Deliverable 2.12

Report from Investigation of multiple hazards at NORM exposure sites in order to develop integrated assessment procedure for NORM and other pollutants (REE, heavy metals)

Work Package 2

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This project has received funding from the Euratom research and training programme 2019-2020 under grant agreement No 900009.

Document information

Project Acronym	**RadoNorm**
Project Title	**Towards effective radiation protection based on improved scientific evidence and social considerations - focus on radon and NORM**
Project Type	**RIA**
EC grant agreement No.	**900009**
Project starting / end date	**1 September 2020 – 31 August 2025**
Work Package No.	**2**
Work Package Title	**Exposure**
Deliverable No.	**2.12**
Deliverable Title	**Report from Investigation of multiple hazards at NORM exposure sites in order to develop integrated assessment procedure for NORM and other pollutants (REE, heavy metals)**
Lead Beneficiary	**DSA**
Contractual Delivery Date	**30.04.2025**
Actual Delivery Date	**30.06.2025**
Type	**Report**
Dissemination level	**PU**
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To be cited as:

Mrdakovic Popic, J., Haanes, H., Jakab, D., Beltran Torres, S., Skipperud, L. (2025): Report from Investigation of multiple hazards at NORM exposure sites in order to develop integrated assessment procedure for NORM and other pollutants (REE, heavy metals). Final version as of 30.06.2025 of deliverable D2.12 of project RadoNorm.

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Acknowledgement

This document is a deliverable of the RadoNorm project. This project has received funding from the Euratom research and training programme 2019-2020 under grant agreement No 900009.

Status of deliverable		
	By	Date
Delivered (Lead Beneficiary)	Jelena Mrdakovic Popic (DSA)	22.06.2025
Verified (WP Leader)	Laureline Février (ASNR)	24.06.2025
Reviewed (Reviewers)	WPL, PC	26.06.2025
Approved (PC)	PC	30.06.2025
Submitted to EC (PC)	PC	30.06.2025

Executive Summary

Naturally occurring radionuclides (NOR) are present in the Earth's crust, atmosphere, and biosphere. Often, NOR is found together with non-radioactive elements such as metals, trace elements and rare earth elements (REE). To be able to correctly understand multiple exposure situations and associated impacts and risks, it is relevant to gain knowledge about patterns of co-occurrence, speciation and mobility of both radionuclides and co-occurring hazardous elements by thoroughly characterising potential exposure sites.

In the RadoNorm project, Work Package 2, Task 2.5 *'Overview of NORM sites and exposure scenarios in Europe and their characteristics'* an investigation of the terrestrial multiple exposure site Fen Complex in Norway, which contains elevated levels of NOR, REE and metals, was conducted with following steps:

- Characterisation of legacy site with multiple NORM and non-radioactive hazards pollution,
- Analysis of correlation between identified hazards (NOR, REE and metals),
- Analysis of correlation and impact of environmental parameters (such as pH, organic matter, water content) on the behaviour of investigated multiple hazards,
- Analysis of exiting environmental impact assessment (EIA) and ecological risk assessment (ERA) procedures as well as uncertainties in these,
- Consideration of knowledge gaps and challenges, in general, for the development of an integrated assessment procedure for identified multiple exposure.

The conducted investigation confirmed that existence of NOR in multiple exposure situation was rather a rule than an exception, while assessment approaches have historically been developed on the basis of separate international and national regulations and practical approaches. An important finding was that high levels of natural radioactivity (Th) most often also involve high levels of REE. Moreover, the obtained results highlight this multiple exposure site with respect to co-variation and behaviour of the elemental hazards, which commonly depend on both their inter-relationships and how they are affected by environmental conditions. An overview of currently used impact assessment procedures, including uncertainty overview in different steps, and possible steps for further work on integrated impact assessment were included.

The soil concentration levels (from moderately to significantly elevated in comparison to average Norwegian and worldwide levels) and observed distribution patterns for all investigated hazardous elements largely reflected the mineral composition of the main rock in the background. However, anthropogenic mining activities have also impacted most localities through exposure of underlying bedrock in waste rock deposits and debris from mining to geological processes such as weathering, erosion, deposition and alteration. Correlation with environmental factors like pH, organic matter content (OM), water content (WC), cation exchange capacity (CEC), altitude and slope of the ground at sampling localities, showed a high-level of complexity, with inter-dependences between different elements and variables at wide ranges of elemental concentrations. This suggested that adaptation of theoretically and laboratory developed procedures for characterisation of multiple exposures and an Integrated Environmental Impact Assessment for real-life conditions would need to be developed with consideration of all sources of uncertainties. Given the high levels of NOR and REE at Fen complex, the site should thus be further investigated in such a way in the future.

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1. Introduction

In radiation protection and radioecology, the focus has been to assess exposure and effects from radionuclides, while mostly disregarding other simultaneously present hazardous elements. When naturally occurring radionuclides (NOR) are found in a material that is anthropogenically treated in some way, which may include exploitation, energy sectors or processing, it is termed naturally occurring radioactive material (NORM) (IAEA, 2022). Within the RadoNorm project, Work Package 2 (Exposure), Task 2.5 (Overview of NORM sites and exposure scenarios in Europe and their characteristics), one of the objectives has been to provide improved characterisation of multiple exposure sites containing NORM, but also other non-radioactive hazardous elements, such as rare earth elements (REE), iron, other metals and trace elements. Obtained results from this task and worldwide publications have shown that the occurrence of other potential elemental hazards is rather the rule than the exception, and the potential additional, synergistic or even antagonistic effects should be accounted for when assessing exposure and effects (ICRP, 2019; Salbu, 2015; Sneve et al., 2020).

To be able to correctly understand multiple exposure situations, it is relevant to improve knowledge about patterns of co-occurrence, speciation and mobility of both radionuclides and co-occurring hazardous elements and thoroughly characterise sites. More detailed, analysis of correlation between hazards and with environmental parameters should contribute to a better understanding of such relationships, how they can influence each other's mobility, bioavailability and general behaviour. Finally, impacts should also be assessed, preferably in a holistic manner. Currently, this is done in consistent assessment procedures, while an integrated assessment procedure needs to be developed to face the remaining challenges in research and regulation.

The main aim of the work presented here is to explore possible relationships between NOR and non-radioactive hazards, as well as their behaviour dependence on different relevant environmental parameters (Figure 1). It is foreseen that improved characterisation, as well as addressing still existing knowledge gaps and uncertainties, will help in developing an integrated risk and impact assessment for terrestrial ecosystems.

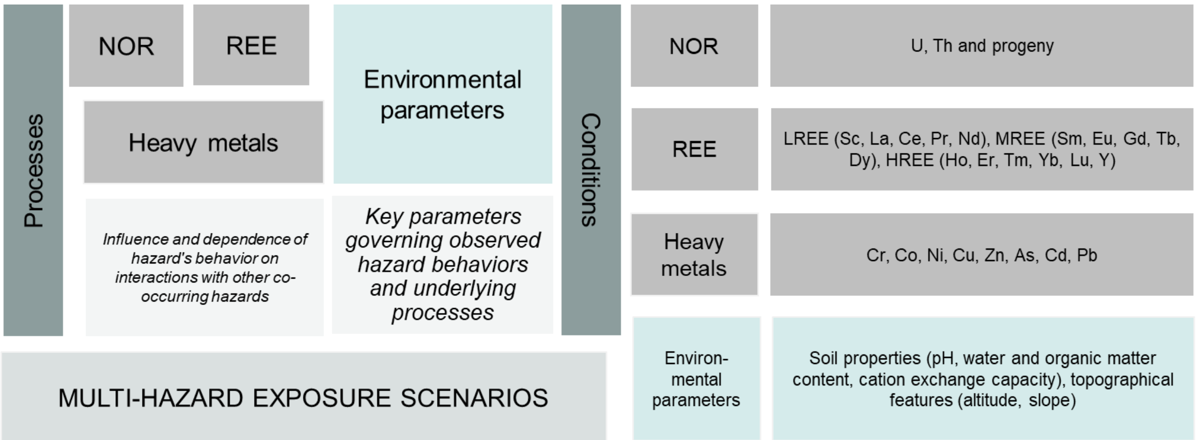

Figure 1 Schematic representation of the done multiple exposure characterisation.

The structure of the present report is as follows:

- Introduction to the present work, with background information and main objectives of the work are given in Section 1,
- Details on multiple exposure situations containing NORM, common NORM sites characteristics, and existing challenges related to assessment, management and control of different present

hazards are provided, together with overview of existing previous similar and/or relevant work, in Section 2,

- Section 3 contains an overview of exposure characterisation of legacy site in Norway with multiple exposure, and analysis of correlations between hazardous elements and their behaviour due to environmental parameters,
- Section 4 provides an overview of EIA, ERA and an analysis of uncertainties in model-based environmental risk assessments, and proposals for future work on development of integrated impact assessment,
- Main conclusions of the present are given in Section 5,
- References are given in Section 6.

2. Multiple exposure situations containing NORM and other non-radioactive hazards

Naturally occurring radioactive material (NORM) is a radioactive material containing no significant amounts of radionuclides other than NOR (i.e., uranium (^{238}U), thorium (^{232}Th) and their decay products, and potassium (^{40}K)). (IAEA, 2022). In the environment, NOR is commonly found in a wide range of activity concentrations, mainly in low, but also elevated concentrations can be found in certain minerals (e.g., monazite, apatite, uraninite, autunite, rutile) or may arise as the result of diverse human activities during their exploitation (IAEA, 2022; UNSCEAR, 2008). It is of importance to have good knowledge of these radionuclides in different exposure situations and scenarios since some U- and Th-progeny are long-lived and have a high radiotoxicity that may significantly affect workers, the public or the environment, under different conditions (ICRP, 2019, Michalik, 2009).

2.1 Overview of common multiple exposure situations and main identified issues

During the exploitation of raw materials with different content of natural radionuclides in certain industries, technological processes may include elevated temperature, pressure or chemical processes with mass reduction that may result in NORM. Involved materials (beside some raw materials with slightly elevated NOR activity concentrations) can be in the form of solid material – main and by-products, residues and waste – as well as liquid effluents and atmospheric discharges. In some of these materials, NOR can be low (e.g., in purified oil, titanium-dioxide pigment, tiles and refractories) or different grade elevated (e.g., in scale and sludge or other residues with elevated activity concentrations of ^{226}Ra, ^{210}Pb and ^{210}Po in oil and gas industry, in geothermal energy production, non-metals ferrous smelting, coal combustion products). As NORM can potentially affect workers due to accumulation, during the above-mentioned processes in facilities, but also during transportation, maintenance and cleaning, disposal activities and decommissioning processes, radiation protection (RP) control is required (Mrdakovic Popic et al., 2023). The proper requirements for RP of workers, public and environment are given in the Directive 2013/59/Euratom, EU BSS (EC, 2014) and IAEA BSS (IAEA, 2014), where NORM involving industries are considered as **planned exposure situations.**

During the RadoNorm project activities, the identification (mapping) of NORM involving industries across Europe was completed and a detailed overview is provided in Mrdakovic Popic et al., 2023*;* Mrdakovic Popic et al., 2025a, under review; Dvorzhak et al, 2025, under review.

Exposure scenarios including NORM, related to planned exposure situations, were provided by the European Commission in 2002, in the guidance RP 122 Part II (EC, 2002), and included following:

- Workers: transport over long/short distances, indoor storage of moderate quantities of material, outdoor storage of large quantities of material, road construction with NORM material, work

using building materials containing NORM, building construction work using undiluted NORM as unshielded surface cover, disposal on a heap or a landfill, road construction;
- Members of public: NORM as surface layer in public places/sport grounds, dweller of a house built with building materials containing NORM, dweller of a house built with undiluted NORM as unshielded surface cover, dweller of a house near a heap or landfill.

The additional exposure scenarios used in the European countries, obtained within the RadoNorm activities, are provided in Mrdakovic Popic et al., 2023 and include more specific descriptions:

- Workers: exposure of workers to NOR that may happen during prolonged and close contact with raw materials while doing manual or semi-manual transportation, unpacking, and other work that include proximity and contact with large quantities of raw materials, in closed or open spaces, exposure due to inhalation of dust or gamma radiation exposure, exposure of workers in oil and gas industry when handling waste materials and surface-contaminated objects, potential exposure during the cleaning of contaminated equipment, potential exposure during the decommissioning of oil and gas platforms;
- Public: exposure of members of public living in the vicinity of NORM involving industrial facilities which discharge the liquid effluent containing NORM to smaller water bodies (e.g. streams or small lakes), exposure of members of the public due to living in the vicinity of disposal sites for NORM waste, exposure of members of public during recreational time spent in the vicinity of old NORM-affected legacy sites.

In the given exposure scenarios, workers and members of the public may be exposed to ionising radiation from NOR through different exposure pathways, such as exposure due to radon and/or radon progeny, external gamma radiation, exposure due to (inadvertent) ingestion of soil, water or contaminated foodstuff and exposure due to inhalation of radon, thoron and dust containing natural radionuclides. Additionally, it is known that hazards beyond radioactive ones are commonly present in the considered exposure scenarios (e.g., REE, heavy metals, organics), with similar exposure routes as given above. Non-radioactive hazards may even be the major health or environmental protection concern, but these are still considered separately – both in assessment, monitoring and regulatory control (separate legislations, regulatory authorities).

Furthermore, legacy sites containing NORM exist all over the world; they often present regional or national challenges, further complicated by commonly having diverse characteristics (Sneve et al., 2020). It should be emphasized that no definition of a '*legacy site'* has been internationally recognised, nor for a NORM affected legacy site, and since the term has already been considered differently in national jurisdictions, it may be interpreted differently in various social contexts. This was confirmed in the exercise of RadoNorm Task 2.5 by participant countries as well in RadoNorm project Deliverable 2.11 (Mrdakovic Popic et al., 2025b). One of the definitions for *Legacy sites*, proposed and used by the Nuclear Energy Agency (NEA) Expert Group on Legacy Management is '*sites that have not completed remediation, but which have radioactive and possibly other type of contamination that is of concern to the regulator*' (NEA, 2019). Still, a common characteristic of legacies is that such sites were contaminated in the past when no legal requirements or standards similar or comparable to current practices, standards or RP requirements existed. NORM affected legacy sites are often old (e.g., several decades or more), abandoned, with ownership and responsibilities often questionable or difficult to assign, and they usually also involve contaminated land and waters, sometimes comprise a variety of facilities with different amounts of residues and/or waste containing mixed hazards (Haanes et al., 2019; Mrdakovic Popic et al., 2011; Sneve et al., 2020; Vandenhove et al., 2006). Taking into consideration legacy sites characteristics and specifications for radiation protection at existing exposure situations provided by the EU BSS (EC, 2014) and International Atomic Energy Agency BSS (GSR Part 3, IAEA, 2014), such NORM affected legacy sites fall under the category of **existing exposure situations**.

Exposure situations and different scenarios, commonly considered depending on the state of legacy sites, are:

- Workers: workers involved in clean-up or remediation activities, transport of NORM containing waste;
- Public: people living or doing different types of activities in the vicinity of NORM affected legacy sites.

European countries with NORM affected legacies (taking into consideration different national definitions or absence of definition of legacy sites), identified in 2021 during an exercise within the RadoNorm project work of Task 2.5 are given in Figure 2.

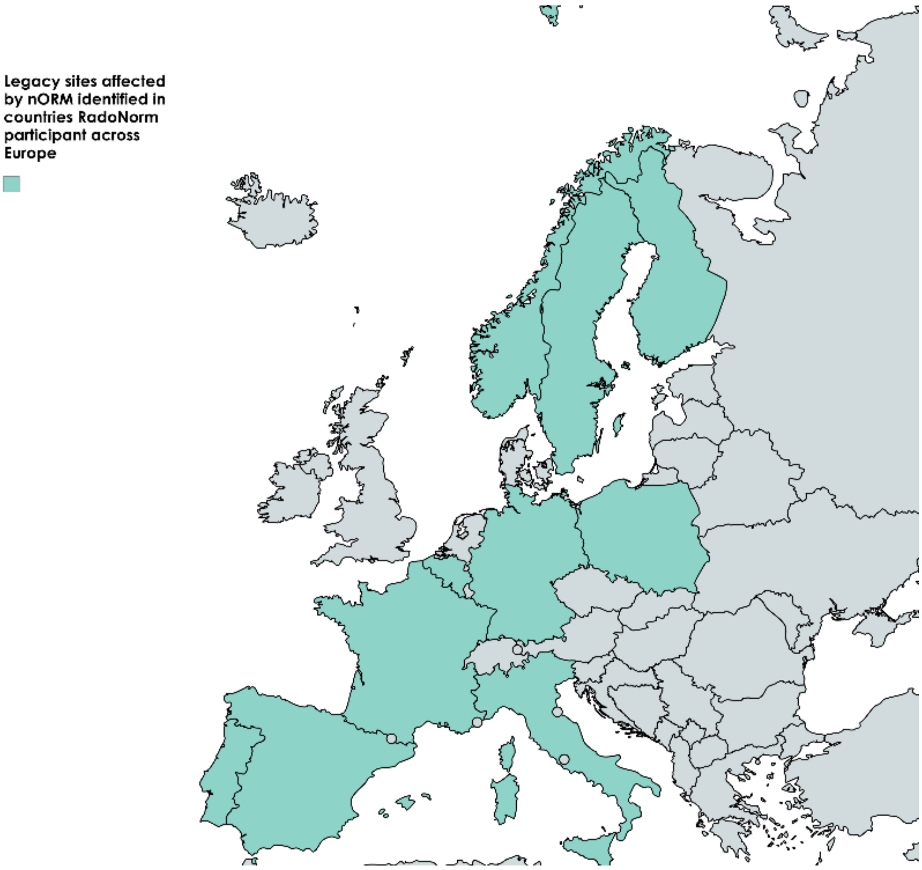

Figure 2 European countries, RadoNorm participants of WP2 Exposure, Task 2.5., with identified NORM affected legacy sites.

The main identified issue of potential concern that is addressed by this report, and which is valid for both types of NORM involving exposure situations, is the presence of hazards other than NOR, which in many cases may involve a higher risk for both workers and public health as well as environment *per se* (ICRP, 2019). Such exposure situations may include a range of metals, organic compounds, asbestos, but also other materials depending on site. Related to this, one of the important questions raised during the RadoNorm work (Task 2.5) was how to holistically consider all the hazards at these multiple exposure situations.

In the NORM industries, within facilities, occupational, health and safety measures (e.g., proper work equipment, dust masks) together with radiation protection measures (e.g., limitation of working hours with NORM, maximal possible distance, using radiation dose dosimeters) usually ensure overall protection of workers. Still, there are questions about improving joint, integrated monitoring programs and impact/risk assessment for the public and the environment.

When it comes to legacy sites, the situation is more complex. Many times, hazards are unknown, which highlights the requirement for a proper site characterisation. The question is how to harmonise globally and how to conduct characterisation of sites with multiple hazards in a practical, holistic way. Furthermore, it is of interest to industry operators, waste managers and regulators that public and environmental impact/dose/risk assessments are conducted, preferably for all identified hazards in one common procedure (knowledge gaps remains, although some models are in the developing or validation phase, e.g., Vives i Batlle, 2025).

2.2 Work towards development of integrated assessment of multiple hazards

Historically, legacy sites have not been concerned with RP issues, as the presence of NORM was not seen as a major issue compared with the levels of other potential hazards (e.g., toxic chemical elements like certain metals or organic compounds). Impact and risk assessments, in general, contain the same steps (see Section 4), which, however, to date are still conducted separately for NORM and for other hazardous elements.

Based on previously identified knowledge gaps (Gilbin et al., 2021; Mrdakovic Popic, 2015; Salbu, 2015; Sneve et al., 2020), it was concluded that NORM (but also other artificial radionuclides) and non-radioactive hazards should be assessed through a common risk assessment such that consistent assumptions are employed and consistent criteria used in the evaluation of risk.

At present, when reviewing the cited literature above, some differences in considerations emerge:

- Separate approaches have historically been developed on basis of separate international and national regulations and are used in different institutions for control of RP and environmental control, respectively.
- In waste or discharge considerations, NORM is present in large volumes in most cases, while this can differ for mixed hazards in several situations.
- Protection objectives and, hence, assessment endpoints and timeframes, are usually different when NORM and other non-radioactive hazards are compared.
- Sometimes, the lack of a common understanding and terminology used to address issues can result in confusion, discrepancies and errors, which can essentially lead to mistrust. An example of this is the term of ‘toxicity index’, which is used for chemicals, but not for radioactive contaminants.
- Differing mechanisms of producing effects (harms) to the health of humans or biota – as the exposure pathways of NORM containing long-lived radiotoxic elements include both external and internal exposure that can damage living tissue, potentially leading to cells damage, mutations, and in long terms cancers, depending on the delivered dose. On the other side, non-radioactive hazards (although they can be quite different) cause harm through chemical toxicity (e.g., affecting cellular functions, damaging tissues, and may accumulate in the body), which can lead to organs disfunction, effects on immune system – depending on the nature of hazard.
- Different benchmark concentrations are used. However, standards have been based on single stressor assessments with numbers of extrapolations being applied, e.g., extrapolating from one organism to a range of organisms that may be present in an environment, from acute to chronic effects, from laboratory to field conditions, or from isolated single species test systems to complex ecosystems.

3. Exposure characterisation

3.1 Fen Complex Norway: Legacy site affected by NOR, iron, REE and metals

3.1.1. Study site

Work presented in this report is conducted on samples and measurements done in geologically well-known Fen Complex, Norway. The special geology of Fen Complex is due to the magmatic origin of its carbonatitic bedrock and the posterior hydrothermal alteration. These geochemical processes resulted in the most abundant present bedrock types in the area (Figure 3): ferro-dolomite-carbonatite - FDC (rauhaugite), red rock (rødbergite) and søvite (calcite carbonatite ($CaCO_3$) named after Søve, a locality in the area). FDC is naturally enriched in iron (Fe), ^{232}Th and REE, while red rock is more enriched in Fe and ^{232}Th (Dahlgren, 2019); however, the boundaries between these bedrocks are not strictly defined. In søvite, the calcite matrix is enriched in niobium (Nb) minerals and somewhat ^{238}U, while ^{232}Th is present in lower concentrations compared with those in the red rock and FDC.

Most of the Fen complex is covered by Holocene superficial deposits with a varying thickness along the complex from 0 m to over 50 m (Dahlgren, 2019). Part of the complex was exploited for iron during 1652 – 1926, leaving behind abandoned mines, open pits and scattered mining waste deposits. Also, a mining facility in Fen Bay at the shore of lake Norsjø, still has mining waste deposits.

In the northern part of Søve, ferro-niobium was produced from søvite, having one facility, which operated from 1953 to 1965, where the mineral was crushed, and separated from water by centrifugation, while the concentrated heavy minerals were transported to another facility for further metallurgical process to obtain the final product (Dowdall et al., 2012). After decommissioning in 1965, the washery and the mining waste deposit were covered with sand layers as a limited remediation measure, but these eroded with time leading to radioactive waste resurfacing at public areas (Dahlgren, 2005; Mrdakovic Popic et al., 2011), thus increasing exposures and risk of contamination spread.

The sampling area in Rullekoll is a small, wooded area surrounded by houses situated 1 km from the mining hill. The main bedrock in this locality is the red rock with damtjernite (containing RRE- bearing xenoliths, heavy minerals like phlogopite, magnetite, apatite) (Dahlgren, 2019).

Torsnes is a locality in the north of Fen Complex at the shore of lake Norsjø. The main bedrock in the sampling area in Torsnes is søvite with the presence of mafic dyke in the north-east.

Nowadays, the wooded areas in the Fen Complex are freely accessible areas, where visitors have recreative walks; there is a mechanic shop operating in the former mining facility in Søve; and there are agricultural lands and human settlements within Fen Complex. Most mine openings are fenced in. Also, in this area, REE exploitation is planned, and the exploration activities are already ongoing. Because of the natural enrichment of REE in the red rock and FDC, Fen Complex is considered one of the most important REE deposit in Europe (Dahlgren, 2019). Additionally, thorium exploitation is planned in close future.

This exposure site is therefore of importance with respect to radiation protection of existing legacies, but also regarding ongoing exploration and future exploitation of REE and possibly Th.

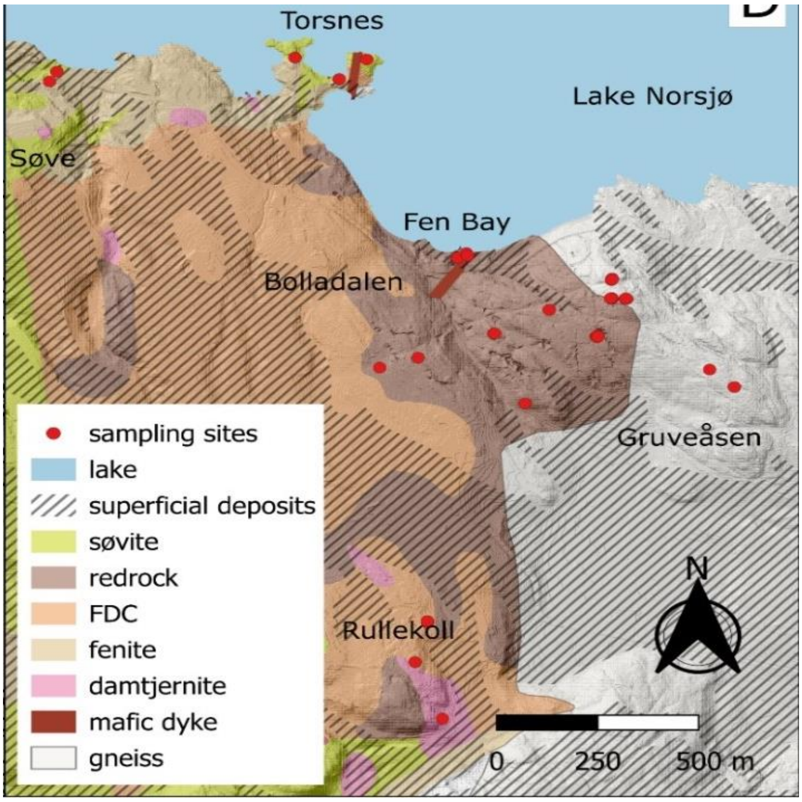

Figure 3 Fen Complex: Satellite photo, investigated locations with bedrock types within the complex.

3.1.2. Data compilation and methodology

The current dataset encloses the data acquired during past investigations in the area, executed by Mrdakovic Popic et al. (2011), Haanes and Gjelsvik (2021) and Valle (2013). In total, 109 records of superficial soil samples (up to 10 cm depth) were compiled for five localities (Figure 3): Fen (F) and Søve (S), former mining facilities, Bolladalen - Gruveåsen former mining area (BG), Torsnes (T) and Rullekoll (R) woods nearby houses. At these main localities, sampling was conducted at numerous localities in the time periods of 2008-2020. The complied information encompasses sample information, soil properties (pH, organic matter content, water content, cation exchange capacity) and concentration of NOR (^{238}U, ^{232}Th, ^{226}Ra, ^{228}Ra), REE and selected metals (Cr, Co, Cu, Ni, Zn, As, Cd, Pb). REEs, comprising of lanthanides considered together with scandium (Sc) and yttrium (Y) were classified according to their atomic number and grouped as light (LREE: lanthanum – La, cerium – Ce, praseodymium – Pr, neodymium – Nd, Sc), medium (MREE: samarium – Sm, europium – Eu, gadolinium – Gd, terbium – Tb, dysprosium – Dy) and heavy REEs (HREE: holmium – Ho, erbium – Er, thulium – Tm, ytterbium – Yb, lutetium – Lu, Y).

The elemental concentration was obtained by ICP-MS, the activity concentrations of ^{228}Ra and ^{226}Ra were obtained by gamma spectrometry using high purity Germanium (HPGe) detectors.

To complement the information regarding the distribution of the hazards in different fractions, results from conducted soil sequential extraction (Tessier et al., 1979; Mrdakovic Popic et al., 2014; Mrdakovic Popic et al., 2012) were also included in the analysis.

3.1.3. Statistical analysis

Distributions of variables significantly deviated from normal, with varying levels and direction of skewness and numerous extreme values. Consequently, neither a uniform nor individual transformation could account for the skewness or be used for an approach to achieve normal distributions. The correlation between concentrations of NOR, REE, metals and soil properties was therefore calculated using Spearman's rank, which is applicable to non-normally distributed data and robust to extreme values (outliers). A hierarchical cluster analysis was applied to group variables into cluster according to their similarity, based on the similarity matrix where the lower the distance the higher the similarity between a variable pair. The chosen clustering method is the Ward's method which minimizes the variance within each cluster. A Principal Component Analysis (PCA) was applied to reveal key relationships of co-variation among the variables.

3.2. Results of soil analyses

The first step in site characterisation was to assess the concentrations of NOR, REE and metals in soil at different localities and across the Fen Complex. To be able to better compare hazardous elements behaviours, co-occurrences and possible effects of environmental parameters, total soil and soil fractions concentrations and soil parameters were analysed together. While details of the current study are in pipeline for peer-review publication, an overview of the main results is presented in the current report.

Soil range values for elemental concentration of all REEs, in groups according to their abundance (see colour codes, different scales on individual graphs) are presented in Figure 4. Soil range values for elemental concentrations of NOR and metals, together with ranges for median values of REE, are given at Figure 5 for the whole Fen Complex (colour coded). In general, wide ranges of all investigated elements were observed. The upper range values for ^{232}Th and LREE were particularly high, exceeding with several orders of magnitude the values reported for crustal abundance (Wedepohl, 1995), average Norwegian soils (Nygård et al., 2011), and soils in other European countries (Salminen et al., 2005). The concentrations of ^{238}U, MREE, Zn, and Cr ranged up to two orders of magnitude higher than crustal abundance (Wedepohl, 1995). Metal concentrations, except Cd, were in the upper national range, with some metals exceeding the national average in Norway (Nygård et al., 2011).

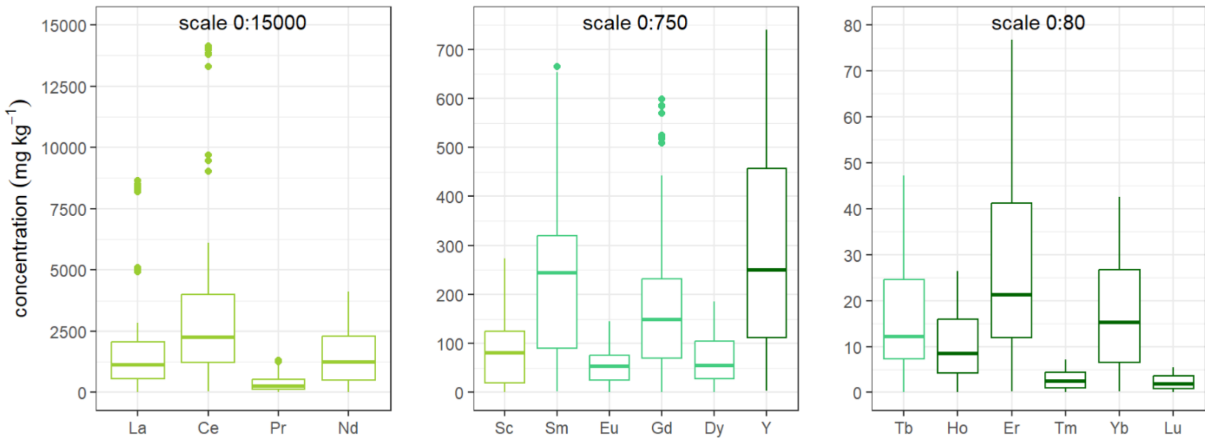

Figure 4 Boxplot showing the elemental concentration of REEs in Fen Complex Complex; REE groups differentiated by colour. LREE (light green): La, Ce, Pr, Nd, Sc; MREE (green): Sm, Eu, Gd, Tb, Dy and HREE (dark green): Ho, Er, Tm, Yb, L.

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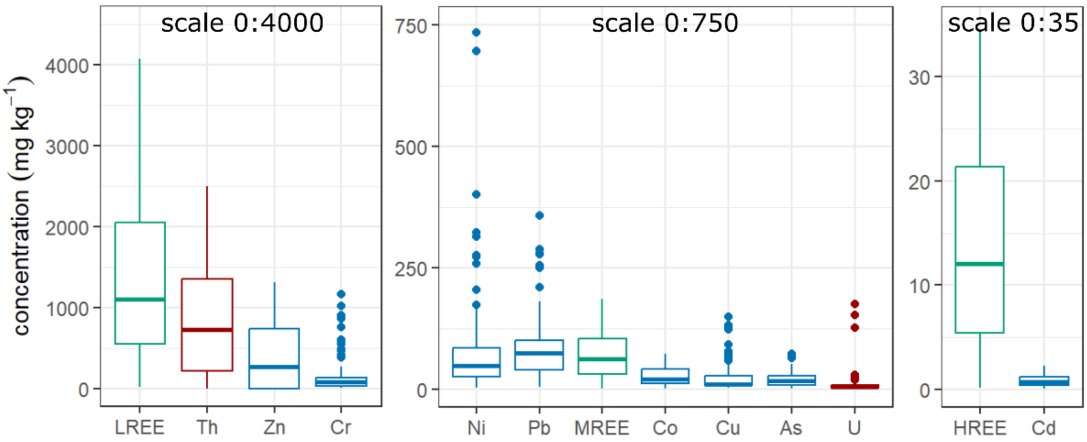

Figure 5 Boxplot showing the elemental concentration of U, Th (red), metals (blue) and the median concentration of LREE, MREE and HREE (green) in Fen Complex.

The high spatial variability of most element hazards was observed at all localities, which align with previous results (Mrdakovic Popic et al., 2011; Mrdakovic Popic et al., 2014, Valle, 2013). Most likely, both underlying bedrock and anthropogenic activities have affected investigated localities. Due to mining, waste rocks consisting of bedrock naturally enriched with NOR, REE and iron, have in large deposits been exposed to geological processes (weathering, erosion, deposition, alteration) and environmental features at several of the sampling localities, like BG, F and S. Soil concentrations of NOR and REE were high at localities BG, F and R, and correlated with the most abundant rocks of these localities (red rock and FDC), which are known to be enriched with Th and REE. Similarly, consistently lower NOR (especially ^{232}Th) and REE were measured at localities S and T, which correspond with bedrock, søvite at these localities. Moreover, significantly higher activity concentrations of ^{238}U were measured at locality S in comparison to other localities, suggesting also an effect from previous mining activities and enriched waste slag from Nb extraction mixed with soil in some spots at this locality.

Based on soil sequential extraction results (Figure 6), it was observed that the mobility of ^{232}Th, REE, Cr and As was low, i.e. 70-94% of these elements were rather inert, irreversibly bound to the soil residual fraction. The consistently similar fractionations, observed across investigated localities - together with the residual fraction dominance - suggested that these hazards exhibited an overall stronger immobility as well as weaker responsiveness to changes in environmental conditions. Results for ^{232}Th were in accordance with its chemical nature, as ^{232}Th is considered to be one of the least mobile elements in continental weathering, having high sorption tendency (Choppin et al., 2002; Termizi Ramli et al., 2005).

On contrary, ^{238}U, Co, Cu and Cd were present at considerable levels, 45-62%, in pH- and redox-sensitive soil fractions and fractions associated with soil organic compounds, implying higher mobilities, transport under some environmental conditions and possible uptake in biota. Similarly, other studies have previously found U to be more mobile than Th (Haanes and Gjelsvik, 2021). Under oxidizing conditions, ^{238}U is present as uranyl ion (UO_2^{2+}) which can form water-soluble complexes depending on pH and other present ligands (Langmuir, 1978). It has been explained that complex sorption and dissolution mechanisms determine the environmental behaviour of ^{238}U (Langmuir, 1978; Echevarria et al., 2001; Koch-Steindl and Pröhl, 2001).

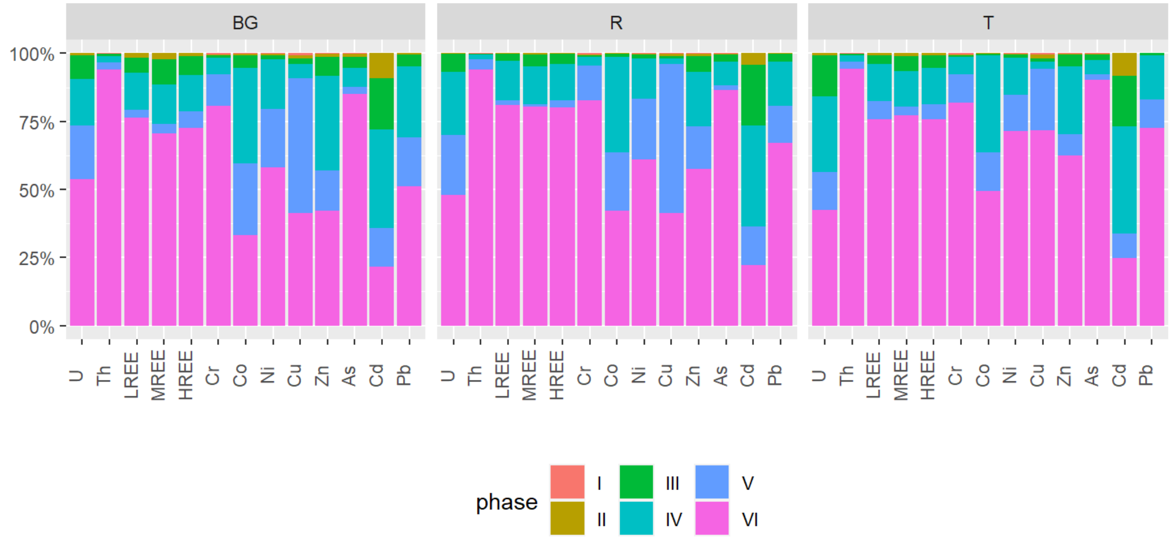

Figure 6 Results of sequential extraction of soil from Bollaladen–Gruveåsen (BG), Rullekoll (R) and Torsnes (T).

Beside total soil and soil fractions analyses of hazardous elements, to understand the behaviour of investigated elements and how these depend on environmental conditions, slope and altitude at each Fen locality, soil pH, OM, water content and CEC were measured and results evaluated across all localities (numerical values and statistical analyses are in detail provided in paper in prep.). These parameters were chosen because the slope and altitude of localities influence weathering and erosion, soil pH controls speciation and solubility of hazards and other soil components, OM enhances complexation of hazards and influence their mobility / retention, water content influences the leaching of hazards, while CEC affects soil ability to bind or exchange cations.

As seen for the hazardous elements investigated, a high degree of variability in environmental factors was observed, particularly at the localities with intensive mining in the past. This variability may be due both to different levels of mining activity but also to the large spatial scale of sampling points. As an illustrative example, values of pH varied most at locality F, ranging between 3.5 and 7.5, with extreme values on the lower end corresponding to strongly acidic conditions, which may be expected since the washery and shipping point of the iron legacy mines were situated at the F locality. Moreover, CEC was particularly high in locality T, which could be associated with the high clay content of this locality, just as the lower CEC in locality S could be attributed to its low clay fraction.

3.3. Co-relationship between investigated hazards and environmental parameters

Considering the enrichment of hazardous elements in Fen Complex, it was important to analyze their co-variation and whether their behaviours vary among localities or with environmental conditions. Since bedrock and important soil parameters like pH differ between the localities, there may be multiple geochemical processes ongoing, and interactions are plausible between numerous elements. We therefore started by looking at separate correlations among elements groups and with environmental factors.

A strong correlation was noticed between REEs and ^{232}Th ($r>0.8$, $p<0.05$, Figure 7). This is one of the main findings of the study, demonstrating that where REE is high, also natural radioactivity is high, although solubility can be reduced with OM and increasing pH, thus affecting mobility. The correlation was expected since Th and REE have the same mineralogic origin as hosted on phosphate minerals (such as monazite, bastnäsite, synchysite, apatite), which are known to be weathering resistant and retained in the soil. These shared a similar behaviour, being the most abundant elements in Fen Complex, but with overall low mobility in soil.

A positive correlation between U and Th/REE concentrations was also observed, but intermediate (r: 0.5-0.6, $p<0.05$, Figure 7) due to less consistency at low U levels, where associated Th/REE concentrations considerably scatter. This may imply different mineralogical origins but could also be affected by environmental conditions like pH, which significantly varied over the area. In strongly acidic conditions, U mainly existed as weekly-retained and highly soluble cation, UO_2^{2+} (U(VI)), which sorption was limited due to intensified cation competition for a reduced number of exchange sites.

The soluble, more mobile fraction of REEs, ^{232}Th, ^{238}U and metals Cd, Cu and Cd were found to be adsorbed onto Fe/Mn oxides, organic matter and clay minerals, depending on the cation exchange capacity of the soil, as shown by the positive correlation with CEC, and depending on soil pH, as shown by the negative correlation (Figure 7).

There was a moderate but statistically significant negative correlation ($r<-0.50$, $p<0.05$, Figure 7) between soil pH and ^{232}Th, ^{228}Ra, REEs, Cr, Co, Ni, and As, related to the speciation and complexation of these elements. The correlation of Th with pH was expected, locality BG has the lowest pH (Figure 8) and, by far, the highest Th levels due to the bedrock being redrock. For REE, primarily bound with the organic part of the Fe-OM colloids, pH controls mobility and co-transport of Th.

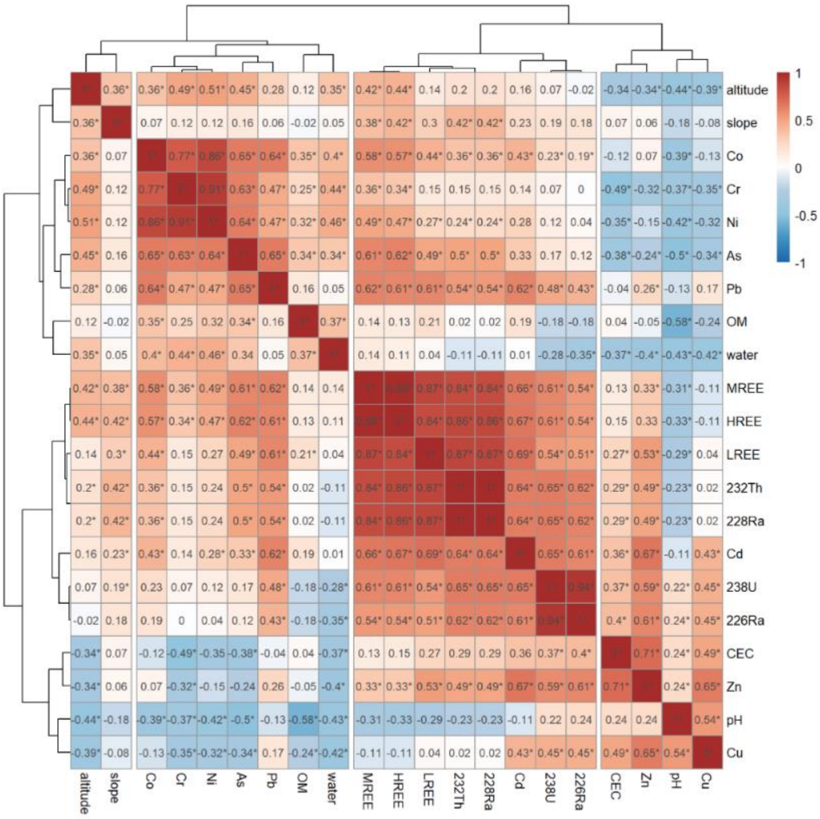

Figure 7 Correlogram, Heatmap and Cluster Dendrogram combined displaying the Spearman correlation coefficient, colour coded ranging from perfect negative correlation (–1) in dark blue to perfect positive correlation (+1) in dark red with the colour intensity representing the strength of the correlation. The statistically significant correlations – within a 95% of confidence interval (i.e. with p-values below 0.05) – are marked with a black star (*).

Additionally, other statistically significant correlations were observed between several elements, both positive and negative, as well as between elements and environmental conditions (Figure 7). This showed the complexity of element ranges, co-variation and interactions at Fen complex (*publication in prep*.).

To explore more in detail the correlations between hazardous elements and between hazardous elements and environmental factors, PCA was performed for all investigated localities of Fen Complex (Figure 8). This helped in assessing the high degree of heterogeneity at different scales, as demonstrated in environmental parameters and hazardous element concentrations. Differentiation was clear and depended on locality and thus bedrock that differed between localities, which was further related to the underlying geology, the specific anthropogenic activities and inherent characteristics (altitude, slope, influence of the mining activities) at each sampling locality. It is important to notice the high influence of the bedrock type that was found at these localities.

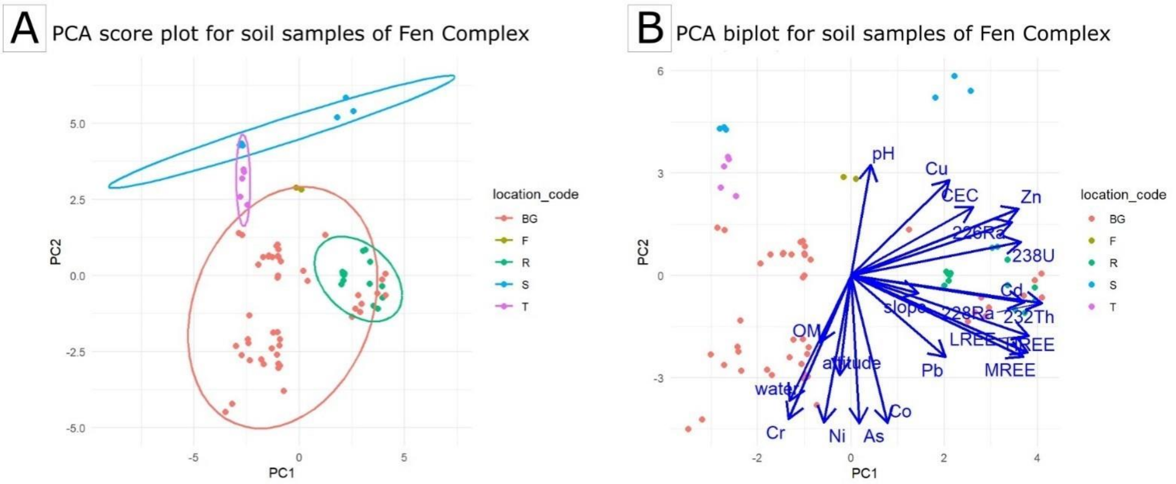

Figure 8 PCA, based on the Spearman correlation matrix for radionuclides, REEs and metals in Fen Complex.

The clusters in the heatmap (Figure 7) reflected correlated hazardous elements and environmental parameters, patterns that could be seen for all sampling localities in Fen Complex also in the PCA (Figure 8). In summary, the PCA showed that correlations were related to bedrock and mineralogy, with a dependency on pH values, with the first two principal components together explaining 61% of the variance. The first principal component (PC1, x-axis), explaining 34% of variance, showed mainly elements of the same mineralogic origin, including the most abundant elements in Fen Complex, whereas the second principal component (PC2, y-axis), explaining 28% of the variance, reflected geochemical behavior influenced by the environmental parameters like soil pH, water content and altitude, and it was grouped to a large extent according to locality and hence bedrock. There was a variation within each locality (Figure 8), as for localities BG and F, which shared the same bedrock, different ranges of pH led to different correlations between hazardous elements and environmental parameters. Locality R, where the impact of mining activities was low, was the only locality where LREE, ^{232}Th, ^{228}Ra were not associated with HREE; however, there was a strong correlation between ^{232}Th, ^{228}Ra and OM that was not found at localities BG and F, which were more heavily affected by mining activities.

4. Moving towards developing integrated risk assessments within Environmental Impact Assessment

Environmental Impact Assessment (EIA)

EIA is a structured process to identify, predict, evaluate and possibly mitigate any negative human health effects and ecological impacts of a proposed or planned project or action. The main purpose is to provide information for decision-making regarding future environmental (and social) consequences, in order to prevent or minimize adverse effects/impacts (Sadler and McCabe, 2002).

According to Fonseca (2022), there are numerous types of EIA processes, which vary depending on size, scale and sector, as well as on the decision-making context, and thus, may differ in assessment scope, information sources, procedures, etc. Despite this, the typical EIA consists of the following steps:

1. *Screening*: determines whether a proposal should be subjected to EIA, and if so, the appropriate level of assessment;
2. *Scoping*: defines key issues and impacts for the EIA, develops and evaluates alternatives for the key aspects of the proposed project, and prepares terms of reference (ToR) to specify impacts, alternatives, scales, methods to consider, which stakeholders to consult etc.;
3. *Impact analysis*: identifies and predicts likely health impacts and environmental effects of the proposal and evaluate their significance, usually consisting of the sub-stages: baseline studies (to establish a baseline), impact identification and prediction, significance evaluation;
4. *Mitigation and impact management*: develops strategies to avoid, reduce or compensate for adverse impacts, identifies mitigation measures with expected reducing impact;
5. *Reporting*: reports the findings of the EIA in a structured and accessible manner for decision-makers and other interested parties;
6. *Review and quality check*: examines the adequacy of the EIA report and assess whether it complies with legislation and expected assessment scope (ToR) and whether the information provided is complete, accurate and sufficient for decision-making.
7. *Decision-making*: approves or rejects the evaluated proposal and sets the terms and conditions for which it may proceed;
8. *Follow-up*: checks on the implementation's compliance with the approval's terms and conditions and monitor the impacts of the proposed project and whether mitigation measures are properly implemented and effective.

Public involvement, which is to inform the diverse stakeholder groups (general public, relevant industry and waste managers, research communities, authorities) about the proposal and gain their inputs through consultations and hearings, may occur throughout the entire process (Sadler and McCabe, 2002; Fonseca, 2022).

The EIA process is not necessarily linear but iterative: depending on the outcomes of each stage, the process may be redirected to an earlier step or even result in the project withdrawal if unacceptable impacts are defined.

In the context of NORM sites, EIA plays a core role in supporting decision-making concerning potential future mining and extraction activities. EIA is also an important tool for planning and managing remediation activities at contaminated legacy sites.

Environmental Risk Assessment (ERA)

ERA is an integral part of EIA and can be applied at multiple stages, offering an analytical tool to characterize human health effects, ecological impacts and associated risks by providing quantitative or qualitative estimates of their likelihood and consequences.

There is a general agreement that ERA is best addressed in four steps:

1. *Problem Formulation* defines the assessment and develops plan for analysing and characterizing risk. It sets the goals, purpose, scope, structure, approaches, activities, available resources and participants involved in the risk assessment. Its key output is the conceptual model, which identifies actually or potentially exposed human and non-human individuals, populations or ecosystems and receptors, known or suspected stressors, their observed or hypothesized effects, and assessment endpoints.
2. *Exposure Analysis* estimates concentrations, doses or extent of contact with stressors and corresponding exposures in terms of intensity, spatial scale (geographical dimension), temporal scale (duration, frequency, timing) and pathways.
3. *Effect Analysis* involves two stages, 1) determination of adverse effects a stressor may cause (hazard identification), and 2) quantification of the relationship between exposure level and response (exposure-response analysis), in terms of effect incidence and severity.
4. *Risk Characterization* synthesizes exposure and effect analysis results to estimate the risk to each endpoint, assess uncertainties, presents findings to support risk management, decision-making and communication (Suter II et al., 2003; U.S. EPA, 2003).

The logical chain of these steps applies to both human health and ecological risk assessments in spite of their methodologies being developed and often used independently. Their integration has obvious advantages, including recognition of interdependencies, ensuring consistent and comparable outputs, improved scientific quality and robustness through exchange and combination of data and models, increased efficiency through reducing redundancy and using common data sources, models, assumptions and resources (Suter II et al., 2003).

Integration applies throughout the whole ERA process. In Problem Formulation, it aligns assessment goals, endpoints and conceptual models and consider shared or parallel concerns for both human and ecological receptors due to common sources, stressors, and exposure pathways. Common models for contaminant release, transport, fate and contact with stressors, shared parameter values, units and spatial-temporal scales enable consistent characterization of exposure, while effects can be analysed considering shared modes of actions, biomarkers and indicators, exposure-response relationships and extrapolation techniques in the Exposure and Effect Analysis phases. In Risk Characterization, risk estimates for human and ecological endpoints are combined using consistent and unified methodologies for risk and uncertainty assessment, same lines of evidence and presentation format as well as comparable metrics enabling joint interpretation (Suter II et al., 2003).

Integration in ERA involves not only coordination across human health and ecological domains, but also to the consideration of multiple hazards or stressors, enabling a more comprehensive understanding of cumulative risks. Unlike *aggregate risk assessments* that sum risks form single stressors across multiple pathways, *multiple-stressor cumulative risk assessments* consider combined impacts and interactions. These may apply for hazards acting by the same mode of action and causing the same effect or address diverse effects from different types (e.g. chemical, biological, radiological, other physical) of hazards, though the latter is methodologically and computationally more complex (U.S. EPA, 2003).

Uncertainty in EIA and ERA

Uncertainty is inherent in EIA due to its predictive nature (Larsen, 2022). While many uncertainties in EIAs – those are associated with baseline and input data, models etc. – are quantifiable, some are subjective and therefore not easily quantified. A certain level of subjectivity is inherently linked to the choices in methods, assumptions, interpretation of assessment outcomes, or to the determination of what constitutes a significant impact. These all can introduce variability in the predictions and significant deviations in predicted and realized impacts, as demonstrated by several post-audit studies. Dealing with uncertainty in EIA, however, is not only a matter of prediction accuracy but also how it is presented and communicated (Tenney et al., 2006). Despite legal requirements (e.g., Directive 2014/52/EU), uncertainty is often underreported or inconsistently presented in EIA reports, potentially leading to misguided decisions because of false confidence in predictions, consideration of incomplete evidence, or undermining precautionary principles.

Incorporating risk assessment into EIA enables a more explicit treatment of uncertainty, it supports evaluating the quality and reliability of the outcomes, guiding risk management, prioritizing actions effectively, strengthening transparency and accountability in decision-making and building creditability among stakeholders.

For NORM and legacy sites, uncertainty is particularly pronounced due to the high variability in hazardous element concentrations even on small spatial scales, incomplete and insufficient monitoring data and limited understanding of site-specific conditions, as well as the nature of long-term exposures often associated with latent health or ecological effects. Multiple hazard scenarios typically introduce greater and more complex uncertainties than single-hazard situations (Rullens et al., 2022), due to:

- increased model complexity – incorporating multiple interacting hazards expands the dimensionality of the model; hazards in real ecosystems may vary in timing, duration, magnitude and spatial extent, requiring analysis across broad spatial and temporal gradients;
- more pronounced data gaps on combined effects – while individual hazards are often well-documented, combined hazard datasets are generally sparse or incomplete thus limiting robustness in estimations of interaction effects and increasing reliance on extrapolation;
- complex hazard interactions – combined effects are often non-additive but predominantly either synergistic or antagonistic, and the nature of interactions may also vary across hazard gradients (Vanhoudt et al., 2012).

Preparatory steps towards developing an integrated ERA within EIA

Having outlined the significance and key challenges of EIA and ERA – in particular regard to multiple exposure scenarios at NORM and legacy sites – the following sections present the preparatory steps taken toward developing integrated ERAs and their main findings:

- A methodological review was conducted on methods used to assess combined effects/impacts and risks in multiple exposure scenarios, with emphasis on those applicable to radioactive hazards alongside non-radioactive contaminants (Section 4.1).
- Sources and types of uncertainties in ERA-EIA were reviewed, followed by an overview of available methods for analysing and addressing uncertainty in model-based ERA, applicable to both human and non-human exposure, effects and risks (Section 4.2).

4.1. Review of effect/impact and risk assessment procedures to be used in multiple exposure scenarios that includes radioactive and other non-radioactive hazards

The approaches applicable for assessing combined health effects or ecological impacts of multiple hazards (see an overview in **Fehler! Verweisquelle konnte nicht gefunden werden.**, which is primarily based on Vanhoudt et al. (2012) and Vandenhove et al. (2012)) can be broadly grouped into two categories depending on their analytical orientations: *bottom-up approaches* or *component-based approaches* start with individual components of the mixture, evaluate the effects/impacts one by one at a time and subsequently use mathematical models to predict combined effects/impacts based on the individual ones; *top-down approaches* or *whole mixture approaches* test the entire mixture without identifying individual interactions between the individual hazards.

Accounting for the biological effects/impacts introduces an additional perspective in the analyses: *biology-based approaches* can test the effect/impact of hazards on specific endpoints and organisms or biological agents.

Table 1: Overview of procedure groups applicable to assessing combined effects/impacts of multiple hazards

	Component-based approaches	**Whole mixture approaches**	**Biology-based approaches**
Application and advantages	To predict mixture effects based on (known or assumed) data and information on mixture composition and individual components. Can reveal additive interactions as well as synergisms or antagonisms of observed effects. Able to use existing ecotoxicity data on single substances without needing specific toxicity experimentation on the mixtures. Mostly used for simple mixtures with known composition.	To test the effect of a whole mixture without attempting the complete identification of the composition of the mixture, e.g. when determination of mixture composition would be impossible or inefficient. Suitable when only fragmented knowledge of the mixture components is available or when identifying the primary contributor to the mixture's toxic effect is not necessary. Often used for site-specific and retrospective studies. More adapted to the assessment of acute effects.	To aid understanding of organisms' integrated responses to hazard mixtures. Can estimate the hazardous effects on growth, reproduction and survival over the life cycle of exposed organisms. Provide information of the possible physiological mode of actions. Often used for studies, where it is not ethically feasible to test compounds on humans or protected species, and hence, there is a need for extrapolation from data on surrogate species.
Limitations	Require concentration, dose and toxicity information of each individual component to predict mixture effects. The mixture composition as well as the effect and the mode of action of each single component must be precisely known as corresponding assumptions are often not realistic. Do not give information on the mechanisms that may drive the interactions; development of real mechanistic models would be very-resource intensive. Do not incorporate the fact that mixture effects can differ in time and endpoint considered or that there may be dose-dependent variation in interactions. Mode of action can also be species- and endpoint-specific and many hazards can have several modes or mechanisms of action. Cannot handle sequential or pulsed exposure profiles.	Will not identify the hazards inducing the effect and do not reveal mechanisms and interacting effects (synergism, antagonism etc.). The results obtained are exclusively applicable to the actual investigated mixture and cannot be extrapolated to other mixtures. Do not consider potential changes in mixtures (depending e.g. on location, time-seasons, different organisms). Since the exposure situation is highly dynamic, frequent re-testing of the mixture concerned is needed. It is also difficult to find suitable control (chemically/structurally similar but uncontaminated matrix) against which the mixture could be tested. Largely unsuitable for prospective approaches and have no predictive power as hazard mixture must be available for direct experimentation.	Require understanding effects of single compounds in sufficient detail. There is no such thing as an ideal biotest or a specific biomarker for stress, as biological species differ from each other in their sensitivity towards a hazard, and radiosensitivity particularly varies extensively between species. While for vertebrates metabolic and physiological information are often available or at least can be inferred, the data availability is limited for invertebrates. Typically, have high data demand for parametrization of the models.
Example models	*Mixture toxicity models*: • *Concentration Addition (CA)* – assumes that hazards exert effects on a similar mode of actions • *Independent Action (IA)* – assumes that hazards act on a different mode of action,	*Whole Effluent Toxicity (WET) tests / Direct Toxicity Assessments (DTAs)*: • *Toxicity Identification and Evaluation (TIE)* – sequentially extracts components from the mixture and tests the toxicity of the remainder to	*Toxicokinetic (TK) models* – describe how a toxicant enters, moves through and is eliminated (via uptake, distribution, metabolization and excretion) from an organism over time: • *Data-based toxicokinetic (DBTK) models* – describe the experimental kinetic data

	Component-based approaches	Whole mixture approaches	Biology-based approaches
	contributing to the common effect independently • *Two-step prediction (TSP) model* – combines CA and IA in a stepwise manner to deal with mixtures containing hazards that have similar and different modes of actions	determine the cause of toxicity in the removed fraction • *Effect Directed Analysis (EDA)* – test the toxicity of the total extraction fractions and individual hazards to determine the toxic components of the mixture	• *Physiology-based toxicokinetic (PBTK) models* – describe the mechanistic processes *Toxicodynamic (TD) models* – capture how the internal toxicant concentration causes biological effects in an organism *Integrated TK and TD models* – predict time-dependent effects of toxicants having interaction either at TK or TD level*:* • *Physiology-based toxicokinetic and toxicodynamic (PBTK/TD) models* • *Dynamic Energy Budget (DEB) inc. the effect of toxicants (DEBtox)*

The choice of methods is strongly dependent on the level of available information on the composition of the mixture of multiple hazards. The complexity of analysis can also be affected and increase gradually in response to the evolvement of data availability, assessment scope and required level of accuracy and precision (Vandenhove et al., 2012). At the most basic level, assessments acknowledge the potential relevance of mixture effects without applying detailed analysis. With the progression in assessment depth, simplified assumptions about the mixture can be applied. At more advanced stages, explicit considerations are incorporated, and more detailed modelling techniques are used. The highest level of assessments includes mechanistic explanation of interactions.

Developed methods are available to assess risk of multiple hazards with applicability for radioactive and non-radioactive (chemical) hazardous elements, that can be positioned along this gradual scale of complexity. At a more basic level, Garnier-Laplace et al. (2009) propose a screening-level approach based on the independent analysis of exposure and effect of hazards, with risk quantified using a single indicator, the Potentially Affected Fraction (PAF). This method ranks hazardous elements according to their relative contribution to the total risk, thereby enabling their prioritization for future investigations. Beaumelle et al. (2017) advance to an effect-based mixture risk assessment approach that integrates mixture toxicity models with Species Sensitivity Distributions (SSDs), allowing the derivation of a joint risk indicator multi-substance PAF (msPAF) that reflect the hazards' combined effect. At the highest level of complexity, Vives i Batlle (2025), introduces a mechanistic risk assessment model based on Ordinary Differential Equations (ODEs) that simulate TK and TD processes, including internal concentrations, damage and recovery over time, enabling dynamic predictions of combined effects of multiple hazards. To the best of our knowledge, these models have not yet been applied to NORM and legacy sites in real-world settings, Vives i Batlle (2025) explicitly proposes such scenarios as a relevant and realistic context for future application.

4.2. Overview of uncertainties in environmental risk assessments

4.2.1. Types and sources of uncertainty

As stated by Skinner et al. (2014), different types of uncertainty require different management approaches, making it essential to first identify its specific kinds. Developing a typology helps categorization of the various ways and dimensions among uncertainty, enhances communication among analysist, decision makers and stakeholders and supports identification and prioritization of the most resource-effective and efficient assessment strategies (Walker et al., 2003).

There is no generally accepted typology, and existing ones often suffer from inconsistent terminology, conflicting definitions or narrow domain-specific focus (Skinner et al., 2014). Nevertheless, uncertainty is commonly categorized among three key dimensions:

- *nature* of uncertainty distinguishes whether the uncertainty is due to the imperfection of our knowledge or is due to the inherent variability of the phenomena described,
- *location* of uncertainty refers to where the uncertainty manifest itself,
- *level* of uncertainty0 refers to where the uncertainty manifest itself along the spectrum of different levels of knowledge.

Nature of uncertainty

Within the dimension of nature, generally two types of uncertainty are distinguished:

- *Type I* or *epistemic uncertainty*, i.e. *uncertainty*: refers to lack, incompleteness or imperfection of knowledge concerning the system of interest, whether quantitative or qualitative.
- *Type II* or *aleatory uncertainty*, i.e. *variability*: refers to natural variability arising from the inherent heterogeneity and randomness displayed in human and natural systems. It can occur across space or over time, within or between individuals, or due to environmental and socioeconomic factors (Walker et al., 2003; U.S. EPA, 2011)

A key distinction is that while epistemic uncertainty can be reduced by improving knowledge through further research and empirical efforts, variability cannot be reduced, only better characterized, as it is an inherent property of the phenomena of concern or system being studied (Skinner et al., 2014; Walker et al., 2003; U.S. EPA, 2011).

The importance of distinction between these categories has been widely acknowledged, for instance, the National Research Council explicitly recommended that “uncertainty and variability should be kept conceptually separate in the risk characterization” (National Research Council, 2009). Nevertheless, in practice, distinguishment is not always straightforward. Many parameters and data in radiological assessment exhibit both epistemic uncertainty and intrinsic variability (Oughton et al. 2008). For instance, radionuclide activity concentrations at NORM sites reflect epistemic uncertainty due to sparse spatial and temporal data, data below detection limits, reliance on modelled or literature-derived estimates in the absence of direct measurements, and assumptions such as secular equilibrium within decay chains. At the same time, variability results from spatial heterogeneity, fluctuating temporal dynamics, changing biological uptake and radionuclide-specific behaviours.

Location of uncertainty

The location dimension may involve many types and sub-types of uncertainties, as illustrated for example by Skinner et al. (2014) through their weight-of-evidence-based assessments of peer-reviewed literature. In model-based assessments, typically five generic uncertainty locations are distinguished (as outlined e.g. by Walker et al. (2003)):

- *Context uncertainty* arises from the underlying conditions, assumptions and boundaries set during problem formulation – specifically, which aspects and parts of the real world to be considered or excluded from the model. It is also linked to how problems are framed, what is the intended use of results and required level of process detail, temporal and spatial resolutions (IAEA, 1989). This uncertainty often results from incomplete understanding of real-world systems, biased or overly narrowed contextual assumptions or conflicting perceptions of assessors, decision makers and stakeholders.
- *Model uncertainty* encompasses both the *conceptual model uncertainty* (i.e., model structure uncertainty), and the *computational model uncertainty* (i.e., model technical uncertainty). The conceptual model defines the investigated system's structure and behaviour, its key components (i.e. distinct compartments) and their interactions between them, and relevant processes and mechanisms to be considered. The computational model is the mathematical formulation and software implementation of the conceptual model, using equations, algorithms, numerical solvers and parameter values to generate quantitative outputs.
    - Conceptual model uncertainty arises from incomplete knowledge, reliance on assumptions, and often subjective system representation choices about which processes and interaction to include or exclude.
    - Computational model uncertainty arises from translating conceptual processes into mathematical form, often involving simplifications and assumptions that may not fully capture real-world behaviour, and numerical approximations, introduced when solving complex mathematical equations.
- *Parameter uncertainty* relates to lack of knowledge and the natural variability in model constants and variables.
  Parameters are often classified by function: source-, physical transport-, biological-transfer, or exposure-related (as in IAEA (1989)). But they can also be categorized according to how their values are obtained and the degree of certainty associated with them (as outlined by Walker et al. (2003)):
    a) exact parameters, i.e., universal constants, considered free from uncertainty;
    b) fixed parameters, i.e., parameters well established based on physical laws, treated as effectively exact;
    c) a priori chosen parameters, i.e., parameters set to assumed invariant due to lack of data or unfeasibility of calibration; their uncertainty is estimated from prior experience;
    d) calibrated parameters, i.e., parameters essentially unknown from previous investigations or cannot be applied to the specific assessment context, determined by calibration, e.g. by fitting models to observed data.
- *Input* or *data uncertainty* is associated primarily with the data describing the system or its external drivers that have an influence on the system and its performance; includes
    a) *availability,* reflecting the incompleteness, scarcity or absence of data;
    b) *precision*;
    c) *reliability,* reflecting the trustworthiness of the data (Skinner et al., 2014).
- *Model outcome uncertainty* is the accumulated uncertainty caused by the uncertainties in all locations that are propagated through the model and reflected in the level of confidence in the resulting estimates.

Level of uncertainty

There is a variety of distinct levels of knowledge, graduating from perfect knowledge to the "unknown unknowns". According to Skinner et al. (2014), the level of uncertainty is attributed to two factors, which defines the degree of understanding and confidence in the likelihood of an event occurring and in the severity of outcomes should that event occur.

Recognized levels of uncertainty include:

- *Determinism* – a practically unattainable idealized state of complete certainty in both the likelihoods and severities of outcomes;
- *Statistical uncertainty* – when uncertainty can be expressed in statistical terms and probabilistically, there is a reasonable confidence about the likelihood but there is limited knowledge about the consequences, i.e. the severity of events;
- *Scenario uncertainty* – when there is a range of possible outcomes with known severity, but their likelihood cannot be quantified due to unclear mechanisms;
- *Recognized ignorance* – when there is limited understanding of mechanism and functional relationships, neither likelihood nor severity of outcomes can be reliably defined;
- *Total ignorance* – where one is unaware of what is unknown, essentially a complete absence of awareness regarding both likelihoods and severities of outcomes (Skinner et al., 2014; Walker et al., 2003).

4.2.2. Analysis and consideration of uncertainty in model-based environmental risk assessments

In the following, we briefly introduce two core approaches used in both human health and ecological risk assessments, which methods have conceptual differences in how they consider uncertainty and variability (see the synthesis in **Fehler! Verweisquelle konnte nicht gefunden werden.**, which is mainly based on U.S. EPA (2001) and U.S. EPA (2014)). According to the U.S. EPA Risk Assessment Guidance (U.S. EPA, 2001), *Deterministic Risk Assessment* (DRA) uses single numerical values, i.e., point estimates to represent variables in the risk equation. Consequently, a single estimate of risk is calculated as an output, which can be an upper bound estimate, i.e., the risk expected if the case of reasonable maximum exposure (RME) for conservative (worst case) scenarios still reasonably likely to occur, or an average risk corresponding to the central tendency exposure (CTE) (best estimate scenario). On the contrary, *Probabilistic Risk Assessment* (PRA) treats inputs and parameters of the risk equation as random variables that can be defined mathematically by probability distributions, and through their combination provides a probability distribution of risks as an output, which enables quantitative characterization of variability and/or uncertainty. The central tendency of the risk distribution (e.g., arithmetic or geometric mean, 50^{th} percentile – median) may be characterized as the CTE risk estimate, whereas the high-end of the risk distribution (e.g. 95^{th} percentile) will represent the RME risk estimate (U.S. EPA, 2001).

Table 2: Characterisation of Deterministic Risk Assessment (DRA) and Probabilistic Risk Assessment (PRA)

	Deterministic Risk Assessment		Probabilistic Risk Assessment	
	Advantage	**Disadvantage**	**Advantage**	**Disadvantage**
Purpose and use	Useful to serve as a screening method for identifying situations in which risks are so low that no further action is needed. Suitable for conservative regulatory estimates.	Decisions need to be done solely on the basis of point estimates providing a limited insight into the full range of possible outcomes and complicating the comparison of possible risk outcomes and corresponding management options.	Support comprehensive risk evaluations and decision-making (e.g., when a screening-level DRA indicates that risks are probably higher than a level of concern and more refined assessment is needed, or when the consequences of using point estimates of risk are unacceptably high). Supports decision-making under uncertainty.	May add no or minimal value in certain situations (e.g., when a screening-level DRA indicates that risks are negligible, or when the cost of averting the exposure and risk is smaller than the cost of implementing a PRA).

	Deterministic Risk Assessment		Probabilistic Risk Assessment	
	Advantage	Disadvantage	Advantage	Disadvantage
			Offers site-specific assessment.	
Computational method	Simple, easy to implement even with basic tools (e.g. spreadsheets). Requires less time to complete, less resource intensive. Its methodology is typically well-established.	Computational simplifications may result in uncertainties and errors, as, for example, it may not adequately represent of all aspects of the phenomena under investigation or may not capture interactions among input variables or combined effects.	Uses more sophisticated models and is suitable for complex assessments, including those of aggregate and cumulative exposures and time-dependent individual exposure, dose and effect analyses.	Requires advanced assessment tools (special computer software) that may even need to be custom designed for specific applications. Requires more computational time, resources and expertise from both the assessor and the decision maker. Communication of a more complex PRA process is challenging, which may generate mistrust of the assessment and its outcomes. Approaches and methods are less standardized.
Use of information and data	Works even with limited data by assuming central ("typical") or upper bound (conservative) estimates for variables. The use of predetermined default values provides a procedural consistency that allows for feasible and traceable risk assessments.	The (selected) point estimates used to represent variables may not reflect reality and decrease the representativeness of data. Often considers non-site-specific data. Account for temporal and spatial variation cannot be explicitly taken. The use of conservative, precautionary defaults may lead to overly conservative and unrealistic outcomes.	Can make more complete use of available data when defining inputs to the risk estimation by facilitating the use of distributions. Makes better use of available site-specific data to reflect real life conditions. Incorporates information on data quality, variability and uncertainty into risk models, such additional information allows for better informed decisions. Has the availability to reveal the limitations and strengths of data that could be critical in identifying steps and measures for improved data collection or research.	Demands the availability of more information and extensive data to generate credible and representative probability distributions. Highly dependent on the quality and extent of data.
Provision of results	Provides single risk estimates as an output, which results can be more easily interpreted and communicated. May present qualitative information regarding the robustness of the estimates.	Does not determine where the point risk estimates lie within the range of possible risks; a point estimate of risk often relies on a combination of central tendency and upper bound input values, so it is difficult to anticipate what percentile is represented by the risk estimate, the likelihood of risk occurring or	Provides the entire range of estimated risks as well as the likelihood of values within the range. Can provide estimate of the probability of occurrence associated with a particular risk level of concern and a more quantitative expression of the confidence in risk estimates.	Because of their complexity, PRA results can be more easily misinterpreted and misused in the decision-making. Presenting and communicating the more complex PRA results and the impact of those results on the decisions is challenging.

	Deterministic Risk Assessment		Probabilistic Risk Assessment	
	Advantage	**Disadvantage**	**Advantage**	**Disadvantage**
		exceeding a particular level is unknown. Lacks expression of confidence. Has difficulties in quantification of the impact of data and model limitations on the quality of the results.		
Variability and uncertainty represen-tation	Uncertainty is usually addressed and compensated for through conservativism, which can ensure high-level of protection and support precautionary in the decision-making.	Provides an estimation of the exposures and resulting risks that addresses variability and uncertainty only in a qualitative manner, does not separate or quantify variability and uncertainty. The degree of conservativism cannot be estimated or communicated, resulting in unquantified uncertainty in risk statements. Identification and prioritizing key contributors to variability and uncertainty can be difficult and time-consuming.	Can provide a quantitative description of the degree of variability and uncertainty in predicted exposures or risk estimates thus allowing a more comprehensive characterization of risk. Allows the separation of variability from uncertainty facilitating more transparent understanding. Can effectively identify the key contributors to variability and uncertainty in predicted exposures or risk estimates and assess their impact on the potential decision alternatives.	A more comprehensive characterization of variability and uncertainty can lead to a decrease of clarity.

According to the U.S. EPA guidelines (U.S. EPA, 2001 and U.S. EPA, 2014), a tiered approach is advocated for risk assessment – a graduated hierarchical approach that progresses from simple to more complex analyses. Assessments at a lower tier typically begin with DRA, performing conservative screening assessment to quickly identify negligible-risk scenarios where no further assessment is needed. This methodology is therefore intentionally biased toward overestimating exposure resulting in a worst-case estimate of risk. It only progresses to a higher tier applying increasingly more complex risk assessment methods, such as PRA, if non-negligible risks have been determined, the current level of analysis is insufficient to support the risk management decisions, and the added complexity (or refinement within the current tier) would provide sufficient benefit to justify the additional effort. The level of complexity should be aligned with the risk assessment objectives, risk management goals and decision-making context, with more sophisticated assessments introduced commensurate with the significance of the problem and expected consequences of the risks.

A tiered approach also reflects a graduated approach to uncertainty assessment. At lower tiers, uncertainty can be addressed implicitly through conservative assumptions, resulting in precautionary, often worst-case estimates. Higher tiers incorporating probabilistic methods can explicitly quantify uncertainty and variability and enable sensitivity analysis to identify the most influential parameters, which can help prioritizing data collection and uncertainty reduction efforts and assess the robustness of results under different scenarios.

Extensive literature exists on uncertainty analysis methods and their application in model-based assessments (e.g. Refsgaard et al., 2007, Mesa-Frias et al., 2012, Maxim, 2014). There are existing software tools to assess the radiological impacts on non-human biota, such as the ERICA Tool (Beresford et al., 2007; Brown et al., 2008; Larsson, 2008) and humans, even specifically arising from

NORM and radioactively contaminated legacy sites as NORMALYSA Tool (IAEA, 2023). These assessments mainly focus on the quantifiable aspects, while the qualitative insights are historically largely under-addressed or even neglected despite growing recognition of their significance (Chen et al., 2007; van der Sluijs et al., 2005). Unlike quantitative uncertainty, qualitative uncertainty analysis lacks a standardized numerical framework and follows a more subjective logic (Chen et al., 2007). Systematic qualitative uncertainty analysis methods do exist (e.g. Chen et al., 2007) and can complement the quantitative tools in multidimensional uncertainty assessment (e.g. van der Sluijs, et al., 2005).

4.3. Plan for future research

Future research activities should be planned internationally in order to develop an integrated impact and risk assessment procedure(s). Based on here provided practical and theoretical outcomes, certain steps which should be included are provided as follows:

- Conduct human and non-human risk assessments (both DRA and PRA) using real-world data from NORM and legacy sites and explore their integration into a holistic risk assessment framework that jointly address human health and ecological risks under complex multi-hazard conditions.
- Apply existing ERA methodologies addressing multiple, co-occurring radioactive and non-radioactive hazards – ranging from screening-level approaches to more sophisticated mechanistic methods – using real-world data from NORM and legacy sites.
  Comparatively evaluate these methodologies how they perform when applied to real-world data from NORM and legacy sites, in terms of data requirements, computational complexity, accuracy, robustness, ability to consider uncertainties, practical feasibility and suitability under real-life complex environmental conditions.
- Systematically assess the impact of uncertainty within the integrated risk assessment framework and its propagation throughout the stages of ERA/EIA processes. This includes quantifying the contribution of different uncertainty sources and identifying the most dominant factors driving variability in the assessment outcomes, and evaluating how enhanced knowledge, i.e., the inclusion of site-specific information may reduce uncertainty. In addition to quantitative analysis, the potential for identifying, characterizing and handling of qualitative uncertainties should be also explored within the context of integrated risk assessment. This includes examining how e.g., expert judgement, model assumptions and data gaps contribute to the overall uncertainty.
- Special attention should be given to how uncertainties (either qualitative or quantitative) affect the interpretation of assessments, and how they could be systematically reported to support more robust decision-making and transparent risk communication – particularly in multi-hazard scenarios at NORM and legacy sites, where limited historical data, heterogeneous contamination and long-term and complex environmental dynamics introduce substantial uncertainty.

5. Conclusions

Co-occurrence of radionuclides and non-radioactive hazardous elements at various NORM and legacy sites is a common phenomenon. To facilitate the regulatory decision-making and management of such sites, environmental impact assessment is a crucial step. Currently, there are separate approaches for impact and risk assessment of radionuclides and other non-radioactive elements. Reasons for this lie in historical differences in legislation, methodology, site characterisation, impact and effects analysis, and risk evaluation. However, it has been internationally recognised that an integrated approach to impact assessment would be beneficial.

A comprehensive site characterisation, at five localities of the Fen complex in Norway, showed enrichment of the soil with ^{232}Th, ^{228}Ra, REE but also moderate levels of ^{238}U, ^{226}Ra, As and several metals (Cu, Cu, Zn, Pb). Similar distribution patterns of ^{232}Th, REE and As among soil fractions, with a dominant irreversibly bound fraction were observed across investigated locations. ^{238}U and investigated metals were distributed somewhat differently, with approximately 50% content in available soil fractions, indicating higher mobility and bioavailability in soil.

Correlation analyses of investigated element hazards and environmental parameters showed, in general, complex patterns that differed between localities according to mineralogy, bedrock and pH. However, uncertainties should be highlighted as they might affect the impact assessment process. Still, a statistically significant negative correlation between soil pH on one hand and ^{232}Th, ^{228}Ra, REEs, Cr, Co, Ni, and As on the other hand, is probably related to the differences between bedrock in speciation and complexation of these elements. This was one of the main findings of the study, demonstrating that where REE was high, so would also the natural radioactivity be high, although solubility might be reduced with the presence of organic matter and increasing pH. Moreover, principal components analysis of all parameters at the same time showed that the most significant correlations were related to bedrock and mineralogy, either explaining around equal parts of variance, with a dependency on pH, for NOR, REE and metals. A trend showing higher distributions (%) of investigated hazardous elements in available soil fractions was observed for the localities with former mining, possibly indicating elevated weathering. However, it was difficult to make final conclusions as the relationship to mineralogy and dominant bedrock was strong.

Additionally, a literature overview of the most important steps and general features in EIA was provided. Advantages and challenges of integrated assessment procedures, which would take into consideration both radioactive and non-radioactive hazards, as well as impacts and possible risks for humans and biota, were analysed. The integrated assessment procedure would have obvious advantages, including recognition of interdependencies, ensuring more realistic outputs, improved scientific quality and robustness, increased efficiency. However, due to differences between hazards and their complex behaviour, many challenges are identified: from available multiple exposure data, different scientific methodologies, different impact assessment paradigms, development of suitable models and their validation to adaptation to existing legislation, implementation in regulatory practice and public communication of holistic risks. Sources and types of uncertainties, followed by an examination of the available approaches for analysing and considering of these uncertainties within model-based ERA, must be keep in mind in development and use of integrated assessment.

Based on available findings from literature and from this study, the preparatory steps for the development of integrated ERAs were proposed as a) conducting human and non-human risk assessments using real-world data to further explore conditions for their practical integration into a holistic risk assessment framework, b) applying existing ERA methodologies and real-world data to address multiple, co-occurring radioactive and non-radioactive hazards, c) systematic assessment of the impact of uncertainty (quantitative, qualitative and communication of both) within the integrated risk assessment framework and its propagation throughout the stages of ERA/EIA processes.

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