LITERATURE REVIEW AND COMPILATION OF ECOTOXICITY DATA OF FLUORIDE, RARE EARTH ELEMENTS, AND NATURALLY OCCURRING RADIONUCLIDES

Contaminants of concern associated with the Kvanefjeld and Tanbreeze mining projects in South Greenland

Scientific Report from DCE – Danish Centre for Environment and Energy No. 483 2022

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Data sheet

Series title and no.:	Scientific Report from DCE – Danish Centre for Environment and Energy No. 483
Category:	Scientific advisory report
Title:	Literature review and compilation of ecotoxicity data of fluoride, rare earth elements, and naturally occurring radionuclides
Subtitle:	Contaminants of concern associated with the Kvanefjeld and Tanbreeze mining projects in South Greenland
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Publisher:	Aarhus University, DCE – Danish Centre for Environment and Energy ©
URL:	http://dce.au.dk/en
Year of publication:	February 2022
Editing completed:	January 2022
Referee:	Lis Bach
Quality assurance, DCE:	Vibeke Vestergaard Nielsen
Financial support:	Aarhus University, The Ministry of Environment of Denmark, and Greenland Institute of Natural Resources (GINR)
Please cite as:	Hansen, V., Gustavson, K., Asmund, G. & Jia, Y. 2022. Literature review of ecotoxicity data of fluoride, rare earth elements and naturally occurring radionuclides. Aarhus University, DCE – Danish Centre for Environment and Energy, 83 pp. Scientific Report No. 483. http://dce2.au.dk/pub/SR483.pdf

Abstract: This report by the Danish Centre for Environment and Energy (DCE) and Greenland Institute of Natural Resources (GINR) aims to gather worldwide information on excitotoxicity data on fluoride (F), rare earth elements (REEs), and naturally occurring radionuclides (NORs) that can potentially be used in Greenland. The report is not explicitly focused on Kvanefjeld and Tanbreeze mineral exploration projects in south Greenland but on the mining and milling of minerals containing REEs, NORs, and F in general in Greenland.

The report provides relevant information to be used as a basis for environmental regulation and monitoring of mining projects and establish guidelines on concentrations threshold values for the REEs, NORs, and F in water, soil, dust, vegetation, berries, birds, and fish both in the mining area but also for the surrounding environment. This report can also be used for teaching or as guide for the general public, politicians, authorities, education, industry, and other Greenland stakeholders.

Keywords:	Greenland, mineral resources, environmental impacts, biological radiation effects, environmental quality criteria, toxicity threshold values, clearance, and exemption values, human health
Layout:	Graphic Group AU Silkeborg
Front and back page photo:	Violeta Hansen
ISBN:	978-87-7156-662-8
ISSN (electronic):	2244-9981
Number of pages:	83
Internet version:	The note is available in electronic format (pdf) at http://dce2.au.dk/pub/SR483.pdf

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Summary

Since previous mining projects in Greenland have not involved the exploitation of minerals containing rare earth elements (REEs), naturally occurring radionuclides (NORs), and fluoride (F) there was a need to build up specific knowledge at the Danish Centre for Environment and Energy (DCE) and Greenland Institute of Natural Resources (GINR) on ecotoxicity data on these contaminants. No Greenlandic environmental quality criteria guidelines exist for F, REEs, and NORs, and hence this report focuses on these elements.

This report by DCE and GINR aims to gather worldwide information on excitotoxicity data on F, REEs, and NORs that can potentially be used in Greenland. The report is not explicitly focused on Kvanefjeld and Tanbreeze mineral exploration projects in south Greenland but on the mining and milling of minerals containing REEs, NORs, and F in general in Greenland.

The report provides relevant information to be used as a basis for environmental regulation and monitoring of mining projects and establish guidelines on concentrations threshold values for the REEs, NORs, and F in water, soil, dust, vegetation, berries, birds, and fish both in the mining area but also for the surrounding environment. This report can also be used for teaching or as guide for the general public, politicians, authorities, education, industry, and other Greenland stakeholders.

Abbreviations

AD	Adaptation to radiation effects
AMAP	Arctic Monitoring and Assessment Program
BCF	Bioconcentration factor
Bq	Bequerel. Bequerel is the SI derived unit of radioactivity. One becquerel is defined as the activity of a quantity of radioactive material in which one nucleus decays per second.
Bq kg^{-1}	Becquerel per kilogram. It is the unit of specific activity (mass activity) equal to the activity of one Becquerel of the sample with the total mass of one kilogram.
CFs	Concentration factors
CYT	Cytogenetic effects
DCRL	Derived Consideration Reference Level
EC_{50}	EC 50 - Half maximal effective concentration. Concentration of a drug, antibody or toxicant that cause 50% of effect at a specified exposure time
ECHA	European Chemicals Agency
ECOL	Ecological effects
Gy	Gray. The SI unit of the absorbed dose of ionizing radiation, corresponding to one joule per kilogram.
HREEs	Heavy REEs
IADWC	Indicative admissible drinking water concentrations
IAEA	International Atomic Energy Agency
IC25	The 25% maximal inhibitory concentration
ICD	Integrated Criteria Document
ICRP	International Commission on Radiological Protection
ITDI	Indicative tolerable daily intake values
LC_{50}	The Lethal Concentration 50. Concentration of a chemical that cause 50% mortality of a population at a specified exposure time
LLRs	Long-lived radionuclides
LNs	Lanthanides
LREE	Light REE
LOAEL	Lowest Observed Adverse Effect Level
MAC	Maximum acceptable concentration
MB	Morbidity
MT	Mortality
NHB	Non-human biota
NO	No effect
NOAEL	No Observed Adverse Effect Level
NOEC	No Observed Effect Concentration
NOEL	No Observable Effect Level
NORs	Naturally occurring radionuclides
NORM	Naturally occurring radioactive materials
NTP	National Toxicology Program
NWT	Northwest Territories
PNEC	Predicted no effect concentration
PPM	Parts per million
RAR	Risk Assessment Report
RDL	Reference dose level
REO	Rare earth oxide
REPR	Reproduction
RIVM	Rijksinstituut voor Volksgezondheid en Milieu

SF	Safety factor
STIM	Stimulation effects
ΣREE	Sum of REEs
TENORM	Technically enhanced naturally occurring radioactive materials
UDS	Unscheduled DNA synthesis
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
US EPA	United States Environmental Protection Agency
WHO	World Health Organization
$\mu Gy\ h^{-1}$	Microgray per hour. It is a unit of measurement of radiation absorbed dose rate.

1 Introduction

The Danish Centre for Environment and Energy (DCE) and Greenland Institute of Natural Resources (GINR) have long experience in advicing the Greenland authorities on environmental issues. The advisory includes evaluating Environmeltal Impact Assessment (EIA) mining reports to give an accurate and thorough description of the project's environmental impacts. The final EIA report and the so-called 'White Book' containing comments from public consultations, form the basis for the Greenland Government to decide for or against a mining project and define its environmental requirements. DCE and GINR have, on behalf of the Environmental Agency for Mineral Resources Activities (EAMRA), assessed Kvanefjeld's and Tanbreeze's EIA's and supporting technical documents since their submission. DCE and GINR provide recommendations on environmental requirements and conditions to minimize adverse environmental effects concerning the exploitation license.

No Greenland environmental quality criteria guidelines exist for REEs, NORs and F, and hence the following report focuses on these elements.

Kvanefjeld and Tanbreeze deposits contain REEs, NORs, and F, as elements of concern. Hence, there was a need to build up specific knowledge at DCE and GINR on ecotoxicity data on these contaminants.

The Kvanefjeld (Kuannersuit) multi-element deposit is located approximately 7.5 km to the north of Narsaq, South Greenland, at an elevation of 700 m above sea level. Kvanefjeld deposit that, besides rare earth elements (REEs), flour (F), and zinc (Zn), contains naturally occurring radionuclides (NORs) such as uranium (U), thorium (Th), and their daughter products like radium (Ra), radon (Rn), lead (Pb) and polonium (Po). Fluoride occurs in a water-soluble mineral Villiaumite. Because of this, F may be easily mobilized in the environment.

Another REEs deposit is Tanbreeze, located at Killavaat Alannguat (Kringlerne) on the Kangerluarsuk Fjord in South Greenland, 12 km southwest of Narsaq. The Minister of Industry & Mineral Resources in Greenland has recently granted Tanbreeze a mineral exploitation licence. The Tanbreeze project is characterized by relatively high concentrations of zirconium, yttrium, niobium, hafnium, tantalum and heavy rare earth elements (HREEs) in the ore.

Possible sources of environmental contamination associated with mining and milling projects include the dispersion of dust from drilling, blasting, crushing, processing, and transportation of ore and waste rock, discharges of process water to freshwater and fjord, and seepage and aerosol spray dispersion from tailings facilities, etc.

This report by DCE and GINR aims to gather worldwide information on excitotoxicity data on F, REEs and NORs. The report is not explicitly focused on Kvanefjeld and Tanbreeze but on the mining and milling of minerals containing REEs, NORs and F in general. Not all the information gathered here will be relevant to all potential future mining and milling projects in Greenland. Still, it will provide a foundation of knowledge that can be targeted towards a specific project, taking site-specific factors such as ore composition, local environmental conditions, etc. into account.

This report includes information gathered through available literature, databases, and national and international agencies and organizations.

The report provides relevant information to be used as a data basis when setting the environmental requirements and conditions for potential new mines involving mining and milling of minerals containing REEs, NORs, and F in Greenland. The gathered information can also be used to establish threshold values for the REEs, NORs, and F concentrations in water, soil, dust, vegetation, berries, and fish in the mining area and the surrounding environment. This report can also be used in teaching or as a guide for the general public, politicians, authorities, education, industry and other Greenland stakeholders.

This report answers the following questions:

- What are the worldwide background concentrations of F, REEs and NORs?
- To what extent do F, RREs and NORs bioaccumulate in flora and fauna?
- What are the biological effects (ecotoxicity) in marine, freshwater, and terrestrial flora and fauna from exposure to F, REEs and NORs?
- What are the associated human risks from exposure to F, REEs and NORs?
- What available regulatory quality levels and discharge limits exist worldwide for F, REEs and NORs into the environment from industrial sites?

Gathered available literature data for F, REEs and NORs are presented in the five chapters of this report:

Chapter 1 gives an introduction to the report and the different chapters.

Chapter 2 provides a review of available F ecotoxicity data, F environmental background levels, behavior of F in different ecosystems, environmental quality criteria and toxicity threshold values.

Chapter 3 provides a review of available REEs ecotoxicity data, REEs environmental background values, REEs human health aspects, environmental quality criteria and toxicity threshold values.

Chapter 4 provides a review of available NORs levels in the environment, biological radiation effects of non-human biota, ecotoxicity data, NORs associated with human health aspects, guidelines values elsewhere and recommended NORs threshold values for e.g., drinking water, clearance, and exemption values for Greenland.

Chapter 5 includes the main findings of the report and provides recommendations for Greenland.

It is important to report that this report focuses on environmental protection and deals superficially with human health protection as this is not within the advisory field of DCE and GINR. Furthermore, cadmium (Cd), stable lead (Pb) and arsenic (As) are additional contaminants of concern associated with mining and milling in Greenland but are not included in this report. The reason for this is that the Greenlandic environmental quality criteria guidelines for Cd, Pb and As are available in Guidelines for preparing an Environmental Impact Assessment (EIA) report for mineral exploitation in Greenland (MRA 2015). Extensive data on Cd, Pb and As can be found via Arctic Monitoring and Assessment Programme (AMAP, 1998, 2002), the databases of the European Chemicals Agency (EU ECHA), World Health Organisation (WHO), etc.

The project has been financed by DCE, the Ministry of Environment and Food of Denmark and GINR.

2 Fluoride (F)

The Kvanefjeld resource contains the water-soluble villiaumite mineral (NaF). Fluoride at the Kvanefjeld, can be dispersed into the environment via air and water from the proposed mining activities such as drilling and blasting, grinding and crushing, ore processing, tailings, and waste rock disposal. Fluor concentration is expected to be as high as 250 ppm in the water cover of the tailing facility. One of the environmental issues which has been investigated at the proposed Kvanefjeld project is the risk of fluor contamination of drinking water via seepage and spray water dispersion from Taseq tailings facilities and dust dispersion and resuspension.

In this section, available general data on environmental fate and pathways of fluor and ecotoxicological data are described. The source of the following text is EU ECHA 2020: https://echa.europa.eu/da/registration-dossier/-/registered-dossier/14274/5/1

The information from EU ECHA 2020 is primarily taken from the European Union Risk Assessment Report (EU RAR) for HF and the Rijksinstituut voor Volksgezondheid en Milieu (RIVM) in Netherlands, Integrated Criteria Document (ICD) fluorides document (Sloof et al., 1987). It is assumed that these observations are also applicable for NaF.

Some of the references can be found at: https://echa.europa.eu/da/registration-dossier/-/registered-dossier/14274/5/1

Other refererences cited in the report can be found in the reference list, chapter 6 of this report.

2.1 Fluoride environmental fate and pathways water, sediments and soil

Since NaF is of environmental concern at the Kvanefjeld site, the information included in this chapter focuses on this chemical form.

Sources of environmental fluoride are of anthropogenic (e.g. phosphate fertiliser) and natural (e.g. volcanic, weathering, marine aerosols) origin. In the environment NaF ionize and is not subject to biodegradation.

F in the atmosphere

Fluoride, as hydrogen fluoride (HF), is removed rapidly from the atmosphere by wet and dry deposition. Wet and dry deposition rates for fluoride have been reported at levels of ~30 mg m^{-2} and ~17 mg m^{-2} (assumed per year), respectively, in the Netherlands, and which is in accordance with a total deposition of up to 52 mg m^{-2} per year in Poland (Walna et al., 2013 and EU ECHA 2020). The background deposition of F in Greenland is unknown.

F in water

The background fluoride concentrations in surface water will depend on geological, physical, and chemical characteristics. In surface water, at

environmental pH, sodium fluoride dissociates and form Na^+ and F^-. At lower pH values for HF, the proportion of fluoride ion decreases while the proportion of HF_2^- and non-dissociated HF increase (EU ECHA 2020).

In surface water, the F^- concentration depends on the presence of other inorganic elements. For example, surface water, with relatively high concentrations of phosphate and calcium, contains no F^- because insoluble fluoride salts are formed. A large proportion of insoluble fluoride salts are transferred to sediment. Concentrations of F in freshwater have been reported in the range of 0.2 - 9.5 mg L^{-1}, dependent on area and season (Sloof et al., 1989). In European countries, fluoride concentrations of up to 20 mg L^{-1} have been reported (EU ECHA 2020).

In seawater, fluoride is present as free fluoride (F^-) (51%), magnesium fluoride (MgF_2) (47%), calcium fluoride (CaF_2) (2%), and traces of HF. The reported average concentration of total fluoride in seawater was 1.4 mg L^{-1} (EU ECHA 2020).

F in sediment

Concentrations of fluoride as insoluble complexes of up to 200 mg kg^{-1} for marine sediments and up to 450 mg kg^{-1} for river sediments on a dry matter basis have been reported (EU ECHA 2020).

F in soil

In soils, fluoride is immobile as a result of its precipitation and adsorption. F^-, has strong complexation properties (EU ECHA 2020). The Freundlich isotherm can describe the fluoride adsorption in the soil, up to a concentration of 20 mg L^{-1} in acidic soils and up to 10 mg L^{-1} in alkaline soils (EU ECHA 2020). Below a soil pH of 5.5, adsorption is low as fluoride is predominantly found as complexes such as fluorspar, cryolite, apatite and clay minerals. At pH values above 6, the fluoride ion (F^-) is the dominant species. At pH values above 5.5, adsorption is lower due to the reduced electrostatic potential. Fluoride precipitates in excess of calcium ions in soil solution. Hence the concentration of F^- in calcareous soils is low (EU ECHA 2020).

Reported fluoride concentrations in clay soil in the Netherlands were in the range of 330 - 660 mg kg^{-1}, with an average value of over 500 mg kg^{-1}. The concentration of total fluoride in Dutch agricultural soils is correlated with the clay content. Samples of greenhouse soil may have relatively higher fluoride contents due to fluorine-containing phosphate fertilizer (EU ECHA 2020).

F Biodegradation

NaF do not biodegrade (EU ECHA 2020).

F bioaccumulation

Uptake of fluoride into plants from soil is low mainly due to the low bioavailability of fluoride in the soils, and that atmospheric uptake is generally the main route of exposure of plants (Sloof et al. 1987). A relatively high fluoride uptake rate was observed for grass species (EU ECHA 2020).

In aquatic organisms, fluoride accumulates mainly in the exoskeleton and the skeleton with no accumulation in edible tissues. Fluoride biomagnification in the aquatic biota is considered insignificant (EU ECHA 2020).

In the terrestrial environment, fluoride accumulates in the skeleton of vertebrates and invertebrates. The EU RAR (2001) notes that the lowest fluoride levels were in herbivores, with higher levels in omnivores and highest levels in predators, scavengers and pollinators. These findings indicate a moderate degree of biomagnification. Vertebrate species store most of the fluoride in the bones and the teeth (EU ECHA 2020).

2.2 Flouride toxicity and effect levels

Flouride toxicity to aquatic and terrestrial organisms

All available studies were performed with NaF. In some cases, HF was used as both NaF and HF are highly water-soluble and dissociate in the aquatic environment. The toxicity of both chemicals is essentially due to the F^-. The EU RAR noted a clear relationship between aquatic toxicity of NaF, HF and water hardness. Tests performed in water with <50 mg $CaCO_3$ L^{-1} showed higher toxicity than those performed in water with >50 mg $CaCO_3$ L^{-1} mainly due to the precipitation of fluoride as CaF_2. All endpoints are expressed in terms of concentrations of the F^- (EU ECHA 2020).

NaF short-term toxicity to fish

LC_{50} values of 107.5, 92.4, 118.5, 105.1 and 64.1 mg F kg^{-1} at 96, 120, 144, 168 and 192 h respectively are reported for rainbow tout (Camargo & Tarazona, 1991). The same authors report LC_{50} values for brown trout of 164.5, 135.6, 118.5, 105.1 and 97.5 ppm after 96, 120, 144, 168 and 192 h, respectively. The EU RAR for HF reports additional LC_{50} values of 299 mg F L^{-1} (48 h in *Leuciscus idus*); 51 mg F L^{-1} (96 h in *Onchorynkus mykiss*) and 340 mg F L^{-1} (96 h in *Gasterosteus aculeatus*). The ICD reports additional data, with LC_{50} values ranging from 128 - 460 mg F L^{-1} (Sloof et al. 1987). The recommended EU ECHA 2020, NaF effect concentration to be used in risk assessments is 51 mg L^{-1}.

NaF long-term toxicity to fish

In a 21-day test with *Oncorhynchus mykiss*, an LC_5 value of 4 mg F L^{-1} is reported (actual concentration). The test was conducted in water with 12 mg $CaCO_3$ L^{-1}. EU ECHA 2020 recommend a NaF effect concentration of 4 mg L^{-1} to be used in risk assessments.

NaF short-term toxicity to aquatic invertebrates

Camargo & Tarazona (1991) reported that benthic larvae are sensitive to fluoride concentrations at EC_{50} values ranging from 26 - 48 mg F L^{-1}. The EU RAR reported EC_{50} values for Daphnia sp. range from 97 - 352 mg F L^{-1} and EC_{50} values of 10.5 - 39 mg F L^{-1} for marine invertebrates. The recommended EU ECHA 2020, NaF effect concentration to be used in risk assessments is 26 mg L^{-1} for freshwater invertebrates and 10.5 mg L^{-1} for marine water invertebrates.

NaF long-term toxicity to aquatic invertebrates

The EU RAR summarises two reproductive studies of sodium fluoride on *Daphnia magna*. The two studies report NOEC values of 3.7 and 14.1 mg F L^{-1}. The ICD reports EC_{50} values in the range of 10 - 48 mg F L^{-1} (EU ECHA 2020). EU ECHA 2020 recommend a NaF effect concentration of 8.9 mg L^{-1} to be used in risk assessments.

NaF toxicity to freshwater and marine algae

The EU RAR, EC_{50} values for freshwater algae are reported to range from 43 to 122 mg F L^{-1} (EU ECHA 2020). For marine algae, the EC_{50} was 81 mg F L^{-1} in a single study with Skeletonema costatum. NOEC values of 50 - 249 mg F L^{-1} and 50 - 200 mg F L^{-1} are reported for freshwater and marine algae, respectively (EU ECHA 2020). The recommended EU ECHA 2020 NaF effect concentration to be used in risk assessments is 43.0 mg L^{-1} for freshwater algae and 81.0 mg L^{-1} for marine water algae.

Toxicity to microorganisms

The EU RAR summarises and reviews the available data on fluoride toxicity to aquatic microorganisms; NOEC values of 7.1 - 226 mg F L^{-1} are reported.

Sediment's toxicity

No data are available.

Toxicity to soil macro-organisms

The toxicity of NaF, potassium fluoride (KF), CaF_2 and the sodium salt of fluoroacetic acid to *Eisenia fetida* was investigated for 22 weeks (EU ECHA 2020). The earthworms' rate of maturity was reduced considerably when NaF and KF were used. The number of hatchlings was decreased in the presence of NaF, while the number of cocoons was reduced in the presence of a low concentration of NaF and KF (Vogel & Ottow, 1992). Long-term EC_{10} for soil macroorganisms were 1200 mg kg^{-1} soil dry weight (EU ECHA 2020).

Toxicity to soil micro-organisms

The EU RAR (2001) reports NOEC values from 106 to 3000 mg / kg. Long-term EC_{10} for soil microorganisms were 106 mg kg^{-1} soil dry weight (EU ECHA 2020).

Toxicity to terrestrial plants

Based on several fumigation experiments with plants, e.g. ornamental crops, fruit crops, and conifers exposed to HF, NOEC values were calculated for highly sensitive, sensitive and slightly sensitive plant species. NOEC values of between 0.2 - 7.5 mg m^{-3} HF were reported for plant species (EU ECHA 2020).

F toxicity to mammals

Acute Toxicity

The LD_{50} for acute oral toxicity of NaF in male Sprague-Dawley rats was 223 mg kg^{-1} body weight with 95% confidence limits of 177 - 272 mg kg^{-1} body weight. The LD_{50} for NaF in female Sprague-Dawley rats was reported to be 148.5 mg kg^{-1} body weight with 95% confidence limits of 116.3 - 185.5 mg kg-

^{1} body weight, respectively (EU ECHA 2020). The EU ECHA 2020 recommended NaF effect concentration (acute toxicity via oral route) to be used in risk assessments is 148.5 mg kg^{-1} mammal body weight.

Chronic Toxicity

The U.S. National Toxicology Program (NTP, 1990) evaluated the toxicological effects of continuous exposure to 0, 30, 100 or 300 mg L^{-1} NaF in drinking water on 344 male and female rats for six months (EU ECHA 2020).

There were no significant signs of toxicity observed at concentrations of up to 30 mg L^{-1} (EU ECHA 2020).

NaF caused (EU RAR, 2001):

- Weight loss at 300 mg L^{-1}
- Fluorosis of the teeth at 100 and 300 mg L^{-1}
- Minimal hyperplasia of the gastric mucosa of the stomach at 100 and 300 mg L^{-1} (however, one high dose rat of each sex had an ulcer)
- A dose-related increase in fluoride content of bone and urine with increasing fluoride concentration in the drinking water, and a significant increase in fluoride content in the plasma at 300 mg L^{-1}.

Reproduction

Collins et al. (2001) reported that NaF in Sprague-Dawley rats' drinking water levels of up to 250 ppm. Concentrations of 28.4 mg NaF kg^{-1} body weight per day or 12.8 mg fluoride kg^{-1} body weight per day had no adverse effects on reproduction throughout three generations. Mating, fertility and survival indices were not affected. The EU ECHA 2020 recommended NaF effect concentration on fertility via oral route, to be used in risk assessments is 28.4 mg kg^{-1} mammal body weight per day.

Metabolism and distribution

A comparative study (Whitford et al., 1991) of NaF pharmacokinetics on five species such as dog, cat, rat, rabbit and hamster determined major quantitative species differences in fluoride's metabolism. When factored for body weight, the young adult dog's plasma, renal, and extra-renal (calcified tissue) values resemble those of the young adult human most closely (EU ECHA 2020). The five-minutes plasma fluoride concentrations were ordered as follows: dog > rabbit > rat > hamster > cat (concentrations were 110.8 +/- 14.3, 91.3 +/- 3.1, 78.4 +/- 5.3, 69.1 +/- 4.9 and 52.2 +/- 4.8 µmol L^{-1}, respectively). In terms of body weight, the plasma clearances were highest in the hamster, rat and cat (8.60, 7.34 and 7.24 mL min^{-1} kg^{-1}, respectively), intermediate in the rabbit (5.80 mL min^{-1} kg^{-1}), and lowest in the dog (3.50 mL min^{-1} kg^{-1}). This result indicates that the hamster, rat and cat cleared fluoride from their extracellular fluids more than two times faster than did the dog. The plasma clearance of fluoride in the rabbit was 66 % faster than that of the dog (EU ECHA 2020).

2.3 Risk to humans

Fluoride toxicity to humans is shown in Table 2.1. Related to the safe consumption of food and feed, the EU has defined a maximum residue level tolerated in drinking water, food and feed (Table 2.2.).

Table 2.1. NaF toxicity to humans.

NaF	Lethal dose mg NaF kg^{-1} body weight	Toxic dose mg NaF kg^{-1} body weight	No Observed Adverse Effect Level mg NaF kg^{-1} body weight	References
Children	16	5.0	0.05 - 0.07	Ullah et al., 2017
Adult	32	5.0		Ullah et al., 2017
Adult	40 - 80*			Guth et al., 2020

*This corresponds to 1.4 – 2.8 g fluoride for a 70 Kg person.

Table 2.2. Flouride maximum residue levels (MRL) in foodstuff.

Flouride	MRL (mg kg^{-1})
Milk of cattle, goat, sheep, other milk (mg L^{-1})	0.2
Vegetables, fresh or frozen	2
Fruits, including also berries, oilseeds, oil fruits and nuts	2
Cereals	2
Meat	1
Bird eggs	0.2
Drinking water (mg/L)*	1.5

Source: Fluoride - legislation-obligation - ECHA (europa.eu)

* Narsaq/Hjemmestyrets bekendtgørelse nr. 7 af 17. marts 2008: bkg_nr_07-2008_dk (lovgivning.gl).

2.4 F summary

When entering the aquatic environment, fluoride may be present in seawater and surface water as F^-, MgF_2 and / or CaF_2. The uptake of fluoride into organisms depends on the concentration of the F^- in the environment. In surface water, the concentration of F^- is strongly related to the presence of other inorganic contaminants, as for example insoluble fluoride salts are formed in the presence of phosphate and calcium. In soils and sediments fluoride occur as insoluble complexes.

Uptake of fluoride into plants from soil is often low due to the low bioavailability of fluoride in the soil. A relatively high fluoride uptake rate has been observed for grass species.

The ingestion of fluoride via water and food may lead to elevated fluoride levels in animals and humans. Fluoride bioaccumulation and biomagnification in the aquatic and terrestrial invertebrates and fish is of little significance. Fluoride may bioaccumulate in invertebrates' exoskeleton and the skeleton of fish but does not accumulate in edible tissues.

The EU ECHA 2020 recommended NaF effect concentrations to be used in risk assessments for different biota. These values are included in this chapter. In Greenland, there are no environmental quality guidelines values for F in the air, deposited dust or water bodies. The information included in this chapter may be used when establishing specific fluoride quality environmental values for Greenland.

3 Rare Earths Elements (REEs)

REEs refers to the 15 elements with atomic numbers 57 to 71 and termed lanthanides (LNs), together with scandium (Sc) and yttrium (Y). The lanthanides are lanthanum (La), cerium (Ce), praseodymium (Pr), neodymium (Nd), promethium (Pm), samarium (Sm), europium (Eu), gadolinium (Gd), terbium (Tb), dysprosium (Dy), holmium (Ho), erbium (Er), terbium (Tm), ytterbium (Yb) and lutetium (Lu).

The REEs are:

1. Light REEs (LREEs) as Sc and lanthanides with atomic numbers 57 through 64 (e.g. La, Ce, Pr, Nd, Pm, Sm, Eu, Gd)
2. Heavy REEs (HREEs), as Y and lanthanides with atomic numbers 65 through 71 (e.g. Tb, Dy, Ho, Er, Tm, Yb, Lu).

Sometimes, the sum of REEs (∑REEs) is used. The REEs do not occur as native elemental materials in nature, only as part of the host mineral's chemistry (US EPA 2012).

At the Kvanefjeld, the ore contains all REEs. The REEs can be dispersed from the Kvanefjeld site into the environment via air and water from the proposed mining activities such as drilling and blasting, grinding and crushing, ore processing, tailings and waste rock disposal.

Since the last two decades, REEs have been recognized as emerging elements of concern with respect to environmental pollution and human health. Research related knowledge of REEs is limited in comparison with the common elements, such as heavy metals.

REEs have been extensively used in industrial-, modern technological-, agricultural-, medical systems etc. (Table 3.4. of Appendix A). The global production and use of REEs have expanded rapidly since the 1980s. REEs have been used as tracers to investigate mining-related environmental impacts, geochemical processes and food research (Oliveri et al., 2010; Delgado et al., 2012; Drivelos et al., 2014).

Recently, REEs have been recognized as emerging elements of concern with respect to polluting the environment and harm humans (Pagano et al., 2019) and have been detected in the runoff, wastewater and aquatic environment as a consequence of their industrial and agricultural uses (Gonzalez et al., 2014). Since the 1990s, as a result of using Gd in the medical treatment, high Gd concentrations were detected in rivers in Germany (Bau and Dulski, 1996) and elsewhere (Kulaksiz and Bau, 2013) in the estuarine waters, coastal seawater, rivers, groundwater and tap water.

3.1 REEs sources into the environment

REEs in water

The lanthanides are abundant in the Earth's crust (except for promethium) (Ng et al., 2012).

Background values of REEs in groundwater, estuaries, lakes and coastal seas have been extensively studied (Elderfield et al., 1990; Smedley, 1991; Johannesson and Lyons, 1995). Groundwater and river waters do not exhibit uniform REE patterns (Goldstein and Jacobsen, 1988; Elderfield et al., 1990; Sholkovitz, 1995), which might be due to humic substances complexing with REEs (Pourret et al., 2007).

REEs in soil

REEs mobility in soils depends on factors such as pH, organic matter and clay minerals content (Price et al., 1991; Wu et al., 2001a; b). In a study by Pang et al. (2001), approximately 90% of total REEs were fixed into soil surface with the remaining 10% soluble and mobile. In Chinese soil, the REEs range from 157.0 to 197.0 mg kg^{-1} (average 177.0 mg kg^{-1} (except for Pm)). Ding et al. (2004), reported a Chines soil average REEs of 157.3 mg kg^{-1}.

The REEs typically enrich in the surface soil and decreases along with the soil profile. However, the vertical distribution of REEs along the soil profile varies depending on the soil type (Tyler, 2004).

As examples, Appendix A includes:

- Soil REEs baseline values from Bayan Obo REE mining factory in Baotou, Inner Mongolia, China (Table 3.1. of Appendix A).
- REEs concentrations in soils in China compared to the world average soil concentrations (Table 3.2. of Appendix A).
- REEs concentrations in freshwater, seawater, effluent water from the sewage treatment plant, and tap water from different sampling sites in Sweden, Japan and Germany (Table 3.3. of Appendix A).

3.2 Toxicity and bioaccumulation of REEs in marine, terrestrial and freshwater environment

REEs toxicological database is mostly confined to Ce and La, with limited information available for Gd, Nd and other REEs, particularly for heavy REEs. Limited REE toxicity data from EU ECHA is included in Table 3.21. of Appendix A. Toxicity of lanthanides decreases as the atomic number increase (Hirano & Kazuo, 1996; Zielhuis, 2006; Gonzalez et al., 2014).

The levels of REEs in the fauna and flora depends on intake, biodegradation kinetics and bioactivity (Rainbow, 2002). Pagano et al. (2019) reviewed adverse effects of REEs in vertebrates, invertebrates and plants, such as organ pathology, cytogenetics and embryological damage and growth inhibition. REEs have been used, particularly in China, as feed additives and fertilizers, taking advantage of their growth promoter effects (He and Rambeck, 2000). The phenomenon of growth stimulation at low concentrations and inhibition at high concentrations is commonly known as the "hormesis effect" of REEs

and has been documented in flora and fauna tissues in both *in vivo* and *in vitro* studies (e.g. Herrmann et al., 2016; Pagano et al., 2015).

Studies show that REEs contents decrease along with the trophic level in freshwater-, marine- and terrestrial biota, including birds (Brown et al., 2019; MacMillan et al., 2017).

Marine environment

A difference in sensitivity between freshwater and marine organisms may be associated with variation between species rather than exposure to REEs in different matrices (Gonzalez et al. 2014). Ng et al. (2012) reviewed REEs acute toxicity to saltwater organisms (Table 3.5. of Appendix A) and bioaccumulation of REEs by saltwater organisms (Table 3.6. of Appendix A).

In general, all REEs have an almost identical extent of ecotoxicity to the marine algae *Skeletonema costatum* (nominal toxicity values: 4055 – 5009 µg REE L^{-1}) (Tai et al., 2010). In the copepod, *Acartia tonsa*, the toxicity of REEs is ranked as Ce > Sm > Gd > Nd > Pr > La > Dy and in the fish, *Poecilia reticulate*, toxicity is ranked as Pr > Nd > Sm > Gd > Ce > Dy > La (Table 3.5. of Appendix A). Among the three marine species, copepod has the highest sensitivity to REEs, algae are intermediate, and fish is ranked the least sensitive (Ng et al., 2012).

Most of the studies only measured the tissue concentration of marine organisms without reporting it (Table 3.6. of Appendix A). The distribution of REEs in marine organisms follows a general pattern, with higher REEs concentrations in the digestive gland, followed by the gill, gonad and kidney, and the lowest is in the muscle (Table 3.6. of Appendix A).

Pagano et al. (2019) summarized REEs' ecotoxicity to invertebrates (Table 3.7. of Appendix A). The ecotoxicity of REEs to invertebrates is typically related to Ce in oxidized form, Pr, Nd, and Sc in chloride form. The damage is typically associated with reproduction and growth inhibition, offspring damage, and cytogenetic damage of the invertebrates (Table 3.7. of Appendix A).

Seaweed and Zooplankton

Seaweed are primary producers and play an important role in entering REEs into the marine food chain. Scarce data are available for concentrations of REEs in seaweed. Squadrone et al. (2019a) reported a baseline value of ΣREEs (i.e. a sum of all REEs) of 12 mg kg^{-1} in seaweed from the Mediterranean Sea in Northwestern Italy. Mashitah et al. (2012) reported REEs mean values of ∑LREE and ∑HREE of 15.45 µg g^{-1} and 1.67 µg g^{-1}, respectively in seaweed (*Padian Sp.*) with 1-10 folds higher than that in terrestrial plants.

Sakamoto et al. (2008) demonstrated that concentrations of REEs in seaweed are higher than that in seawater. Data on REEs' bioaccumulation in aquatic organisms is summarized in Table 3.8. of Appendix A (Gonzalez et al., 2014). In literature, a typical 0.2-1 mg L^{-1} concentration of REEs is used to identify REEs' bioaccumulation in seaweed and zooplankton (Table 3.8. of Appendix A) (Gonzalez et al., 2014).

Bivalves and Fish

Squadrone et al. (2019a) reported REE baseline values for bivalves (0.16 mg kg^{-1}) and fish (0.21 mg kg^{-1}) in the Mediterranean Sea. Marine samples showed several orders of magnitude higher sum of REEs (ΣREE) levels than terrestrial samples (Squadrone et al., 2019a).

Bioaccumulation of REEs in juvenile Arctic char (*Salvelinus alpinus*) under Greenlandic experimental field conditions was studied by Nørregaard et al. (2019). Arctic char exposed to Ce, La and Y for 15 days accumulated REEs at a decreasing level in gill, liver to muscle.

Marine mammals

La and Ce at 0.41 mg kg^{-1} (w.w.) and 0.66 mg kg^{-1} (w.w.), respectively, were detected in the kidney of pregnant female sperm whale (*Physeter macrocephalus*) (Squadrone et al., 2015).

Terrestrial environment

REEs has slight toxicity to mammals (Haley, 1979). REEs toxicity depends on the route of intake, chemical form and the studied species (Zielhuis, 2006).

Adverse effects following REEs include organ pathologies in mammals, cytogenetic and embryonic damage, and growth inhibition (Table 3.9. of Appendix A). To our knowledge, there are no available info on the carcinogenicity of REEs in terrestrial non-human biota (Hirano and Suzuki, 1996; Moriwaki and Yamamoto, 2013).

Sheep and cattle

Zhang et al. (2015) studied the effects of lanthanum (La) on the growing lambs *(Agnus)* via i*n vitro* rumen fermentation and methane (CH_4) production. The addition of $LaCl_3$ of up to 1.0 mmoL kg^{-1} dry mass to feed mixtures showed no negative impact on feeds' *in vitro* rumen fermentation. Apart from this, no data exists for other REEs for sheep and cattles.

Reindeer and muskox

No relevant information is available for reindeer *(Rangifer tarandus)* and muskox *(Ovibos moschatus)* from either scientific reports or popular papers.

Hares

Danezis et al. (2019) investigated the concentrations of La, Ce, Pr, Nd, Sm, Eu, Gd, Tb, Dy, Ho, Er, Tm, Yb, Lu, Sc and Y in muscle and liver tissues of rabbits (*Leporidac*). Liver accumulates more REEs than muscle tissue in wild and commercial rabbits.

Birds

Limited baseline REEs concentrations in birds are available (Brown et al., 2019). No relevant information is available for ptarmigans (*Lagopus muta*). Factors such as gender, age, size and overall forage areas can affect bird tissues' metal levels (Brown et al., 2019). There is currently little toxicological evidence for Y and other REEs in birds.

A high Y concentration in the seagulls *(Larus dominicanus)* were linked to repeated application of fertilizers in intensive agricultural areas dominated by corn and soybean cultivation. Squadrone et al. (2019a) studied the distribution of lanthanides in several wildlife species as including sparrow (*Passer*), hawk (*Accipiter*), tawny owl (*Screech-owl*) and crow (*Corvus*) from Northwestern Italy. They reported low values of the ΣREE in the liver (mean 0.04 mg kg^{-1}) while REEs were undetectable in muscle. Squadrone et al. (2019b) reported the successful use of penguin feathers to identify REEs levels.

Terrestrial flora

The distribution patterns of REEs may differ considerably among various species of plants at the same site and even among the same species (Wyttenbach et al., 1994). REEs induced toxicity to plants is summarized in Table 3.10. and Table 3.11. of Appendix A. The major REEs effects on terrestrial plants are toxicity to root elongation and growth inhibition (Table 3.10. of Appendix A, Pagano et al., 2019). Diversity of soil macrofauna along a gradient of REEs contamination from mining activities was increased at low REEs concentrations and reduced at high REEs concentrations, with various species showing different sensitivities (Pagano et al., 2019; Table 3.11. of Appendix A).

Concentrations of REEs in plants are in the range of µg kg^{-1} (Wyttenbach et al., 1994). Natural background concentrations in mg kg^{-1} dry weight of lanthanide elements in the plant materials in a natural forest ecosystem of northwest Germany were La of 0.15 - 0.25; Ce of 0.25 - 0.55; Sm of 0.02 - 0.04; Eu of 0.005 - 0.015; Tb of 0.005 - 0.015 and Yb of 0.015 - 0.030. (Markert and De, 1991).

Freshwater environment

Some available data for the ecotoxicity of REEs to freshwater fish is shown in Table 3.12. of Appendix A.

The embryotoxicity tests revealed that La^{3+} and Yb^{3+} retarded the zebrafish embryos hatching (0.01 – 1.0 mmol L^{-1}), reduced the body length of larvae, and killed the larvae (Table 3.12. of Appendix A, Pagano et al., 2019). Heavy rare earth Yb was more toxic than light rare earth La. Compared to heavy metal pollutants such as Cu and Cd, La and Yb caused lesser acute toxicity to zebrafish embryos (Cui et al., 2012).

Ng et al. (2012) reviewed the aquatic effects of lanthanides and presented detailed information on the acute and chronic toxicity of lanthanides to freshwater organisms (Table 3.13. and Table 3.14. of Appendix A).

Reported REEs toxicity values for freshwater algae, cladoceran (*Daphnia magna, Daphnia carinata, Ceriodaphnia dubia*) were 450 – 4,400 µg REE L^{-1}; 43.2 – 24,000 µg REE L^{-1}, and 44 – 4,400 µg REE L^{-1}, respectively in the water of all hardness and all forms of REEs (dissolved / total) (Table 3.13., Appendix A). Acute toxicity of total REEs to cladocerans in the freshwater environment (water hardness: 210 mg L^{-1} as $CaCO_3$) is ranked as Nd > Gd > Sm > Pr ~ Dy > Ce > La (Table 3.13. of Appendix A). Toxicity of lanthanum to algae can be indirectly associated with the removal of phosphate from the growth medium through the formation of $LaPO_4$ (Stauber 2000). The shrimp (*T. platyurus)* is the most resistant to lanthanide (Table 3.13. of Appendix A, Appendix A).

An increase of water hardness from 40 - 48 mg·L^{-1} to 210 mg·L^{-1} as $CaCO_3$ reduced the toxicity of total REEs to the cladocerans by five folds (Table 3.14. of Appendix A). When water hardness increased from 18 – 124 mg·L^{-1} as $CaCO_3$, toxicity values increased from 0.01 – 191 µg REE L^{-1} to 278 – 1665 µg REE ·L^{-1} in *Hyallela azteca.* Chronic toxicity (7 d, LC_{50}) of each dissolved REE to *H. azteca* (Borgmann et al. 2005) is ranked as the follow: Tm > La > Nd > Lu > Ce > Pr > Sm > Tb > Yb > Eu > Ho > Gd > Dy > Er.

Data on the bioaccumulation of REEs in freshwater organisms is shown in Table 3.15. of Appendix A. The bioconcentration factor (BCF) of REEs in freshwater fish as common carp (*Cyprinus carpio)*, is high in organs involved in metabolism and detoxification, intermediate in gill, and low in muscle and skeleton (Tu et al. 1994; Sun et al. 1996) (Table 3.15., Appendix A). As reported by Weltje et al. (2002b), BCF of REEs (La – Lu) in freshwater plants and mollusc is between 10,000 and 100,000 L· kg^{-1} dry weight (duckweed Lemna minor: 10,000; pondweed Potamogeton pectinatus: 5,000 – 300,000; snail soft tissue: 5,000 – 200,000; bivalve soft tissue: 3,000 -30,000).

3.3 Threshold concentrations of REEs

Background and threshold concentrations of REEs in drinking water

Chronic exposure studies were carried out with mice for Sc and Y (Schroeder and Mitchener 1971). The rate of administration via drinking water was 0.5 mg kg^{-1} body weight per day. As a consequence of the exposure growth was retarded, and body weight diminished. This dose level may be considered as a Lowest Observed Adverse Effect Level (LOAEL). From these LOAEL's, indicative tolerable daily intake values (ITDI's) for humans were calculated by division with a factor of 1000 (a factor of 10 for intra-species variability insensitivity, a factor of 10 for interspecies extrapolation, and a factor of 10 for extrapolation from LOAEL). According to the procedure mentioned above, indicative admissible drinking water concentrations (IADWCs) for drinking water were calculated for Sc and Y, resulting in IADWC's of 2 mg L^{-1}. Threshold values of 2 µg L^{-1} for each individual REEs are thus recommended for drinking water in the Netherlands (de Boer et al., 1996). For La, Ce, Tb and Yb, no literature data is available from which IADWC values could directly be derived. Thus, the IADWCs for these four elements were based on the similar toxicological behavior as mentioned above, wherefore the IADWC's for these elements, as for Sr and Y, were also 2 mg L^{-1}.

In Dutch drinking water, REEs in the dissolved form were observed up to concentration levels of 30 mg L^{-1}. REEs were only present in the water of phreatic winnings of moderate depths. In surface water (rivers Rhine and Meuse, Lake I Jsselmeer), REEs were present up to a concentration level of 1 mg L^{-1} (de Boer et al., 1996).

Threshold concentrations of REEs in food

In 1991, China developed standards to limit total Rare earth oxides (REOs) in cereals and vegetables. In cereals and fresh vegetables, the national limit standard for REOs is 2.0 and 0.7 mg kg^{-1}, respectively (Jiang et al., 2012). No information has been found on what criteria were followed in setting up the standard limiting values.

Threshold concentrations of REEs in soil

A critical threshold concentration of a soil content of 30 mg kg^{-1} in dry weight was determined for REEs (Tang et al., 2004; Redling 2006). However, this value seems to be much lower than the background concentrations in China and the world (see Table 3.2. of Appendix A).

Concerning environmentalquality criteria, the threshold concentration of La was suggested to be 42 mg kg^{-1} and 83 mg kg^{-1} in red soil and paddy soil, respectively, based on a 10% yield decrease in the rice pot experiments.

Water and sediment quality criteria of REEs

Herrmann et al. (2016) derived freshwater and sediment quality-limiting values for La by applying NOEC values based on measured concentrations. By compiling PNEC data for water and sediment and multiplying a factor of 10, it results in a threshold concentration of 4 µg L^{-1}. Table 3.16. shows literature data on calculated freshwater & sediment quality for La.

Table 3.16. Calculation of quality targets by the added-risk approach for lanthanum (after Herrmann et al., 2016)

Matrix	Background concentrations (Europe)	MPA or PNEC	MPC
Freshwater (µg L^{-1})	<0.002	4	4
Sediment (mg kg^{-1})	5	5	

PNEC = predicted no effect concentration. MPA = maximum permissible additions (maximum permissible criteria). MPC = maximum permissible concentration.

Emission standards of pollutants for REE mining industry

In relation to a REEs project in Canada, effluent quality criteria guidelines were established (Avalon, 2013). The quality criteria were calculated for all the REEs, based on existing toxicity data from the United States Environmental Protection Agency (US EPA) ECOTOX database and the report by Ng et al. (2012). According to the Avalon (2013) and Ng et al. (2012), the criteria were calculated by applying a safety factor (SF) of 10 to the LC_{50} value at a water hardness at 18 mg L^{-1}. Avalon (2013) considered this method conservative as the toxicity decreases strongly with increasing water hardness. For example, is the LC_{50} level of 651 µg L^{-1} at a more realistic water hardness of 124 mg L^{-1} for Ce compared to 32 µgL^{-1} at a water hardness at 18 mg L^{-1}. The LC_{50} con

It is important to note that the safety factor (SF) 10 most likely does not take into account the number of test species and the quality of the toxicity tests, meaning that in some occasions the SF probably has to be 1000, while in other occasions it has to be less than 10. Besides, the hardness of the water contributes a lot with respect to the toxicity of the elements. Normally, the toxicity decreases with increased hardness. Since the hardness of water in Greenland primarily is zero, this needs to be considered when establishing water quality criteria in Greenland (Jeppesen 1987).

Table 3.17. Proposed water quality criteria and comparison with predicted tailings concentrations for a rare earth metals project in Canada (Nechalacho) (Avalon, 2013).

Element	LC_{50} Concentration ($\mu g\ L^{-1}$) [at hardness 18 mg L^{-1} $CaCO_3$]	Aquatic Organism Affected	Proposed Water Quality Objective (LC 50/10) ($\mu g\ L^{-1}$)	Predicted Final Tailings Concentration ($\mu g\ L^{-1}$)
Cerium	32	*Hyalella Azteca* (amphipod crustacean)	3.2	< 0.7
Dysprosium	162	*Hyalella Azteca*	16.2	0.048
Erbium	191	*Hyalella Azteca*	19.1	0.026
Europium	112	*Hyalella Azteca*	11.2	0.023
Gallium	>1000	*Hyalella Azteca*	>100	0.13
Gadolinium	150	*Hyalella Azteca*	15	0.05
Hafnium	>1000	*Hyalella Azteca*	100	0.263
Holmium	143	*Hyalella Azteca*	14.3	0.022
Lanthanum	18	*Hyalella Azteca*	1.8	0.21
		Oncorhynchus mykiss (Rainbow trout)	2.0	
Lutetium	29	*Hyalella Azteca*	2.9	0.017
Niobium	26	*Hyalella Azteca*	2.6	0.126
Neodymium	55	*Hyalella Azteca*	5.5	0.17
Praseodymium	35	*Hyalella Azteca*	3.5	0.06
Scandium	29	*Hyalella Azteca*	2.9	1.22
Samarium	74	*Hyalella Azteca*	7.4	0.05
Tantalum	2	*Hyalella Azteca*	0.2	0.032
Terbium	84	*Hyalella Azteca*	8.4	0.025
Thulium	0.01[a]	*Hyalella Azteca*	0.001[a]	0.020
Ytterbium	69	*Hyalella Azteca*	6.9	0.024
Zirconium	>1000	*Hyalella Azteca*	100	0.44

[a]The LC_{50} level at a more realistic water hardness of 124 mg L^{-1} is 739 $\mu g\ L^{-1}$; the corresponding water quality objective at that level would be 74.0 $\mu g\ L^{-1}$.

3.4 Risk to humans

Figure 1 of Appendix A presents the use of REEs and their health risk. Figure 1, Table 3.18., and Table 3.19. of Appendix A include information on the toxicity of REEs to humans.

Geographical studies have been conducted on residents in REEs mining districts, reporting on the association between REEs residential exposures and adverse health effects. The studies have shown REEs bioaccumulation in scalp hair, excess REEs urine levels and defective gene expression (Pagano et al., 2019). Liang et al. (2014) reported data up to 8.9 $\mu g\ g^{-1}$ of REEs in the hair for mining workers and 4.7 $\mu g\ g^{-1}$ REEs for farmers near the mining areas. High levels of REEs were detected in hair of children living in a REEs mining area in China.

It has been proved that an oral intake of 3 g of lanthanum carbonate per person per day for four years has no toxic effects (Joy et al., 2003; Locatelli et al., 2003; Harrison and Scott, 2004). A few reports show that REEs exhibit hepatotoxic and neurotoxic effects (Basu et al. 1982; Pałasz and Czekaj 2000). The residents in the Chinese REEs mining and processing regions have been investigated in a set of studies assessing scalp hair REEs accumulation as a function of distance from mining sites and considering gender and age (Pagano et al., 2019). Two studies reported excess hypertension risk among housewives residing in Shanxi Province due to REEs hair levels and REEs-containing

indoor air pollution (Table 3.18. of Appendix A). Table 3.19. of Appendix A shows that Ce and La compounds may affect oxidative stress and gene regulation for human cells.

Occupational exposure to REEs poses health risks to miners and workers working in REEs production (Pagano et al. 2019). At least three reports indicate health-related injuries to workers exposed to REEs (Chen et al., 2005; Liu et al., 2015; Li et al., 2016). A summary of working related REE exposures is shown in Table 3.20.

Table 3.20. Working - REE exposures – case- studies and case-control studies (after Pagano et al., 2019).

Occupation	Main Effects/Endpoints	Reference
Movie projectionist	Pneumoconiosis; interstitial lung disease	Sabbioni et al., 1982; Vocaturo et al., 1983; Porru et al., 2001; Waring and Watling. 1990
Exposure to carbon arc lamps in photoengraving laboratory	Pneumoconiosis; comparison of REE lung levels with 11 non-exposed workers	Pairon et al., 1995; Dufresne et al., 1994
Glass polisher	Dendriform pulmonary ossification	Yoon et al., 2005
Manufacturing Ce and La oxide	Excess Ce and La urine levels	Li et al., 2016
REE miners	Excess REE hair levels and dysregulation of protein expression	Liu et al., 2015
e-waste processing	Decreased hemoglobin concentration in REE-exposed	Henríquez-Hernández et al., 2017

3.5 Summary – Rare Earth Elements (REE)

The toxicity of REEs decreases with the increase of the atomic number and depends on e.g. the route of entry into the body, chemical speciation of REEs, non-human biota, etc. The background concentration of REEs decrease from freshwater-, to marine- and terrestrial biota (including birds).

The biological effects of REE on non-human biota may include mammalian organ disease, cytogenetic and neurological damage, and growth inhibition. To date, no documentation is found concerning the carcinogenicity of REEs in either non-human biota or humans. Concerning human health, most studies focus on Ce and La, with very limited data available for the other REE elements.

The REEs concentration in soil surface can be up to a couple of hundred mg kg^{-1}, whilst human activities can increase REE levels in the soil for more than ten folds. To our knowledge, no ecotoxicity data are available for REEs effect on terrestrial animals such as ptarmigans, sheep, beef cattle, reindeer and muskox. Very limited data are available for hares / rabbits. Limited research data are available on REEs toxicity to aquatic organisms. Most studies on bioaccumulation of REEs in aquatic organisms contain only three or four REEs. Moreover, large uncertainty exists regarding EC_{50} data from various ecotoxicity studies. Up-to-date, available data for marine organisms are even more scarce than for freshwater. The sensitivity between REEs toxicity to freshwater- and marine organisms has been associated with variation between species rather than to exposure to REEs in different exposure matrices. Scarce information is available for the REEs effects on seaweed and zooplankton.

Literature studies show that higher levels of REEs can be accumulated in seaweed than in terrestrial plants and that seaweed bioaccumulate REEs from seawater. There are only a few studies available regarding the levels of REEs in marine fish (e.g., 0.2 mg kg^{-1} ∑REE) and bivalves (e.g., 0.1 mg kg^{-1} ∑REE). Marine fish and bivalves samples can accumulate REEs several orders of magnitude higher than terrestrial samples.

No guideline values are available worldwide for either ∑REE, or REEs. Only fragmental data are available for quality criteria for REEs in water and sediment.

4 Naturally Occurring Radionuclides (NORs)

NORs of concern associated with mining projects in Greenland include uranium (U), thorium (Th), radium (^{226}Ra, ^{228}Ra), radon and thoron (^{222}Rn, ^{220}Rn), lead (^{210}Pb) and polonium (^{210}Po). As a result of the mining projects, the naturally occurring radionuclides (NORs) may be dispersed into the environment via air (e.g. dust, water droplets dispersion from the tailings facilities) and water (e.g. proposed liquid discharges, tailings facilities, waste rock depot, etc.). The units for radioactive elements are given in activity concentration as Bq g^{-1}.

4.1 Sources of NORs into the environment

There are two main contributors to natural radiation exposure: cosmic rays that come from outer space and the Sun's surface and radionuclides that originated in the earth's crust (Table 4.1. of Appendix B, UNSCEAR 2000). Radionuclides are present everywhere in the environment, including soil, sediments, building materials, air, water, foods and in the human body. In the environment, natural radiation varies widely, depending on the location. Cosmic rays are for example more intense at higher altitudes, while uranium and thorium concentrations in soils are elevated in localized areas. Natural radiation sources can also vary as a result of human activities. Particulary, natural radionuclides can be released to the environment from mining and mineral processing of uranium, extraction of rare earth elements, oil and gas industry, etc. Activity concentrations of naturally occurring radionuclides (NORs) in the Arctic environment ecosystems are given in Tables 4.2. and 4.3. of Appendix B (EPIC 2001).

4.2 Effects and risks to biota due to exposure to radiation

Emission of NORs to the atmosphere, terrestrial and aquatic environment from mining and milling is important to control to avoid unwanted environmental impacts (Hansen et al., 2017). Environmental contamination may occur, and as a result, non-human biota may be exposed to radiological, chemical, and physical stressors through ingestion (food or water), inhalation, cell membrane-mediated uptake, cutaneous absorption and biotic uptake / trophic transfer (Hansen et al. 2017).

The radiation effects in biota (Table 4.4.) depend on differences in exposure, biota recovering capacity, type of reproduction, sensitivity at critical periods of ontogenesis, duration of lifespan, the season of exposure and other factors. Tables 4.5., 4.6., 4.7., 4.8., and 4.9. of Appendix B includes data for the Arctic environment on radiation effects on terrestrial animals (Russian / FSU data) (EPIC, 2011, ANNEX A. EPIC Database).

Table 4.4. Radiation effects in biota (UNSCEAR 2000; EPIC 2001).

Radiation effects	
Morbidity (MB)	It worsens organisms' physiological characteristics, effects on the immune system, blood system, nervous system, etc.
Reproduction (REPR)	Negative changes in fertility and fecundity, resulting in reduced reproductive success.
Mortality (MT)	It shortens lifetime as a result of combined effects on different organs and tissues of the organism.
Cytogenetic effects (CYT)	Radiation effects on the cellular level.
Ecological effects (ECOL)	Changes in biodiversity, ecological successions, predator-prey relationships.
Stimulation effects (STIM)	Activation of defense mechanisms in organisms, but without their exhaustion.
Adaptation to radiation effects (AD)	Adaptation of some wild organisms to the conditions of chronic irradiation.
No effects (NO)	

Dose effect relationship for Arctic biota provide a scale of severity of radiation effects at different chronic radiation exposure levels (Table 4.10.; EPIC 2001). In organisms, radionuclides' behavior follows the biochemical pathways of stable analogous elements (Table 4.11., UNSCEAR 2016, IAEA 2014, 2017). Non-uniform distribution of radionuclides in the body results in increased exposure of individual organs, leading to organ dysfunction. Species also exhibit radiosensitivity differences. Particulary species are more vulnerable to radiation exposure. For example, the adult lethal doses of acute exposure are ranging from few Grays (mammals) to thousands of Grays (insects, microorganisms) (UNSCEAR 2000, Sazykina, T.G., 2017). The radiosensitivity of organisms also varies considerably at different stages of life cycle development. For example, few Grays of radiation exposure can cause mortality at the early development stages of insects, whereas the adult insects of the same species are very radioresistant (EPIC 2001).

Table 4.10. Dose effects relationships for Arctic biota (EPIC 2001).

Dose (µGy/h)	**Effects**
4.17E-02 - 4.17E-01	Natural radiation background for Arctic biota
4.17E+00 - 2.08E+01	Minor cytogenetic effects. Stimulation of the most sensitive species
2.08E+01 - 4.17E+01	Threshold for minor effects on morbidity in sensitive vertebrate animals
8.33E+01 - 2.08E+02	Threshold for effects on reproductive organs of vertebrate animals, decrease of embryo's survival
2.08E+02 - 4.17E+02	Threshold for life shortening of vertebrate animals. Threshold for effects in invertebrate animals
4.17E+02 - 4.17E+03	Life shortening of vertebrate animals; chronic radiation sickness
4.17E+03 - 4.17E+04	Acute radiation sickness of vertebrate animals. Damage to eggs and larva of invertebrate animals
>4.17E+04	Acute radiation sickness of vertebrate animals. Damage to eggs and larva of invertebrate animals

Table 4.11. Bioaccumulation of radionuclides in organs and tissues.

Radionuclide	**Tissue & Organ**	**References**
^{238}U	Kidney, bones, lungs	UNSCEAR 2016,
^{230,232}Thorium	Bones	IAEA 2014,
^{226,228}Radium	Bones	IAEA 2017
^{222}Radon	Lungs	
^{210}Lead	Bones	
^{210}Polonium	Spleen, kidney, liver, and lymph nodes	

Arctic low temperatures and extreme seasonal variations in light can cause environmental stress to organisms and make them more vulnerable to contaminants (AMAP 1998). Due to low temperatures, especially during the winter, radiation effects in poikilotherm (cold-blooded) Arctic organisms are expected to occur more slowly than organisms from temperate environments. The repair of radiation damage in cells and tissues is not effective at low Arctic temperatures (EPIC 2001).

The development of embryos and young poikilothermic Arctic organisms occurs slowly. Hence, they are prone to receive a higher dose during their radiosensitive life stage than similar organisms from temperate environments. Arctic species are long-lived and exhibit deferred age of first reproduction in females compared with similar species in a temperate climate. The high content of fat or fatty oils in Arctic animal tissues that prevents their body from freezing and provide support during periods of starvation may increase animal radiosensitivity since chemical products of fats peroxidation via irradiation are toxic for organisms (EPIC 2001).

4.3 Naturally Occurring Radionuclides (NORs)

Uranium (U)

Uranium (U) is a heavy naturally occurring element. It decays by emitting an alpha (α) particle from its nucleus. The bioavailability and toxicity of U depend on its speciation and several environmental key factors. U occurs naturally in the +2, +3, +4, +5, or +6 valence states, but it is most common in a hexavalent form in the environment (UNSCEAR 2016: Hansen et al., 2017). In oxic natural waters, U is present mainly in the U(VI) state (oxidized), as uranyl ion UO_2^{2+} (free ion). Conditions that favor the formation of the free ion UO_2^{2+} include low pH and low concentrations of natural organic matter.

Metal toxicity in aquatic systems is better correlated with the concentration of free ion than with total metal concentration. The speciation of U at given conditions (e.g. hardness, pH, temperature, etc.) is more indicative of toxicity than its total concentration. Speciation-based toxicity studies conclude that UO_2^{2+} is the toxic chemical species (UNSCEAR 2016; Hansen et al., 2017).

U in soil

In soils, the concentration of U varies with location and geology. The average background ^{238}U activity concentration in soil is 15 Bq kg^{-1} with a typical activity range of 10–50 Bq kg^{-1} (UNSCEAR 2016).

U partitioning depends on soil/sediment pH, Eh, oxygen content, Kd, redox state, organic content, temperature, etc. (Zhang and Brady, 2002; UNSCEAR 2016). Adsorption of U by soils and single-mineral phases is generally low at pH <3, increases rapidly with increasing pH (pH = 3 to 5), reaches a maximum in adsorption in the pH range from 5 to 8, then decreases with increasing pH at pH >8 (the U.S. EPA 2008; Hansen et al., 2017). This trend is related to the pH-dependent surface charge properties of the soil minerals and the complex aqueous behaviour of dissolved U (uranium (VI)).

U in air

Background U concentration in air is typically low, varying from location to location. From the atmosphere, uranium can deposit on soil, plants and water via dry or wet deposition.

Average worldiwide U concentration in the air were 1 µBq m^{-3} (UNSCEAR 2016). Reported airborne concentrations of ^{238}U were 0.3 µBq m^{-3} (near the Argonne National Laboratory, Illinois, United States), 0.25 µBq m^{-3} (Tokyo), 1.25 µBq m^{-3} (southern Ontario) (UNSCEAR 2016).

U in water

U is present in surface water, groundwater, drinking water and drilled water at variable concentrations depending on local geology. ^{238}U average concentratios in water worldwide are 15 mBq L^{-1} for groundwater, 12.4 mBq L^{-1} for surface water, 12.4 mBq L^{-1} for public water supplies and 6.5 mBq L^{-1} for bottled mineral water (UNSCEAR 2016). In seawater, ^{238}U average cocentration is 42 mBq L^{-1}, often bound by ligands or associated with suspended particles. In Canada, the mean ^{238}U concentrations in surface water and groundwater (some treated) supplies were about 50 mBq L^{-1} (southern-central British Colombia), 124 mBq L^{-1} (south-eastern Manitoba), 65 mBq L^{-1} (Kitigan Zibi First Nation community in Quebec) and 5 mBq L^{-1} (Ontario) (UNSCEAR 2016). Mean ^{238}U in Canada's drinking water varies with location and local geology, from 5 to 750 mBq L^{-1} (UNSCEAR 2016).

U in plants and animals

The most available chemical forms of U in plants are phosphate, carbonate, sulphate or citrate. U concentrates mainly in plant roots.

In aquatic biota such as crustaceans, molluscs and fish, the bioconcentration factor from water is low (UNSCEAR 2016).

U concentration factors (CFs) for fish vary from 0.01 to 20 (UNSCEAR 2016). The U CFs values depend on e.g. the uranium concentration in the water, behaviour of the organisms (pelagic species accumulate approximately 10 times less than benthic species) and the tissues considered (bone 200–8,000 and kidneys > liver and gills > muscles 1.5–24 > digestive system > gonads) (UNSCEAR 2016).

The Codex Alimentarius (Codex 2013) includes guideline levels applied to radionuclides contained in food, destined for human consumption and traded internationally. These guideline levels are based on an intervention exemption level of 1 mSv in a year. A value of 100 Bq kg^{-1} is given for ^{235}U. However, this guideline level excludes ^{238}U (Codex 2013).

U toxicity to non-human biota

U is a highly toxic heavy metal. Natural U consists of three isotopes — 234, 235, and 238 — with half-lives ranging from hundreds of millions to billions of years. As a result of these long half-lives, the specific activities of the U isotopes are very low. Thus, U chemical toxicity predominates over its radiotoxicity (UNSCEAR 2016; Hansen et al., 2017). The activity-effect relationships for U in water are given in Table 4.12. (EPIC 2001).

Table 4.12. Activity-effects relationships for ^{238}U in aquatic organisms (EPIC 2001).

^{238}U in water (Bq L^{-1})	Effect on aquatic biota
0.6	No observable effects.
6.2	About 10% decrease in growth of microalgae.
12.3	Decrease in growth of microalgae by 50-60%. Some decrease in Daphnia reproduction.
61.5	Signs of acute toxicity in Daphnia, decrease of reproduction. Degeneration (sex transformation) in gonads of fish.
123	Decrease of bacterial growth, decrease of phytoplankton biomass; Total cessation of Daphnia reproduction. Degeneration in gonads of fish.
307.5	Rapid regeneration (sex transformation) in gonads of adult fish.
1230	Acute lethal concentration for Daphnia, fish, cessasion of phytoplankton growth, oppression of bacterial mineralization processes.

U risk to humans

The below information is from Kreuzer et al., 2011 (Wismut Cohort, 2011) and UNSCEAR 2016.

Chemical and radiological toxicities of U depend on its chemical form and exposure route (UNSCEAR 2016). Intake of U into the human body is primary via ingestion and inhalation, with transfer through the skin as a minor route.

The general public intake U mainly through ingestion of water or foodstuffs. Occupational exposure arises primarily through inhalation of dust containing U or following injury.

In blood, U is present mainly as uranyl ions complexed to proteins (e.g. transferrin and albumin) or bicarbonate anions. In the human body, U distributes in the skeleton, kidneys, and soft tissues. Human and animal data show that urinary excretion is rapid. About two-thirds of U in the blood is excreted in the first 24 hours and 10% over the next five days. Most of the remaining U is excreted over a few months. A small percentage of the U may be retained for years (UNSCEAR 2016).

The behaviour of U, its distribution, and its clearance in the lungs depend on U compounds' solubility. Soluble compounds such as uranyl fluoride, uranium tetrachloride, uranyl nitrate are less toxic to the lungs but more toxic to distal organs due to easier absorption of the uranium from the lungs into the blood and systemic transport. Insoluble uranium particles with a longer residence time in the lung confer more risk than the more soluble particles with a shorter residence time (UNSCEAR 2016).

U miners and millers

Miners are exposed to internal contamination due to inhalation of long-lived radionuclides (LLR) from uranium ore dust, inhalation of radon and its progeny and to external gamma radiation. The main source of radiological exposure of uranium miners is radon and its decay products.

Many epidemiological studies of uranium miners show an association of accumulated exposure to radon and its decay products with lung cancer risk. Uranium millers are subjected to inhalation of radon and radon decay products, inhalation of LLR from uranium ore dust and external exposure to gamma radiation.

Potential health effects from uranium exposure are summarized in Kreuzer et al., 2011 (Wismut Cohort, 2011) and UNSCEAR 2016 and includes, but not to be limited to:

Non-cancer diseases include injury to the kidneys, bone metabolism, respiratory diseases, liver, brain, reproduction, endocrine system and metabolism, immune and haematopoietic system, cardiovascular system, effects on DNA, cytogenetic damage, tumorigenic potential, lethal effects.

Cancer diseases as respiratory cancers (lung, laryngeal), urological cancers (kidney, bladder, prostate), digestive cancers (stomach, intestine, liver, pancreas), cancers of the brain and central nervous system, lymph-hematopoietic malignancies (leukemia, lymphoma, multiple myeloma).

Thorium (Th)

Thorium (Th) is a lithophilic element, highly abundant in the earth crust, in about three times the concentrations of uranium. Lithophile element remains on or closes to the surface because it combines readily with oxygen, forming compounds that do not sink into the Earth's core. In nature, thorium occurs in four isotopic forms, thorium-228, thorium-230, thorium-232, and thorium-234. Thorium-228 is the decay product of thorium-232, and thorium-234 and thorium-230 are decay products of uranium-238. The half-life of Th^{232} is 14 billion years. In solution, thorium occurs in 4^{+} oxidation state and may be strongly adsorbed by ion exchangers. Geochemical behaviour of Th is similar to rare earth elements (e.g. cerium), zirconium and uranium (Hansen et al., 2017).

Th in air

In the atmosphere, Th, mainly as ThO_2, occurs from natural and anthropogenic sources. Natural thorium sources in the air include windblown terrestrial dust and volcanic eruptions. Anthropogenic thorium sources in the air include mining, milling, tin processing, phosphate fertilizer production, coal industry, etc. Depending on the meteorological conditions, the particle size and density and the chemical form of thorium particles, thorium is removed from the atmosphere by wet and dry deposition. Small aerodynamic diameters thorium particles (< 10-micron aerodynamic diameter) may travel long distances from their emission source (Hansen et al., 2017).

Th in water

Sources of Th to surface waters and groundwater may include acidic leaching of uranium tailings and liquid effluent discharges from uranium and thorium mining, tin processing, phosphate rock processing, and phosphate fertilizer production. The concentrations of dissolved thorium in water with high pH (>8) are expected to be very low, and the concentration may increase with decreasing pH. Soluble thorium ions in water hydrolyze at pH above three to form $Th(OH)_4$ precipitate or hydroxy complexes, e.g. $Th(OH)_2^{+2}$, $Th_2(0H)_2^{+6}$, $Th_3(OH)_5^{+7}$. The hydroxy complexes may be adsorbed on particulate matter, e.g. goethite. Th has a strong affinity to humic acid and other organic ligands. Adsorption of thorium to suspended particles or sediments depends on the particle size; e.g. high adsorption and subsequent thorium removal from the aqueous phase are obserced with decreasing particles size. The reported concentration of dissolved Th in an alkaline lake was up to 4.9 dpm L^{-1} (dpm= disintegration per minute, 1 Bq = 60 dpm) while about $1.3x10^{-5}$ dpm L^{-1} in seawater (Hansen et al., 2017).

Th in soil

Worldwide Th average concentration in soil were 6 µg g^{-1} (Kabata-Pendias and Pendias, 2000). Th content of the soil depends on soil matrix. For example, concentration in soil increases with an increase in clay content, and its mobility is low. This is possibly due to clay's high sorption capacity (Harmsen and de Haan, 1980). Depending on the soil sorption characteristics (e.g. low sorption) and soil chemistry, thorium may form soluble complexes with carbonate, humic materials or other ligands, thorium mobility in these soils will increase.

Th in non-human biota

Literature results show Th soil to aboveground parts of the plant transfer factor values less than 0.01, indicating that it does not bio-concentrate in aboveground parts of the plants (Garten, 1978). Concentrations of Th in plants such as lichens, mosses range from 0.3 to 1.7 mg kg^{-1} depending on the species and sampling location (Shtangeeva, manus, unpublished). Under low soil pH, plants grown near uranium tailings had a plant / soil concentration ratio of about 3 (Ibrahim and Whicker, 1988). Plant roots adsorb thorium from the soil, but the transport from the root to the aboveground parts of the plant is low, as indicated by l00-fold higher concentrations of all three isotopes (thorium-228, thorium-230 and thorium-232) in the root compared to the aboveground parts (Sheepard 1980, Shtangeeva, manus, unpublished).

Bio-concentration of thorium can be significant in lower trophic level aquatic non-human biota (NHB), but thorium does not biomagnify as the trophic level increases. Most thorium body burden in fish was found in the gastrointestinal tract (Poston, 1982).

Th in human body

Th can enter the human body via inhalation of dust and / or water and food ingestion. Some inhaled thorium will remain in the lungs, and some will leave the body via feces and urine within days. Most of the ingested thorium from water or food will leave the body in the feces. The small amount of thorium left in the body may enter the bloodstream and deposit in the bones, where it may remain for many years (ATSDR 1990).

Inhaled thorium in dust can cause an increased risk of developing lung disease, including lung cancer and / or pancreatic cancer. Liver disease and some types of cancer have been found in people injected in the past with thorium to take special X-rays. Bone cancer is also a potential health effect through the storage of thorium in the bone (ATSDR, 1990).

Radium (Ra)

In the environment, there are four naturally occurring radium isotopes: ^{226}Ra (^{238}U series), ^{223}Ra (^{235}U series) and ^{224}Ra and ^{228}Ra (^{232}Th series). Radium-226 (half-life = 1600 years) and ^{224}Ra (half-life = 3.66 days) undergo decay by alpha particle emission. Radium-228 (half-life = 5.75 years) decays through beta emission to ^{228}Th. Radium is an alkaline earth element and occurs in the environment in the $Ra2^{+}$ oxidation state (IAEA 2014).

Ra in roks and soil

Observed ^{226}Ra concentrations in various rocks were 15–2220 Bq kg^{-1} (shale), 629–1040 Bq kg^{-1} (bitumen), 48–137 Bq kg^{-1} and 148–1480 Bq kg^{-1} (volcanic and phosphate rocks), 0.037–185 Bq kg^{-1} (granites), 55 Bq kg^{-1} (clay rocks), 7–55 Bq kg^{-1} (sandstone), 9.2–15 Bq kg^{-1} (sedimentary rocks), 5–18 Bq kg^{-1} (lime), and

26–30 Bq kg^{-1} (carbonate). In soil, Ra behaves similarly to Ca, Sr, and Ba. Reported average worldwide ^{226}Ra concentration in soil were 32 Bq kg^{-1} (UNSCEAR 2008). ^{226}Ra concentrations in soil vary across a wide range and depend on soil type, soil inhomogeneity, U and Th concentrations, and Ra retention in the topsoil. For example, Switzerland (10–900 Bq kg^{-1}), China (2–440 Bq kg^{-1}), Ramsar, Iran (80–50 000 Bq kg^{-1}) and The Komi Republic of the Russian Federation (259–71000 Bq kg^{-1}). In areas with low natural background, ^{226}Ra concentrations in soil vary from 3.7 to 126 Bq kg^{-1} (IAEA 2014).

Radium is known to be most strongly absorbed by ion exchange on clay minerals, organic materials, and mineral oxides, especially in near neutral and alkaline pH conditions (IAEA 2014).

Ra in groundwater

Sources of radium in groundwater include natural and industry sources such as mining, oil and gas, geothermal energy production, etc. ^{226}Ra and ^{228}Ra activity concentrations have a wide range in groundwater, and depend on e.g. source characteristics in the study region and the ionic strength of the groundwater. ^{226}Ra concentrations in groundwater in Helsinki varied from <4-9470 mBq L^{-1} for ^{226}Ra and <18.5-570 mBq L^{-1} ^{228}Ra, while in thermal springs in Japan, a range of 1300-7840 mBq L^{-1} were reported (IAEA 2014). In the Western USA near exploitation of geothermal water a range of 1.5-55 500 mBq L^{-1} ^{226}Ra, while in Australia, Yeelirrie, proximal to uranium deposits a range of 18.5-33 400 mBq L^{-1} ^{226}Ra in groundwater were reported (IAEA 2014).

Ra in freshwater

Ra sources in freshwater include groundwater inflow, sediment resuspension, resolubilization of sediment-bound radionuclides and from the air through precipitation and particle deposition. In surface water radium can be found dissolved in a pH range of 3-10 (IAEA 2014). In sulphate-bearing waters, precipitation and dissolution of calcium, strontium, and barium sulphates may control the concentration of dissolved radium (IAEA 2014).

Background ^{226}Ra activity concentrations in river water range between 0.5 and 20 mBq L^{-1}, while in lakes ^{226}Ra activity concentrations are within a range of 0.5–15 mBq L^{-1} (IAEA 2014). Baseline ^{226}Ra values in surface waters around U mining / milling sites in northern Saskatchewan shows a range of 5.0 to 20 mBq L^{-1}, with values that fall between 5.0 and 3 000 mBq L^{-1} in mining areas (IAEA 2014). For example, ^{226}Ra is found in a concentration range of 7.4–222 mBq L^{-1} in river water in the vicinity of the U mining area at Pocos de Caldas, Brasilia (IAEA 2014).

Ra in seawater

^{226}Ra distributes along the open ocean profile and depends on its supply from continents and the seabed and its removal via radioactive decay and particle scavenging. In the surface ocean waters, ^{226}Ra vary from 0.12-0.26 mBq L^{-1}, while in Ocean deep water, 0.25-0.6 mBq L^{-1} (IAEA 2014). ^{226}Ra activity concentrations in estuaries waters are one order of magnitude higher than in the surface ocean. In the open ocean ^{228}Ra distribution is restricted to its sources, surface and near-bottom waters in contact with sediments (IAEA 2014).

Ra in air

In the air Ra occurs in resuspended soil particles (IAEA 2014). Based on the worldwide ^{226}Ra soil concentration of 32 Bq kg^{-1}, the air concentrations can be estimated to be 1.5 µBq m^{-3}(IAEA 2014). However, ^{226}Ra concentration in the

air vary with location, climate, etc. (IAEA 2014). IAEA (2014) reports a range of values for temperate environments of 0.6–32 μBq m-3 (e.g. United States of America, Germany and Poland, respectively).

Ra in terrestrial plants

Radium accumulates in plants via root uptake from soil and foliar uptake (e.g. uptake via plant leaves) from the atmospheric deposition. Literature studies show that the highest concentrations of Ra are found in the roots, followed by the stem and grain (IAEA 2014). In plants ^{226}Ra activity concentrations depend on its concentration and inhomogeneity in the soil, soil properties (e.g. organic matter content, pH, sulphate content, etc), plants' ability to accumulate Ra and environmental characteristics governing plants' external contamination. ^{226}Ra activity concentrations range from 0.01-0.9 Bq kg^{-1} dry weight in cereals and 0.7 Bq kg^{-1} dry weight in root vegetables in the UK to 9.5 Bq kg^{-1} dry weight in leguminous vegetables in Brazil (IAEA 2014). In Rio de Janeiro area black bean, cabbage, cauliflower, carrot and tomato ^{226}Ra activity concentrations range from 0.131 to 0.157 Bq kg^{-1} fresh weight and, in fruit species, potato, spinach, and watercress range from 0.01–0.04 Bq kg^{-1} fresh weight (IAEA 2014). A study on an area contaminated by U mine process effluents in Ontario, Canada, reported similar data with the highest Ra concentrations in roots and leafy vegetables and the lowest in fruits (IAEA 2014). Plants grown in high background areas have significantly higher ^{226}Ra concentrations compared to those from normal background areas. For example, a range of 33.5–2064 Bq kg^{-1} dry weight was reported for ^{226}Ra concentration in the grass for the Komi region in Northwestern Russia (IAEA 2014).

Ra in terrestrial animals

Ra transfers to animals mainly via ingestion of feed and water. Accumulation in animals depends on the concentration in food and water and seasonal variations in the diet patterns of animals and their position in the trophic chain. Accumulation of radium descend from insectivores (soil animals) > mice rodent > carnivores > ungulates (IAEA 2014). Partition studies in cattle from normal background area, show that radium distributes mainly in teeth and bone and to a lesser extent in soft tissues e.g. muscle (IAEA 2014). A similar distribution pattern was observed for Arctic animals such as caribou, reindeer, elk and moose, with the highest ^{226}Ra concentration in bone (3.2 -13.5 Bq kg^{-1} fresh weight) and the lowest (0.01-0.05 Bq kg^{-1} fresh weight) in soft tissues, e.g. muscle (IAEA 2014).

Ra in freshwater biota

Algae, mussels and some fish species (e.g. Northern Pike, Arctic grayling) may be used as indicators species for environmental Ra. High freshwater Ra concentrations equate to elevated Ra concentrations in biota. ^{226}Ra content in whole-body fish in uncontaminated surface water ranges from 0.68 to 1.12 Bq kg^{-1} fresh weight, while fish in surface waters affected by U milling, have ranges from 0.33 to 10.4 Bq kg^{-1} fresh weight with higher Ra in bone than muscle (IAEA 2014).

Ra in marine biota

Most of the available data is on ^{226}Ra in marine biota, with limited data on ^{228}Ra in marine biota (IAEA 2014). ^{226}Ra and ^{228}Ra accumulate in tissues with high calcium content (e.g. bones), while low activity concentrations are reported in soft tissues. In several seas around the world, ^{226}Ra activity concentrations ranged between 0.05 and 1.9 Bq kg^{-1} fresh weight in phytoplankton and multicellular marine algae, <0.1 to 0.3 Bq kg^{-1} fresh weight in filter

feeders, while in marine mammals and fish muscle the concentrations were below 1 Bq kg^{-1} fresh weight. In the North Atlantic Sea ^{226}Ra activity concentration in the muscle of plaice, halibut, cod and round nose grenadier ranged from 0.16 to 0.9 Bq kg^{-1} fresh weight (IAEA 2014).

Ra in drinking water and food

Water and food ingestion are the main pathways of radium transfer to humans. UNSCEAR (2000) assumes a worldwide reference concentration of ^{226}Ra of 0.5 mBq L^{-1} in drinking water.

In food products, UNSCEAR (2000) adopted the following worldwide ^{226}Ra values as reference concentrations: 5 Bq kg^{-1} in milk, 15 Bq kg^{-1} in meat, 80 Bq kg^{-1} in grain, 50 Bq kg^{-1} in leafy vegetables, 30 Bq kg^{-1} in root vegetables and 100 Bq kg^{-1} in fish.

Radon (Ra-222) and thoron (Rn-220)

Radon (Ra-222) and thoron (Rn-220) are radionuclides in gas form originating from the decay of radium-226 and thorium-232. Thoron has a short half-life of 55 seconds, whereas the half-life of radon is 3.8 days. It means that their radioactivity dissipates in a matter of few days if no radium-226 and thorium-232 are present. Radon is continually being formed in soil and released to air as a result of the long half-lives of uranium-238 and radium-226 and their abundance in the Earth's crust. Radon decays via alpha emission into short half-life radon daughters. The radon daughters are chemically reactive and attach almost immediately to aerosol particles in the atmosphere.

The radon concentration in soil is a function of a) radium concentration, b) soil moisture content, c) soil particle size and d) the rate of exchange of soil-entrapped air pockets with the atmosphere (Papp et al., 2010). The radon release rate from e.g. soil depends on its moisture content, porosity and meteorological conditions. If the moisture content is very low, the radon release is low due to the effect of re-adsorption of radon atoms on surfaces in the pores. If the moisture content increases slightly, the radon release increases up to certain moisture content, above which the release of radon decreases again owing to a decreasing diffusion rate in water-filled pores. High porosity of the material increases the diffusion rate. The diffusion rate and thereby the release rate of radon from the soil are influenced also by meteorological conditions such as rainfall, snowfall, freezing and variations in atmospheric pressure.

Groundwater in contact with sediment and rock containing radium will be a source of radon. In groundwater radon transportation is determined primarily by diffusion patterns and the water's mechanical flow. The solubility of radon in water is relatively low and with its short radioactive half-life, much of it will decay before being released from the groundwater.

Extensive study results on human health effects from exposure to radon are included in
Kreuzer et al., 2011 (Wismut Cohort, 2011); Kusiak, et al., 1993; UNSCEAR 2016.

Lead -210 (210Pb) and Polonium-210 (^{210}Po)

Pb-210 and Po-210 are products of the decay of uranium-238. ^{210}Po has a half-life of 138 days and emits alpha particles, which carry high energy (5.408 MeV). Industrial activities such as mining, oil and gas can enhance the activity concentration of ^{210}Po in the environment. Polonium-210 is also produced in milligram amounts in nuclear reactors by bombarding stable bismuth-209 with neutrons, resulting in bismuth-210 (^{210}Bi). ^{210}Bi decays with a half-life of five days to ^{210}Po via the emission of a beta particle. About 100 grams of ^{210}Po is believed to be produced in a nuclear reactor each year (NRC 2019).

In the environment polonium occurs in oxidation states of −2, +2, +4 and +6. Under oxic freshwater conditions polonium occurs in a Po^{4+} oxidation state, while Po^{2+} predominates under reducing seawater conditions (IAEA 2017). Polonium is highly surface reactive, and adsorbs onto mineral surfaces, particulates and colloids. Its partitioning may be controlled by its speciation, nature and concentration of particulate matter, dissolved organic matter, other ligands, salinity, etc. In rivers, ^{210}Po is associated with suspended particulate matter and bottom sediments, while in seawater with biological particles such as bacteria and microplankton. In contrast ^{210}Pb in seawater is associated with the inorganic particulate fraction such as clay minerals. Groundwater typically has less than 40 mBq / L, with a maximum value of 19 Bq / L ^{210}Po (IAEA 2017).

^{210}Pb and ^{210}Po in the atmosphere

^{210}Pb and ^{210}Po concentrations in the atmosphere depend on their sources. In the atmosphere, the main source of ^{210}Pb and ^{210}Po is radon-222 (^{222}Rn) exhalation from the soil. Additional sources include release from surface oceans, volcanic eruptions, resuspension of soil dust, biomass burning, and certain industries. Annual global fluxes for ^{210}Pb and ^{210}Po are estimated to be 3.64×10^{16} Bq and 3.82×10^{15} Bq, respectively (IAEA 2017). ^{210}Pb and ^{210}Po are removed from the atmosphere by wet and dry deposition (IAEA 2017).

^{210}Pb and ^{210}Po in soil

Pathways of ^{210}Pb and ^{210}Po in terrestrial environment is shown in Fig. 4.1. ^{210}Pb and ^{210}Po activity concentrations in undisturbed soils (i.e., undisturbed by industry, weathering, leaching or dissolution by groundwater, forest fires and volcanic activities, etc.) depends on the soil's type and chemistry, uranium concentration, soil depth profile and the addition of phosphate fertilizers, etc.

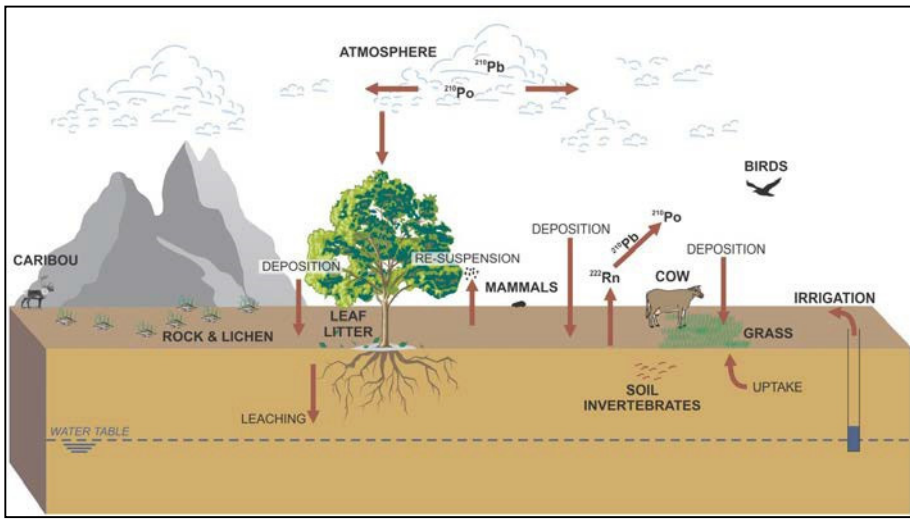

Figure. 4.1. Patwhays of ^{210}Pb and ^{210}Po in the environment. Source: IAEA 2017.

Worldwide background ^{210}Po concentrations in surface soil range from 8 to 220 Bq kg^{-1} dry weight while near industrial areas as uranium mines ^{210}Po concentrations in surface soil of 22000 Bq kg^{-1} dry weight were reported (Table 4.13. in Appendix B, IAEA 2017). A summary of available partition coefficients (K_d) (L kg^{-1}) values for lead and polonium for different types of soils are given in the IAEA report (Table 6.1 of IAEA 2017), with polonium Kd values lower than those for the lead.

^{210}Pb and ^{210}Po in plants

Accumulation of ^{210}Pb and ^{210}Po in plants depend on physical, chemical and biological processes such as uptake mechanism (e.g. the decay of ^{222}Rn accumulated in the plant, root uptake from the soil, and foliar uptake of ^{210}Pb and ^{210}Po deposited on the leaves), their concentration and chemical speciation in soils, type of soil and plant, the rate of deposition onto leaves, weathering (rain wash-off, exfoliation), etc. Reported activity concentrations of ^{210}Po in terrestrial plants from non-industrial areas were 1.1-30.0 Bq kg^{-1} dry weight (grass), 34 Bq kg^{-1} dry weight (blueberry) and 0.08 (vegetables) (Table 4.14. of Appendix B, Source: IAEA 2017). The highest observed 210Pb and 210Po activity concentrations (d.w.) were in stem, roots and leaves in blueberry, and roots, rhizome, stem and leaves (lingonberry). Reported 210Pb and 210Po activity concentrations in wild berries in Finland were 0.7-1.7 and 2.5-3.2 (Bq kg-1 dry weight) (Blueberry) and 1.4-3.2 and 2.2.-7.5 (Bq kg-1 dry weight (Lingonberry), respectively (IAEA 2017).

Transfer factors values (TFs) are an empirical measure of plant contamination under steady-state conditions. The highest observed transfer factors (TFs) values are 1.0 for polonium in pasture vegetation and 25 for lead in leafy vegetables (IAEA 2017).

^{210}Pb and ^{210}Po in animals

Transfer of polonium to animals is mainly via food and to a lesser extent water ingestion. In animals, ^{210}Pb is accumulated in bones, while ^{210}Po is distributed mainly in soft tissue such as the spleen, liver and kidneys. The ^{210}Po:^{210}Pb ratio exceeds unity in the soft tissues. However, this can change with time due to the continuous decay of ^{210}Pb, resulting in supported ^{210}Po and the difference in biological half-lives of these radionuclides in the different tissues (IAEA 2017).

^{210}Pb and ^{210}Po in groundwater, surface water and drinking water

Sources of ^{210}Po and ^{210}Pb in groundwater include rainwater, atmospheric deposition, their concentrations in soils, anthropogenic releases and decay of parent isotopes dissolved in the groundwater. ^{210}Po and ^{210}Pb are particle reactive under most groundwater conditions and associate to colloids and particles (e.g. clays and compounds of manganese and iron).

Activity concentrations of ^{210}Po of 16 Bq L^{-1} have been recorded in a well in Finland and up to 19 Bq L^{-1} in brines in Australia (IAEA 2017).

The sources of lead and polonium in surface waters include in situ decay of parent isotope, surface runoff, fluvial inputs, atmospheric deposition and anthropogenic inputs. In surface water ^{210}Po may be uptaken by biomass particulates and scavenged from the water column as these particulates settle to the bottom sediments. The cycling of iron and manganese influences ^{210}Po behaviour in the sediment as the sediment's redox conditions change (IAEA 2017).

^{210}Pb and ^{210}Po in marine water

In the marine environment ^{210}Pb and ^{210}Po originate from atmospheric deposition, in situ decay of soluble ^{226}Ra, ^{222}Rn exhaled from the seafloor, and river, sediment and anthropogenic discharges. In coastal waters, polonium and lead are associated with suspended particulate matter, and only minor fractions are in the dissolved phase. Adsorption onto particulates or uptake by phytoplankton and zooplankton remove ^{210}Pb and ^{210}Po from the water column. Some of the ^{210}Pb and ^{210}Po incorporated in the biogenic particulate materials in the surface water may settle on the seafloor adding ^{210}Pb and ^{210}Po to the sediment layer. Reported concentrations of dissolved ^{210}Po in coastal areas were 0.5 Bq m^{-3}, with the Po: Pb ratio near one. In suspended particulate matter, ^{210}Po concentrations are generally higher than ^{210}Pb and hence the the Po: Pb ratio is greater than 1. Dissolved ^{210}Pb occurs in the ocean's upper layer up to 2.5 Bq m^{-3}, while in deeper layers concentrations up to 1 Bq m^{-3} has been reported. Dissolved ^{210}Po in the upper layer is around 1 Bq m^{-3}, and while the concentrations increase in the mesopelagic layer, the concentrations decrease towards the abyssal sea-floor. The estimated ^{210}Po mean residence times in marine surface water is 6–12 months and around two years in the deep-sea layers (IAEA 2017).

^{210}Pb and ^{210}Po in marine and freshwater biota

Compared to terrestrial organisms relatively high concentrations of ^{210}Po have been reported in marine biota. ^{210}Po concentrations in marine biota are higher than those of ^{210}Pb with ^{210}Po:^{210}Pb ratios of 1–100. In fresh and marine biota ^{210}Po is distributed mainly in soft tissues as kidney, liver and other digestive system organs. In contrast, higher concentrations of ^{210}Pb have been measured in the shells of the molluscs (IAEA 2017; Hansen et al., 2020).

^{210}Po in humans

Except for radon, ^{210}Po (5.304 MeV) is the largest contributor to the radiological dose to humans via food ingestion (especially seafood and marine mammals, Hansen et al., submitted 2021), when compared to anthropogenic and naturally occurring radionuclides. ^{210}Po is not a hazard to the outside of the body, but enter the human body, mainly via food consumption and to a lesser extent via water ingestion, inhalation and in vivo contributions from ^{222}Rn dissolved in the body and from ^{210}Pb and ^{226}Ra in bones. Its effective biological half-life is in the range of 20 – 60 days. Its human gastrointestinal uptake fraction (f1) has been reported to 50% (ICRP, 1993), while other studies report a range from a few percent to 90% (IAEA 2017). Much of ingested Po-210 is excreted via the feces, urine and hair. The remaining portion enters the bloodstream and concentrates in the liver and a lesser extent, in the kidneys (IAEA 2017).

Inhaled polonium is deposited on the mucosal lining along the respiratory tract. While some of the inhaled will stay in the lungs, it may also be removed from the respiratory tract via mucociliary clearance by coughing and swallowing phlegm into the digestive tract. If not removed the inhaled Po-210 can increase the risk of lung cancer (IAEA 2017).

4.4 Recommended dose rates for protecting the biota from ionizing radiation

IAEA (1992) concluded that:

''Chronic dose rates of 40 µGy h^{-1} to even the more radiosensitive species in terrestrial ecosystems are unlikely to cause measurable detrimental effects in populations and that up to this level adequate protection would therefore be provided".

''In the aquatic environment it would appear that limiting chronic dose rates to 400 µGy h^{-1} or less to the maximally exposed individuals in a population would provide adequate protection for the population".

ICRP (2008) has developed Derived Consideration Reference Levels (DCRLs). DCRLs are zones of dose rates that can be used in making decisions on scenarios which require further evaluation. For the aquatic environment the doses are in the range of 40-400 µGy h^{-1}. For the terrestrial environment the doses are in the range of 4-40 µGy h^{-1} for pine trees, birds and mammals, and 40-400 µGy h^{-1} for grass and invertebrates (ICRP 2008).

UNSCEAR (2008), found that reproductive effects are the more sensitive indicator of radiation response for aquatic organisms. Overall consideration of the data available led to the conclusion that chronic irradiation at dose rates up to 400 µGy h^{-1} to a small proportion of the individuals in an aquatic population would not have any detrimental effects at the population level. UNSCEAR (1996) suggests that dose rates below 40 µGy h^{-1} are unlikely to cause a loss of reproductive capacity of terrestrial biota.

4.5 Short and long-term freshwater quality guidelines for uranium for the protection of aquatic life

Short-term exposure guidelines were derived using severe effect data (e.g. lethality) of defined short-term exposure periods (24 - 96-hours) (Canadian Water Quality Guidelines (CWQG), 2011). These guidelines provide estimations of the highest concentration that do not cause lethal effects (or other defined effects) to aquatic organisms in acute exposures (e.g. spill events to aquatic receiving environments and infrequent releases of short-lived / nonpersistent substances). Short-term exposure guidelines for uranium are 33 µg·L^{-1} (Table 4.15.). However, organisms may also be exposed to continuous concentrations (e.g. long-term exposure). Therefore, long-term guidelines identify the maximum levels in the aquatic ecosystem that protect all forms of aquatic life for indefinite exposure periods (≥7 day's exposures for fish and invertebrates, ≥24 hours for aquatic plants and algae). As can be seen in Table 4.15., the long-term exposure WQG for the protection of freshwater life is 15 µg·L^{-1} for uranium.

Table 4.15. Canadian Water Quality Guidelines (CWQG) for Uranium (CWQG, 2011).

	Long-Term Exposure Guideline(µg·L^{-1})	**Short-Term Exposure Guideline(µg·L^{-1})**
Freshwater	15	33

4.6 Guidelines for drinking water quality

With regard to drinking water guideline values have been developed by the World Health Organization for gross alpha and beta activity and for natural radionuclides (Table 4.16.).

Table 4.16. Drinking water quality guidelines for radionuclides developed by the World Health Organization (WHO 2011).

Radionuclide	(Bq L^{-1})
Screening level:	
Gross alpha activity	0.5
Gross beta activity	1
Guidance level:	
Pb-210	0.1
Po-210	0.1
Ra-226	1
Ra-228	0.1
Th-228	1
Th-230	1
Th-232	1
U-234	1
U-238	10

The WHO guideline value for uranium in drinking water is 15 µg L^{-1} (WHO 2012). This value is based on uranium chemical toxicity.

Radiological quality of drinking water

Recommended values for guidance activity concentrations of NORs in drinking water in Greenland are listed in Table 4.17. The recommended guidance levels for radionuclides correspond to a reference dose level (RDL) equal to 0.1 mSv (for each radionuclide listed) from a yearly consumption of drinking water. A guideline value for radon in drinking water is not deemed necessary, but the recommendations given by the EU and Nordic countries are included in the table.

Table 4.17. Recommended guidlines for activity concentrations of NORs in Greenland drinking water. (Hansen et al., 2017).

Radionuclide	Half-life	Decay mode	Adult dose coefficient (DC) for ingestion (mSv Bq^{-1})	Guidlines (Bq L^{-1}) due to radiation	Guidlines (µg L^{-1}) due to chemical toxicity
Total uranium (U-238, U-235 and U-234)			-	10	15
Th-228	1.91 years	alpha (100%)	7.20E-05	2	
Th-230	75,400 years	alpha (100%)	2.10E-04	0.6	
Th-232	14 billion years	alpha (100%)	2.30E-04	0.6	
Ra-226	1600 years	alpha (100%)	2.80E-04	0.5	
Po-210	138.4 days	alpha (100%)	1.20E-03	0.1	
Pb-210	22.3 years	beta (100%)	6.90E-04	0.2	
Rn-222	3.824 days	alpha (100%)		100	
Ra-228	5.75 years	beta (100%)		0.1	
Gross alpha				0.5	
Gross beta				1	

The radiological guidance level for ^{210}Pb should not be confused with the chemical guidance level for Pb of 0.01 mg L^{-1}. A ^{210}Pb concentration at a radiological guaidance level of 0.2 Bq L^{-1} would correspond to a total lead concentration of only 7×10^{-8} µg L^{-1}.

4.7 Exemption and clearance threshold values for Greenland

Naturally occurring radioactive materials (NORM) and technically enhanced naturally occurring radioactive materials (TENORM) waste are any materials that arise from NORM industrial activities and in which the activity concentrations of radionuclides are above those set out in Table 4.18. Materials and waste with activity concentrations at or below clearance and exemption threshold values set out in Table 4.18. are deemed not radioactive. A single set of threshold values for clearance and for exemption are recommended to apply to Greenland (Table 4.18.).

Table 4.18. Clearance and exemption threshold values for activity concentrations of radionuclides for NORM industrial activities.

Radionuclide	**Solid (Bq g^{-1})**	**Liquid (Bq L^{-1})**	**Air (Bq m^{-3})**
U-238 Series (all progeny)	0.3	1	0.001
U-238 (U-238, Th-234, Pa-234m, U-234)	10	10	0.01
Th-230	10	5	0.001
Ra-226 (in equilibrium with its progeny)	0.3	5	0.01
Pb-210 (in equilibrium with Bi-210)	0.3	1	0.01
Po-210	0.3	1	0.01
Th-232 Series (all progeny)	0.3	1	0.001
Th-232	10	1	0.001
Ra-228 (in equilibrium with Ac-228)	0.3	5	0.005
Th-228 (in equilibrium with all its progeny)	0.3	1	0.001
K–40	17	n/a	n/a

Source: Hansen et al., (2021).

Activity concentrations of naturally occurring radionuclides resulting in a 0.3 mSv y^{-1} for any member of the public above the natural background level of radiation were used to define the radionuclide clearance and exemption threshold values as included in Table 4. 18.

In practice, NORM/TENORM waste, usually contain a mixture of different naturally occurring radionuclides. To determine if a material, which contains a mixture of radionuclides, is below the clearance and exemption level, a simple summation formula can be used:

$$\sum_{i=1}^{n} \frac{Ci}{CU,i} \leq 1$$

Where:

Ci is the specific activity of a given radionuclide *i* (Bq g^{-1})
CU, i is the clearance threshold value in the table 4.18. of a given radionuclide *i* (Bq g^{-1})
n is the number of radionuclides in the mixture.

In the above expression, the ratio of the concentration of each radionuclide to the clearance threshold value is summed over all radionuclides in the mixture. If the sum is less than one, the material complies with the clearance requirements.

Discrete NORM / TENORM sources are small in size and generally exceed the concentrations criteria in Table 4.18. Because of the possibility of high radiation dose rates close to the source, the clearance and exemption threshold values (Table 4.19.) for discrete sources are lower than those in Table 4.18.

Table 4.19. Clearance and exemption threshold values for activity of radionuclides for discrete NORM /TENORM sources.

Radionuclide	Bq
Uranium ore (in equilibrium with all progenies)	1000
U-238 (partitioned) (in equilibrium with Th-234 and Pa-234m)	10 000
Th-230 (no progeny)	10 000
Ra-226 (in equilibrium with its progeny)	10 000
Pb-210 (in equilibrium with Bi-210)	10 000
Po-210	10 000
Th-232 series (all progeny)	1000
Ra-228 (in equilibrium with Ac-228)	100 000
Th-228 (in equilibrium with its short-lived progeny)	10 000
K–40	1 000 000

To determine if a discrete NORM / TENORM source that contain a mixture of different naturally occurring radionuclides is below the clearance and exemption level, a simple summation formula can be used:

$$\sum_{i=1}^{n} \frac{Ci}{CU,i} \leq 1$$

Where:

Ci is the activity of radionuclide *i* (Bq)
CU, i is the clearance threshold value in the table 4.19. of radionuclide *i* (Bq)
n is the number of radionuclides in the mixture.

In the above expression, the ratio of the activity of each radionuclide to the clearance threshold value is summed over all radionuclides in the mixture. If the sum is less than one the material complies with the clearance requirements.

Surface contamination

Limits for surface radioactive contamination on equipment, tools or scrap surfaces intended for release from radiological restrictions are based on personal radiation exposure pathways to a maximum annual dose of 0.3 mSv. Discrete NORM / TENORM sources with surface contamination less than the Table 4.20. can be released without radiological restrictions.

Table 4.20. Surface contamination clearance and exemption threshold values for discrete NORM/TENORM sources.

	Limit
Dose rate	0.5 μSv h^{-1} at 50 cm
Surface contamination	1 Bq cm^{-2} averaged over a 100 cm^2 area

A thin window radiation detector is recommended when monitoring alpha / beta / gamma sources of surface contamination. The clearance and exemption threshold values provided in table 4.20. are only applicable to fixed surface contamination. Loose surface contamination must be removed entirely and / or all accessible surfaces stripped to ensure complete removal.

The clearance and exemption threshold values included in Table 4.18. and Table 4.19. relate to the long-lived parent radionuclide in equilibrium with its progenies. Where the equilibrium has been disturbed by partitioning of decay series, the activity of each long-lived radionuclide must be found and compared to its appropriate clearance and exemption threshold value.

It shall be underlined that the considerations above focus on radiation protection and that health aspects other than radiation, like chemical toxicity, may be prominent. The chemical risk may thus be well above the radiological risk.

4.8 NORs Summary

NORs of concern associated with the Kvanefjeld project include U, Th, ^{226}Ra, ^{228}Ra, ^{222}Rn, ^{220}Rn, ^{210}Pb, and ^{210}Po. The NORs may be dispersed into the environment via air (e.g. dust, water droplets dispersion from the tailings facilities) and water (e.g. proposed liquid discharges to the Bredefjord, tailings facilities, waste rock depot, etc.). Non-human biota may be exposed to NORs through ingestion of food and to a lesser extent to ingestion of water, inhalation, cell membrane-mediated uptake, cutaneous absorption and biotic uptake / trophic transfer.

Radiation effects in biota depend on differences in exposure, biota recovering capacity, type of reproduction, sensitivity at critical periods of ontogenesis, duration of lifespan, the season of exposure and other factors. The distribution of NORs in the biota body is not uniform and may result in increased exposure of individual organs, leading to these organs' dysfunctions. Species also exhibit radiosensitivity differences. Particulary species are more vulnerable to radiation exposure. Arctic low temperatures and extreme seasonal variations in light can cause environmental stress to organisms and may make them more vulnerable to contaminants. Due to low temperature, especially during the winter, radiation effects in poikilotherm Arctic species are expected to occur more slowly as opposed to species from temperating environments. The repair of radiation damage in cells and tissues is not effective at low Arctic temperature.

The bioavailability and toxicity of uranium depend on its speciation and several environmental key factors. Uranium chemical toxicity predominates over its radiotoxicity. Speciation-based toxicity studies conclude that UO_2^{2+} is the toxic chemical species, and it is often the target concentration to indicate its toxicity. Uranium concentrates mainly in plant roots. In non-human biota uranium distributes in the skeleton, kidneys and soft tissues. Animal data show that urinary excretion is rapid. About two-thirds of uranium in the blood is excreted in the first 24 hours and 10% over the next five days. Most of the remaining uranium is excreted over a few months and a small percentage of the uranium may be retained for years. The behaviour of uranium, its distribution, and its clearance in the lungs depend on the solubility of the uranium compounds. Soluble compounds such as uranyl fluoride, uranium tetrachloride, uranyl nitrate are less toxic to the lungs but more toxic to distal organs

due to easier absorption of the uranium from the lungs into the blood and systemic transport. Insoluble uranium particles with a longer residence time in the lung confer more risk than the more soluble particles with a shorter residence time.

Literature results indicate that thorium does not bio-concentrate in plants (soil to plant transfer factor values < 0.01). Plant roots adsorb thorium from the soil, but the transport from the root to the above ground parts of the plant is low. Bio-concentration of thorium can be significant in lower trophic level aquatic NHB, but thorium does not biomagnify as the trophic level increases.

Radium is of particular importance due to its relatively long half-live, the short-lived progeny of two isotopes (^{226}Ra and ^{228}Ra), its high mobility in the environment under natural environmental conditions and its tendency to accumulate in bone following uptake into the body. Literature studies show that the highest concentrations of Ra are found in the plant roots, followed by the stem and grain. High pH and soil sulphate content are reported to decrease Ra concentrations in plants. Ra transfers to animals mainly via ingestion of food and water. Accumulation in animals depends on its concentration in food and water and seasonal variations in the diet patterns of animals and their position in the trophic chain. The accumulation of Racan be presented as descending from insectivores (soil animals) > mice rodent > carnivores > ungulates. ^{226}Ra and ^{228}Ra accumulate in tissues with high calcium content (e.g. bones), while low activity concentrations are reported in soft tissues. Partition studies in cattle show that radium distributes mainly in teeth and bone and to a lesser extent in soft tissues, e.g. muscle. A similar distribution pattern was observed for Arctic animals such as caribou, reindeer, elk and moose, with the highest ^{226}Ra concentration in bone (3.2 -13.5 Bq kg^{-1} fresh weight) and the lowest (0.01-0.05 Bq kg^{-1} fresh weight) in soft tissues, e.g. muscle.

The highest observed transfer factors (TFs) values are 1.0 for ^{210}Po in pasture vegetation and 25 for ^{210}Pb in leafy vegetables. In animals ^{210}Pb is accumulated in bones, while ^{210}Po is distributed mainly in soft tissues such as the spleen, liver and kidneys. The ^{210}Po:^{210}Pb ratio exceeds unity in the soft tissues, suggesting that ^{210}Po is not supported by ^{210}Pb. Compared to terrestrial organisms, relatively high concentrations of ^{210}Po were reported in marine biota. ^{210}Po concentrations in marine biota are higher than those of ^{210}Pb with ^{210}Po:^{210}Pb ratios of 1–100.

The International Atomic Energy Agency (IAEA) recommends chronic dose rates of 40 µGy h^{-1} to terrestrial biota and 400 µGy h^{-1} to the aquatic environment. These dose rates are unlikely to cause measurable detrimental effects in populations. The International Commission on Radiological Protection (ICRP) has developed Derived Consideration Reference Levels (DCRLs). DCRLs are zones of dose rates that can be used in making decisions on scenarios which require further evaluation. For the aquatic environment the doses are in the range of 40-400 µGy h^{-1}. For the terrestrial environment the doses are in the range of 4-40 µGy h^{-1} for pine trees, birds and mammals and 40-400 µGy h^{-1} for grass and invertebrates.

There are no environmental quality guidelines values of NORs in the environment in Greenland, and thus data in this report may be used as a starting point for further work.

5 Conclusions and recommendations

Following data from the scientific literature and national and international organisations are compiled in the present report:

- Environmental behaviour and concentrations of F, REEs and NORs,
- The fate and toxicity on biota and humans,
- Flouride maximum residue levels (MRL) in foodstuff,
- Threshold concentrations of REEs in food and soil, background and threshold concentrations of REEs in drinking water, water and sediment quality criteria of REEs**,** emission standards of pollutants for REE mining industry, etc.
- Recommended guidance activity concentrations of NORs in drinking water in Greenland, environmental threshold and guidelines values (e.g. clearance and exemption threshold values, etc.), recommended dose rates for protecting the biota from ionizing radiation, short and long-term freshwater quality guidelines for uranium for the protection of aquatic life, etc.

Data in the report can be used as a basis to establish environmental quality and guidelines values for the F, REEs and NORs concentrations for specific mineral exploration and exploitation projects in Greenland.

Uptake of fluoride into plants from soil is often low due to the low bioavailability of fluoride in the soil. Fluoride bioaccumulation and biomagnification in the aquatic and terrestrial invertebrates and fish are of little significance. Fluoride may bioaccumulate in invertebrates' exoskeleton and the skeleton of fish but does not accumulate in edible tissues. Studies on fluoride toxicity on terrestrial organisms are sparse, and effect concentrations are not available.

The toxicity of REEs decreases with the increase of the atomic number. In the environment the concentration of REEs decreases from freshwater biota to the marine and to terrestrial biota. To our knowledge, no ecotoxicity data are available for REEs effect on terrestrial animals such as sheep, beef cattle, reindeer and muskox. Limited research data are available on REEs toxicity to aquatic organisms. No environmental quality guidelines values are available worldwide for either ∑REE, or individual REEs. Due to the lack of environmental quality guidelines values, it is difficult to assess environmental impacts associated with industry and compare research results worldwide. The lack of evidence in the bioavailability, bioaccumulation, and the biological and eco-toxicological effect of the REEs calls for future research on the subjects.

Radiation effects in biota depend on differences in exposure, biota recovering capacity, type of reproduction, sensitivity at critical periods of ontogenesis, duration of lifespan, the season of exposure and other factors. Species also exhibit radiosensitivity differences. Particulary organisms are more vulnerable to radiation exposure. Arctic low temperatures, and extreme seasonal variations in light can cause environmental stress to organisms and may make them more vulnerable to contaminants. Due to low temperature, especially

during the winter, radiation effects in poikilotherm Arctic organisms are expected to occur more slowly as opposed to organisms from temperating environments. The repair of radiation damage in cells and tissues is not effective at low Arctic temperature. Uranium chemical toxicity predominates over its radiotoxicity. Uranium concentrates mainly in plant roots. In biota uranium distributes in the skeleton, kidneys and soft tissues. Thorium does not bioconcentrate in plants. Plant roots adsorb thorium from the soil, but the transport from the root to the aboveground parts of the plant is low. Bio-concentration of thorium can be significant in lower trophic level aquatic biota, but thorium does not biomagnify as the trophic level increases. Highest concentrations of Ra are found in the plant roots, followed by the stem and grain. Ra accumulation in animals can be presented as descending from insectivores (soil animals) > mice rodent > carnivores > ungulates. ^{226}Ra and ^{228}Ra accumulate in tissues with high calcium content (e.g. teeth and bone), and low activity concentrations are reported in soft tissues. In animals, ^{210}Pb is accumulated in bones, while ^{210}Po is distributed mainly in soft tissue such as the spleen, liver and kidneys. Compared to terrestrial organisms, relatively high concentrations of ^{210}Po were reported in marine biota. ^{210}Po concentrations in marine biota are higher than those of ^{210}Pb with ^{210}Po:^{210}Pb ratios of 1–100.

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APPENDIX A. Rare Earths Elements

Sources REE into the environment

Table 3.1. Baseline value of REEs in the soil in the area of Inner Mongolia, China where Bayan Obo REE mining factory, Baotou is located (unit: mg kg^{-1}) (After Guo et al., 2013).

	La	Ce	Pr	Nd	Sm	Eu	Y
Baseline value	32.8	49.1	5.68	19.2	3.81	0.81	17.0

Table 3.2. Average concentrations and distribution of REEs in the soils of China in comparison with the world soil (unit: ppm, can be converted to be mg kg^{-1} directly) (Source: Wang and Sun 1991).

Element	Guangzhou, Hainan island	Xiangjiang River valley	Liaoshong Plain	Tianjing Region	Northing China	Mount Everest	Entire China	World soil*
No. of Replicates	18	96	96	36	104	14	364 average	-
La	44.2	39.7	34.4	35.1	30.0	30.5	34.9	40
Ce	99.7	92.5	70.7	74.1	57.6	67.2	74.3	50
Nd	53.0	30.3	35.1	32.5	28.6	31.7	32.5	35
Sm	6.53	5.95	5.90	6.21	5.79	5.39	5.92	4.5
Eu	1.26	1.11	1.17	1.28	1.18	0.78	1.16	1
Tb	0.89	0.81	0.77	0.79	0.81	0.66	0.80	0.7
Yb	3.42	2.80	2.52	2.34	2.76	2.09	2.67	3.0
Lu	0.42	0.47	0.42	0.47	0.43	0.37	0.44	0.4
ΣREE	240	197	172	177	150	160	177	156

*Bowen, 1979.

ΣREE is the total content of 14 types of REEs

Table 3.3. Concentration of dissolved REE (> 0.2 µm fraction) in river and stream waters from Germany, Japan and Sweden, in tap water from Berlin-Steglitz, and in the clear water effluent of the sewage treatment plant "Ruhleben" in Berlin (unit: µg L^{-1}) [After Bau and Dulski, 1996].

River	**Västerdalälven**	**Toshibetsu**	**Rhein**	**Elbe**	**Mosel**	**Wupper**	**Spree**	**Havel**
Sample site	Lima, Dalarna	Aikappu, Hokkaido	South of Düsseldorf	Nienstetten, Hamburg	Wasser-billi-gerbrück	Leverkusen	Berlin	Potsdam
Country	Sweden	Japan	Germany	Germany	Germany	Germany	Germany	Germany
Date	05.05.94	06.11.95	03.05.95	10.05.95	05.05.95	05.08.95	20.07.95	13.03.95
La	0.14	0.0240	0.0193	0.0165	0.0080	0.0027	0.0019	0.0022
Ce	0.23	0.0367	0.0226	0.0248	0.0117	0.0046	0.0030	0.0034
Pr	0.04	0.0080	0.0048	0.0037	0.0023	0.0008	0.0005	0.0006
Nd	0.19	0.0368	0.0193	0.0138	0.0108	0.0046	0.0021	0.0025
Sm	0.04	0.0096	0.0045	0.0028	0.0027	0.0009	0.0005	0.0007
Eu	0.01	0.0025	0.0011	0.0007	0.0006	0.0003	0.0001	0.0001
Gd	0.04	0.0122	0.0086	0.0057	0.0046	0.0326	0.0068	0.1061
Tb	0.01	0.0019	0.0008	0.0006	0.0005	0.0002	0.0001	0.0002
Dy	0.04	0.0120	0.0003	0.0033	0.0033	tracer	0.0007	0.0012
Ho	0.01	0.0028	0.0012	0.0008	0.0008	0.0005	0.0002	0.0003
Er	0.02	0.0089	0.0004	0.0029	0.0029	0.0024	0.0008	0.0014
Tm	0.00	Tracer	0.0005	tracer	0.0005	0.0005	0.0002	0.0003
Yb	0.02	0.0094	0.0039	0.0039	0.0033	0.0042	0.0015	0.0024
Lu	Tracer	0.0017	tracer	0.0008	tracer	0.0009	tracer	Tracer
Gd_{SN}/gd^*_{SN}	1.2	1.2	1.9	1.9	1.7	30	12.4	126

River	**Dhünn-UC**	**Dhünn-LC**	**Seawater**	**Tap water**	**Effluent**
Sample site	Altenberg	Bergisch-Gladbach	Koljöfjord, Skagerrak	Berlin-Steglitz	Ruhleben, Berlin
Country	Germany	Germany	Sweden	Germany	Germany
Date	03.12.95	03.12.95	22.08.95	20.07.95	14.03.96
La	0.0021	0.0038	0.0360	0.0009	0.0012
Ce	0.0056	0.0073	0.0514	0.0023	0.0041
Pr	0.0007	0.0012	0.0083	0.0003	0.0004
Nd	0.0032	0.0054	0.0313	0.0012	0.0026
Sm	0.0009	0.0015	0.0056	0.0003	0.0005
Eu	0.0002	0.0004	0.0009	0.0001	0.0001
Gd	0.0014	0.0112	0.0066	0.0063	1.1144
Tb	0.0002	0.0003	0.0009	0.0001	0.0001
Dy	0.0010	0.0019	tracer	0.0011	0.0010
Ho	0.0003	0.0005	0.0011	0.0004	0.0003
Er	0.0009	0.0018	0.0034	0.0022	0.0015
Tm	Tracer	Tracer	tracer	0.0004	tracer
Yb	0.0013	0.0026	0.0035	0.0036	0.0023
Lu	0.0003	0.0006	0.0006	tracer	0.0008
Gd_{SN}/gd^*_{SN}	1.6	6.9	1.33	12	1681

* Contaminated. “Effluent” is clear water effluent from sewage treatment plant, “Ruhleben”, Berlin. $Gd_{SN}/Gd^*_{SN} = GD_{SN}/(0.33Sm_{SN} + 0.67Tb_{SN})$. The unit was pmol kg^{-1} for the data presented originally in Bau and Dulski 1996. In the present review, all concentrations have been converted from pmol kg^{-1} to µg L^{-1}. 0.2 µm filter was used for all samples, except for Skagerrak seawater (0.8 µm); Skagerrak seawater is coastal surface (0.5 m depth) water originating in the Baltic Sea.

Table 3.4. Main use of lanthanides (After Gonzalez et al., 2014. Also refers to US EPA 2012; Lacour et al., 2005; Hutchison et al., 2004 Monafo et al., 1976; Pang et al., 2001).

Element	Atom no.	REE groups	Primary applications
La	57	Light	Batteries (electric cars, laptops); catalysts for petroleum refining; high-tech digital cameras; X-ray films; lasers; communication devices.
Ce	58	Light	Catalysts; polishing; metal alloys; lens polishers (for glass, television faceplates, mirrors, optical glass, silicon microprocessors and disk drives), medical applications.
Pr	59	Light	Improved magnet corrosion resistance; pigment; searchlights; airport signal lenses, photographic filters
Nd	60	Light	Magnets for laptops; lasers; fluid-fracking catalysts; electric motors, communication devices
Pm	61	Light	Fluid-cracking catalysts; β radiation source
Sm	62	Light	High-temperature magnets; reactor control rods; electric motors
Eu	63	Light	Liquid crystal displays (LCDs); fluorescent lighting; glass additives; communication devices
Gd	64	Light	Magnetic resonance imaging contrast agent; glass additives
Tb	65	Heavy	Phosphors for lighting and display; electric motors
Dy	66	Heavy	Highest power magnets, lasers, electric motors
Ho	67	Heavy	Highest power magnets
Er	68	Heavy	Lasers; glass colorant
Tm	69	Heavy	High-power magnets
Yb	70	Heavy	Fiber-optic technology; solar panels; alloys (stainless steel); lasers, radiation source for portable X-ray units
Lu	71	Heavy	X-ray phosphors
Sc	21	Light	Aerospace industry components; sports equipment; mercury vapor lamps.
Y	39	Heavy	Red phosphors for color television picture tubes, heat- & shock-resistant glass, superconductors, additive in alloys.
Ln-mixtures		Heavy	Fertilizer

Bioaccumulation of REE in marine, terrestrial and freshwater environment

Marine environment

Table 3.5. Acute toxicity of lanthanides to marine organisms (After Ng et al., 2012).

REE elements	Organism	Species	Effect	Toxicity value (µg REE L^{-1})	References
La	Algae	*Skeletonema costatum*	Growth	4054.5	(Tai et al. 2010)
Ce	Algae	*Skeletonema costatum*	Growth	4155.2	(Tai et al. 2010)
Nd	Algae	*Skeletonema costatum*	Growth	4375	(Tai et al. 2010)
Sm	Algae	*Skeletonema costatum*	Growth	4313.5	(Tai et al. 2010)
Eu	Algae	*Skeletonema costatum*	Growth	4432.3	(Tai et al. 2010)
Gd	Algae	*Skeletonema costatum*	Growth	4686	(Tai et al. 2010)
Tb	Algae	*Skeletonema costatum*	Growth	4536.3	(Tai et al. 2010)
Dy	Algae	*Skeletonema costatum*	Growth	4593.9	(Tai et al. 2010)
Ho	Algae	*Skeletonema costatum*	Growth	4829.9	(Tai et al. 2010)
Er	Algae	*Skeletonema costatum*	Growth	4962.1	(Tai et al. 2010)
Tm	Algae	*Skeletonema costatum*	Growth	4866	(Tai et al. 2010)
Yb	Algae	*Skeletonema costatum*	Growth	4935.7	(Tai et al. 2010)
Lu	Algae	*Skeletonema costatum*	Growth	5008.5	(Tai et al. 2010)
Ce	Copepod	*Acartia tonsa*	Survival	150	(Bowmer et al. 1992) [SD]
Dy	Copepod	*Acartia tonsa*	Survival	3600	(Bowmer et al. 1992) [SD]
Gd	Copepod	*Acartia tonsa*	Survival	520	(Bowmer et al. 1992) [SD]
La	Copepod	*Acartia tonsa*	Survival	1040	(Bowmer et al. 1992) [SD]
Nd	Copepod	*Acartia tonsa*	Survival	850	(Bowmer et al. 1992) [SD]
Pr	Copepod	*Acartia tonsa*	Survival	920	(Bowmer et al. 1992) [SD]
Sm	Copepod	*Acartia tonsa*	Survival	420	(Bowmer et al. 1992) [SD]
Ce	Fish	*Poecilia reticulata*	Survival	11200	(Hooftman et al. 1992) [SD]
Dy	Fish	*Poecilia reticulata*	Survival	15400	(Hooftman et al. 1992) [SD]
Gd	Fish	*Poecilia reticulata*	Survival	10800	(Hooftman et al. 1992) [SD]
La	Fish	*Poecilia reticulata*	Survival	47000	(Hooftman et al. 1992) [SD]
Nd	Fish	*Poecilia reticulata*	Survival	9600	(Hooftman et al. 1992) [SD]
Pr	Fish	*Poecilia reticulata*	Survival	4500	(Hooftman et al. 1992) [SD]
Sm	Fish	*Poecilia reticulata*	Survival	10600	(Hooftman et al. 1992) [SD]

Table 3.6. Bioaccumulation of REEs by marine organisms (After Ng et al., 2012)

REE element	Organism	Species	Tissue	Tissue conc. (μg REE g^{-1}dw)	References
La	Cephalopod	*Nautilus macromphalus*	Digestive Gland	0.3	(Pernice et al. 2009)
La	Cephalopod	*Nautilus macromphalus*	Pericardial appendages	0.1	(Pernice et al. 2009)
Ce	Cephalopod	*Nautilus macromphalus*	Digestive Gland	0.3	(Pernice et al. 2009)
Ce	Cephalopod	*Nautilus macromphalus*	Pericardial appendages	0.2	(Pernice et al. 2009)
Nd	Cephalopod	*Nautilus macromphalus*	Digestive Gland	0.2	(Pernice et al. 2009)
Nd	Cephalopod	*Nautilus macromphalus*	Pericardial appendages	0.2	(Pernice et al. 2009)
La	Cephalopod	*Nautilus pompilius*	Digestive Gland	1	(Pernice et al. 2009)
La	Cephalopod	*Nautilus pompilius*	Pericardial appendages	0.2	(Pernice et al. 2009)
Ce	Cephalopod	*Nautilus pompilius*	Digestive Gland	1.6	(Pernice et al. 2009)
Ce	Cephalopod	*Nautilus pompilius*	Pericardial appendages	0.4	(Pernice et al. 2009)
Nd	Cephalopod	*Nautilus pompilius*	Digestive Gland	0.9	(Pernice et al. 2009)
Nd	Cephalopod	*Nautilus pompilius*	Pericardial appendages	0.3	(Pernice et al. 2009)
La	Mussel	*Mytilus edulis*	Kidney	0.4	(Lobel et al. 1991)
La	Mussel	*Mytilus edulis*	Digestive Gland	0.3	(Lobel et al. 1991)
La	Mussel	*Mytilus edulis*	Gills	0.2	(Lobel et al. 1991)
La	Mussel	*Mytilus edulis*	Mantle	0.2	(Lobel et al. 1991)
La	Mussel	*Mytilus edulis*	Foot	0.1	(Lobel et al. 1991)
Ce	Mussel	*Mytilus edulis*	Kidney	0.5	(Lobel et al. 1991)
Ce	Mussel	*Mytilus edulis*	Digestive Gland	0.5	(Lobel et al. 1991)
Ce	Mussel	*Mytilus edulis*	Gills	0.2	(Lobel et al. 1991)
Ce	Mussel	*Mytilus edulis*	Mantle	0.2	(Lobel et al. 1991)
Ce	Mussel	*Mytilus edulis*	Foot	0.1	(Lobel et al. 1991)
La	Mussel	*Mytilus edulis*	Soft tissue	3.7	(Riget et al. 1996)
Ce	Mussel	*Mytilus edulis*	Soft tissue	4.7	(Riget et al. 1996)
Eu	Mussel	*Mytilus edulis*	Soft tissue	0.02	(Riget et al. 1996)
Ce	Scallop	*Chlamys varia*	Digestive Gland	10.6	(Bustamante and Miramand 2005)
Ce	Scallop	*Chlamys varia*	Kidney	1.9	(Bustamante and Miramand 2005)
Ce	Scallop	*Chlamys varia*	Gills	5.4	(Bustamante and Miramand 2005)
Ce	Scallop	*Chlamys varia*	Gonad	5.7	(Bustamante and Miramand 2005)
Ce	Scallop	*Chlamys varia*	Muscle	0.3	(Bustamante and Miramand 2005)
Ce	Scallop	*Chlamys varia*	Non-organ tissue	4	(Bustamante and Miramand 2005)
Ce	Scallop	*Chlamys varia*	Digestive Gland	2.2	(Bustamante and Miramand 2005)
Ce	Scallop	*Chlamys varia*	Kidney	0.2	(Bustamante and Miramand 2005)
Ce	Scallop	*Chlamys varia*	Gills	0.1	(Bustamante and Miramand 2005)
Ce	Scallop	*Chlamys varia*	Gonad	1.5	(Bustamante and Miramand 2005)
Ce	Scallop	*Chlamys varia*	Muscle	0.04	(Bustamante and Miramand 2005)
Ce	Scallop	*Chlamys varia*	Non-organ tissue	0.6	(Bustamante and Miramand 2005)
La	Scallop	*Chlamys varia*	Digestive Gland	7.9	(Bustamante and Miramand 2005)
La	Scallop	*Chlamys varia*	Kidney	1.8	(Bustamante and Miramand 2005)
La	Scallop	*Chlamys varia*	Gills	4	(Bustamante and Miramand 2005)
La	Scallop	*Chlamys varia*	Gonad	5.1	(Bustamante and Miramand 2005)
La	Scallop	*Chlamys varia*	Muscle	0.3	(Bustamante and Miramand 2005)
La	Scallop	*Chlamys varia*	Non-organ tissue	2.8	(Bustamante and Miramand 2005)
La	Scallop	*Chlamys varia*	Digestive Gland	0.2	(Bustamante and Miramand 2005)
La	Scallop	*Chlamys varia*	Kidney	0.2	(Bustamante and Miramand 2005)
La	Scallop	*Chlamys varia*	Gills	0.4	(Bustamante and Miramand 2005)
La	Scallop	*Chlamys varia*	Gonad	0.7	(Bustamante and Miramand 2005)
La	Scallop	*Chlamys varia*	Muscle	0.03	(Bustamante and Miramand 2005)
La	Scallop	*Chlamys varia*	Non-organ tissue	0.2	(Bustamante and Miramand 2005)
Nd	Scallop	*Chlamys varia*	Digestive Gland	5.4	(Bustamante and Miramand 2005)

Nd	Scallop	*Chlamys varia*	Kidney	0.7	(Bustamante and Miramand 2005)
Nd	Scallop	*Chlamys varia*	Gills	2	(Bustamante and Miramand 2005)
Nd	Scallop	*Chlamys varia*	Gonad	3.5	(Bustamante and Miramand 2005)
Nd	Scallop	*Chlamys varia*	Muscle	0.1	(Bustamante and Miramand 2005)
Nd	Scallop	*Chlamys varia*	Non-organ tissue	1.8	(Bustamante and Miramand 2005)
Nd	Scallop	*Chlamys varia*	Digestive Gland	0.9	(Bustamante and Miramand 2005)
Nd	Scallop	*Chlamys varia*	Kidney	0.1	(Bustamante and Miramand 2005)
Nd	Scallop	*Chlamys varia*	Gills	0.3	(Bustamante and Miramand 2005)
Nd	Scallop	*Chlamys varia*	Gonad	1.1	(Bustamante and Miramand 2005)
Nd	Scallop	*Chlamys varia*	Muscle	0.01	(Bustamante and Miramand 2005)
Nd	Scallop	*Chlamys varia*	Non-organ tissue	0.3	(Bustamante and Miramand 2005)

Table 3.7. Ecotoxicity of REEs to invertebrates (After Pagano et al., 2019).

Organisms	**Species**	**Main effects/Endpoints**	**References**
Echinoidea	Sea urchin embryos and sperm	Comparative toxicities of several REEs to early development, fertilization success, offspring damage, cytogenetic damage and oxidative stress	Oral et al., 2010; Pagano et al., 2015, 2016; Gravina et al., 2018
Water flea	*Daphnia pulex*	CeO_2 uptake in gut content, independent on feeding conditions	Auffan et al., 2013
Roundworm	*Caenorhabditis elegans*	Reproduction and growth inhibition	Zhang et al.,(2010
Rmphipod and zebra mussel	*Gammarus roeseli* and *Dreissena polymorpha*	nCeO_2 decreased size of the lysosomal system, catalase activity and lipoperoxidation in D. polymorpha	Garaud et al., 2015; González et al., 2015
Diatom, harlequin fly, African clawed frog and sharp-ribbed newt	*Nitzschia palea, Chironomus riparius, Xenopus laevis* and *Pleurodeles waltl*	nCeO_2-associated comparative toxicities	Blaise et al., 2008
Roundworm	*Caenorhabditis elegans*	$PrCl_3$, $NdCl_3$ and $ScCl_3$ induce behavioral deficits and neural damage	Xu et al., 2017
Various invertebrates and microbiota		Comparative toxicities of REEs in four invertebrates and in two bacteria species	Bour et al., 2015

Seaweeds and Zooplankton

Table 3.8. Bioaccumulation of REEs in aquatic organisms (mg L^{-1} unless otherwise specified) [After Gonzalez et al., 2014 with modifications].

Elements	Organisms	Types	Exposure concentrations	References
La, Ce	Blue mussel	*Mytilus edulis*	Unpolluted site	Lobel et al., 1991
La, Gd	Common carp	*Cyprinus carpio*	0.5	Tu et al., 1994 (*)
La, Ce, Nd	Variegated scallop	*Chlamys varia*	Unpolluted site vs polluted site	Bustamante and Miramand, 2005
La, Ce, Pr, Nd, Sm	Common carp	*Cyprinus carpio*	(mixed) Ce: 0.27; La 0.30; Nd 0.29; Pr 0.06; Sm 0.25	Sun et al., 1996
La, Gd	Chlorella	*Chlorella vulgaris*	1	Sun et al., 1997
La–Lu	Fennel pondweed; Common duckweek	*Potamogeton pectinatus; Lemna minor*; Mollusc (7 spp)	Unpolluted site vs polluted site	Weltje et al., (2002)
La, Ce, Eu	Blue mussel	*Mytilus edulis*	Unpolluted site vs polluted site	Riget et al., (1996)
Sm, Eu, Gd, REEs mixture	Bacteria	*Arthrobacter nicotianae, Streptomyces albus*	66.5 µM each metal	Tsuruta, (2006)
La, Ce, Nd	Bellybutton nautilus, chambered nautilus	*Nautilus macromphalus, Nautilus pompilius*	Unpolluted site vs polluted site	Pernice et al., (2009)

(*) Data obtained from secondary references.

Terrestrial environment

Table 3.9. Eco-toxicity of REEs for mammals and vertebrates (After Pagano et al., 2019).

Species	**Main effects/Endpoints**	**References**
Rats	Yttrium-induced renal damage. Neurotoxicological effects following $LaCl_3$ oral administration. CeO_2 fine particles, by intratracheal instillation, induce: Multiple respiratory damage. Lung, liver, kidney, spleen and brain accumulation. Lung fibrosis. CeO_2 intravenous infusion results in multiple oxidative stress.	Hayashi et al., 2006; He et al., 2008; Toya et al., 2010; Nalabotu et al., 2011; Tseng et al., 2012; Geraets et al., 2012. Ma et al., 2012; Nemmar et al., 2017. Hardas et al., 2014; Kawagoe et al., 2008.
Mice	$CeCl_3$ oral administration induces lung and liver toxicity.	Kawagoe et al., 2008
Rat lung alveolar macrophages	$LaCl_3^-$, $CeCl_3^-$, and $Nd_2O_3^-$induced cytotoxicity.	Palmer et al., 1987
Mouse bone marrow cells	Intraperitoneal Pr_6O_{11} and Nd_2O_3 induce chromosomal aberrations.	Jha and Singh, 1995

Table 3.10. Eco-toxicity of REEs to plants (After Pagano et al., 2019).

	Species	Main effects/Endpoints	References
Seven plant species		Comparative toxicities to root elongation of n-CeO_2, n-La_2O_3, n-Gd_2O_3 and nano-Yb_2O_3	Ma et al., 2010
Cucumber	*Cucumis sativus*	Comparative toxicities of nanoparticulate/bulk Yb_2O_3 and $YbCl_3$	Zhang et al., 2012
Coriander	*Coriandrum sativum*	n-CeO_2-induced growth inhibition	Morales et al., 2013
Six plant species	-	Comparative phitotoxicities of Y, La and Ce	Thomas et al., 2014
Western waterweed	*Elodea nuttallii*	La-associated oxidative effects, nutrients and metabolic changes	Zhang et al., 2015a
Lactuca plants	-	Species-specific toxicity of n-CeO_2	Zhang et al., 2015b
Diatoms	*Skeletonema costatum*	Same growth inhibition induced by 13 lanthanoids	Tai et al., 2010
Broad bean	*Vicia faba*	La-associated shift from hormetic to inhibitory affects	Wang et al., 2011

Table 3.11. Exposure of agricultural plants to REEs (After Gonzalez et al., 2014).

	Exposure concentrations	Agricultural plants	Types	References
La	(100–700 μg g^{-1})	Red rice	*Oryza sativa L.*	Fashui et al., 2000
Ce	(2.5–30 μg ml^{-1})	Red rice	*Oryza sativa L.*	Fashui, 2002
La, Ce	La (0.5–25 mg L^{-1}) Ce (0.5–25 mg L^{-1})	Wheat	*Triticum aestivum L.*	Hu et al., 2002
La	(0.10–2.5 mg kg^{-1})	Maize	*Zea mays L.*	Hu et al., 2006
La, Ce	La (0.2–5 μM) Ce (0.5–5 μM)	Maize, mung bean	*Zea mays L.; Vigna radiata L.*	Diatloff et al., 2008
La, Ce, Pr, Nd, Gd	La (0.01–10 mM) REE mix (0.01–10 mM)	Durumwheat	*Triticum durum L.*	d'Aquino et al., 2009a
La, Y, Ce	La (4.4–208.7 mg kg^{-1}) Ce (25.7–1209 mg kg^{-1}) Y (10.2–910 mg kg^{-1})	Switchgrass, Common milkweed, Radish, Radish, Tomato	*Panicum virgatum L.,* *Asclepias syriaca L.,* *Desmodium canadense L.,* *Raphanus satuvus L.,* *Solanum lycopersicum L.*	Thomas et al., 2014

Table 3.12. Eco-toxicity of REEs for freshwater fish (After Pagano et al., 2019).

Species	Main effects/Endpoints	Concentration	References
Goldfish (Carassius auratus)	CeO_2-induced changes of enzymatic activities (AChE, SOD, and CAT)	20-320 mg L^{-1} CeO_2	Xia et al., 2013
Zebrafish embryos	La and Yb affect the development of zebrafish embryos	0.01-1 mmmol L^{-1} La^{3+} and Yb^{3+}	Cui et al., 2012

Table 3.13. Acute toxicity of REEs to freshwater organisms (Modified from Ng et al., 2012).

REE element	Organism	Species	Effect	Toxicity value (µg REE L^{-1})	References
Sm	Algae	*Pseudokirchneriella subcapitata*	Growth	430 – 4,300[a]	(Blaise et al., 2008)
Er	Algae	*Pseudokirchneriella subcapitata*	Growth	440 – 4,400[a]	(Blaise et al., 2008)
Ho	Algae	*Pseudokirchneriella subcapitata*	Growth	440 – 4,400[a]	(Blaise et al., 2008)
Ce	Algae	*Pseudokirchneriella subcapitata*	Growth	8,302 - 15,470[b]	(Van Hoecke et al., 2009)
La	Algae	*Selenastrum capricornutum*	Growth	450	(NICNAS, 2001)
La	Cladoceran	*Daphnia carinata*	Survival	49	(Barry and Meehan, 2000)
La	Cladoceran	*Daphnia carinata*	Survival	1232.4	(Barry and Meehan, 2000)
La	Cladoceran	*Daphnia carinata*	Survival	1180	(Barry and Meehan, 2000)
La	Cladoceran	*Daphnia carinata*	Survival	484.5	(Barry and Meehan, 2000)
La	Cladoceran	*Daphnia carinata*	Survival	43.2	(Barry and Meehan, 2000)
Ce	Cladoceran	*Daphnia carinata*	Survival	NOEC: 8139.5	(Gaiser et al., 2009)
Ce	Cladoceran	*Daphnia carinata*	Survival	NOEC: 813953	(Van Hoecke et al., 2009)
La	Cladoceran	*Daphnia carinata*	Survival	24,000	(Den-Ouden, 1995)[SD]
Ce	Cladoceran	*Daphnia carinata*	Survival	22,000	(Den-Ouden, 1995)[SD]
Pr	Cladoceran	*Daphnia carinata*	Survival	9,000	(Den-Ouden, 1995)[SD]
Nd	Cladoceran	*Daphnia carinata*	Survival	1,400	(Den-Ouden, 1995)[SD]
Sm	Cladoceran	*Daphnia carinata*	Survival	7,600	(Den-Ouden, 1995)[SD]
Gd	Cladoceran	*Daphnia carinata*	Survival	6,800	(Den-Ouden, 1995)[SD]
Dy	Cladoceran	Daphnia carinata	Survival	9,100	(Den-Ouden, 1995)[SD]
La	Cladoceran	Daphnia carinata	Survival	NOEC > 63270	(Watson-Leung, 2009)
La	Cladoceran	*Ceriodaphnia dubia*	Survival	80	(Stauber, 2000)
La	Cladoceran	*Ceriodaphnia dubia*	Survival	5,000	(NICNAS, 2001)
Sm	Cnidarian	*Hydra attenuata*	Morphology	430 – 4,300[a]	(Blaise et al., 2008)
Er	Cnidarian	*Hydra attenuata*	Morphology	440 – 4,400[a]	(Blaise et al., 2008)
Ho	Cnidarian	*Hydra attenuata*	Morphology	44 – 440[a]	(Blaise et al., 2008)
Sm	Shrimp	*Thamnocephalus platyurus*	Survival	> 43,000[a]	(Blaise et al., 2008)
Er	Shrimp	*Thamnocephalus platyurus*	Survival	> 44,000[a]	(Blaise et al., 2008)
Ho	Shrimp	*Thamnocephalus platyurus*	Survival	> 44,000[a]	(Blaise et al., 2008)
Ce	Shrimp	*Thamnocephalus platyurus*	Survival	4,069,767	(Van Hoecke et al., 2009)
La	Fish	*Melanotaenia duboulayi*	Immobilization	NOEC: 127	(Stauber, 2000)
La	Fish	*Melanotaenia duboulayi*	Immobilization	NOEC < 600	(NICNAS, 2001)
Ce	Fish	*Danio rerio*	Hatching	NOEC: 162791	(Van Hoecke et al., 2009)
La	Fish	*Danio rerio*	Survival	23,000	(Den-Ouden, 1995)[SD]
Ce	Fish	*Danio rerio*	Survival	22,000	(Den-Ouden, 1995)[SD]
Pr	Fish	*Danio rerio*	Survival	25,000	(Den-Ouden, 1995)[SD]
Nd	Fish	*Danio rerio*	Survival	21,000	(Den-Ouden, 1995)[SD]
Sm	Fish	*Danio rerio*	Survival	22,000	(Den-Ouden, 1995)[SD]
Gd	Fish	*Danio rerio*	Survival	19,000	(Den-Ouden, 1995)[SD]
Dy	Fish	*Danio rerio*	Survival	25,000	(Den-Ouden, 1995)[SD]
La	Fish	*Oncorhynchus mykiss*	Survival	NOEC > 63270	(Watson-Leung, 2009)

IC25 - The 25% maximal inhibitory concentration. EC 50 - Half maximal effective concentration. LC 50 - lethal concentration 50. *S = static; SR = static-renewal; M = measured; N = nominal; D = dissolved (pass through ≤ 0.45 µm filter); T = total; SD = secondary data, whereas unmarked references are primary data

[a] Exact endpoint values are not reported in the article, instead a range is reported according to the EU Directive 93/67/EEC

[b] EC50 range is given for nanoparticles with different sizes (14, 20, 29 nm) (Van Hoecke et al., 2009)

Table 3.14. Chronic toxicity of REEs to freshwater organisms (After Ng et al., 2012).

REE elements	Organism	Species	Hardness (mg $CaCO_3$ L^{-1})	pH	Temp	Effect	Toxicity value (µg REE L^{-1})	References
La	Duckweed	*Lemna minor*	54.8	5.1	25	Growth	NOEC: 1.39	(Weltje et al., 2002a)
La	Cladoceran	*Daphnia carinata*	160	7.5	20	Survival	LOEC: 39	(Barry and Meehan, 2000)
La	Cladoceran	*Daphnia carinata*	160	7.5	20	Age at maturity	LOEC: 39	(Barry and Meehan, 2000)
La	Cladoceran	*Daphnia carinata*	160	7.5	20	Brood size	LOEC: 30	(Barry and Meehan, 2000)
La	Cladoceran	*Daphnia magna*	88	7.6	20	Weight	8.7[a]	(Lürling and Tolman, 2010)
La	Cladoceran	*Daphnia magna*	88	7.6	20	Length	15.6[a]	(Lürling and Tolman, 2010)
La	Cladoceran	*Daphnia magna*	88	7.6	20	Survival	NOEC = 1,001	(Lürling and Tolman, 2010)
La	Cladoceran	*Daphnia magna*	88	7.6	20	Brood size	NOEC = 1,001	(Lürling and Tolman, 2010)
La	Cladoceran	*Daphnia magna*	88	7.6	20	Growth	NOEC = 1,001	(Lürling and Tolman, 2010)
Ce	Cladoceran	*Daphnia magna*	249	7.4	20	Survival	30035 –57872[b]	(Van Hoecke et al., 2009)
Ce	Cladoceran	*Daphnia magna*	249	7.4	20	Brood size	16686 –34756[b]	(Van Hoecke et al., 2009)
Nd	Cladoceran	*Daphnia magna*	210	7.6-8.7	NA	Survival	NOEC: 1,600	(Den-Ouden, 1995)[SD]
Nd	Cladoceran	*Daphnia magna*	210	7.6-8.7	NA	Fitness	NOEC: 1,600	(Den-Ouden, 1995)[SD]
Dy	Cladoceran	*Daphnia magna*	210	NA	NA	Reproduction[d]	NOEC < 200	(Den-Ouden, 1995)[SD]
Dy	Cladoceran	*Daphnia magna*	210	7.9-8.5	NA	Survival	NOEC > 2,100	(Den-Ouden, 1995)[SD]
Dy	Cladoceran	*Daphnia magna*	210	7.9-8.5	NA	Fitness	NOEC > 2,100	(Den-Ouden, 1995)[SD]
La	Cladoceran	*Ceriodaphnia dubia*	84.9	7.9	25	Survival	842	(NICNAS, 2001)
La	Cladoceran	*Ceriodaphnia dubia*	84.9	7.9	25	Brood size	154	(NICNAS, 2001)
La	Cladoceran	*Ceriodaphnia dubia*	40-48	7.9	25	Survival	510	(Borgmann et al., 2005)
La	Cladoceran	*Ceriodaphnia dubia*	40-48	7.9	25	Brood size	430	(Borgmann et al., 2005)
Ce	Amphipod	*Hyallela azteca*	124	7.2-9.0	24-25	Survival	651	(Borgmann et al., 2005)
Ce	Amphipod	*Hyallela azteca*	18	6.4-8.7	24-25	Survival	32	(Borgmann et al., 2005)
Dy	Amphipod	*Hyallela azteca*	124	7.2-9.0	24-25	Survival	897	(Borgmann et al., 2005)
Dy	Amphipod	*Hyallela azteca*	18	6.4-8.7	24-25	Survival	162	(Borgmann et al., 2005)
Er	Amphipod	*Hyallela azteca*	124	7.2-9.0	24-25	Survival	929	(Borgmann et al., 2005)
Er	Amphipod	*Hyallela azteca*	18	6.4-8.7	24-25	Survival	191	(Borgmann et al., 2005)
Eu	Amphipod	*Hyallela azteca*	124	7.2-9.0	24-25	Survival	717	(Borgmann et al., 2005)
Eu	Amphipod	*Hyallela azteca*	18	6.4-8.7	24-25	Survival	112	(Borgmann et al., 2005)
Gd	Amphipod	*Hyallela azteca*	124	7.2-9.0	24-25	Survival	599	(Borgmann et al., 2005)
Gd	Amphipod	*Hyallela azteca*	18	6.4-8.7	24-25	Survival	150	(Borgmann et al., 2005)
Ho	Amphipod	*Hyallela azteca*	124	7.2-9.0	24-25	Survival	755	(Borgmann et al., 2005)
Ho	Amphipod	*Hyallela azteca*	18	6.4-8.7	24 - 25	Survival	143	(Borgmann et al., 2005)
La	Amphipod	*Hyallela azteca*	124	7.2-7.9	24 - 25	Survival	1665	(Borgmann et al., 2005)
La	Amphipod	*Hyallela azteca*	18	6.4-8.7	24 - 25	Survival	18	(Borgmann et al., 2005)
Lu	Amphipod	*Hyallela azteca*	124	7.2-9.0	24 - 25	Survival	1054	(Borgmann et al., 2005)
Lu	Amphipod	*Hyallela azteca*	18	6.4-8.7	24-25	Survival	29	(Borgmann et al., 2005)
Nd	Amphipod	*Hyallela azteca*	124	7.2-9.0	24-25	Survival	511	(Borgmann et al., 2005)
Nd	Amphipod	*Hyallela azteca*	18	6.4-8.7	24-25	Survival	55	(Borgmann et al., 2005)
Pr	Amphipod	*Hyallela azteca*	124	7.2-9.0	24-25	Survival	441	(Borgmann et al., 2005)
Pr	Amphipod	*Hyallela azteca*	18	6.4-8.7	24-25	Survival	35	(Borgmann et al., 2005)

Sm	Amphipod	*Hyallela azteca*	124	7.2-9.0	24-25	Survival	846	(Borgmann et al., 2005)
Sm	Amphipod	*Hyallela azteca*	18	6.4-8.7	24-25	Survival	74	(Borgmann et al., 2005)
Tb	Amphipod	*Hyallela azteca*	124	7.2-9.0	24-25	Survival	693	(Borgmann et al., 2005)
Tb	Amphipod	*Hyallela azteca*	18	6.4-8.7	24-25	Survival	84	(Borgmann et al., 2005)
Tm	Amphipod	*Hyallela azteca*	124	7.2-9.0	24-25	Survival	739	(Borgmann et al., 2005)
Tm	Amphipod	*Hyallela azteca*	18	6.4-8.7	24-25	Survival	0.01	(Borgmann et al., 2005)
Yb	Amphipod	*Hyallela azteca*	124	7.2-9.0	24-25	Survival	278	(Borgmann et al., 2005)
Yb	Amphipod	*Hyallela azteca*	18	6.4-8.7	24-25	Survival	69	(Borgmann et al., 2005)
La	Amphipod	*Hyallela azteca*	145-208	8.0-8.6	23	Survival	NOEC: 7	(Watson-Leung, 2009)
La	Amphipod	*Hyallela azteca*	145-208	8.0-8.6	23	Growth	NOEC: 7	(Watson-Leung, 2009)
La	Insect	*Chironomus dilutus*	138-179	8.1-9.1	21-23	Survival	NOEC: 880	(Watson-Leung, 2009)
La	Insect	*Chironomus dilutus*	138-179	8.1-9.1	21-23	Weight	NOEC: 880	(Watson-Leung, 2009)
La	Insect	*Hexagenia spp.*	101-125	8.1-8.5	22	Survival	NOEC < 3	(Watson-Leung, 2009)
La	Insect	*Hexagenia spp.*	101-125	8.1-8.5	22	Weight	NOEC < 3	(Watson-Leung, 2009)
Dy	Fish	*Danio rerio*	210	NA	NA	Weight	NOEC: 3,000	(Den-Ouden, 1995)[SD]
Dy	Fish	Danio rerio	210	6.7-8.4	NA	Survival	NOEC: 2,600	(Den-Ouden, 1995)[SD]
Dy	Fish	Danio rerio	210	6.7-8.4	NA	Fitness	NOEC: 3,800	(Den-Ouden, 1995)[SD]

EC 50 - Half maximal effective concentration. LC 50 - lethal concentration 50. *M = measured; N = nominal; D = dissolved (pass through ≤ 0.45 µm filter), T = total; SD = secondary data, whereas unmarked references are primary data. NOEC = No observed effect concentration; [a] EC50 of La is calculated from the measured percentage of La (0.001 %) leached from Phoslock® (Lürling and Tolman, 2010); [b]EC50 range is given for nanoparticles with different sizes (14, 20, 29 nm) (Van Hoecke et al., 2009).

Table 3.15. Bioaccumulation of REEs by freshwater organisms (After Ng et al., 2012).

REE elements	Organism	Species	Duration	Hardness (mg $CaCO_3$/L)	pH	Tissue	Exposure conc.[a] (µg l^{-1})	Tissue conc.[b] (µg REE g^{-1})	BCF[c] (L kg^{-1})	References
La	Duckweed	Lemna minor	48 h	54.8	5.1-5.6	Whole plant	1.39	1.7	826.3[d]	(Weltje et al., 2002a)
La	Fish	*Cyprinus carpio*	45 d	53-60	6	Skeleton	500	2.8	5.6	(Tu et al., 1994)
La	Fish	*Cyprinus carpio*	45 d	53-60	6	Muscle	500	1.3	2.6	(Tu et al., 1994)
La	Fish	*Cyprinus carpio*	45 d	53-60	6	Gill	500	7	13.9	(Tu et al., 1994)
La	Fish	*Cyprinus carpio*	45 d	53-60	6	Internal organs	500	38.9	77.8	(Tu et al., 1994)
Gd	Fish	*Cyprinus carpio*	45 d	53-60	6	Skeleton	500	2.3	4.6	(Tu et al., 1994
Gd	Fish	*Cyprinus carpio*	45 d	53-60	6	Muscle	500	1.6	3.2	(Tu et al., 1994
Gd	Fish	*Cyprinus carpio*	45 d	53-60	6	Gill	500	5.3	10.7	(Tu et al., 1994
Gd	Fish	*Cyprinus carpio*	45 d	53-60	6	Internal organs	500	42.3	84.6	(Tu et al., 1994
Ce	Fish	*Cyprinus carpio*	43 d	53-60	6	Muscle	270	0.05	0.2	(Sun et al., 1996)
Ce	Fish	*Cyprinus carpio*	43 d	53-60	6	Skeleton	270	1.6	5.9	(Sun et al., 1996)
Ce	Fish	*Cyprinus carpio*	43 d	53-60	6	Gill	270	3.5	12.8	(Sun et al., 1996)
Ce	Fish	*Cyprinus carpio*	43 d	53-60	6	Internal organs	270	164.2	608	(Sun et al., 1996)
La	Fish	*Cyprinus carpio*	43 d	53-60	6	Muscle	300	0.2	0.8	(Sun et al., 1996)
La	Fish	*Cyprinus carpio*	43 d	53-60	6	Skeleton	300	1.1	3.7	(Sun et al., 1996)
La	Fish	*Cyprinus carpio*	43 d	53-60	6	Gill	300	4.1	13.5	(Sun et al., 1996)
La	Fish	*Cyprinus carpio*	43 d	53-60	6	Internal organs	300	180.6	602	(Sun et al., 1996)
La	Fish	*Cyprinus carpio*	43 d	53-60	6	Muscle	250	0.3	1.1	(Sun et al., 1996)
La	Fish	*Cyprinus carpio*	43 d	53-60	6	Skeleton	250	1.4	5.5	(Sun et al., 1996)
La	Fish	*Cyprinus carpio*	43 d	53-60	6	Gill	250	4	16	(Sun et al., 1996)
La	Fish	*Cyprinus carpio*	43 d	53-60	6	Internal organs	250	176.3	705	(Sun et al., 1996)
Ce	Fish	*Danio rerio*	7 d	97.7	7.2	Liver	500	1350[c]	2700[e]	(Johnston et al., 2010)

*All references are primary data. [a] Concentration is nominal and total of lanthanide in the exposure medium. [b] Tissue concentration is expressed in wet weight, except otherwise described. [c] BCF is calculated from total or nominal REE concentrations in the water and expressed in wet weight, except otherwise described. [d] BCF is dynamic i.e., accumulation not in equilibrium yet, is calculated as described in Weltje et al (2002) and expressed based on fresh weight of plant. [e] Values are calculated from dry weight of tissue.

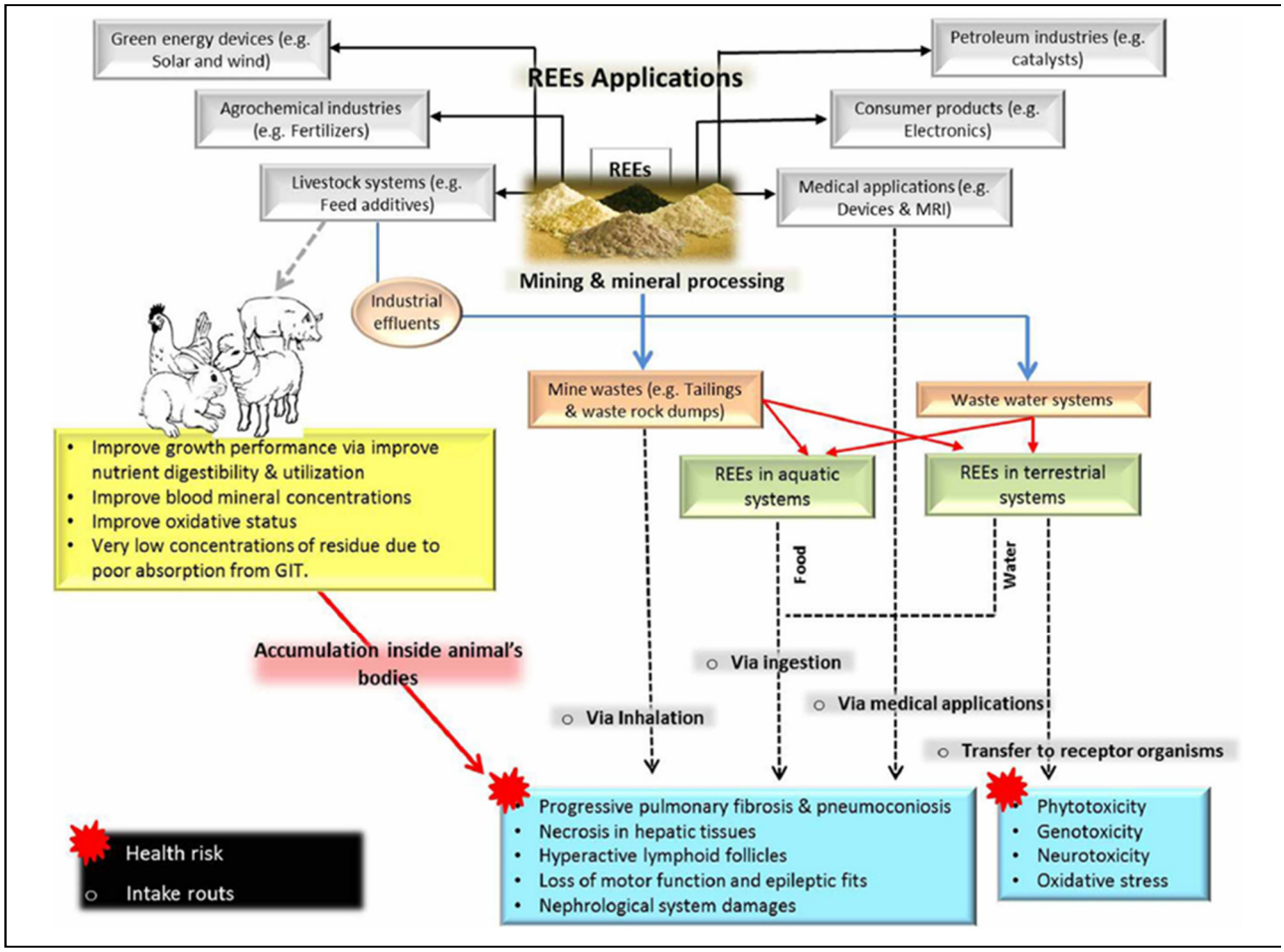

Figure 1. Use of REEs and their health risk (After Abdelnour et al., 2019).

Table 3.18. Toxicity of REEs to human (After Pagano et al., 2019).

Exposed populations	Main Effects/Endpoints	References
Children and mothers living in a REE mining area (Jiangxi, China)	The REE hair level of young children in mining area was significantly higher than in their mothers	Peng et al., 2003
Residents in South Jiangxi Province, China	Significantly lower serum total protein and globulin from both HREE and LREE areas, and albumin from the LREE area	Zhu et al., 2005
Housewives exposed to indoor air pollution	Excess REE levels were found in scalp hair as related to indoor air pollution and associated with hypertension risk	Wang et al., 2017a, 2018
Residents in Baiyun Obo mining area, China	Excess scalp hair levels of REEs, heavy metals and uranium	Hao et al., 2015
Fujian and Shandong Provinces	Excess REE accumulation in soil and vegetables	Li et al., 2013
Zhuzhou (industrial city in central China)	Excess REE levels in street dust and associated health risk	Sun et al., 2017

Table 3.19. Eco-toxicity of REEs for human cells (After Pagano et al., 2019).

Species	Main effects/Endpoints	References
Human spermatozoa	$nCeO_2$	Préaubert et al., 2018
BEAS – 2B cells	$nCeO_2$-induced oxidative stress	Park et al., 2008
HepG2 and HT – 29 cell lines	$CeCl_3$ and $LaCl_3$ affect gene regulation detected by RT-PCR based arrays	Benedetto et al., 2018

HepG2 is a human liver cancer cell line.

HT – 29 is a human colon adenocarcinoma cell line.

Table 3.21. REE toxicity (After EU ECHA).

Cerium (Ce)	
Hazard for Aquatic Organisms	
Freshwater	600 µg L^{-1}
Marine water	60.9 µg L^{-1}
Sewage treatment plant (STP)	60.9 mg L^{-1}
Short–term toxicity to fish	LL50 (4 days) 100 mg L^{-1} NOELR (4 days) 100 mg L^{-1}
Short–term toxicity to aquatic invertebrates	LL50 (48 h) 100 mg L^{-1} NOELR (48 h) 100 mg L^{-1}
Long–term toxicity to aquatic invertebrates	NOELR (21 days) 100 mg L^{-1} LOELR (21 days) 100 mg L^{-1}
Toxicity to aquatic algae and cyanobacteria	EL10 (72 h) 67 mg L^{-1} EL50 (72 h) 100 mg L^{-1} NOELR (72 h) 25 mg L^{-1} LOELR (72 h) 50 mg L^{-1}
Toxicity to microorganisms	EC50 (3 h) 1 g L^{-1} NOEC (3 h) 1 g L^{-1}
Gadolinium (Gd)	
Short–term toxicity to aquatic invertebrates	EC50 (48 h) 430 µg L^{-1} NOEC (48 h) 430 µg L^{-1}
Acute toxicity	LD50 2 000 mg kg^{-1} bw (rat)
Neodymium (Nd)	
Short–term toxicity to aquatic invertebrates	EC50 (48 h) 430 µg L^{-1} NOEC (48 h) 430 µg L^{-1}
Yttrium (Y)	
Short–term toxicity to fish	LC50 (4 days) 610 µg L^{-1}
Short–term toxicity to aquatic invertebrates	EL50 (48 h) 100 mg L^{-1}

APPENDIX B Naturally Occuring Radionuclides

Table 4.1. Decay of radionuclides of natural origin and physical data (UNSCEAR, 2000).

Element	Isotope	Half-life	Decay mode	Energy released (MeV)
^{238}U series				
Uranium	^{238}U	4.47 10^{9} a	alpha (100%)	4.3
Thorium	^{234}Th	24.1 d	beta (100%)	0.3
Protactinium	^{234m}Pa	1.17 m	beta (99.8%), IT	2.3 (beta); 0.07 (IT)
Uranium	^{234}U	2.45 10^{5} a	alpha (100%)	4.9
Thorium	^{230}Th	7.54 10^{4} a	alpha (100%)	4.8
Radium	^{226}Ra	1600 a	alpha (100%)	4.9
Radon	^{222}Rn	3.824 d	alpha (100%)	5.6
Polonium	^{218}Po	3.05 m	alpha (99.98%), beta (0.02%)	6.1 (alpha); 0.3 (beta)
Lead	^{214}Pb	26.8 m	beta (100%)	1.0
Bismuth	^{214}Bi	19.9 m	alpha (0.02), beta (99.98%)	5.6 (alpha); 3.3 (beta)
Polonium	^{214}Po	164 µs	alpha (100%)	7.9
Lead	^{210}Pb	22.3 a	beta (100%)	0.06
Bismuth	^{210}Bi	5.013 d	beta (100%)	1.2
Polonium	^{210}Po	138.4 d	alpha (100%)	5.4
Lead	^{206}Pb	stable		
^{232}Th series:				
Thorium	^{232}Th	1.405 10^{10} a	alpha (100%)	4.1
Radium	^{228}Ra	5.75 a	beta (100%)	0.05
Actinium	^{228}Ac	6.15 h	beta (100%)	2.1
Thorium	^{228}Th	1.912 a	alpha (100%)	5.5
Radium	^{224}Ra	3.66 d	alpha (100%)	5.8
Radon	^{220}Rn	55.6 s	alpha (100%)	6.4
Polonium	^{216}Po	0.145 s	alpha (100%)	6.9
Lead	^{212}Pb	10.64 h	beta (100%)	0.6
Bismuth	^{212}Bi	60.55 m	alpha (36%), beta (64%)	6.2 (alpha) 2.3 (beta)
Polonium	^{212}Po	0.299 µs	alpha (100%)	8.8
Thalium	^{208}Tl	3.053 m	beta (100%)	1.8
Lead	^{208}Pb	stable		
^{235}U series:				
Uranium	^{235}U	7.038 10^{8} a	alpha (100%)	4.7
Thorium	^{231}Th	25.52 h	beta (100%)	0.4
Protactinium	^{231}Pa	32760 a	alpha (100%)	5.1
Actinium	^{227}Ac	21.77 a	alpha (1.4%), beta 98.6%)	5.0 (alpha) 0.05 (beta)
Thorium	^{227}Th	18.72 d	alpha (100%)	6.1
Francium	^{223}Fr	21.8 m	alpha (0.006%) beta (99.994%)	5.3 (alpha) 1.1 (beta)
Radium	^{223}Ra	11.44 d	alpha (100%)	5.9
Radon	^{219}Rn	3.96 s	alpha (100%)	6.9
Polonium	^{215}Po	1.781 ms	alpha (99.99977%) beta (0.00023%)	7.5 (alpha) 0.7 (beta)
Lead	^{211}Pb	36.1 m	beta (100%)	1.4

Bismuth	^{211}Bi	2.14 m	alpha (99.7%), beta (0.3%)	6.8 (alpha) 0.6 (beta)
Thalium	^{207}Tl	4.77 m	beta (100%)	1.4
Lead	^{207}Pb	stable		

IT: Internal transition.

Background activity concentrations of NOR

Table 4.2. Activity concentrations of NOR in surface seawater, and marine organisms in the Arctic (EPIC 2001).

Radionuclide	Sea water (Bq m^{-3})	Crustaceans (Bq kg^{-1})	Molluscs (Bq kg^{-1})	Fish (Bq kg^{-1})
^{210}Po	0.2 - 1.6	15 - 60	15 – 41	0.02 - 5 (muscles) 7.4 - 33 (liver) 0.7 - 8 (bone)
^{210}Pb	0.4 - 2.5	1.5 - 2.6	0.2 - 0.4	0.007 - 0.09 (muscles) 0.4 - 0.9 (liver) 0.3 - 4.8 (bone)
^{226}Ra	1.5 - 1.7			0.007 - 0.2 (flesh)
^{238}U	44			0.0025 - 1.1

Table 4.3. Activity concentrations of NOR in freshwater, freshwater sediments and freshwater fish in the Arctic (EPIC 2001).

Radionuclide		(Bq kg^{-1} or Bq L^{-1})	Range (Bq kg^{-1} or Bq L^{-1})
^{210}Pb	Water	0.004	0.0009-0.017
^{210}Po	Water	0.002	0.0002-0.008
^{232}Th	Water	0.0003	0.00005-0.004
^{238}U	Water	0.003	0.0005-0.01
^{226}Ra	Bottom sediments	45	12-170
^{232}Th	Bottom sediments	40	14-94
^{238}U	Bottom sediments	50	10-100
^{210}Pb	Fish	0.006	0.004-0.007
^{210}Po	Fish	0.6	0.14-1.15
^{232}Th	Fish	0.02	-
^{238}U	Fish	0.03	-

Table 4.5. Radiation effects on terrestrial animals (Russian/FSU data), Source: EPIC 2011, ANNEX A. EPIC Database.

Type of organism	Latin name Common name	Location	Nuclide	Activity in soil (Bq m^{-2})	Activity in tissues (Bq kg^{-1})	Dose rate (Gy d^{-1})	Effect	Effect code
Mammal	*Microtus oeconomus* P.Tundra vole	Local area with high natural radioactivity (Komi AR of Russia, radium stationary site, 3 hectares)	Ra-226 Po-210 U-238 Th-232	70.7 Bq Ra g^{-1} soil ash; 25.7 Bq Po210 g^{-1}; 50 Bq U kg^{-1}	2.5 x E^{-12} (g Ra/g live weight)	External: 8000 microR hr^{-1} (300 mGy y^{-1}); Internal: 12-40 mGy y^{-1}; from Rn gas 350 mGy per day	Period of reproduction of voles reduces to 4 months (control – 6 months). Number of females involved in reproduction reduced by 2-5 times as compared with control. Number of embryos per pregnant female decreased by a factor 2 (6.3 in control, 3.8 in the radium stationary site, N= 2900.The population size was supported by migrations of animals from neighboring areas, disturbed by regular hay cuttings. The average number of migrants found in the radioactive sites was about 30%. After 1 month of living on contaminated site, newcomers became as much contaminated as residential animals.	REPR
Mammal	*Microtus oeconomus* P.Tundra vole	Local area with high natural radioactivity (Komi AR of Russia, uranium-radium stationary site, 12 hectares)	Ra-226, Po-210, U-238, Th-232	up to 70000 Bq kg^{-1} (Ra-226); 25700 Bq kg^{-1} (Po-210); 140 Bq kg^{-1} (U-238); 40 Bq kg^{-1} (Th-232)	1.5E^{-12} (g Ra g^{-1} live weight); 2.3E^{-8} (g U g-1 weight)	External 4000 microR hr^{-1}; internal 40 mGy y^{-1}; from Rn gas 350 mGy per Day	Young mice had considerably lower weight of liver in % to body weight (N>1000). Differences with control were about 2 times for mice of 1 month old. The effect became statistically insignificant for mice over 1 year old. Histological analysis revealed numerous abnormalities in liver structure of adult mice - spots of necrosis, hemorrhagic, glycemic infiltrations. Synthesis of albumins by liver was lower than in control (p<0.01). Abnormalities in the blood of mice: leucocytes decreased by 48 -52%; erythrocytes decreased by 20-22%; hemoglobin - by 15-17%. (N>1000, results were statistically reliable). Changes in blood composition of 3-5% of mice were large enough to be considered as signs of radiation sickness.	MB
Mammal	*Microtus oeconomus* P.Tundra vole	Local area with high natural radioactivity (Komi AR of Russia, radium stationary site, 3 hectares)	Ra-226, Po-210, U-238, Th-232.	70.7 Bq g^{-1} soil ash (Ra); 25.7 Bq g^{-1} (Po-210); 50 Bq kg^{-1} (U).	2.5 x 10^{-12} (g Ra g^{-1} live weight)	External 8000 microR hr^{-1} (300 mGy y^{-1}); internal 12-40 mGy y^{-1}; from Rn gas 350 mGy per day	The sexual maturity of young males of voles was considerably inhibited up to 9 months of age. In radium site 50% of males reached maturity at age 1 month and 100 % at age over 9 months (N=540). In control about 50% of males reached maturity at age of 1 month and 100% at age over 3 months. (Lifespan of voles is 1-1.5 years). Lower gonad weight and numerous degenerative transformations in gonads were observed.	REPR

Table 4.5. Radiation effects on terrestrial animals (Russian/FSU data), Source: EPIC 2011, ANNEX A. EPIC Database.

Type of organism	Latin name Common name	Location	Nuclide	Activity in soil (Bq m^{-2})	Activity in tissues (Bq kg^{-1})	Dose rate (Gy d^{-1})	Effect	Effect code
Mammal	*Microtus oeconomus* P.Tundra vole	Local area with high natural radioactivity (Komi AR of Russia, radium stationary site, 3 hectares)	Ra-226 Po-210 U-238 Th-232	70.7 Bq Ra g^{-1} soil ash; 25.7 Bq Po210 g^{-1}; 50 Bq U kg^{-1}	2.5 x E^{-12} (g Ra/g live weight)	External: 8000 microR hr^{-1} (300 mGy y^{-1}); Internal: 12-40 mGy y^{-1}; from Rn gas 350 mGy per day	Period of reproduction of voles reduces to 4 months (control – 6 months). Number of females involved in reproduction reduced by 2-5 times as compared with control. Number of embryos per pregnant female decreased by a factor 2 (6.3 in control, 3.8 in the radium stationary site, N= 2900.The population size was supported by migrations of animals from neighboring areas, disturbed by regular hay cuttings. The average number of migrants found in the radioactive sites was about 30%. After 1 month of living on contaminated site, newcomers became as much contaminated as residential animals.	REPR
Mammal	*Microtus oeconomus* P.Tundra vole	Local area with high natural radioactivity (Komi AR of Russia, uranium-radium stationary site, 12 hectares)	Ra-226, Po-210, U-238, Th-232	up to 70000 Bq kg^{-1} (Ra-226); 25700 Bq kg^{-1} (Po-210); 140 Bq kg^{-1} (U-238); 40 Bq kg^{-1} (Th-232)	1.5E^{-12} (g Ra g^{-1} live weight); 2.3E^{-8} (g U g-1 weight)	External 4000 microR hr^{-1}; internal 40 mGy y^{-1}; from Rn gas 350 mGy per Day	Young mice had considerably lower weight of liver in % to body weight (N>1000). Differences with control were about 2 times for mice of 1 month old. The effect became statistically insignificant for mice over 1 year old. Histological analysis revealed numerous abnormalities in liver structure of adult mice - spots of necrosis, hemorrhagic, glycemic infiltrations. Synthesis of albumins by liver was lower than in control ($p<0.01$). Abnormalities in the blood of mice: leucocytes decreased by 48 -52%; erythrocytes decreased by 20-22%; hemoglobin - by 15-17%. (N>1000, results were statistically reliable). Changes in blood composition of 3- 5% of mice were large enough to be considered as signs of radiation sickness.	MB
Mammal	*Microtus oeconomus* P.Tundra vole	Local area with high natural radioactivity (Komi AR of Russia, radium stationary site, 3 hectares)	Ra-226, Po-210, U-238, Th-232.	70.7 Bq g^{-1} soil ash (Ra); 25.7 Bq g^{-1} (Po-210); 50 Bq kg^{-1} (U).	2.5 x 10^{-12} (g Ra g^{-1} live weight)	External 8000 microR hr^{-1} (300 mGy y^{-1}); internal 12-40 mGy y^{-1}; from Rn gas 350 mGy per day	The sexual maturity of young males of voles was considerably inhibited up to 9 months of age. In radium site 50% of males reached maturity at age 1 month and 100 % at age over 9 months (N=540). In control about 50% of males reached maturity at age of 1 month and 100% at age over 3 months. (Lifespan of voles is 1-1.5 years). Lower gonad weight and numerous degenerative transformations in gonads were observed.	REPR

Table 4.6. Radiation effects on terrestrial animals (Russian/FSU data), Source: EPIC 2011, ANNEX A. EPIC Database.

Type of organism	Latin name Common name	Location	Nuclide	Dose, (Gy)	Effect	Effect code
Mammal	*Microtus oeconomus* P. Tundra vole	Local area with high natural radioactivity (Komi AR of Russia, Ukhta region; radium stationary sites, 1 and 12 hectares; uranium-radium stationary site, 3 hectare)	Ra-226, Po-210, U-238, Th-232.	External 3-30 mGy y^{-1}; internal 12-40 mGy y^{-1}; from Rn gas 13 mGy per year	Negative changes in blood of young voles (1 month old). Main characteristics of blood were worse than in control: hemoglobin (gramm %) 13.6±0.6; erythrocytes (per mm^3) (7.3±0.3) E^6; leucocytes (per mm^3) (4±0.5) E^3. Control numbers: 16.3±0.2; (9.2±0.3) E^6. (7.6±0.2) E^3, respectively. Differences were statistically reliable (N=5000, P<0.001) Negative changes in blood of adult voles (3 months old), signs of chronic radiation disease. Main characteristics of blood became near normal, however the leuco-formulae were all wrong: young leucocytes were 150-190% of the control, large forms of lymphocytes were 200% of the control, number of neutrophils exceeded the number of lymphocytes (in the control inverse proportion), reticulocytosis (up to 85-95%). Differences were statistically reliable (N=5000, P<0.001).	MB
Mammal	*Microtus oeconomus* P. Tundra vole	Local area with high natural radioactivity (Komi AR of Russia, Ukhta region; radium stationary sites, 1 and 12 hectares)	Ra-226, Po-210, U-238, Th-232	External 3-30 mGy y^{-1}; internal 12-40 mGy y^{-1}; from Rn gas 13 mGy per year	Abnormalities in erythrocytes in the blood of voles from radium sites: micronuclei in erythrocytes were found in 2-9% of animals (N=5000); percentage of abnormal erythrocytes in blood of these animals was 0.2-0.4% (results statistically reliable). Micronuclei in erythrocytes were not found in the control voles. Presence of micronuclei is indicator of somatic mutation. Abnormalities in lymphocytes in the blood of voles from uranium-radium sites: micronuclei in lymphocytes were found in 22-82.6% of animals in different years (N total =5000); percentages of abnormal lymphocytes in white blood of these animals were 21-40% (results statistically reliable). Presence of micronuclei is indicator of somatic mutation. Micronuclei in lymphocytes were not found in the control voles.	CG

Table 4.7. Radiation effects on terrestrial animals (Russian/FSU data), Source: EPIC 2011, ANNEX A. EPIC Database.

Type of organism	Latin name Common name	Location	Nuclide	Activity in soil (Bq m^{-2})	Activity in tissues (Bq kg^{-1})	Dose (Gy)	Effect	Effect code
Mammal	*Lutra Otter* lutra.	Local area with high natural radioactivity (Komi AR of Russia, rivers within the thorium area, northern taiga). Ecological and Radio-ecological studies of 1950s.	Th, Ra, U, Rn	riverside soil: Th ($10E^{-3}$ g g^{-1}, d.w.); Ra:($4E^{-12}$ g g^{-1}); U:($3E^{-6}$ g g^{-1}); river water: Rn ($1E^{-8}$ g L^{-1}); Ra ($8E^{-12}$ g L^{-1}); U ($1E^{-8}$ g L^{-1}); Th ($1.5E^{-7}$ g L^{-1})	Ra: $90E^{-14}$ g/g W.W. (bones); (20-30) E^{-14} g g^{-1} (liver, kidney). U: $2E^{-7}$ g g^{-1} (kidney); $2E^{-8}$ g g^{-1}(bones). Th: $2E^{-6}$ g g^{-1} (bones, kidney), $1E^{-6}$ (muscle, liver)	External gamma radiation from soil up to 1000 microR h^{-1}	Weight of adult otters permanently lived within area of high thorium concentrations was somewhat lower (7.4 ± 0.6 kg, N=4) than that in the control (8.3 ± 0.8 kg, N=3) summer period. Exposed otters had less numbers and shorter periods of swimmings in the river per day comparing with control animals. The total time spent in water per day was shorter than that in control (exposed otters spent in water 4.9± 0.3 hours per day (N=4); control 5.5 ± 0.17 hours day^{-1} (N=3)). Exposed otters had less numbers and shorter periods of swimmings in the river per day comparing with control animals. The total time spent in water per day was shorter than that in control (exposed otters spent in water 4.9± 0.3 hours per day (N=4); control 5.5 ± 0.17 hours day^{-1} (N=3). Measurements of hair density at different parts of otter's body did not revealed significant differences with the control in hair density on the back and sides of animals. However, for one male otter from 4 animals lived within radioactive local area, the density of hair on belly was about 2 times lower than that in the control. Microscopic study of skin and hair cover revealed un-development or atrophy of some hair follicules in belly side of animal. Decrease of hair density could lead to a worse thermoregulation of animals, which resulted in shorter periods of swimming/hunting and decrease in weight of animals.	MB

Table 4.8. Radiation effects on terrestrial animals (Russian/FSU data), Source: EPIC 2011, ANNEX A. EPIC Database.

Type of organism	Latin name Common name	Impact	Nuclide	Activity in soil (Bq m^{-2})	Activity in tissues (Bq kg^{-1})	Dose (Gy)	Effect	Effect code
Mammal	*Lutra Otter* lutra.	Local area with high natural radioactivity (Komi AR of Russia, rivers within the thorium area, northern taiga). Ecological and Radio-ecological studies of 1950s.	Th, Ra, U, Rn	riverside soil: Th ($10E^{-3}$ g g^{-1}, d.w.); Ra:($4E^{-12}$ g g^{-1}); U:($3E^{-6}$ g g^{-1}); river water: Rn ($1E^{-8}$ g L^{-1}); Ra ($8E^{-12}$ g L^{-1}); U ($1E^{-8}$ g L^{-1}); Th ($1.5E^{-7}$ g L^{-1})	Ra: $90E^{-14}$ g g^{-1}, w.w. (bones); (20-30) E^{-14} g g^{-1} (liver, kidney). U: $2E^{-7}$ g g^{-1} (kidney); $2E^{-8}$ g g^{-1} (bones). Th: $2E^{-6}$ g g^{-1} (bones, kidney), $1E^{-6}$ (muscles, liver)	External gamma radiation from soil up to 1000 microR h^{-1}	The density of otter population in thorium area was about 33% lower than that in the control (in thorium area there was 0.31 otter animals per 1 km of river; in the control 0.21 otters per 1 km of river.	MT, ECOL
Bird	Grouse birds (big): *Tetrao urogallis* (great grouse) *Lyrurus tetrix* (black grouse)	Local area with high natural radioactivity (Komi AR of Russia, rivers within the thorium area, northern taiga). Ecological and Radio-ecological studies of 1950-1963. Total number of grouse birds analyzed: N=840 (control), N=340 in radioactive area	Th, Ra, U, Rn		U: $80E^{-9}$ g g^{-1}, w.w. (bones); $120E^{-9}$ (internal organs); (12-15) E^{-9} (muscle); Ra: (520-590) E^{-14} g g^{-1} (bones); $300E^{-14}$ (internal organs); (35-40) E^{-14} (muscle); Th: (80-120) E^{-8} g g^{-1} (bones); (120-150) E^{-8} g g^{-1} (internal organs); (200-250) E^{-8} g g^{-1} (muscle).	External gamma radiation from soil up to 1000 microR/ h	Within radioactive area, the average weights of big grouse birds (wood and black grouses) were somewhat lower than those in control. Average weights of wood grouses from irradiated population (N=21): male 3.5 kg, female 1.7 kg (control 4.3 and 2 kg respectively, N=186). Average weights of black grouses from irradiated population (N=l 1): male 1.1 kg, female 0.9 kg (control 1.4 and 1.05 kg respectively, N=45). Within radioactive area, the infestations of big grouse birds (wood and black grouses) with parasites of feather and gastro-intestine were higher than those in control. Exposed birds: 10- 15% free from feather parasites, 30% free from endo-parasites (Nematode), in control 30-50% and 60-70% respectively. Heavily infested with feather parasites were 35-50% of exposed birds (control 8-9%). Heavily infested with endo-parasites (Nematode) were 25-30% of exposed birds (control 2-7%).	MB

Table 4.9. Radiation effects on terrestrial animals (Russian/FSU data), Source: EPIC 2011, ANNEX A. EPIC Database.

Type of organism	Latin name Common name	Impact	Nuclide	Activity in tissues (Bq kg^{-1})	Dose (Gy)	Effect	Effect code
Bird	Grouse birds (big): *Tetrao urogallis* (great grouse) *Lyrurus tetrix* (black grouse)	Local area with high natural radioactivity (Komi AR of Russia, rivers within the thorium area, northern taiga). Ecological and Radio-ecological studies of 1950-1963. Total number of grouse birds analyzed: N=840 (control), N=340 in radioactive area	Th, Ra, U, Rn.	U: $80E^{-9}$ g g^{-1}, w.w. (bones); $120E^{-9}$ (internal organs); (12-15) E^{-9} (muscle); Ra: (520-590) E^{-14} g g^{-1} (bones); $300E^{-14}$ (internal organs); (35-40) E^{-14} (muscle); Th: (80-120) E^{-8} g g^{-1} (bones); (120-150) E^{-8} g g^{-1} (internal organs); (200-250) E^{-8} g g^{-1} (muscle).	External gamma radiation from soil up to 1000 microR h^{-1}	The populations of big grouse birds within the thorium area were smaller than those in the control. In thorium area the numbers of great grouses per lkm2 were 13-25% lower, and black grouse 14-35% lower than those in the control.	MT, ECOL
Bird	Grouse birds (big): *Tetrao urogallis* (great grouse) *Lyrurus tetrix* (black grouse)	Local area with high natural radioactivity (Komi AR of Russia, rivers within the thorium area, northern taiga). Ecological and Radio-ecological studies of 1950-1963. Total number of grouse birds analyzed: N=840 (control), N=340 in radioactive area	Th, Ra, U, Rn.	U: $80E^{-9}$ g g^{-1}, w.w. (bones); $120E^{-9}$ (internal organs); (12-15) E^{-9} (muscle); Ra: (520-590) E^{-14} g g^{-1} (bones); $300E^{-14}$ (internal organs); (35-40) E^{-14} (muscle); Th: (80-120) E^{-8} g g^{-1} (bones); (120-150) E^{-8} g g^{-1} (internal organs); (200-250) E^{-8} g g^{-1} (muscle).	External gamma radiation from soil up to 1000 microR h^{-1}	The populations of big grouse birds within the thorium area were smaller than those in the control. In thorium area the numbers of great grouses per lkm2 were 13-25% lower, and black grouse 14-35% lower than those in the control.	MT, ECOL

Table 4.13. ^{210}Po activity concentration in surface soil (Bq kg^{-1}, d.w)

Brazil	27–74*
Canada	20–22 000*
Germany	11–210**
India	4–220
Malaysia	8
Spain	16–780***
United Kingdom	51
USA	10–60
	70–15 000*
Worldwide	8–220

Near a *uranium mine, **coal plant, ***phosphate factory. Source: IAEA 2017

Table 4.14. Activity concentrations of ^{210}Po in terrestrial plants

		Activity conc. of ^{210}Po (Bq kg^{-1}, d.w.)
Areas undisturbed by humans		
Germany	Grass and hay	1.1–29.6
	Grass	22–160
	Heather	20–88
	Juniper	20–38
	Blueberry	34
India	Tobacco	0.1–3.3
Portugal*	Vegetables	0.084
	Fruits	0.06
	Trees	0.37
United Kingdom	Grass	6.5–29
	Lichens	290–370
World data	Grass	2.0–35
	Tobacco	5.6–57
Areas disturbed by humans		
Brazil, uranium mine at 1 km	Vegetables	1–8
Canada, uranium mine at Key Lake	Spruce needles	84
	Labrador tea	1300
Spain, near phosphoric acid factory	*Spartina* spp.	5.1–40.6
United Kingdom, Cotswolds	Grass	4–25

*activity concentrations based on w.w., Source IAEA 2017.

LITERATURE REVIEW AND COMPILATION OF ECOTOXICITY DATA OF FLUORIDE, RARE EARTH ELEMENTS, AND NATURALLY OCCURRING RADIONUCLIDES

Contaminants of concern associated with the Kvanefjeld and Tanbreeze mining projects in South Greenland

This report by the Danish Centre for Environment and Energy (DCE) and Greenland Institute of Natural Resources (GINR) aims to gather worldwide information on excitotoxicity data on fluoride (F), rare earth elements (REEs), and naturally occurring radionuclides (NORs) that can potentially be used in Greenland. The report is not explicitly focused on Kvanefjeld and Tanbreeze mineral exploration projects in south Greenland but on the mining and milling of minerals containing REEs, NORs, and F in general in Greenland.

The report provides relevant information to be used as a basis for environmental regulation and monitoring of mining projects and establish guidelines on concentrations threshold values for the REEs, NORs, and F in water, soil, dust, vegetation, berries, birds, and fish both in the mining area but also for the surrounding environment. This report can also be used for teaching or as guide for the general public, politicians, authorities, education, industry, and other Greenland stakeholders.

ISBN: 978-87-7156-662-8
ISSN: 2244-9981