1, Department of Environmental Sciences, University of California Riverside, Riverside, CA (2005).

[105] NEXIA SOLUTIONS, User Guide for ReCLAIM, V3.0, Report 9084, Issue 02 (2008).

[106] PACIFIC NORTHWEST LABORATORY, Residual Radioactive Contamination from Decommissioning, Final Report, NUREG/CR-5512, Volume 1, PNL-7994, prepared for the US Nuclear Regulatory Commission, Washington, DC (1992).

[107] YU, C., et al., User's Manual for RESRAD Version 6, Environmental Assessment Division; Argonne National Laboratory, Argonne, IL (2001).

[108] YU, C., et al., Data Collection Handbook to Support Modelling Impacts of Radioactive Material in Soil, Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, IL (1993).

[109] YU, C., GNANAPRAGASAM, E., CHENG, J.-J., BIWER, B., Benchmarking of RESRAD-OFFSITE: Transition from RESRAD (onsite) to RESRAD-OFFSITE and Comparison of the RESRAD-OFFSITE Predictions with Peer Codes, ANL/EVS/TM/06-3, DOE/EH-0708 (2006).

[110] YU, C., et al., User's Manual for RESRAD-OFFSITE Version 2, ANL/EVS/TM/07-1, DOE/HS-0005, NUREG/CR-6937 (2007).

[111] YU, C., et al., Development of Probabilistic RESRAD 6.0 and RESRAD-BUILD 3.0 Computer Codes, Environmental Assessment Division, Argonne National Laboratory, Argonne, IL, ANL/EAD/TM-98, NUREG/CR-6697, prepared for the US Nuclear Regulatory Commission, Washington, DC (2000).

[112] LEPOIRE, D., et al., Probabilistic Modules for the RESRAD and RESRAD-BUILD Computer Codes, User Guide, ANL/EAD/TM-91, NUREG/CR-6692, prepared for US Nuclear Regulatory Commission, Washington, DC (2000). (Current manual – Version 3: https://resrad.evs.anl.gov/docs/ANL-EAD-03-1.pdf)

[113] KAMBOJ, S., et al., Probabilistic Dose Analysis Using Parameter Distributions Developed for RESRAD and RESRAD-BUILD Codes, ANL/EAD/TM-89 NUREG/CR-6676, prepared for US Nuclear Regulatory Commission, Washington, DC (2000).

[114] DECISIONEERING INCORPORATED, CRYSTAL BALL Version 4.0, forecasting and risk analysis tool for spreadsheet users (1996).

[115] BROWN, J.E., et al., The ERICA Tool, Journal of Environmental Radioactivity, 99: 1371–1383 (2008).

[116] MARKKANEN, M., Radiation dose assessments for materials with elevated natural radioactivity, Report No. STUK-B-STO 32, Finnish Centre for Radiation and Nuclear Safety, Helsinki (1995).

[117] SIMUNEK, J., VAN GENUCHTEN, M.TH., The CHAIN 2D Code for Simulating the Two-Dimensional Movement of Water, Heat and Multiple Solutes in Variably Saturated Porous Media, Version 1.1, Research Report No. 136, US Salinity Laboratory, USDA, ARS, US Department of Agriculture, Riverside, CA (1994).

[118] UNITED STATES ENVIRONMENTAL PROTECTION AGENCY, EPA's Composite Model for Leachate Migration with Transformation Products, EPACMTP, Background Document, Office of Solid Waste, Washington, DC (1995).

[119] UNITED STATES ENVIRONMENTAL PROTECTION AGENCY, EPA's Composite Model for Leachate Migration with Transformation Products, EPACMTP, User's Guide, Office of Solid Waste, Washington, DC (1995).

[120] LIU, S., MILLS, W.B., BAENA, B., Multimedia Exposure Assessment Modeling Including Fate and Transformation Products (MULTIMED_ DP 1.0): Implementation, Tests and User's Manual, Technical Report prepared for US Environmental Protection Agency, Athens, GA (1995).

[121] SALHOTRA, A.M., et al., Multimedia Exposure Assessment Model (MULTIMED 2.0) for Evaluating the Land Disposal of Wastes – Model Theory, Technical Report prepared for the US Environmental Protection Agency, Athens, GA (1995).

[122] SHARP-HANSEN, S., et al., A Subtitle D Landfill Application Manual for the Multimedia Exposure Assessment Model (MULTIMED 2.0), Technical Report prepared for the US Environmental Protection Agency, Athens, GA (1995).

[123] PACIFIC NORTHWEST LABORATORY, MILDOS – A Computer Program for Calculating Environmental Radiation Doses From Uranium Recovery Operations, NUREG/CR-2011, PNL-3767, prepared for the US Nuclear Regulatory Commission, Washington, DC (1981).

[124] PACIFIC NORTHWEST LABORATORY, Methods for Estimating Radioactive and Toxic Airborne Source Terms for Uranium Milling Operations, NUREG/CR-4088, PNL-5338, prepared for the US Nuclear Regulatory Commission, Washington, DC (1984).

[125] EUROPEAN COMMUNITIES, Radiation Protection 122, Practical Use of the Concepts of Clearance and Exemption – Part II, Application of the Concepts of Exemption and Clearance to Natural Radiation Sources, Office for Official Publications of the European Communities, Luxembourg (2002).

[126] INTERNATIONAL ATOMIC ENERGY AGENCY, Derivation of Activity Concentration Values for Exclusion, Exemption and Clearance, Safety Report Series No. 44, IAEA, Vienna (2005).

[127] ESPINOSA, A., ARAGON, A., STRADLING, N., HODGSON, A., BIRCHALL, A., Assessment of doses to adult members of the public in Palomares from inhalation of plutonium and americium, Radiat. Prot. Dosim. **79**(1–4) (1998) 161–164.

[128] ECKERMAN, K.F., RYMAN, J.C., External Exposure to Radionuclides in Air, Water, and Soil, Federal Guidance Report No. 12, EPA-402-R-93-081, US Environmental Protection Agency, Office of Radiation and Indoor Air, Washington, DC (1993) (superseded by US EPA, Federal Guidance Report No. 15: External Exposure to Radionuclides in Air, Water and Soil, EPA 402/R19/002, August 2019).

[129] ECKERMAN, K.F., et al., Cancer Risk Coefficients for Environmental Exposure to Radionuclides, Federal Guidance Report No. 13, EPA-402-R-99-001, US Environmental Protection Agency, Office of Radiation and Indoor Air, Washington, DC (1999) (superseded by US EPA, Update to the Federal Guidance Report No. 13, April 2002, https://www.epa.gov/radiation/federal-guidance-report-no-13-cd-supplement).

[130] KERRY ROWE, R., QUIGLEY, R.M., BRACHMAN, R.W.I., Barrier Systems for Waste Disposal Facilities, 2nd Edition, CRC Press (2004).

[131] PAYNE, T.E., et al., Trench 'Bathtubbing' and Surface Plutonium Contamination at a Legacy Radioactive Waste Site, Environ. Sci. Technol. **47** (2013) 13284–13293.

[132] BROWN, J.E., et al., The ERICA Tool, J. Environ. Radioact. **99** (2008) 1371–1383.

[133] BROWN, J.E., et al., A new version of the ERICA tool to facilitate impact assessments of radioactivity on wild plants and animals, J. Environ. Radioact. **153** (2016) 141–148.

[134] UNITED NATIONS SCIENTIFIC COMMITTEE ON THE EFFECTS OF ATOMIC RADIATION, Effects of radiation on the environment, A/AC.82/R.549, UNSCEAR 1996 Report, United Nations, Vienna (1996).

[135] UNITED NATIONS SCIENTIFIC COMMITTEE ON THE EFFECTS OF ATOMIC RADIATION, Sources and effects of ionizing radiation, Volume I: Sources, Report to the General Assembly, UNSCEAR 2008 Report, United Nations, Vienna (2008).

[136] UNITED STATES DEPARTMENT OF ENERGY, A graded approach for evaluating radiation doses to aquatic and terrestrial biota, US Department of Energy, Final Technical Standard No. DOE-STD-1153-2002, Washington, DC (2002).

[137] JONES, D., et al., Principles and issues in radiological ecological risk assessment, J. Environ. Radioact. **66**(1–2) (2003) 19–39.

[138] NORWEGIAN GEOTECHNICAL INSTITUTE, Kartlegging av omfang og kostnader ved eventuell senere opprydning av radioaktivt materiale ved Søve gruver, Report No. 20091927-00-14-R (2009) (in Norwegian).

[139] INSTITUTE FOR ENERGY TECHNOLOGY, Radiologisk kartlegging av området rundt tidligere Søve gruver. Institutt for energiteknikk, IFE Report No. IFE/KR/F – 2006/174 (2006) (in Norwegian).

[140] YU, C., et al., RESRAD-OFFSITE: A new member of the RESRAD family of codes, Radioprotection **44** (2009) 659–664.

[141] AVILA, R., BROED, R., PEREIRA, A., ECOLEGO – A toolbox for radioecological risk assessment, Proceedings of the International Conference on the Protection from the Effects of Ionizing Radiation, IAEA-CN-109/80, Stockholm (2003), pp. 229–232.

[142] DE JONG, P., VAN DIJK, W., Modeling Gamma Radiation Dose in Dwellings Due to Building Materials, Health Physics **94**(1) (2008) 33–42.

[143] WORLD HEALTH ORGANIZATION, WHO Handbook on Indoor Radon, A Public Health Perspective, WHO, Geneva (2009).

[144] VANDENHOVE, H., et al., Radiation Protection 115: Investigation of a possible basis for a common approach with regard to the restoration of areas affected by lasting radiation exposure as a result of past or old practice or work activity, CARE final report (1999).

[145] BLOMMAERT, W., MANNAERTS, K., PEPIN, S., Application of an environmental remediation methodology: Theory vs. Practice. Reflections and two Belgian case studies. International Conference on Radioactive Waste Management and Environmental Remediation, ICEM2011-59184 (2011) 1011–1021.

[146] VANMARCKE, H., Sanering van de omgevingsbesmetting met Radium-226 te Olen en Geel, SCK/CEN, Mol (1997) (in Dutch).

[147] GOLDER ASSOCIATES INC., Uranium Mill Tailings Radon Flux Calculations, Project No. 073-81694.23, August 2010 (2010).

[148] LI, Z., et al., A Particle Size Distribution Model for Tailings in Mine Backfill, Metals **12**(4) 594 (2022) 1−11.

[149] INSTITUT DE RADIOPROTECTION ET DE SÛRETÉ NUCLÉAIRE, Expertise globale du bilan décennal environnemental d'AREVA NC: stockage de Bellezane et impact environnemental à l'échelle du bassin versant du Ritord, IRSN Report No. DEI/2007-01, IRSN, Paris, January 2007 (2007).

[150] AREVA, Etablissement de Bessines: Bilan décennal environnemental 1994 – 2003, AREVA, December 2004 (2004).

[151] AREVA-COGEMA, Site de Bellezane: Expertise hydrogéologique et préconisations pour le site de stockage de sédiments et de boues, AREVA-COGEMA, RTr00046a, May 2006 (2006).

[152] INSTITUT DE RADIOPROTECTION ET DE SÛRETÉ NUCLÉAIRE, Guide méthodologique – Gestion des sites potentiellement pollués par des substances radioactives, IRSN, Paris, December 2011 (2011).

[153] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Age-dependent Doses to the Members of the Public from Intake of Radionuclides, Part 5: Compilation of Ingestion and Inhalation Coefficients, ICRP Publication 72 (1996).

[154] NIRAS/ONDRAF, Element dependent environmental input parameters for the biosphere model, NIRAS/ONDRAF report in the frame of the project for a near surface disposal of category A waste at Dessel, NIROND-TR 2008-26 E V3 (2011).

[155] Bundesministerium für Umwelt, Naturschutz und nukleare Sicherheit, Berechnungsgrundlagen zur Ermittlung der Strahlenexposition infolge bergbaubedingter Umweltradioaktivität (Berechnungsgrundlagen - Bergbau), BMU, Berlin (1999). (Assessment principles for estimation of radiation exposure resulting from mining-related radioactivity in environment; German Federal Ministry for Environment, Nature conservation and Reactor Safety, 1999).

[156] BAES, C.F. III, SHARP, R.D., A proposal for estimation of soil leaching and leaching constants in assessment models, J. Environ. Qual. **12** (1983) 17−28.

[157] LEDOUX, E., SCHMITT, J.M., Etude du fonctionnement hydrogéochimique de l'ancien site minier de Bellezane (Limousin, France), No R100119EL, BGM/DGS RT 10/004, Centre de Géosciences, Ecole des mines de Paris, Fontainebleau (2010).

[158] SMITH, G.M., SNEVE, M.K., MARKAROV, V.G., Making the link between radiological assessment, nuclear safety assessment and environmental impact assessment, as applied to unloading of the Lepse spent fuel storage vessel. Proceedings of the International Congress of the International Radiation Protection Association, IRPA-10, Hiroshima, Japan, 14−19 May 2000, Ref. No. EDB-01:040460, 8 pp. (2000).

[159] SNEVE, M.K., et al., Development of the regulatory guidance documents within the Lepse Regulatory Project, Norwegian Radiation Protection Authority Report No. 2000:9, Østerås (2000).

[160] EUROPEAN COMMISSION, Support in the development of regulatory procedures for licensing Lepse waste management operations, European Commission Report EUR 19896, Luxembourg (2001).

[161] SHANDALA, N.K., et al., Regulatory supervision of sites for spent fuel and radioactive waste storage in the Russian Northwest, J. Radiol. Prot. **28**(4) (2008) 453−465.

[162] SNEVE, M.K., SHANDALA, N.K., SMITH, G.M., Progress in Norwegian-Russian Regulatory Cooperation in Management of the Nuclear Legacy, Proceedings of the Waste Management Conference WM2008, 24−28 February 2008, Phoenix, Arizona, Paper No. 8289 (2008) 11 pp..

[163] CHIZHOV, K., et al., Radiation situation dynamics at the Andreeva Bay site for temporary storage of spent nuclear fuel and radioactive waste over the period 2002–2016, J. Radiol. Prot. **38**(2) (2018) 480−509.

[164] NORWEGIAN RADIATION PROTECTION AUTHORITY, Regulatory Improvements Related to the Radiation and Environmental Protection during Remediation of the Nuclear Legacy Sites in North West Russia, Report of work completed by NRPA and FMBA of Russia in 2007, NRPA Report No. 2008:7, Østerås (2008).

[165] NORWEGIAN RADIATION PROTECTION AUTHORITY, Progress report on regulatory cooperation program between the Norwegian Radiation Protection Authority and the Federal Medical Biological Agency of Russia: Projects and other activities completed in 2008–2009 and plans for 2010–2011, NRPA Report No. 2011:7, Østerås (2011).

[166] RANKIN, R., What is optimum humidity? Respir. Care Clin. N. Am., **4**(2) (1998) 321–328.

[167] Propuesta de gestion integral de los residuos de la mineria del uranio en la Provincia de Córdoba, Informe IF-40M-020 (2005) (in Spanish).

[168] Eco-regiones de la Argentina. Administración de Parques Nacionales. PRODIA. BID.

[169] COMISIÓN NACIONAL DE ENERGÍA ATÓMICA, Estudio de plancton, bentos, y nectos del gradiente Río Cajón-Rio San Antonio, Informe Técnico de la situación de la biota-Mangeud-Pucheta-Noviembre (2000) (in Spanish).

[170] FERNANDES, H.M., et al., Management of uranium mill tailings: geochemical: processes and radiological risk assessment, J. Enviro. Radioact. **30**(1) (1996) 69–95.

[171] FERNANDES, H.M., FRANKLIN, M.R., GOMIERO, L.A., Critical analysis of the waste management performance of two uranium production units in Brasil, Part I: Poços de Caldas production center, J. Enviro.l Manage. **87** (2008) 59–72.

[172] FERNANDES H.M., FRANKLIN M.R., VEIGA L.H., Acid rock drainage and radiological environmental impacts, A study case of the Uranium mining and milling facilities at Poços de Caldas, Waste Management **18** (2008) 169–181.

[173] GROHMANN C.H., RICCOMINI C., ALVES F.M., SRTM-based morphotectonic analysis of the Poços de Caldas Alkaline Massif, southeastern Brazil, Computers & Geosciences **33**(1) (2007) 10–19.

[174] AMARAL, E.C.S., CARVALHO, Z.L., GODOY, J.M.O., Transfer of ^{226}Ra and ^{210}Pb to forage and milk in a Brazilian high natural radioactivity region, Radiat. Prot. Dosim. **24** (1988) 119–121.

[175] VASCONCELLOS, L.M.H., AMARAL, E.C.S., VIANNA, M.E., PENNA FRANCA, E., Uptake of ^{226}Ra and ^{210}Pb by food crops cultivated in a region of high natural radioactivity in Brazil, J. Enviro. Radioact. **5** (1987) 287–302.

[176] AMARAL, E.C.S., et al., The Environmental Impact of the Uranium Industry: is the Waste Rock a Significant Contributor? Radiat. Prot. Dosim. **22**(3) (1988) 165–171.

[177] AZEVEDO, H.L.P., AMARAL, E.C.S., GODOY, J.M., Evaluation of the ^{226}Ra transport by river sediments surrounding the Brazilian uranium mining and milling facilities, Environ. Pollut. **51** (1988) 259–268.

[178] BARCELLOS, C., AMARAL, E., ROCHEDO, E., Radionuclide transport by Poços de Caldas Plateau rivers, Brazil, Environ. Technol. **11** (1990) 533–540.

[179] FERNANDES, H.M., et al., Environmental impact assessment of uranium mining and milling facilities: A study case at the Poços de Caldas uranium mining and milling site, Brazil, J. Geochem. Explor. **52** (1995) 161–173.

[180] AMARAL, E.C.S., AZEVEDO, H.L.P., MENDONÇA A.H., Pre-operational environmental survey at the uranium mine and mill site, Poços de Caldas, Minas Gerais, Brazil, Sci. Total Enviro. **42** (1985) 257–266.

[181] PRADO, V.C.S., O Impacto da Produção de Concentrado de urânio sobre a Qualidade da Água dos Rios-Um estudo de caso na área do complexo minero-industrial do planalto de Poços de Caldas, COPPE-UFRJ, Rio de Janeiro, Brasil (1994) (in Portuguese).

[182] DOLCHINKOV, N.T., Influence of uranium mines in the formation of natural background radiation, Science. Business. Society **1**(5) (2016) 55–58.

[183] FLORJANČIČ, A.P., et al., Rudnik urana Žirovski vrh, Doneski 1. Didakta, Radovljica (2000) (in Slovenian).

[184] KRIŽMAN, M., BYRNE, A.R., BENEDIK, L., Distribution of ^{230}Th in milling wastes from the Žirovski vrh uranium mine (Slovenia), and its radioecological implications, J. Environ. Radioact. **26** (1995) 223–235.

[185] ŠTROK, M., et al., Surveillance results of the radioactivity in the Žirovski vrh uranium mine environment for the year 2010, IJS-DP-10680, "Jožef Stefan" Institute, Ljubljana (2011).

[186] ŠTROK, M., SMODIŠ, B., Fractionation of natural radionuclides in soils from the vicinity of a former Uranium mine Žirovski vrh, Slovenia, J. Environ. Radioact. **101** (2010) 22–28.

[187] ŠTROK, M., SMODIŠ, B., Natural radionuclides in milk from the vicinity of a former uranium mine, Nucl. Eng. Des. **241** 4 (2011) 1277–1281.

[188] ŠTROK, M., SMODIŠ, B., Comparison of two sequential extraction protocols for fractionation of natural radionuclides in soil samples, Radiochimica Acta **98** (2010) 221–229.

[189] ŠTROK, M., SMODIŠ, B., Levels of ^{210}Po and ^{210}Pb in fish and molluscs in Slovenia and the related dose assessment to the population, Chemosphere, **82** (2011) 970–976.

[190] ČERNE, M., SMODIŠ, B., ŠTROK, M., JAĆIMOVIĆ, R., Accumulation of ^{226}Ra, ^{238}U and ^{230}Th by wetland plants in a vicinity of U-mill tailings at Žirovski vrh (Slovenia), J. Radioanal. Nucl. Chem. **286** (2010) 323–327.

[191] ČERNE, M., SMODIŠ, B., ŠTROK, M., Uptake of radionuclides by a common reed (Pharagmites australis Trin. ex Steud.) grown in the vicinity of the former uranium mine at Žirovski vrh, Nucl. Eng. Des. **241**(4) (2011) 1282–1286.

[192] ČERNE, M., SMODIŠ, B., Estimation of absorbed dose rates to wetland plants from the vicinity of the former Uranium mine at Žirovski vrh, Slovenia using ERICA tool, International Conference Nuclear Energy for New Europe 2010, Portorož, Slovenia, 6–9 September 2010, Proceedings. Ljubljana, Nuclear Society of Slovenia (2010).

[193] PARIDAENS, J., VANMARCKE, H., Inventarisatie en karakterisatie van verhoogde concentraties aan natuurlijke radionucliden van industriële oorsprong in Vlaanderen (2001) (in Dutch).

[194] INTERNATIONAL ATOMIC ENERGY AGENCY, INTERNATIONAL UNION OF RADIOECOLOGISTS, Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments, Technical Reports Series 364, IAEA, Vienna (1994).

[195] VANMARCKE, H., ZEEVAERT, TH., GOVAERTS, P., Dosisevaluatie van het slibbekken Veldhoven, SCK-CEN, Belgium (1993) (in Dutch).

[196] VANMARCKE, H., ZEEVAERT, TH., SWEECK, L., Radiologische evaluatie van de stortplaats Spoorwegstraat, SCK-CEN Report (1996) (in Dutch).

[197] NOUAILHETAS, Y., et al., Radiological question concerning the monazite sand cycle wastes in Brazil, Radiat. Prot. Aust. **11** (1993) 177–181.

[198] LAURIA, D.C., ROCHEDO, E.R.R., The legacy of monazite processing in Brazil, Radiat. Prot. Dosim. **114**(4) (2005) 546–550.

[199] AGUDO, E.G., GONÇALVES, S., FRANCISCO, J.T., SHINOMYIA, C.N., The use of radium isotopic ratio in groundwater as a tool for pollution source, Fourth International Conference Low-Level Measurements of Actinides and Long-Lived Radionuclides in Biological and Environmental Samples (1992).

[200] MAGALHÃES, M.H., ZENARO, R., LAURIA, D.C., The radium concentration in groundwater at a waste disposal site in Brazil. Is it naturally occurring or a contamination process? United States: Elsevier, The Natural Radiation Environment VII, Book Series (2005).

[201] PASCHOLATI, E.M., et al., Survey of environmental gamma radiation around Itu. Publicado, 4th Meeting on Nuclear Applications, XI ENFIR/IV ENAN Joint Nuclear Conferences (1997).

[202] WU, Q., et al., A Report of Survey and Assessment of Radioactive Pollution Resulting from Exploitation of Bayan Obo Ores, Tsinghua University (2009).

[203] WU, Q., et al., The use and management of NORM residues in processing Bayan Obo ores in China, In: Naturally Occurring Radioactive Material (NORM VI), Proceedings of NORM VI symposium, 22–26 March 2010, Marrakech, Morocco, IAEA, Vienna (2011). Pp. 65–78.

[204] INNER MONGOLIA RADIOACTIVE ENVIRONMENT MANAGEMENT INSTITUTE, A Report of Baotou Radioactive Environmental Quality (Year 2006) (2007).

[205] INNER MONGOLIA RADIOACTIVE ENVIRONMENT MANAGEMENT INSTITUTE, The Monitoring Data of Baotou Radioactive Environmental Quality (Year 2006) (2007).

[206] MICHALIK, B., CHAŁUPNIK, S., WYSOCKA, M., SKUBACZ, K., Behaviour of radium discharged with waste waters into the surface settling pond, 7th International Mine Water Association Congress, Ustron, Poland (2000) 443–452.

[207] JAGIELAK, J., BIERNACKA, M., HENSCHKE, J., SOSINSKA, A., Radiation Atlas of Poland, Biblioteka Ochrony Środowiska, Warsaw (1992) (in Polish).

[208] JAGIELAK, J., BIERNACKA, M., HENSCHKE, J., SOSINSKA, A., Radiation Atlas of Poland, Biblioteka Ochrony Środowiska, Warsaw (1998) (in Polish).

[209] MICHALIK, B., SKUBACZ, K., CHAŁUPNIK, S., LEBECKA, J., Radioactive deposits in Polish coal mines and its influence on the natural environment. International Symposium on the Natural Radiation Environment, NRE-VI, Montreal, Canada (1995) p. 136.

[210] CHAŁUPNIK, S., MICHALIK, B., WYSOCKA, M., SKUBACZ, K., MIELNIKOW, A., Contamination of settling ponds and rivers as a result of discharge of radium-bearing waters from Polish coal mines, J. Environ. Radioact. **54** (2001) 85–98.

[211] SKUBACZ, K., et al., Assessment of the radiological impact of the discharge of radium-bearing waters into Golawiecki river, Wiadomosci Gornicze 5/93, Katowice (1993) (in Polish).

[212] WARDASZKO, T., GRZYBOWSKA, D., Radium in Polish rivers, Proceedings of TENR I conference, Szczyrk, Central Mining Institute, Katowice, Poland, p. 124 (1996).

[213] WYSOCKA, M., CHAŁUPNIK, S., MIELNIKOW, A., LEBECKA, J., SKUBACZ, K., Behaviour of radium isotopes released with brines and sediments from coal mines in Poland, 13th Radiochemical Conference, 19–24 April 1998, Marianske Lazne - Jachymov, Czech Republik (1998) p. 47.

[214] ROBLES, B., et al., Radiological investigacion in the boiler's maintenance operations in a Coal-Fired Power Plant, Proceedings of a Conference Held in Buenos Aires, Argentina, 19–24 October 2008, IRPA 12 (2008).

[215] MORA, J.C., et al., Modelling the behaviour of ^{210}Po in high temperature processes. J. Environ. Radioact. **102** 5 (2011).

[216] SIMMONDS, J.R., LAWSON, G., MAYALL, A., Methodology for assessing the radiological consequences of routine releases of radionuclides to the environment. Radiation Protection 72, European Commission Report EUR 15760, Luxembourg (1995).

[217] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Assessing Dose of the Representative Person for the Purpose of the Radiation Protection of the Public, ICRP Publication 101a (2006).

[218] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Radionuclide Transformations: Energy and Intensity Emissions, ICRP Publication 38 (1983). (superseded by ICRP Publication 107, Nuclear Decay Data for Dosimetric Calculations (2008))

[219] EUROPEAN COMMISSION, MARINA II – Update of the MARINA project on the radiological exposure of the European Community from radioactivity in North European marine waters, Radiation Protection 132, EC, Luxembourg (2003).

[220] ECKERMAN, K.F., WOLBARST, A.B., RICHARDSON, A.C.B., Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion, Federal Guidance Report No. 11, EPA-520/1-88-020, US Environmental Protection Agency, Office of Radiation Programs, Washington, DC (1988).

[221] UNITED STATES GOVERNMENT, National Emission Standards for Hazardous Air Pollutants, 40 CFR 61.93(a), Emission Monitoring and Test Procedures, Washington, DC (2002).

[222] UNITED STATES ENVIRONMENTAL PROTECTION AGENCY, Risk Assessment, Radionuclide Table Slope Factors Download Area, 16 April 2001, US EPA, Washington, DC (2001).

[223] VAN GENUCHTEN, M.TH., Convective-Dispersive Transport of Solutes Involved in Sequential First-Order Decay Reactions, Computers and Geosciences **11**(2) (1985) 129–147.

[224] BATEMAN, H., The solution of a system of differential equations occurring in the theory of radioactive transformations, Proceedings of the Cambridge Philosophical Society **15**(V) (1910) 423–427.

[225] HUYAKORN, P.S., BUCKLEY, J.E., Finite Element Code for Simulating One-Dimensional Flow and Solute Transport in the Vadose Zone, Technical Report prepared for US Environmental Protection Agency, Athens, GA (1987).

[226] ARGONNE NATIONAL LABORATORY, The Uranium Dispersion and Dosimetry (UDAD) Code, Version IX, NUREG/CR-0553, ANL/ES-72, prepared for the US Nuclear Regulatory Commission, Washington, DC (1979).

[227] MOMENI, M.H., YUAN, Y., ZIELEN, A.J., UDAD. Uranium Dispersion & Dosimetry Model, Argonne National Laboratory, Report No. ESTSC/NRC-000045I037000; NESC-824 (1992).

ANNEX I.
EXAMPLES OF CURRENT REGULATIONS RELEVANT TO THE MANAGEMENT AND REMEDIATION OF SITES AFFECTED BY PAST ACTIVITIES

Some examples of current regulations relevant to the management and remediation of sites affected by past activities (e.g. historic NORM sites, historic nuclear sites, legacy sites) are presented in the following Sections I–1 to I–5 of this Annex.

I-1. REGULATIONS IN THE UNITED STATES

By way of illustration of regulatory development, concerning the so-called 'Superfund' sites, the United States Environmental Protection Agency (US EPA) provides substantial regulations and guidance, as follows:

- The basic law was set out in the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA)[93];
- Regulations issued under CERCLA are set out in several documents within the National Oil and Hazardous Substances Pollution Contingency Plan (NCP), covering a range of topics[94];
- An overview regarding remediation and Superfund sites[95];
- The policies are the same for radioactively and chemically contaminated sites;
- More specific guidance and tools for determining the remediation strategy for radioactively contaminated sites[96], used in conjunction with the overall policies on the above websites.

It is notable that the same risk informed approach has been adopted by the US EPA for radioactive and other contaminants. It may be noted that in other regulatory contexts, a dose-based system is applied in relation to control of radiation exposure.

A proposed plan for the remediation of the radioactive waste management complex at the United States Department of Energy (US DOE) Idaho National Laboratory (INL) site is described, along with the roles and responsibilities of the regulatory body and other relevant authorities in Ref. [I–1]. Within this document, three government agencies are identified as responsible for remediation decisions at the site: US DOE, as the lead agency and the party responsible for conducting the selected remedial option(s), is required to issue a proposed remediation plan to fulfil public participation requirements under CERCLA and the NCP. The State of Idaho Department of Environmental Quality (DEQ) and the US EPA provide regulatory oversight. Together, the three organizations are referred to as the Agencies in the context of cleanup at the INL site. Protection standards in relation to radioactive and other contamination have also been set out [I–1].

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[93] An overview of CERCLA is available at: https://www.epa.gov/superfund/superfund-cercla-overview
[94] An overview is available at: https://www.epa.gov/emergency-response/national-oil-and-hazardous-substances-pollution-contingency-plan-ncp-overview#Hazardous Substance Removals
[95] See https://www.epa.gov/superfund/cleaning-superfund-sites
[96] See https://www.epa.gov/superfund/superfund-remedial-design-remedial-action

I-2. REGULATIONS IN BELGIUM

The regulatory framework for historic sites in Belgium can be summarized as follows: As a European Union member, Belgium has transposed the Council Directive 96/29/EURATOM laying down the Basic Safety Standards for the protection of the health of workers and the general public against the dangers arising from ionizing radiation into its own regulations [I−2]. This transposition into Belgian regulations was affected by the Royal Decree of 20 July 2001, which sets forth the general regulation for the protection of the population, workers and the environment against the dangers of ionizing radiations.

According to the above, a historic site is considered as an intervention[97] [I−3] situation in cases of prolonged exposure and is defined as:

— “Any action intended to reduce or avert exposure or the likelihood of exposure due to sources that are not part of a controlled practice or that are out of control as a consequence of an accident” [I−4].

Article 53 of the EURATOM directive [I−2] addresses the question of intervention in cases of lasting exposure. It has been transposed into article 72*bis* of the Belgian Royal Decree, which states:

> “Where the Agency [in Belgium, the Federal Agency for Nuclear Control] has identified a situation leading to lasting exposure resulting from the after-effects of a radiological emergency or a past or old practice or work activity, it shall, if necessary and to the extent of the exposure risk involved, ensure:
>
> (a) that the area concerned is demarcated;
> (b) that arrangements for the monitoring of exposure are made;
> (c) the coordination of the implementation of any appropriate intervention, in agreement with the concerned authorities, including regulation of the access or use of the grounds and buildings located in the demarcated area and including the use of the contaminated or activated materials.”

The intervention levels are expressed as incremental annual effective dose with respect to the natural background:

— Dose < 0.3 mSv: no intervention (unless the intervention work is trivial: application of the ‘as low as reasonably achievable’ (ALARA) principle);
— 0.3 < dose < 1 mSv: intervention rarely justified;
— 1 < dose < 3 mSv: intervention generally justified;
— Dose > 3 mSv: intervention always justified.

It is important to note that there is a separate set of regulations for sites contaminated with non-radioactive hazardous substances. As most historic nuclear sites are also contaminated with

[97] The regulations are still based on the concept of ‘intervention’ and ‘practice’, as defined by the International Commission on Radiological Protection (INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, 1990 Recommendations of the International Commission on Radiological Protection, ICRP Publication 60 (1991)), even though these concepts have been replaced by ‘planned exposure situation’, ‘emergency exposure situation’ and ‘emergency exposure situation’ in ICRP 103 [I−5] (see also GSR Part 3 [I−6]). The concept of ‘intervention’, as applied in this publication, is equivalent to application of the reference level to determine whether or not remediation is justified.

non-radioactive contaminants, these much more detailed and prescriptive regulations also apply. In Belgium, radiation protection and nuclear safety is a competency of the federal state, whereas other environmental issues are a competency of the regions. Each region has its own regulations with respect to sites contaminated with non-radioactive hazardous substances. Further regulation is under development that will better define the liabilities and the administrative procedure for interventions in cases of lasting exposure.

I-3. REGULATIONS IN BRAZIL

In Brazil, the basic standards include an intervention level of 10 mSv in a year, but once remediation starts, an environmental monitoring programme has to be carried out by the responsible party (the owner of the site) and the results have to comply with Brazilian requirements, i.e. a public limit on annual effective dose of 1 mSv and an annual dose constraint of 0.3 mSv. Similarly, a worker monitoring programme is required to be established, approved by the Brazilian Nuclear Commission (CNEN), performed by the responsible party for the site, and verified by CNEN. The dose limit for workers is 100 mSv within 5 years, implying no more than 20 mSv in a year on average.

I-4. REGULATIONS IN BULGARIA

Requirements on the management and supervision of historic mining sites in Bulgaria are set out in 'Preliminary Information on the Implementation of the Obligations Under Article 35 of the Euratom Treaty prepared for the purposes of the verification mission of the EC in Bulgaria' [I−7]. This includes a description of the regulatory body and other relevant authorities, and provides details of the monitoring programmes for the relevant sites.

I-5. REGULATIONS IN CZECH REPUBLIC

The reduction in the uranium production programme in the Czech Republic is a consequence of lower uranium demand and a revision of environmental criteria to limit the adverse environmental effects from the production process. Dismantling and remediation work is based on individual government decrees, and for separate localities, it is specified through technical dismantling and remediation projects. Legislation related to mining and remedial actions has been developed for a long period of time and has gone through many changes.

The following is a list of legislation related to mining and remediation activities in the uranium industry in the Czech Republic:

- Act No. 20/1966 on Medical Care and Public Health;
- Act No. 254/2001 on Waters (Water Act);
- Act No. 44/1988 on Protection and Use of Minerals and Raw Materials (Mining Act);
- Act No. 86/2001 on Protection of Air Environment against Polluting Agents (Act on Air Environment);
- Act No. 388/1991 on State Environmental Fund of the Czech Republic;
- Act No. 17/1992 on the Environment;
- Act No. 114/1992 on Protection of the Nature and Landscape;
- Act No. 100/2001 on the Environmental Impact Assessment (EIA Act);
- Act No. 18/1997 on Peaceful Use of Nuclear Energy and Ionizing Radiation (Atomic Act);

— Act No. 185/2001 on Wastes;
— Act No. 183/20006 Coll. (Construction Act).

This list illustrates the complex and cross-cutting nature of the regulation of uranium facilities and activities. Provisions of the Acts, listed above, are implemented by related Decrees and implemented in line with recommendations and guidelines providing additional detail. These are implemented by a similarly complex range of State regulatory bodies and administrative agencies involved in mining and remedial actions related to the uranium industry.

Approval for remediation is based on a governmental resolution in which the Government sets requests and tasks. The State enterprise, DIAMO, is the only organization that is involved in the operation and remediation of uranium mining and milling sites and it falls within the Ministry of Industry and Trade (MTO). The MTO authorizes all projects and their financing.

Financial resources are allocated each year from the State, according to the specific documentation 'Actualization of the Uranium Industry Contraction Program', which is prepared annually by DIAMO and submitted to the MTO for its authorization. After approval in the State budget, the Ministry of Finance monitors its use.

Preparation of any remediation plan includes the following steps:

(1) Analysis and preparation of a report on the current status at a given site is carried out by DIAMO, and subsequently, submitted (via the MTO) to the Government to make an appropriate decision regarding approval;
(2) The Government issues the Decree, specifying tasks related to remediation of the site;
(3) DIAMO prepares a detailed schedule to carry out the tasks specified in the Government Decree, and specifications are then issued by the MTO;
(4) Risk analysis for the site is carried out by an independent company;
(5) A technical plan of decommissioning is prepared by DIAMO and submitted for review, and then to the MTO for authorization. The plan also includes a social programme;
(6) Technical designs and plans and other documentation are prepared;
(7) An environmental impact assessment is carried out for every facility or technology change;
(8) The State Office for Nuclear Safety (SÚJB) issues a licence for decommissioning of a facility and for site remediation;
(9) Supervision is performed by many authorities, such as local offices for the environment (within the mandate of the Ministry of Environment), Czech Bureau of Mines and its regional offices, and others during all phases of implementation of remediation.

REFERENCES TO ANNEX

[I–1] UNITED STATES DEPARTMENT OF ENERGY, UNITED STATES ENVIRONMENTAL PROTECTION AGENCY, STATE OF IDAHO DEPARTMENT OF ENVIRONMENTAL QUALITY, Proposed Plan for Radioactive Waste Management Complex Operable Unit 7-13/14, October (2007).

[I–2] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, 1990 Recommendations of the International Commission on Radiological Protection, ICRP Publication 60 (1991).

[I–3] EUROPEAN COMMISSION, Official Journal of the European Communities, L 159, Vol. 30 (1996).

[I–4] INTERNATIONAL ATOMIC ENERGY AGENCY, IAEA Safety Glossary: Terminology Used in Nuclear Safety and Radiation Protection (2018 Edition), IAEA, Vienna (2019).

[I–5] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, The 2007 Recommendations of the International Commission on Radiological Protection, ICRP Publication 103 (2007).

[I–6] EUROPEAN COMMISSION, FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS, INTERNATIONAL ATOMIC ENERGY AGENCY, INTERNATIONAL LABOUR ORGANIZATION, OECD NUCLEAR ENERGY AGENCY, PAN AMERICAN HEALTH ORGANIZATION, UNITED NATIONS ENVIRONMENT PROGRAMME, WORLD HEALTH ORGANIZATION, Radiation Protection and Safety of Radiation Sources: International Basic Safety Standards, IAEA Safety Standards Series No. GSR Part 3, IAEA, Vienna (2014).

[I–7] EUROPEAN COMMISSION, Preliminary Information on the Implementation of the Obligations Under Article 35 of the Euratom Treaty prepared for the purposes of the verification mission of the EC in Bulgaria, 10–14 August 2009, Bulgaria (2009).

BIBLIOGRAPHY

The bibliography provides a list of reference publications that contain additional information on specific technical, organizational, financial and safety issues relating to remediation.

CONSEJO DE SEGURIDAD NUCLEAR, Instrucción de Seguridad 13, Instrucción de 21 de marzo de 2007, del Consejo de Seguridad Nuclear, número 13, sobre criterios radiológicos para la liberación de emplazamientos de instalaciones nucleares, BOE nº 109 de 7 de mayo de 2007. (in Spanish).

Forskrift om forurensningslovens anvendelse på radioaktiv forurensning og radioaktivt avfall, FOR-2010-11-01-1394, Lovdata, Oslo (2010) (in Norwegian). Available at: http://www.lovdata.no/cgi-wift/ldles?doc=/sf/sf/sf-20101101-1394.html Last accessed 04.07.12.

Hydrogeochemistry of the U-Mineralized area of Los Gigantes, Córdoba – Universidad de Cagliari (Italia), Comisión Nacional de Energía Atómica (Argentina).

INSTITUT DE RADIOPROTECTION ET DE SÛRETÉ NUCLÉAIRE, Guide méthodologique – Gestion des sites potentiellement pollués par des substances radioactives, IRSN, Paris (2011).

INTERNATIONAL ATOMIC ENERGY AGENCY, Effects of Ionizing Radiation on Plants and Animals at Levels Implied by Current Radiation Protection Standards, Technical Reports Series No. 332, IAEA, Vienna (1992).

INTERNATIONAL ATOMIC ENERGY AGENCY, Measurement and Calculation of Radon Release from Uranium Mill Tailings, Technical Reports Series No. 333, IAEA, Vienna (1992).

INTERNATIONAL ATOMIC ENERGY AGENCY, Validation of models using Chernobyl fallout data from the Central Bohemia region of the Czech Republic – Scenario CB, First Report of the VAMP Multiple Pathways Assessment Working Group, IAEA-TECDOC-795, IAEA, Vienna (1995).

INTERNATIONAL ATOMIC ENERGY AGENCY, Modelling of radionuclide interception and loss processes in vegetation and of transfer in semi-natural ecosystems, Second Report of the VAMP Terrestrial Working Group, IAEA-TECDOC-857, IAEA, Vienna (1996).

INTERNATIONAL ATOMIC ENERGY AGENCY, Validation of models using Chernobyl fallout data from southern Finland – Scenario S, Second Report of the VAMP Multiple Pathways Assessment Working Group, IAEA-TECDOC-904, IAEA, Vienna (1996).

INTERNATIONAL ATOMIC ENERGY AGENCY, Testing of environmental transfer models using data from the atmospheric release of Iodine-131 from the Hanford site, USA, in 1963, Report of the Dose Reconstruction Working Group of the Biosphere Modelling and Assessment (BIOMASS) Programme, Theme 2, IAEA-BIOMASS-2, IAEA, Vienna (2003).

INTERNATIONAL ATOMIC ENERGY AGENCY, Testing of environmental transfer models using Chernobyl fallout data from the Iput River catchment area, Bryansk Region, Russian Federation, Report of the Dose Reconstruction Working Group of the Biosphere Modelling and Assessment Methods (BIOMASS) Programme, Theme 2, IAEA, Vienna, IAEA-BIOMASS-4, IAEA, Vienna (2003).

INTERNATIONAL ATOMIC ENERGY AGENCY, "Reference Biospheres" for solid radioactive waste disposal, Report of BIOMASS Theme 1 of the Biosphere Modelling and Assessment Programme, IAEA-BIOMASS-6, IAEA, Vienna (2003).

INTERNATIONAL ATOMIC ENERGY AGENCY, User's manual for NORMALYSA v.2.3 Description of Program Module Libraries, Mathematical Models and Parameters, Report of Working Group 3 (Task 2), Application of models for assessing radiological impacts arising from NORM and radioactively contaminated legacy sites to support the management of remediation of MODARIA Topical Heading, Remediation of Contaminated Areas, MOdelling and Data for Radiological Impact Assessments (MODARIA) Programme, IAEA, Vienna, (in preparation).

Investigación Hidrogeológica en el área Los Gigantes-Convenio INA-CNEA-Junio (2001).

OATWAY, W.B., MOBBS, S.F., Methodology for estimating the doses to members of the public from the future use of land previously contaminated with radioactivity. National Radiological Protection Board Report NRPB-W36, NRPB, Chilton (2003).

SAFEGROUNDS LEARNING NETWORK, SAFEGROUNDS Approach to managing contaminated land on nuclear-licensed and defence sites, an Introduction, June 2009. (http://www.safegrounds.com/guidance_approach.htm).

SAFEGROUNDS LEARNING NETWORK, SAFEGROUNDS Guide to the comparison of contaminated land management, June 2009. (http://www.safegrounds.com/guidance_options.htm).

SAFEGROUNDS LEARNING NETWORK, SAFEGROUNDS Good practice guidance for site characterization, Version 2, June 2009. (http://www.safegrounds.com/guidance_characterization.htm).

SOUTH AFRICAN NATIONAL NUCLEAR REGULATOR, A Guide Document for the Release of NORM Contaminated Sites from Regulatory Control, GD-1047 Rev 0 (2005).

SWEDISH RADIATION PROTECTION INSTITUTE, Multiple Model Testing using Chernobyl Fallout Data of I-131 in Forage and Milk and Cs-137 in Forage, Milk, Beef and Grain, H. Köhler, S.-R. Peterson, and F.O. Hoffman, Eds., BIOMOVS Technical Report 13, Vol. I–II, Scenario A4, SRPI, Stockholm (1991).

SWEDISH RADIATION PROTECTION INSTITUTE, Long term contaminant migration and impacts from uranium mill tailings, BIOMOVS II Technical Report No. 4, SRPI, Stockholm (1995).

SWEDISH RADIATION PROTECTION INSTITUTE, Long term contaminant migration and impacts from uranium mill tailings, BIOMOVS II Technical Report No. 5, SRPI, Stockholm (1996).

SWEDISH RADIATION PROTECTION INSTITUTE, Wash-off of Sr-90 and Cs-137 from Two Experimental Plots: Model Testing Using Chernobyl Data, BIOMOVS II Technical Report No. 9, SRPI, Stockholm (1996).

SWEDISH RADIATION PROTECTION INSTITUTE, Assessment of the Consequences of the Radioactive Contamination of Aquatic Media and Biota: Model Testing Using Chernobyl Data, BIOMOVS II Technical Report No. 10, SRPI, Stockholm (1996).

SWEDISH RADIATION PROTECTION INSTITUTE, Atmospheric Resuspension of Radionuclides: Model Testing Using Chernobyl Data, BIOMOVS II Technical Report No. 11, SRPI, Stockholm (1996).

UNITED KINGDOM ENVIRONMENT AGENCY, Guidance on the characterization and remediation of radioactively contaminated land. Environment Agency, UK (2002).

UNITED NATIONS SCIENTIFIC COMMITTEE ON THE EFFECTS OF ATOMIC RADIATION, Effects and risks of ionizing radiation, Report to the General Assembly, with Annexes, UNSCEAR, United Nations, New York (1994).

UNITED STATES DEPARTMENT OF DEFENCE, UNITED STATES DEPARTMENT OF ENERGY, UNITED STATES ENVIRONMENTAL PROTECTION AGENCY, UNITED STATES NUCLEAR REGULATORY COMMISSION, Multi-Agency Radiation Survey and Assessment of Materials and Equipment Manual (MARSAME), NUREG-1575, Supp. 1, EPA 402-R-09-001, DOE/HS-0004 (2009).

ZEEVAERT TH., SWEECK L., WALRAVENS J., VANMARCKE H., Radiological impact study of the dumping site D1, SCK-CEN report R-3098 (1996) (in Dutch).

ZHUNUSSOVA, T., SNEVE, M., ROMANENKO, O., SOLOMATINA, A., MIRSAIDOV, I., Threat Assessment Report, Regulatory Aspects of the Remediation and Rehabilitation of Nuclear Legacy in Kazakhstan, Kyrgyzstan and Tajikistan, Strålevern Rapport 2011:5, Norwegian Radiation Protection Authority, Østerås (2011).

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