



# Risk Based Marine Impact Assessment of NORM and Hg from Decommissioning Oil & Gas Infrastructure

Curtin University Oil & Gas Innovation Centre

## Literature Review

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# Literature Review

## Risk Based Marine Impact Assessment of NORM and Hg from Decommissioning Oil & Gas Infrastructure

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# Executive Summary

Australia has a growing liability of non-producing oil and gas infrastructure in the marine environment, subject to decommissioning decisions. Complete removal is the base case for decommissioning, unless better outcomes can be achieved by alternative decommissioning options, such as leaving in place (i.e. *in situ*). The ecological benefit of leaving in place options are well established and related to the provision of hard substrate that allows the formation of epibenthic ecosystems. However, these benefits are challenged by the presence of mercury and Naturally Occurring Radioactive Materials (NORM) contaminants which may result in negative impact to surrounding ecosystems.

This literature review is a comprehensive consolidation and critique of public and available grey literature. It defines the current understanding of the risks that NORM and mercury pose in the marine environment from decommissioned oil and gas subsea pipeline infrastructure, when primarily left in *situ*. The key knowledge gaps and uncertainties are also established, along with a set of recommendations.

The project's aim is to enhance the understanding of the impacts and risks of NORM and mercury to the marine environment from decommissioned oil and gas infrastructure, to support science-based decommissioning decision-making. The project is framed around three problem statements in which the Australian oil and gas industry are seeking to:

- Understand the threshold levels at which NORM in subsea production equipment and mercury in subsea gas transmission systems in the offshore environment become a concern during and after decommissioning, relative to the receiving environment, specifically: levels/dosage/concentration, form/chemical species and likely behaviour (how are they likely to leach over time and what are the likely pathways) if NORM and mercury is present in O&G infrastructure.
- Understand the processes and exposure pathways whereby individual contaminants might (or might not) reach receptors after decommissioning.
- Improve its tools and procedures for measuring mercury and NORM associated with equipment/infrastructure *in situ* to inform decommissioning decisions and long-term monitoring objectives.

This literature review is structured by first defining the current understanding of Australia's regulatory framework which defines the required environmental outcomes for decommissioning options. This section also reviews the behaviour of mercury and NORM in oil and gas systems, including the proposed mechanisms leading to the formation of contaminated products (products in oil and gas systems such as mineral scales, films, oxide layers, and so on that contain concentrations of NORM and mercury) and reviews measurement techniques for mercury and NORM. It steps through the components of an ecological risk assessment as they relate to mercury and NORM in marine ecosystems including the ecosystem receptors they may impact, their behaviour, and key environmental transformations governing their exposure pathways, the impacts they may have on ecosystem receptors, and approaches to understanding and quantifying their risk as it relates to the legislative requirements.

The current practice section compares industry practice with the understanding developed in earlier sections to identify potential gaps in assessing mercury and NORM impact and risk in the marine

environment. This comparison will provide the context for the future drafting of a proposed risk framework.

The knowledge gaps and recommendations section identifies gaps and associated recommendations that can be addressed either in the next assessment (i.e. post literature review) or subsequent NDRI project phases.

The key knowledge gaps identified in this review include:

#### **Understanding thresholds of NORM and mercury in the marine environment**

1. How ecological risk assessments are applied to meet legislative requirements
2. Whether exclusion-type criteria ('thresholds') can be derived for mercury and NORM to inform subsea oil and gas pipeline decommissioning decisions
3. What is the inventory of contaminated products – including the contaminated products' form, species, mass, and distribution in pipelines

#### **Understanding the fate of NORM and mercury**

4. How the speciation and distribution of contaminated products in pipelines affects important environmental transformations controlling exposure pathways
5. How different environmental conditions affect contaminated products, their environmental transformations, and their resulting mobility and bioavailability

#### **Understanding of tools and procedures**

6. What measurements are required for ecological risk assessments
7. What are the current and future technology options for measuring subsea *in-situ* mercury and NORM contamination which meet ecological risk assessment requirements

The focus of the Research Team over the remainder of the current project will be to develop an ecological risk assessment framework that can be generalised to all contexts but provide the flexibility to tailor an assessment to local contexts. Steps to achieve this could include:

- Developing consensus on the management objectives required to meet legislative requirements of decommissioning and the assessment methodologies meeting these requirements. This can take the form of a NERA-issued report (or peer-reviewed journal article) and include input from all stakeholders.
- Identifying the most important factors controlling contaminant exposure pathways and hazard impacts. This will be used to inform the development of a tiered risk assessment framework and identify measurement methodology as well as measurement needs and can be achieved by consolidating reported parameters and their uncertainties used in risk assessments.
- Developing a tiered approach to mercury and radiological risk assessments that trades conservatism for data requirements at different levels.
- Identifying technology options for the *in-situ* measurement of mercury and NORM. This could be achieved by surveying operators and instrument providers to review technology options, and their readiness level.

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# Acronyms

<b>AAS</b>	Atomic absorption spectrometry	<b>HgS</b>	Mercuric sulfide
<b>AES</b>	Atomic emission spectrometry	<b>IAEA</b>	International Atomic Energy Agency
<b>AIMS</b>	Australian Institute of Marine Science	<b>ICP</b>	Inductively coupled plasma
<b>ALARP</b>	As low as reasonably practicable	<b>ICRP</b>	International Commission on Radiological Protection
<b>ANSTO</b>	Australian Nuclear Science and Technology Organisation	<b>IP</b>	Industry participants
<b>ARPANSA</b>	Australian Radiation Protection and Nuclear Safety Agency	<b>ISAB</b>	Independent Scientific Advisory Board
<b>BAF</b>	Bioaccumulation factor	<b>K<sub>d</sub></b>	Partitioning coefficient
<b>Bi</b>	Bismuth	<b>LOD</b>	Limit of detection
<b>Bq</b>	Becquerel, the SI unit for radioactivity that is equal to one disintegration per second	<b>MDL</b>	Method detection limit
<b>CH<sub>3</sub>Hg<sup>+</sup></b>	Methylmercury	<b>MS</b>	Mass spectrometry
<b>COPC</b>	Contaminant of primary concern	<b>NDRI</b>	National Decommissioning Research Initiative
<b>CR</b>	Concentration ratio, the ratio of radionuclide activity in the tissue of biota to the environmental media	<b>NHMRC</b>	National Health and Medical Research Council
<b>CRA</b>	Corrosion resistant alloy	<b>NOEC</b>	No observable effect concentration
<b>CSIRO</b>	Commonwealth Scientific and Industrial Research Organisation	<b>NOPSEMA</b>	National Offshore Petroleum Safety and Environmental Management Agency
<b>CVAAS</b>	Cold-Vapour atomic absorption spectrometry	<b>NOPTA</b>	National Offshore Titles Administrator
<b>DAWE</b>	Department of Agriculture, Water, and the Environment	<b>NORM</b>	Naturally occurring radioactive material
<b>DC</b>	Dose coefficient, relates external or internal radiation to absorbed dose for a given organism and radiation type. Also referred to as a DCC or DCF.	<b>NRMMC</b>	National Resource Management Ministerial Council
<b>DCRL</b>	Derived consideration reference level	<b>O&amp;G</b>	Oil & Gas
<b>DoEE</b>	Department of the Environment and Energy	<b>OES</b>	Optical emission spectrometry
<b>EDX</b>	Energy dispersive x-ray	<b>OM</b>	Organic matter
<b>EDXRF</b>	Energy dispersive x-ray fluorescence	<b>OPGGS</b>	Offshore Petroleum and Greenhouse Gas Storage
<b>EMCL</b>	Environmental media concentration limits	<b>Pb</b>	Lead
<b>EP</b>	Environmental Plan	<b>Po</b>	Polonium
<b>EPBC</b>	Environment Protection and Biodiversity Conservation	<b>pXRF</b>	Portable x-ray fluorescence
<b>ERICA</b>	Environmental Risk from Ionising Contaminants: Assessment and Management	<b>Ra</b>	Radium
<b>FPSO</b>	Floating Production Storage and Offtake	<b>SEM</b>	Scanning electron microscopy
<b>FSANZ</b>	Food Standards Australia and New Zealand	<b>SQG</b>	Australian and New Zealand default sediment quality guideline value
<b>Gy</b>	Gray, standard unit for absorbed radiation	<b>SWASV</b>	Square Wave Anodic Stripping Voltammetry
<b>H<sub>2</sub>S</b>	Hydrogen sulfide	<b>WQG</b>	Australian and New Zealand default water quality guideline value
<b>Hg</b>	Mercury	<b>XAS</b>	X-ray absorption spectrometry
<b>Hg<sup>0</sup></b>	Elemental mercury	<b>XRD</b>	X-ray diffraction
<b>HgCl<sub>2</sub></b>	Mercury chloride		

# Key terms

<b>Benthic</b>	Relating to the sea floor including the sub-surface layers
<b>Contaminated product</b>	Defined for the purposes of this review as a product such as a scale, film, layer of corrosion, sludge etc. formed in oil and gas production systems that contains contaminants subject to an ecological risk assessment.
<b>Coupon</b>	Physical sample of the pipeline removed for contaminant testing
<b>Demersal</b>	Relating to or occurring close to the sea floor
<b>Elemental</b>	A compound comprising of atoms from a single element. For example, $\text{Hg}^0$ .
<b>Environmental media</b>	The sediment or water in which organisms reside
<b>Environmental reference level</b>	The concentration of a contaminant in environmental media below which there is a low risk of a defined hazard impact occurring
<b>Epibenthic</b>	Relating to the surface of the sea floor
<b>Exclusion level or criteria</b>	The activity of a radionuclide or substance below which a regulatory agency has determined that there is unlikely to be an unacceptable risk in any reasonable exposure scenario
<b>Exemption level or criteria</b>	The activity of a radionuclide below which the risk should be sufficiently low as to be of no regulatory concern to humans as defined by the IAEA in their basic safety standards
<b>Guideline value</b>	The concentration of a contaminant in environmental media below which there is a low risk of a defined hazard impact occurring
<b>Inorganic</b>	A compound which does not contain carbon atoms. May include both elemental and ionic species.
<b>Ionic</b>	A compound with an electrical charge due to the loss or gain of electrons. For example, $\text{Hg}^{2+}$ , $\text{MeHgCl}^+$
<b>Inventory</b>	The concentration or specific activity, mass, speciation, and distribution of a contaminated product
<b>Lines of evidence</b>	Different hazard impacts that are assessed together to understand contaminant risk in an ecological risk assessment
<b>Management objectives</b>	Environmental management criteria that contextualises the legislative protection requirements for decommissioning options to an operator's specific asset, ecosystem, and community
<b>Organic</b>	A compound that contains a carbon atom (which may be a 'methyl' group). For example, methylmercury ( $\text{MeHg}^{2+}$ ).
<b>Pelagic</b>	Relating to the water column above the sea floor
<b>Spatial scale</b>	The defined spatial range considered
<b>Specific activity</b>	The radioactivity of a substance expressed by its mass or volume, i.e. $\text{Bq/g}$ or $\text{Bq/L}$
<b>Temporal scale</b>	The defined range of time considered
<b>Threshold level</b>	An undefined concept that could mean the concentration of a contaminant at which a hazard impact is observed to an organism, or may be conflated with exclusion, guideline, or reference level definitions.

# 1 Review Introduction

## 1.1 Purpose & objectives

The project's aim is to enhance the understanding of the impacts and risks of NORM and mercury to the marine environment from decommissioned oil and gas infrastructure, to support science-based decommissioning decision-making.

In fulfilment of the stated project purpose, the identified objectives are:

- **Objective 1:** Consolidate the state of knowledge on protective standards and impact of NORMS/mercury exposure on the marine environment at various concentrations.
- **Objective 2:** Determine the applicable standards and guidelines for the identification and measurement of NORM and mercury types and quantities in oil and gas subsea infrastructure.
- **Objective 3:** Science-based understanding of NORM and mercury location, concentration and amount, speciation, mobility and quantified risk in marine receptors, considering degradation scenarios of in-situ subsea infrastructure when exposed to various decommissioning scenarios.
- **Objective 4:** Develop a contaminants of primary concern (COPC) framework to quantify the combined impact and risk of NORM and mercury to marine receptors from left in-situ subsea infrastructure under various decommissioning scenarios.

This document's purpose is to review and critique the published public domain and available grey literature and data, in relation to the above stated project aim and objectives, to establish the current state of knowledge, the associated key knowledge gaps and uncertainties to frame the research team's focus during the subsequent assessment phase in 2021.

The basis and scope for the assessment phase programme will be attached to this document following consultation with National Decommissioning Research Initiative (NDRI), the Industry Participants (IP) and the Independent Scientific Advisory Board (ISAB) (see Attachment B).

## 1.2 Problem statement and key questions

A further expansion of the project objectives is stated in the NDRI Problem Statement file note (See Attachment A), which identifies three problem statements and an associated set of questions. These problem statements, in the order tabled, relate to the Australian oil and gas industry seeking to:

- **Problem Statement 1:** Understand the threshold levels at which NORM in subsea production equipment and mercury in subsea gas transmission systems in the offshore environment become a concern during and after decommissioning, relative to the receiving environment, specifically: levels/dosage/concentration, form/chemical species and likely behaviour (how are they likely to leach over time and what are the likely pathways) if NORM and mercury is present in O&G infrastructure.
- **Problem Statement 2:** Understand the processes and exposure pathways whereby individual contaminants might (or might not) reach receptors after decommissioning.
- **Problem Statement 3:** Improve its tools and procedures for measuring mercury and NORM associated with equipment/infrastructure in-situ to inform decommissioning decisions and long-term monitoring objectives.

The key questions within these problem statements tabled by NDRI are:

#### **Problem Statement 1**

- What are the threshold levels at which NORM and mercury in decommissioned subsea production infrastructure are considered to be a concern to the environment and hence driving decommissioning decision outcomes?
- What are the published levels/dosage/concentration, form/chemical species and their likely behaviour (i.e. temporal leach rates and likely pathways) if NORM and mercury is present in the O&G infrastructure?

#### **Problem Statement 2**

- What are the exposure pathways (i.e. the processes) whereby contaminants might reach the receptors? What research has been performed on leaching of NORM scale?
- What are the current published works of impact to the environment?
- What are the published (i.e. simulated or measured) mercury/NORM concentrations in sediment after pipeline degradation?
- What levels of NORM and mercury have been reported in the environment?
- What are the current guidelines, protective standards and regulations for mercury/NORM contamination, how do they compare and how suitable are their application to decommissioning subsea oil and gas infrastructure?

#### **Problem Statement 3**

- How well can we identify and measure NORM/mercury in O&G infrastructure today?
- What are the standards, guidelines, procedures and technologies for *in-situ*/subsea measurement?

These questions, in addition to those identified through this literature review, will be used to help identify and capture the knowledge gaps summarised in Section 5.

### **1.3 Problem scope**

The problem scope of work is restricted to the marine environmental impact of two classes of COPC, NORM and mercury and their associated species. The scope includes understanding exposure pathways arising from subsea oil and gas infrastructure internally exposed to production fluids which has then accumulated mercury and/or NORM contamination. Infrastructure may be cleaned prior to decommissioning in a variety of ways and it is the ‘as intended to be left’ condition subject to decommissioning decisions that is investigated here. The scope boundary includes pipelines downstream of the well head from the mudline to mean sea level.

While there is an extensive range of subsea oil and gas infrastructure which can be associated with the above described COPC exposure (e.g. subsea Christmas trees, well head jumpers and spools, subsea heat exchangers, subsea manifolds and numerous types of valve skids, flexible and rigid risers), this problem scope has been constrained around subsea flowlines and pipelines sitting on the seabed transporting either raw well (i.e. three phase), partially or fully processed oil and gas streams.

This focus is driven by the quantity (i.e. length, tonnage) of flowlines and pipelines on the seabed, their exposure and potential contamination with NORM and mercury, the associated uncertainties surrounding the marine environmental impact and risk if they are left *in-situ* and the considerable

expense if they are removed following cessation of production. This report will focus on *in-situ* pipeline decommissioning, the outcomes from which can be used to infer a position on partial or full removal options.

Given the variability of the Australian offshore oil and gas environment, three basins of interest have been nominated to span the expected range of environmental settings and water depths associated with the majority of flowline and pipeline infrastructure. These are:

- Carnarvon Basin, 0-1500 m (water depth)
- Bonaparte Basin, 0-135 m
- Gippsland Basin, 0-400 m

## 1.4 Exclusions

Scope exclusions of this report include the following:

- Topsides infrastructure (anything above mean sea level)
- Onshore facilities, shoreline is the boundary
- Anywhere outside of the stated basins of interest and water depth range

## 1.5 Document structure

This document is structured around three key headings:

- 1) **Current understanding:** defines the problem framing around pipelines and the key basins of interest, uses public domain documents to investigate and capture the state of knowledge on decommissioning regulations, NOPSEMA's (National Offshore Petroleum Safety and Environmental Management Authority) role and Environment Plan (EP) acceptability criteria, radiological and chemical hazard guidelines, relevant ecological risk assessment frameworks, the detailed behaviour of contamination products, published inventories of NORM and mercury in oil and gas production systems, mercury and NORM measurement and monitoring, environmental receptors for the basins of interest, exposure pathways inclusive of degradation mechanisms, hazard characterisation with the key lines of evidence and contaminant environmental reference levels and finally risk assessment methods including tiered assessments and the treatment of cumulative impacts.
- 2) **Current practice:** this section compares and contrasts industry practice, representative pipeline contamination inventories, measurement techniques and risk mitigation techniques employed, insights from publicly available decommissioning environmental plans with published scientific work to establish the presence of NORM and mercury and potential gaps in determining the impact and risk on the environment. This comparison will provide the context for the future drafting of a proposed risk framework.
- 3) **Knowledge gaps and recommendations:** gaps identified with associated recommendations for addressing either in the next assessment (i.e. post literature review) or subsequent phases following the current project.

The Current Understanding and Current Practices subsections contain an upfront statement of relevancy to the project objectives and problem statements followed by the key findings of that section.

## 2 Approach

### 2.1 Overview

Following project kick off and framing, the review phase of the project (i.e. the phase covered by this document) was designed to establish the state of knowledge, collect, collate and review available data to then define the basis and scope of the subsequent assessment phase. The assessment phase would execute more detailed and discrete pieces of mercury and NORM work in support of a better understanding of impact and risk to the environment.

For input to the literature review, the research team has initiated and completed multiple workshops and follow-up meetings with Industry Participants, ISAB members and NDRI to identify and receive all available relevant contamination data and assessment reports, tabled and discussed the potential range of COPC scenarios and clarified the geographical regions of interest, which were expanded from the Bass Strait (Gippsland Basin) and North West Shelf (Carnarvon Basin) to also include North Australia (Bonaparte Basin).

The Research Team's approach for the literature review was to address the problem statement in its entirety using available published and industry provided data and assessment reports so the state of knowledge could be fully articulated and the gaps identified. Due to the limited availability of industry data, the primary focus has been public domain data and documentation to outline the current understanding. The available industry grey data and reports have been used to compare and contrast this understanding to further highlight the knowledge gaps and inform the recommendations captured at the end of the report.

During the project kick off and framing, the Research Team tabled a range of infrastructure scenarios across the basins of interest and sought agreement with the Industry Participants to focus the literature review. There are many key parameters defining these infrastructure production systems driven by the reservoir types, the production fluid compositions and their temperatures and pressures, whether the infrastructure was upstream or downstream of any processing facility, the marine environment etc. As no specific scenarios could be agreed, the Research Team has constrained the literature review to any flowlines and pipelines located on the seabed, exposed to any fluids from oil and gas reservoirs, whether raw well fluid or partially/fully processed, across the three basins and water depth ranges defined in Section 1.3.

This is a desktop review only.

### 2.2 Engagement and consultation

In accordance with the project engagement plan a number of internal and external entities were identified to gather data or information that might not be publicly available with regard to the environmental risk of NORM and mercury in the marine environment; and to access expert knowledge and perspectives.

#### 2.2.1 Industry participants

In addition to the formal group engagement sessions and workshops all Industry Participants have been contacted individually to identify all relevant and available contamination data sets and reports, to secure their release where possible and to clarify their content.

## 2.2.2 NDRI projects: program 1 and 2

The research team has engaged with all the research teams across the entire NDRI program but in particular Dr Dianne McLean of AIMS, to leverage their work around the classification and quantification of fish and other marine organisms around pipelines located in the North West Shelf and Bass Strait, and Professors Mike Tan and Robert Melchers to seek their opinion as to the current level of understanding for subsea pipeline degradation.

## 2.2.3 State and federal regulatory bodies and agencies

The Research Team was not given approval to discuss matters associated with this research with NOPSEMA. This position will be reviewed following the completion of this literature review when specific questions can be framed and agreed. As the majority of the offshore pipeline infrastructure lies in Commonwealth waters, the engagement with State regulators was deemed as a lower priority and therefore not executed until first engaging with NOPSEMA.

The Research Team has engaged with the Australian Radiation Protection and Nuclear Safety Agency (ARPANSA) to discuss a range of matters relating to their published guide ARPANSA (2015a) and to take feedback on what ARPANSA perceive to be key questions and uncertainties in relation to pipeline infrastructure contaminated with NORM.

## 2.2.4 Local and international researchers / subject matter experts

During the literature review the Research Team has engaged with Dr Tom Cresswell and his current Doctor of Philosophy candidate Amy MacIntosh, to share and discuss ANSTO's NORM research to avoid where possible duplication of work, within the confidentiality constraints of their commercial clients.

The Research Team has also sought guidance from Dr Graeme Batley of CSIRO Lucas Heights in regard to the Water Quality Management Framework Commonwealth of Australia (2018) given his key role in its development.

Through facilitation of NDRI, the Research Team has also engaged with Atteris engineering consultancy to discuss their pipeline degradation assessment studies supporting the oil and gas operators.

### 3 Current understanding

The purpose of this section is to identify, review and critique the current understanding of NORM and mercury contaminants in oil and gas pipelines based on public domain sources and is structured around a step wise process to ultimately characterize the COPC hazard to the environment and how the risk can be assessed.

The scope of this section will first provide a high-level overview of oil and gas systems, the key contaminants, and a decommissioning overview. Following a review of the current Australian decommissioning regulatory framework, NOPSEMA's role and its supporting policy and guidance documentation, a summary is provided of the environmental protection requirements and two ecological risk assessment frameworks are tabled.

The behaviour of the contaminants is a detailed identification of the mercury and NORM species and forms leading to the key contamination products likely to occur within oil and gas pipeline systems. Due to the nature of NORM its spatial and temporal variability will also be addressed.

Mercury and NORM measurement techniques are also reviewed given their importance in establishing the COPC species and distribution within a pipeline followed by an investigation of the ecosystem receptors for the basins of interest and various exposure pathways for the key mercury and NORM species and forms. This will then support the sections addressing the hazard characterization, with a dedicated section on contaminant environmental reference levels, and finally risk analysis.

#### 3.1 Australian offshore oil and gas operations overview

The Australian offshore petroleum industry has been operating for approximately 40 years and plays an important role in contributing to Australia's energy security. Australia's oil and gas reserves are outlined in Figure 3.1. It is anticipated that the sector will continue its contribution into the future with offshore petroleum projects expected to deliver for the coming decades. However, some projects in Commonwealth waters are reaching the end of their production lives. Approximately 136 fixed facilities (including pipelines) are expected to commence decommissioning in the next decade (Australian Government, 2018), a sample of these projects expected to reach the end of their operational life are outlined in Figure 3.2. Each of these projects have subsea flowlines and pipelines to either transport hydrocarbons between production wells and offshore facilities and/or export products to shore via trunklines (i.e. large diameter export pipelines) for further processing. These structures will also be required to undergo decommissioning at the end of each field's operational life.

In comparison with decommissioning projects/activities in global regions, with an extensive history of such as the North Sea and the Gulf coast of the US, Australia is still in its infancy with a future decommissioning cost (total abandonment expenses) estimated at more than US\$40.5 billion (~AUD\$53 billion) (Advisian Pty Ltd., 2020). Despite some small projects (i.e. limited subsea development tied back to a Floating Production Storage and Offtake (FPSO)) having been decommissioned, there have been no decommissioning of larger projects (i.e. 50,000 t jacket or semi-submersible and topsides with multiple rigid production flowlines and > 100kms of trunkline to shore) in Australian waters and these larger more complicated projects are expected to test the effectiveness of Australia's policy, regulatory and legislative framework.

The Department of Industry, Innovation and Science has commenced a review with the aim of ensuring that the legislative framework for decommissioning offshore petroleum infrastructure in Commonwealth waters is fit-for-purpose, continues to meet community and industry expectations, and enables Australia to respond to and overcome challenges associated with decommissioning of a range of projects (Australian Government, 2018).

### 3.1.1 Offshore infrastructure and production system overview

As stated in the project objectives, the problem scope has been framed around subsea flowlines and pipelines sitting on the seabed transporting either raw well (i.e. three phase), partially or fully processed oil and gas streams. This infrastructure exists in all Commonwealth oil and gas basins shown in Figure 3.1 but this study focuses on the Gippsland, Carnarvon and Bonaparte basins.

Figure 3.3 provides a generalised arrangement of an offshore production system from the subsea wells, the flowlines, the processing facility, and the export pipelines to an onshore processing plant. The flowlines connect the wells to the processing facility either through rigid steel (i.e. carbon steel or corrosion resistant alloy (CRA)) or unbonded flexible pipe (i.e. a composite structure combining CRA and carbon steel layers separated by thermoplastic). The processing facility can perform simple bulk separation of oil/condensate, water and gas (i.e. partially process), or it can be a full process scheme delivering sales quality gas or liquids.

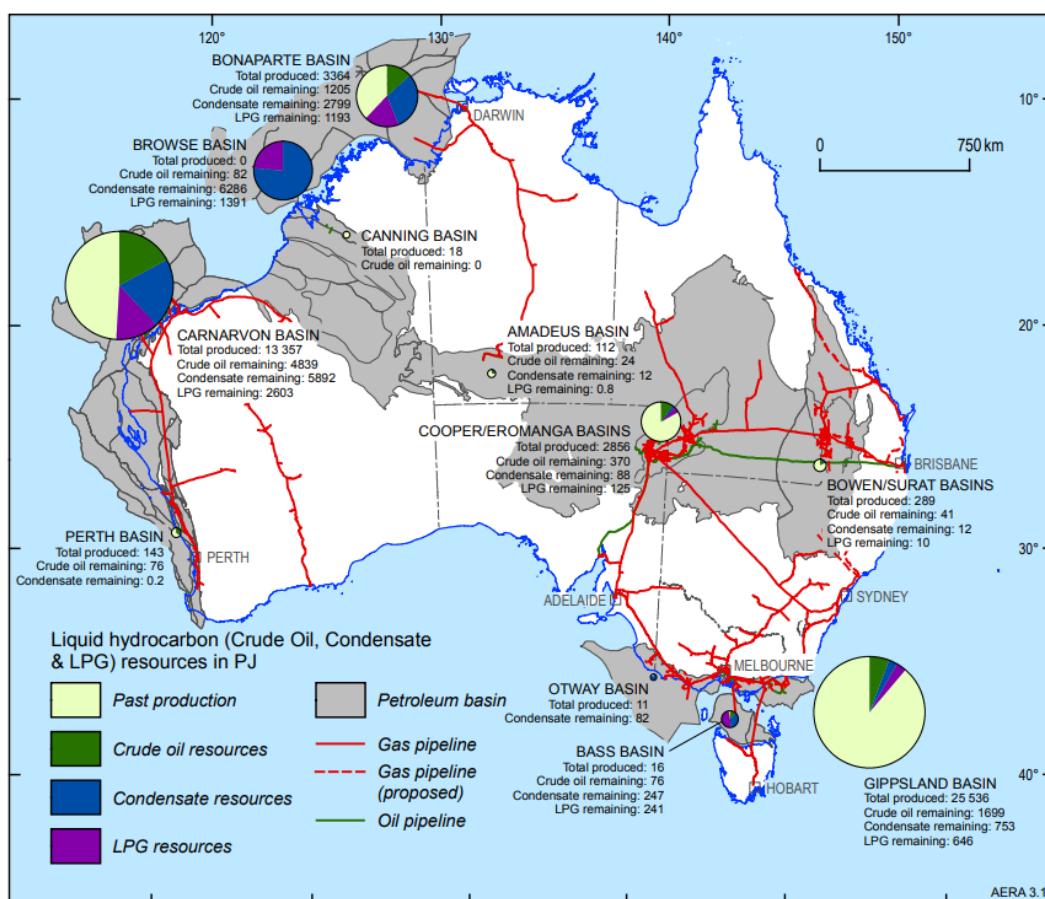
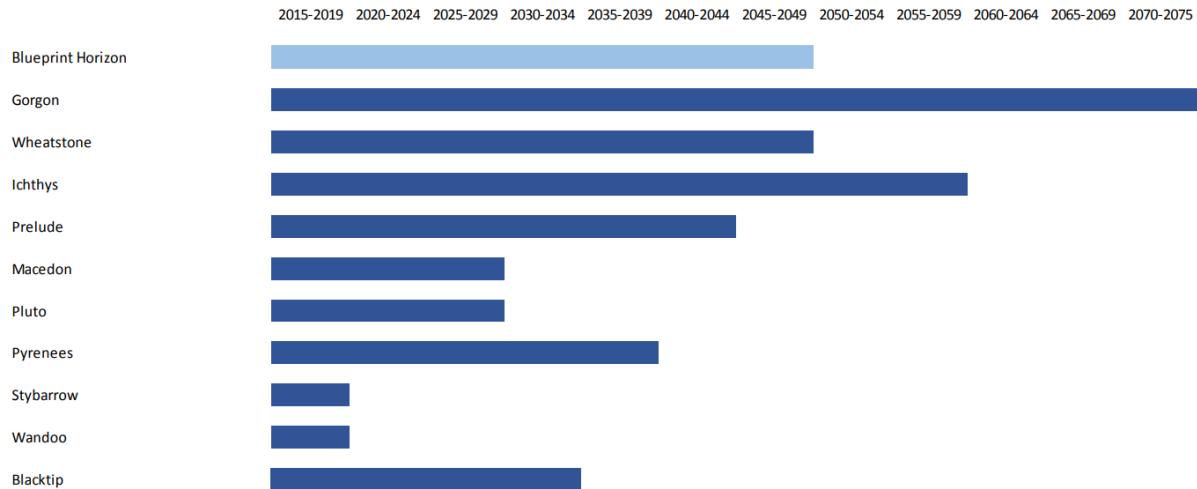


Figure 3.1 Australian crude oil, condensate and naturally-occurring liquefied petroleum gas resources, infrastructure, past production and remaining resources (Source: Geoscience Australia). All values are in the units of Peta Joules (PJ).



**Figure 3.2 Sample of current West Australian oil and gas assets operational life cycle (Bills, 2018).**

These facilities can be supported by a floating or a fixed platform, or, in the case of a direct subsea to beach arrangement, be deemed not required whereby the raw well fluid is transported directly onshore via the subsea pipeline for processing. The sales product type and the market may also drive the need to export the partially or fully processed hydrocarbon, at a substantial volume and distance, from the offshore field to an onshore facility via a trunkline (see Figure 3.3).

Within the broad definition of flowlines, pipelines and trunklines there is a wide diversity of diameters, length and internal/external coating systems. The most comprehensive and publicly available data base for this infrastructure definition is captured within the pipeline licences managed by NOPTA (National Offshore Petroleum Titles Administrator) and is accessible online (NOPTA, 2021). Figure 3.4 captures all the current pipeline licences, both state and federal. However, it is not possible to create a complete and comprehensive inventory of the subsea pipeline infrastructure across all three basins of interest as the pipeline licences do not account for secondary pipelines, which for a subsea development is the majority of the infrastructure between the wells and the offshore processing facilities. This is a gap which may need to be addressed to better quantify the infrastructure (i.e. diameter, wall thickness, length, material, coating systems and hydrocarbon service) for COPC risk assessment on a basin rather than individual pipeline basis.

By way of illustration, from the 29 Commonwealth and 25 State individual pipeline licenses off the Western Australia coastline reported online (NOPTA, 2021), the nominal diameter ranges from 6" to 44", length 2 to 596 km, water depths 10 to 1350 m, full suite of coating systems (i.e. concrete, cold tar enamel, fusion bonded epoxy, polypropylene, epoxy flowcoat, asbestos felt etc.) and materials (i.e. carbon steel, CRA, CRA lined etc.). From this limited inventory of pipelines we estimate there to be in excess of 1.3 million t of steel and in excess of 500,000 t of concrete weight coating alone. This spans the full Carnarvon basin and the West Australian extent of the Bonaparte basin.

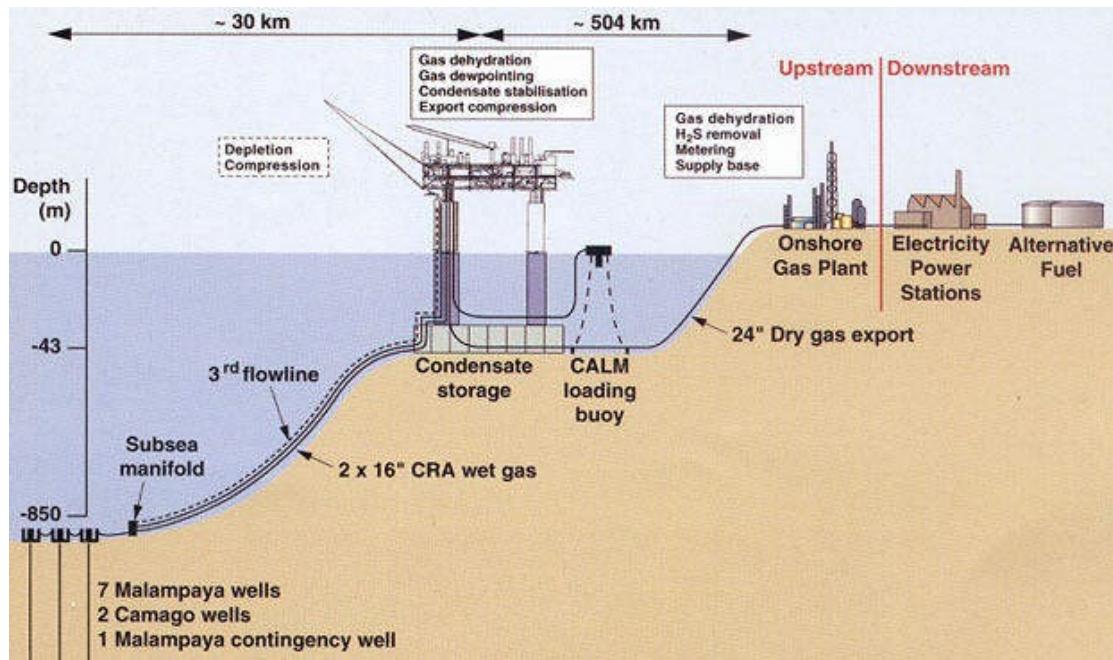


Figure 3.3 Typical subsea field development infrastructure combining offshore and onshore processing with flowlines and pipelines (from <https://www.offshore-technology.com/projects/malampaya/>).

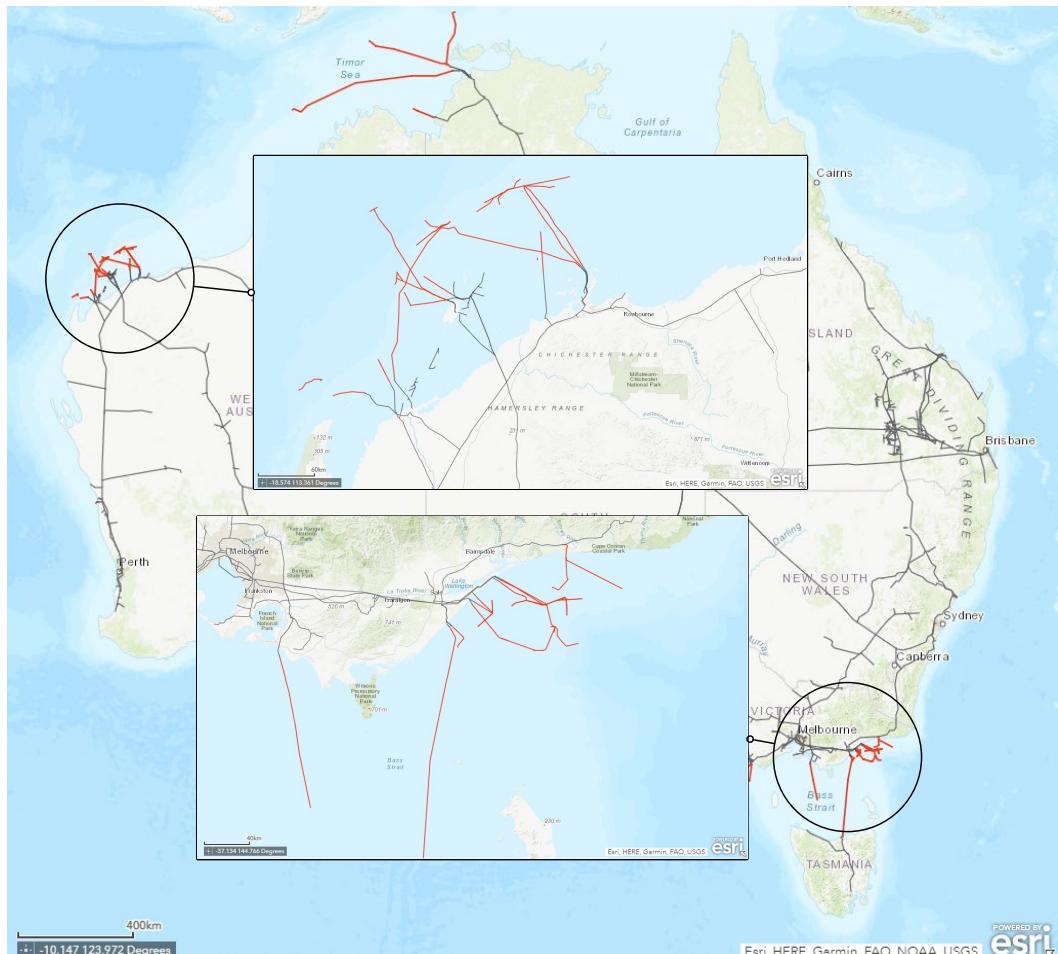


Figure 3.4 Commonwealth and State pipeline licenses created from the NEATS Public Portal (NOPTA, 2021).

### 3.1.2 COPC in production systems

Production from offshore oil and gas reservoirs, while primarily containing a multiphase mixture of oil, gas and water, may also contain a range of NORM and mercury contaminants which can form, deposit and adhere to the inner surface of the subsea flowline and pipeline infrastructure. When elevated beyond naturally occurring concentrations in background environmental media, these deposits are termed contaminants.

NORM contaminants form discrete contaminated products that can be categorised into four main groups: (1) scales of inorganic salts (i.e. BaSO<sub>4</sub>, SrSO<sub>4</sub>, CaSO<sub>4</sub>, CaCO<sub>3</sub>) with elevated radium (<sup>226</sup>Ra and <sup>228</sup>Ra); (2) lead films and depositions that occur from the deposition of <sup>210</sup>Pb from water streams or the decay of <sup>222</sup>Rn; (3) polonium films – <sup>210</sup>Po films may form in the absence of <sup>210</sup>Pb in condensate treatment systems or gas production systems due to its solubility and volatility; and (4) sludge and sands containing low activity levels of <sup>238</sup>U, <sup>232</sup>Th, and decay products. Of these, radium-contaminated scales and lead films are the two most important contamination products for ecological risk assessments. A detailed definition of the origins and behaviour of these grouped species within oil and gas production systems is provided in Section 3.4.

Mercury emanates from the geological formations and is produced with the formation water, oil and gas in various chemical forms (i.e. chemical species) which depend on the pressure, temperature and chemistry of the formation and fluids. It can accumulate to contaminated products inside production systems by many processes, including: (1) deposition of elemental mercury to pipeline surfaces by condensation of mercury vapour; (2) adsorption to steel surfaces; (3) adsorption into corrosion products; (4) mercury accumulated in sludges and as asphaltenes. The accumulation of mercury is dependent on the chemical processes within the production system, including changes of temperature and pressure, and exists in elemental, and ionic and organic forms. A more detailed characterisation of the mercury species and its behaviour is defined in Section 3.3.

### 3.1.3 Pipeline decommissioning overview

There is no Australian industry-wide standard for pipeline decommissioning (Jas et al., 2017) and so is conducted on a case by case basis and is dependent on the experience of the pipeline Operator. Other oil and gas regions of the world have detailed guidance material based on many years of practice under mature legislation and regulation (Oil & Gas UK, 2013). This section will present a high level pipeline decommissioning overview on the key decisions and practices to set the context for the subsequent sections capturing the Australian decommissioning regulatory framework, COPC behaviour, COPC measurement, exposure pathways, hazards and risk.

#### 3.1.3.1 Pipeline decommissioning process

In accordance with Commonwealth regulations the primary environmental permissioning document for decommissioning activities in Commonwealth waters is the Environmental Plan (EP) which must demonstrate to NOPSEMA the requirements defined in Section 3.2.2 are fulfilled for the proposed pipeline decommissioning activities. These activities could support a range of pipeline decommissioning outcomes be that total removal, leave *in situ* or a position in between.

The general process supporting the necessary approvals, field execution and post decommissioning monitoring may be summarised as follows (Manouchehri, 2017):

- Gather the pipeline data (e.g. physical, condition, contamination, environmental)
- Engineering desktop studies and planning including analysing the production history, survey data and inspection, repair, and maintenance records
- Survey the pipeline and neighbouring facilities (if existing survey is not up-to-date)
- Ecological risk assessment
- Comparative assessment of all possible options
- Prepare a detailed cost assessment
- Carry out pipeline pigging, cleaning, and flushing
- Carry out pipeline ends disconnection
- If required, apply corrective measures to the pipeline (e.g. rock dumping, jetting, burial)
- If required, remove and recover the pipeline
- Perform post decommissioning survey
- Develop the strategy/plan for pipeline monitoring (if left *in situ*)

The above process is very general in nature and is dependent on the actual site and pipeline in question. The presence of COPCs could materially impact a number of these steps and potentially the selected decommissioning option (i.e. pigging and cleaning, comparative/risk assessment, costs, monitoring program).

### 3.1.3.2 Pipeline decommissioning – key decisions / options

As previously stated the decommissioning of pipelines can be divided into two main options: leave *in situ*; and removal. Each are likely to influence the level of environmental risk from COPCs. These options may be further divided into sub-categories, each with their own limitations and use cases:

- Leave *in situ* – minimal intervention;
- Leave *in situ* – minor intervention;
- Leave *in situ* – moderate intervention;
- Removal by reverse reeling;
- Removal by reverse S-lay;
- Removal by cut and lift.

For a UK North Sea context these are described in more detail in Table 3.1 and are provided as an indicative illustration of what might be considered in a comparative assessment, all of which would need to be assessed for COPC environmental risk.

**Table 3.1 Overview of pipeline decommissioning options. Adapted from Oil & Gas UK (2013).**

<b>Decommissioning Option</b>		<b>Description</b>
Leave <i>in situ</i>	Minimal Intervention	This option may be suitable for pipelines that remained buried over the duration of its lifetime. Following cleaning, the pipeline is left filled with seawater and the ends open to the sea. Any snagging hazards are removed. The pipeline may be expected to self-bury over time.
	Minor Intervention	Pipeline is filled with treated seawater and capped. Removal or remediation may be required for sections of the pipeline which may have become scoured and therefore pose a hazard to other sea users. Sections would usually be removed by subsea cutting, equipment such as abrasive water jets, diamond wire cutting, reciprocating cutting and hydraulic shears may all be used as part of the decommissioning programme and lifted to the surface by a support vessel. In some cases, remedial trenching and/or rock dumping may be used on exposed sections to remove snagging risk.
	Major Intervention	A pipeline that was first installed on the seabed or originally trenched may have sections that required intervention over its lifetime. In this case the best option may be to de-commission the pipeline <i>in situ</i> and carry out major intervention works over complete removal. The tie-ins at each end will be cleaned and removed and the pipeline or large sections of the pipeline may be trenched below the seabed. Sections may also be removed by the cut and lift or reverse installation methods.
Removal	Reverse Reeling	This method may be used for pipelines with a diameter of 16 inches or less which are not concrete coated. In such cases, the pipe may be reeled onto a recovery reel on a specialist reel vessel. Once the pipeline is on the reel it can be taken to a shore-based facility for reuse/ recycle/ disposal.
	Reverse S-lay	Larger diameter pipelines or concrete coated trunklines are often installed using the S-lay method. This method may be reversed as a potential removal method. It involves the recovering of a pipeline end to the deck of an S-lay vessel. The vessel then moves along the length of pipeline stopping intermittently for a cut to remove a section of pipe from recovered pipeline string on deck of the vessel. The removed sections are then transferred for onshore recycling.
	Cut and Lift	Method may be used for any diameter of pipeline. This option involves the pipeline being cut into sections subsea by cutting tools operated by divers or remotely; sections are then transferred to a surface vessel. This option is often used for shorter sections of pipeline where the use of major removal equipment is unsuitable. Pipeline bundles may also be re-floated and towed to shore.

## 3.2 Australian regulatory framework for decommissioning

This section considers the legislative setting that underpins decommissioning decision making around contaminant risk. Australia's suite of environmental legislation have complicated interconnections related to jurisdictional boundaries and overlapping scopes. How these interact with the principle legislation governing offshore oil and gas activities is explored to identify what legislation's standards and guidelines are applicable and identify how ecological impact and risk is considered. In doing so this section partly addresses objectives 2 and 3 and problem statement 1.

### Key findings:

- The *OPGGS Act* and *Environment Regulations* define broad outcome-based criteria based on acceptability and ecologically sustainable development principles for decommissioning options other than complete removal.
- Operators are required to create criteria that contextualises the legislative protection requirements to their specific asset, ecosystem, and community, and demonstrate how such criteria will be met.
- No ecological risk assessment methodology is prescribed, however, the Water Quality Management Framework and ARPANSA's Radiation Protection Series are Commonwealth-developed methodologies agreed to by all federal, state and territory governments representing best practice ecological risk assessment approaches.
- Exemption criteria (such as for the safe transport of radioactive material or for dredging) are specific to their activities so cannot be used in the context of decommissioning contaminated pipelines. Environmental reference levels (such as water and sediment quality guidelines or dose-rate reference levels) may be used to understand hazard impacts such as toxicity to organisms.

### 3.2.1 Overview

Australia's regulatory framework for offshore oil and gas infrastructure decommissioning is provided by the Offshore Petroleum and Greenhouse Gas Storage Act 2006 (*OPGGS Act*) and Offshore Petroleum and Greenhouse Gas Storage (Environment) Regulations 2009 (*Environment Regulations*) which outlines statutory requirements for all offshore oil and gas activities and details how environmental impacts and risks should be considered.

The base case for decommissioning in Australia is complete removal. However, alternatives may be accepted if titleholders can demonstrate equal or better environmental outcomes, and the alternative option complies with all other legislative and regulatory requirements, e.g. the Sea Dumping Act (DISER, 2018). All proposals are considered by the regulator NOPSEMA, who have developed a range of guidance documentation that details how they assess environmental impacts and risks and determine what is acceptable under the *OPGGS Act* and *Environment Regulations*. However, the process of environmental management including ecological risk assessments is not prescriptive, rather being driven by environmental outcomes (i.e. environmental impacts and risks will be at an acceptable level). Titleholders are required to identify, assess, and control environmental impacts and risks to the satisfaction of the regulator. A range of Commonwealth legislation and guidelines,

discussed below, provides more specific guidance for conducting ecological risk assessments or that detail contaminant-specific environmental risk considerations.

### 3.2.2 Legislative requirements for decommissioning

The *OPGGS Act* is the principle legal instrument governing offshore oil and gas activities, including their decommissioning. *OPGGS Act s. 573(3)* outlines the requirement for titleholders to remove all structures, equipment, and other property that is neither used nor to be used in connection with the operations in the title area. Specific criteria for environmental protection are required to be met before a decommissioning activity can be accepted and a title surrendered. These are outlined in the *OPGGS Act s. 270* which requires that decommissioning activities have to meet the satisfaction of NOPSEMA:

- (c)(i) been removed, or caused to be removed, all property brought into the surrender area by any person engaged or concerned in operations authorised by the title,
- (c)(ii) made arrangements that are satisfactory to NOPSEMA in relation to that property
- (e) provided for the conservation and protection of the natural resources in the surrender area
- (f) made good any damage to the seabed or subsoil in the surrender area caused by any person engaged or concerned in the operations authorised by the title.

Alternative decommissioning options to complete removal are possible under the *OPGGS Act s. 270(c)(ii)*, including *in-situ* or partial decommissioning approaches. These alternatives are acceptable if the arrangements are acceptable to NOPSEMA, in relation to the property. This has been defined by DISER as the alternative decommissioning option delivering equal or better environmental, safety and well integrity outcomes compared to complete removal, and that the approach complies with all other legislative and regulatory requirements (DISER, 2018).

The *OPGGS Act* is supported by a number of regulations, with environmental management for petroleum activities being regulated by provisions in the *Environment Regulations*. The object of these regulations (*Environment Regulations r. 3*) is to ensure that any petroleum activity or greenhouse gas activity carried out in an offshore area is:

- carried out in a manner consistent with the principles of ecologically sustainable development set out in section 3A of the *EPBC Act*; and
- carried out in a manner by which the environmental impacts and risks of the activity will be reduced to as low as reasonably practicable; and
- carried out in a manner by which the environmental impacts and risks of the activity will be of an acceptable level.

To achieve this, the *Environment Regulations (rr. 6-26)* sets out a requirement that all oil and gas activities must have an Environment Plan accepted by NOPSEMA before they can take place. The Environment Plan defines the activities being undertaken and outlines how operators will manage their activities' environmental impacts and risks to be as low as reasonably practicable and of an acceptable level.

The criteria for acceptance of an Environment Plan are given in the *Environment Regulations r. 10A* that the plan:

- a) is appropriate for the nature and scale of the activity; and

- b) demonstrates that the environmental impacts and risks of the activity will be reduced to as low as reasonably practicable; and
- c) demonstrates that the environmental impacts and risks of the activity will be of an acceptable level; and
- d) provides for appropriate environmental performance outcomes, environmental performance standards and measurement criteria; and
- e) includes an appropriate implementation strategy and monitoring, recording and reporting arrangements; and
- f) does not involve the activity or part of the activity, other than arrangements for environmental monitoring or for responding to an emergency, being undertaken in any part of a declared World Heritage property within the meaning of the *EPBC Act*; and
- g) demonstrates that:
  - i. the titleholder has carried out the consultations required by *Division 2.2A*; and
  - ii. the measures (if any) that the titleholder has adopted, or proposes to adopt, because of the consultations are appropriate; and
- h) complies with the [OPGGS] *Act* and the regulations.

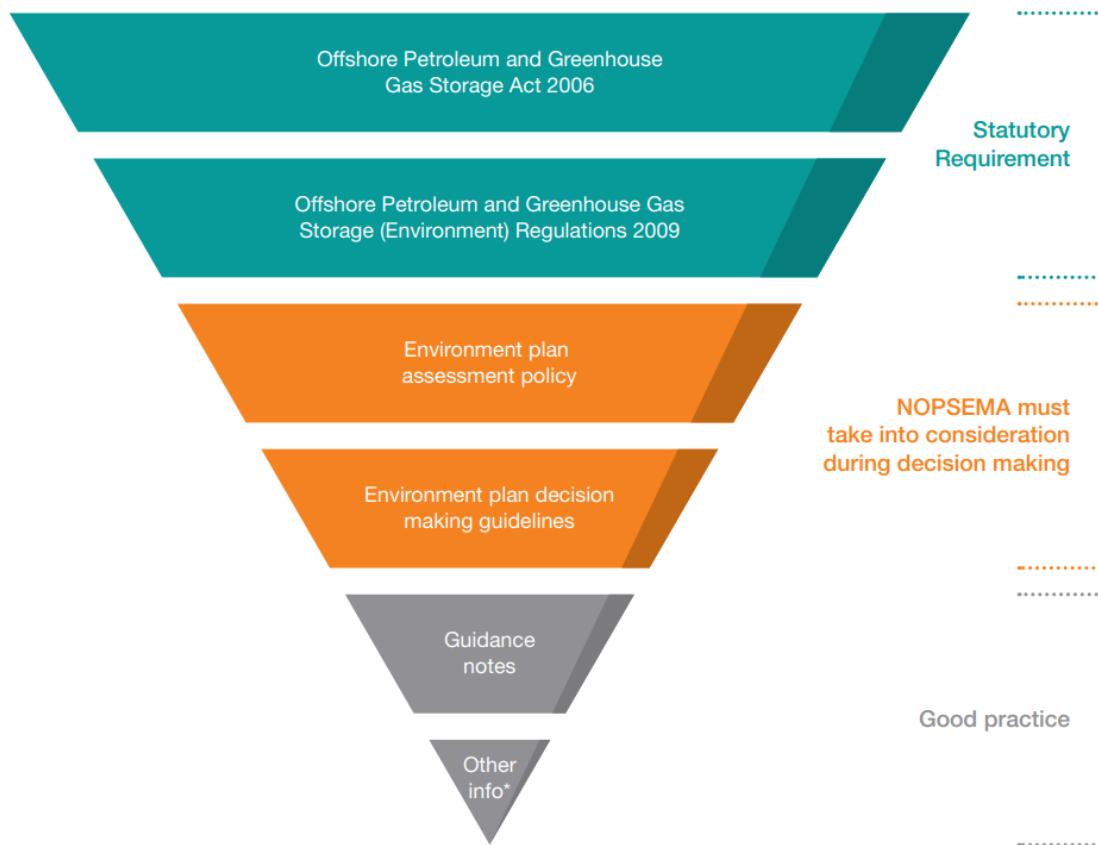
The *Environment Regulations r. 10A(h)* means that Environment Plans relating to decommissioning activities are also assessed against the requirements outlined in the *OPGGS Act s. 270* (i.e. made good any damage to the seabed).

To meet the requirements for acceptance, the *Environment Regulations rr. 13(1-7)* specifies what should be included in an Environment Plan:

1. Description of the activity
2. Description of the existing environment
3. Describe environmental receptors of particular values and sensitivities
4. Description of relevant requirements, including legislative requirements
5. Details of environmental impacts and risks, an evaluation of these impacts and risks, and details of control measures for these impacts and risks
6. Consideration for risks arising directly or indirectly from the activities
7. Set environmental performance standards, outcomes, and measurement criteria for control measures

At the conclusion of activities in an Environment Plan, a titleholder must notify NOPSEMA that the activity or activities to which the plan relates have ended, and that all of the obligations under the environment plan have been completed. Once the notification is accepted, the Environment Plan ends and is no longer in force for the activity. For the purpose of decommissioning activities, acceptance and completion of an Environment Plan provides for the *OPGGS Act s. 573(3)* requirements. This includes whether or not infrastructure was removed or an alternative decommissioning option was selected, so long as approval was granted for the activity, and completion of the activity, by NOPSEMA (DISER, 2018).

To maintain transparency in their decision and assessment procedures, NOPSEMA has developed a range of guidelines, guidance notes, and fact sheets for industry. These range from policies and guidelines which govern their decision-making process, to guidance notes, fact sheets, information papers, and reports (Figure 3.5).



\*Other information includes brochures, environment alerts, bulletins, information papers, articles in the Regulator magazine, forms and other published material.

**Figure 3.5 Levels and weight of documentation used by NOPSEMA in assessing environmental plans. Taken from NOPSEMA.**

### 3.2.3 NOPSEMA's role, supporting policy and guidance documentation

Most important to decommissioning considerations are those relating to environment plan assessment:

- Environment plan assessment policy (NOPSEMA, 2020b)
- Environment plan decision making guidelines (NOPSEMA, 2019)
- Environment plan content requirement guidance note (NOPSEMA, 2020c)
- Section 572 Maintenance and removal of property policy (NOPSEMA, 2020d)

The objectives of the *Environment Regulations* provide a basis for which one can assess decommissioning options where contaminants are present in oil and gas infrastructure. The concepts of ALARP and 'acceptable level' have been detailed to some extent in guidance developed for industry by NOPSEMA (NOPSEMA, 2020a) and are summarised below:

**ALARP:** 'ALARP is achieved when all practicable measures to reduce environmental impacts or risks are implemented and any further measures not implemented are demonstrated to be grossly disproportionate in "cost" when compared to the reduction in impact or risk reduction gained.'

**Acceptable:** 'NOPSEMA can decide whether impacts or risks are acceptable where a duty holder has suitably defined an acceptable level or impact or risk and proposed measures to reduce the

consequence, severity or likelihood of those impacts or risks to that defined level while referring to the relevant Australian laws and conventions, published guidance, industry standards, and best practice guides.'

Demonstration of acceptable level is an object of the *Environment Regulations* (r. 3) and a criterion for acceptance in Environmental Plans (*Environment Regulations r. 10A(c)*). Important factors influencing NOPSEMA's decisions around acceptability includes consideration of Australian environmental management laws as well as other non-legislated requirements that may apply to the activity (NOPSEMA, 2019). These may encompass Australia's obligations as signatories to international conventions and treaties, environmental requirements outlined by other Acts, such as the *Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act)* and *Australian Radiation Protection and Nuclear Safety Act 1998 (ARPANS Act)*, or national strategies, conventions, and guidelines including the Charter: National Water Quality Management Strategy (Commonwealth of Australia, 2018) and associated Australian and New Zealand Water Quality Guidelines (ANZG, 2018).

### 3.2.4 Other relevant legislation

The OPGGS Act and its regulations are the primary instrument for regulating decommissioning in Commonwealth waters; however, other relevant information may be relevant to meet acceptability requirements. For example, the Environment Plan Assessment Policy (NOPSEMA, 2020b) states that information provided in an Environment Plan is considered as well as '*other relevant information such as peer reviewed scientific literature, relevant Department of the Environment and Energy (DoEE) [now Department of Agriculture, Water, and Environment] policies, guidelines, plans of management and other material relating to matters protected under Part 3 of the EPBC Act relevant to the activity*'. This review considers a number of relevant acts, conventions, and national strategies that are related to contaminants and environmental protection. These are summarised below and in Table 3.2.

#### 3.2.4.1 Environment Protection and Biodiversity Conservation Act 1999

The *EPBC Act* is Australia's key environmental protection legislation for activities likely to affect areas of national environmental significance. It provides a legal framework to protect and manage nationally and internationally important flora, fauna, ecological communities, cultural, and heritage places. The *EPBC Act* also provides a mechanism to meet Australia's international obligations as signatory and party to various treaties. The *EPBC Act* includes a process for environmental assessments and approvals, which is managed by the Commonwealth Department of Agriculture, Water, and the Environment (DAWE).

In 2014, the Commonwealth Minister for the Environment endorsed NOPSEMA's environmental management authorisation process under the *EPBC Act*, making NOPSEMA the sole Commonwealth environmental management regulator for offshore oil and gas activities. This streamlining process meant that NOPSEMA's environmental management regulatory functions incorporate the full scope of matters protected by the *EPBC Act*. However, any conditions for decommissioning applied to environmental approvals granted under the *EPBC Act* prior to the streamlining process must still be met.

The *EPBC Act* has defined eight objectives:

1. provide for the protection of the environment, especially matters of national environmental significance
2. conserve Australian biodiversity

3. provide a streamlined national environmental assessment and approvals process
4. enhance the protection and management of important natural and cultural places
5. control the international movement of plants and animals (wildlife), wildlife specimens and products made or derived from wildlife
6. promote ecologically sustainable development through the conservation and ecologically sustainable use of natural resources
7. recognise the role of Indigenous people in the conservation and ecologically sustainable use of Australia's biodiversity
8. promote the use of Indigenous peoples' knowledge of biodiversity with the involvement of, and in cooperation with, the owners of the knowledge.

Ecologically sustainable development is defined by the following five principles (*EPBC Act* s. 3A (a-e)):

- a. decision-making processes should effectively integrate both long-term and short-term economic, environmental, social and equitable considerations;
- b. if there are threats of serious or irreversible environmental damage, lack of full scientific certainty should not be used as a reason for postponing measures to prevent environmental degradation;
- c. the principle of inter-generational equity--that the present generation should ensure that the health, diversity and productivity of the environment is maintained or enhanced for the benefit of future generations;
- d. the conservation of biological diversity and ecological integrity should be a fundamental consideration in decision-making;
- e. improved valuation, pricing and incentive mechanisms should be promoted.

The *EPBC Act* was principally designed to protect matters of national environmental significance, of which nine are defined:

- world heritage properties
- national heritage places
- wetlands of international importance (as listed under the Ramsar Convention)
- listed threatened species and ecological communities
- migratory species
- nuclear actions (including uranium mines)
- Commonwealth marine areas
- the Great Barrier Reef Marine Park
- a water resource, in relation to coal seam gas development or large coal mining development

While matters of national environmental significance are a focus of the *EPBC Act*, its objectives offer broader protection goals for the environment. This is in recognition that ecosystems are complex and dynamic and the effective means of protecting biodiversity is by maintaining their functions (Commonwealth of Australia, 2019).

### 3.2.4.2 Environment Protection (Sea Dumping) Act 1981

The *Sea Dumping Act* meets Australia's obligations as a signatory to the *Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter* (as amended in 2006), also known as the *London Protocol*. The *Sea Dumping Act* aims to minimise pollution threats by:

- prohibiting ocean disposal of waste considered too harmful to be released in the marine environment and
- regulating permitted waste disposal to ensure environmental impacts are minimised.

Operators may apply to DAWE for a permit to dump material or place an artificial reef at sea. The permit application involves similar considerations around environmental impact and stakeholder consultation as the *OPGGS Act* requires when considering alternative decommissioning scenarios (APPEA, 2017). However, this requirement functions alongside provisions of the *OPGGS Act* and regulations meaning that one does not substitute for the other (DISER, 2018).

### 3.2.4.3 Australian Radiation Protection and Nuclear Safety Act 1998

The *Australian Radiation Protection and Nuclear Safety Act 1998 (ARPANS Act)* is Australia's legislation to ensure radiation protection and nuclear safety. The *ARPANS Act* established the ARPANSA as the regulator for radiation and nuclear safety matters. This includes protection of human health and the environment.

ARPANSA has developed the Radiation Protection Series which includes fundamentals, codes, and guides to promote practices protecting human health and the environment. The Fundamentals for Protection Against Ionising Radiation (ARPANSA, 2014) and Code for Radiation Protection in Planned Exposure Situations, RPS C-1, (ARPANSA, 2020) outline the principles and requirements for radiation safety. These are supported by the Guide for Radiation Protection of the Environment, RPS G-1, which outlines a process to assess the risk of radiation to non-human biota (e.g. marine organisms in this case) (ARPANSA, 2015b). Specific advice for NORM materials is detailed in the Code of Practice and Safety Guide for Radiation Protection and Radioactive Waste Management in Mining and Mineral Processing (ARPANSA, 2005) and the Safety Guide – Management of Naturally Occurring Radioactive Materials (ARPANSA, 2008b) which both reference NORM in the oil and gas industry.

ARPANSA's radiation protection series of requirements meets Australia's obligations as a member nation of the International Atomic Energy Agency (IAEA) (IAEA, 2014d; IAEA, 2018). The IAEA also provides guidance that is specific to NORM material (IAEA, 2003; IAEA, 2013) and radiation safety in the oil and gas industry (IAEA, 2003). The Code for Radiation Protection in Planned Exposure Situations (ARPANSA, 2020) details the relevant management objectives for NORMs hazards in decommissioning scenarios. They are:

- 3.1.12 The Responsible Person must ensure protection of people and the environment from exposure to radiation by the application of radiation control measures that are designed to optimise protection, taking into account:
  - a. the exposures controlled
  - b. the societal and economic factors
  - c. the impact on beneficial uses of radiation
- 3.3.1 The Responsible Person must ensure that disposal of radioactive material is only carried out in accordance with an authorisation issued by the relevant regulatory authority that takes protection of the public and the environment into account.

These requirements only apply to material that is not exempted or excluded, which are given in the National Directory for Radiation Protection (ARPANSA, 2017b) and based on the IAEA Basic Safety Standards (IAEA, 2004a). Exclusion criteria define exposures whose magnitude or likelihood is essentially not amenable to control through legislation. Exemption is defined by a regulatory authority

*a priori* for practices or sources within a practice where the exposure and the potential exposure due to the source or practice are too small to warrant the application of those aspects or that this is the optimum option for protection irrespective of the actual level of the doses or risks (IAEA, 2004a).

For NORM-contamination these criteria are:

- Exclusion criteria: 1 Bq/g for each of the parent and daughter radionuclides for a natural decay series.
- Exemption criteria: 10 Bq/g for the radionuclides  $^{226}\text{Ra}$ ,  $^{228}\text{Ra}$ , and  $^{210}\text{Pb}$  (total activity including progeny assumed to be in secular equilibrium) to a total activity of  $10^4$  Bq  $^{226}\text{Ra}$  or  $^{210}\text{Pb}$  or  $10^5$  Bq  $^{228}\text{Ra}$ .

For substances that do not meet this criteria, exposure should be treated as a planned exposure situation under RPS C-1 (ARPANSA, 2020). Other exemption levels exists for radiation risk from specific activities including the transportation of radioactive material and dumping of dredged material at sea:

- Code for the Safe Transport of Radioactive Material (Transport Code): 0.4 Bq/cm<sup>2</sup> or 0.04 Bq/cm<sup>2</sup> (e.g.  $^{210}\text{Po}$  only) averaged over 300 cm<sup>2</sup> (ARPANSA, 2008a; IAEA, 2018)
- National Assessment Guideline for Dredging: 35 Bq/g total radioactivity in material (Commonwealth of Australia, 2009)

These exemption criteria are specific to their activities so cannot be used in the context of decommissioning contaminated pipelines.

It should be noted that these values are typically derived to limit human exposure based on plausible exposure pathways. Substances that have an activity concentration below these criteria are not 'regulated' under the *ARPANS Act* but may still pose a risk to ecological receptors and should be assessed in accordance with the relevant legislation (such as the *EPBC Act*, *Environment Regulations*, or relevant state environmental protection legislation). The Guide for Radiation Protection of the Environment, RPS G-1 (ARPANSA, 2015b) provides fundamental concepts for the assessment of radiation impacts to non-human biota.

The *ARPANS Act* does not establish ARPANSA as a regulator for industries managing environmental risks from NORMs but intend for the relevant regulatory authorities to use the Radiation Protection Series of codes and guides in the regulation of radiation practices in planned exposure situations.

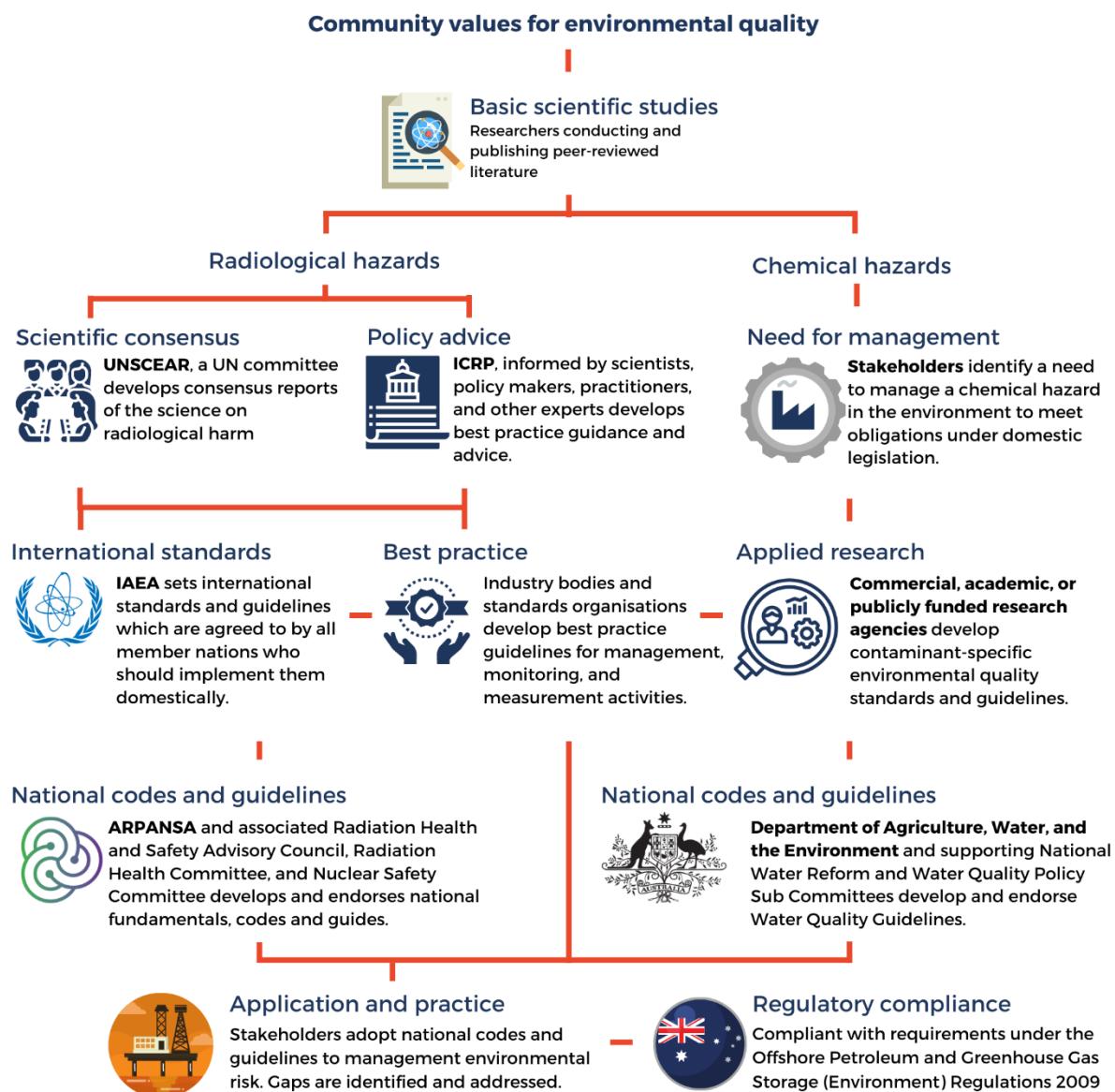
### 3.2.4.4 National Water Quality Management Strategy

The National Water Quality Management Strategy (including the Water Quality Management Framework, Sediment Quality Guidelines, and Water Quality Guidelines) facilitates water quality management for the productive and sustainable use of Australia's water resources and to protect community values, such as aquatic (marine and freshwater) ecosystems. The strategy is adopted by all Australian governments (i.e. the Commonwealth and state and territory governments) and have been developed with consideration to basic research, international best practice, community values, and industry needs (Figure 3.6).

The Water Quality Management Framework outlines a step-wise guide for the integrated environmental management and risk assessment of threats to aquatic ecosystems (ANZG, 2018). This framework aligns with the process outlined in AS/NZS ISO 31000: Risk management – Principles and guidelines. The framework can incorporate all types of pressures and stressors, including of a

radiological or chemical nature. The framework provides default water and sediment quality guidelines for contaminants, which indicate the concentration below which there is a low risk of unacceptable effects occurring, and are recommended to be used with other lines of evidence to protect aquatic ecosystems. These guidelines are typically used as threshold or trigger values, beyond which more detailed assessments are undertaken.

The strategy, framework, and guidelines are non-mandatory; however, they provide a robust process for contaminant risk assessment in aquatic ecosystems and may help to define acceptability for contaminants in decommissioning scenarios.



**Figure 3.6 The development of environmental management guidelines and standards for chemical and radiological hazards which support legislative requirements for environmental protection.**

Table 3.2 Legislation complimentary to the *OPGGS Act* and *Environment Regulations* and relevant for environment management of COPCs.

Legislation	Description
<b><i>Environment Protection and Biodiversity Conservation Act 1999</i></b>	Commonwealth's key piece of environmental legislation. Most activities that will or are likely to have a significant impact on matters of national environmental significance require approval from the Minister for the Environment and Energy under the <i>EPBC Act</i> before they can proceed. Meets part of Australia's commitments under the Convention on Biological Diversity, Convention on Migratory Species, Ramsar Convention on Wetlands, World Heritage Convention, and the Sustainable Development Goals. Defines the principles of ecologically sustainable development ( <i>EPBC Act</i> s. 3A) which are referenced as a management objective of the <i>Environment Regulations</i> (r. 3). In 2014 approval of activities under the <i>Environment Regulations</i> were also considered to satisfy <i>EPBC Act</i> requirements, eliminating the need for dual approvals.
<b><i>Australian Radiation Protection and Nuclear Safety Act (1998)</i></b>	Commonwealth legislation aimed at protecting the health and safety of people, and to protect the environment, from the harmful effects of radiation. Establishes ARPANSA as an advisory agency empowered to promote uniformity of radiation protection and nuclear safety policy and practices across all Australian jurisdictions. Defines exemption and exclusion criteria for radioactive substances. Meets Australia's obligations as a member state of the IAEA.
<b><i>Environment Protection (Sea Dumping) Act 1981</i></b>	Aims to address marine pollution by: (a) prohibiting ocean disposal of waste considered too harmful to be released; (b) regulating the disposal of controlled waste and the creation of artificial reefs, to ensure that environmental impacts are minimised; (c) prohibiting incineration at sea of wastes and other matter. Meets Australia's obligations under the London Protocol.
<b><i>UN Convention on Biological Diversity, Sustainable Development Goals, Ramsar Convention on Wetlands, the Convention on Migratory Species, World Heritage Convention</i></b>	Australia is a signatory to a range of international conventions and treaties. Some are met by specific legislation (e.g. the Basal Convention, London Protocol, ARPANS Act) while others are met by Australia's environmental management framework and strategies which includes the <i>EPBC Act</i> , National Water Quality Management Strategy, and the National Strategy for the Conservation of Australia's Biological Diversity.

Legislation	Description
<b>Minamata Convention on Mercury (2017)</b>	<p>A convention to protect human and environmental health from anthropogenic emissions and releases of mercury and mercury compounds. Australia signed the convention in 2013 but has not yet ratified it. Oil and gas are potentially affected by Article 9 which focuses on 'significant anthropogenic point sources'. However, a Regulatory Impact Statement produced by the Commonwealth Government found that there will be "no regulatory burden on business or the community, as existing regulatory frameworks broadly align with international obligations under the Convention" (DAWE, 2020). The Regulatory Impact Statement specifically referenced adherence to the 'ANZECC Water Quality Guidelines', the reporting in the National Pollutant Inventory, and the regulatory framework that exists under the <i>OPGGS Act</i> and <i>Environment Regulations</i> as meeting convention requirements for pollution control.</p>
<b>National Water Quality Management Strategy (2018)</b>	<p>A strategy developed and endorsed by Commonwealth and state and territory governments to preserve water quality so that it is 'fit for purpose'. Includes a water quality management framework and environmental reference levels for toxicity impacts of contaminants (ANZG, 2018). Enables environmental managers, industry, communities, and regulators to understand and assess the potential risk pressures and stressors to ecosystem receptors (Commonwealth of Australia, 2018).</p>

### 3.2.5 Summary of environmental protection requirements

The high-level environmental protection requirements that need to be met in decommissioning activities (defined by the *OPGGS Act*, *Environment Regulations*, and *EPBC Act*) can be summarised as:

- Making good any damage to the seabed and subsoil and providing for the protection of natural resources (*OPGGS Act s. 270 c(ii)*)) before a title may be relinquished (i.e. whether infrastructure is removed completely or decommissioned *in situ*. Additionally, deliver equal or better environmental outcomes if adopting a decommissioning option other than complete removal (DISER, 2018; NOPSEMA, 2020d), and
- Reduce environmental impacts and risks of decommissioning activities to as low as reasonably practicable and of an acceptable level (*Environment Regulations r. 3*), where principles of ecologically sustainable development (*EPBC Act s. 3A*) are integral to defining acceptable levels of impact and risk (NOPSEMA, 2019), including:
  - giving fundamental consideration to the conservation of biological diversity and ecological integrity
  - giving consideration to long-term risks and intergenerational equity
  - adopting a precautionary principle where uncertainty of impacts and risks exists

Meeting these requirements is challenging, as recognised by NOPSEMA, because ‘each environmental receptor, including different plants, animals, ecological communities, and the social, economic and cultural features of those, have a different level of sensitivity or resilience to the changes (impacts and risks) caused by a petroleum activity. Accordingly, consultation with relevant persons is an important part of establishing context for defining an acceptable level and successfully demonstrating it will be met.’ (NOPSEMA, 2019).

Operators are effectively being asked to create criteria that contextualises the legislative protection requirements to their specific asset, ecosystem, and community, and demonstrate how they will be met. We refer to these criteria as the management objectives for decommissioning activities.

Two relevant ecological risk assessment frameworks exist in Australia that may help understand the risk and impacts of mercury and NORM contaminants against these management objectives: the Water Quality Management Framework (which includes water and sediment quality guidelines) and ARPANSA’s Guide for Radiation Protection of the Environment (ANZG, 2018; ARPANSA, 2015b). These provide the process and environmental reference levels to understand and interpret risk, respectively. Fundamentally they follow the typical components of an ecological risk assessment, which includes: (i) problem identification (i.e. understanding NORMs or mercury behaviour, forms, and concentrations in the contaminated product, and the ecosystem receptors they may impact); (ii) hazard characterisation; (iii) exposure characterisation; and (iv) a quantitative, qualitative, or combination thereof risk assessment process (Suter II, 2006). This is the general format adopted for the rest of the literature review.

### 3.3 Behaviour of mercury in oil and gas systems

This section reviews the behaviour of mercury in oil and gas systems. An understanding of mercury in oil and gas systems is important to understanding potential consequences to ecosystem receptors. This will also develop an understanding that can be applied to assess how different decommissioning scenarios affect the risk of mercury to marine ecosystems. This section partly meets project objectives 3 and 4 and problem statement 2.

#### Key findings:

- Mercury naturally occurs and cycles in the marine and terrestrial environments, and atmosphere.
- A wide range of elemental, ionic, and organic mercury species exist in oil and gas systems and likely partition to form discrete contamination products.
- Mercury cannot penetrate steel and will only form contaminated products on surfaces.
- Internally-coated pipelines are not reported to accumulate mercury in contamination products. There is uncertainty around mercury's accumulation in CRA and unbonded flexible pipelines.
- Mercury that remains in oil and gas pipelines that are left in the ocean will eventually be released into the marine environment as the pipeline corrodes.

#### 3.3.1 Background and forms

Mercury is a volatile metal that has a complicated cycle in the environment. Mercury is found in all domains, the geosphere, atmosphere, and hydrosphere, and is transported between these because of biogeochemical transformations and partitioning. Mercury concentrates in oil and gas basins through natural geological processes, creating the potential for mercury-contamination in oil and gas systems and production fluids. It is found in water, oil, and gas phases within production systems, and exists in various chemical species depending on the environments' pressure, temperature and chemistry. Different chemical species of mercury have different behaviours in oil and gas systems because of different physical and chemical properties. These properties lead to its partitioning throughout oil and gas systems and ultimately accumulation in different contaminated products.

Mercury exists as elemental, ionic, and organic species, each with different chemical properties.

##### 3.3.1.1 Elemental mercury

Elemental mercury ( $Hg^0$ ) is also known as colloidal mercury, liquid silver, quicksilver, and hydrargyrum. It has a relative molecular mass of 200.59 g/mol, a melting point of  $-38\text{ }^\circ C$ , a boiling point of  $356\text{ }^\circ C$ , and a density of  $13.534\text{ g/cm}^3$  at  $25\text{ }^\circ C$ .

Elemental mercury is the most volatile form of mercury. It has a vapour pressure of 0.3 Pa at  $25\text{ }^\circ C$  and transforms into the vapour phase at typical room temperatures. It is relatively insoluble in water (56 µg/L at  $25\text{ }^\circ C$ ) and hydrochloric acid, but is soluble in lipids, nitric acid, pentane (2.7 mg/L), and sulfuric acid when boiled.

##### 3.3.1.2 Ionic mercury compounds

Ionic mercury occurs as salts of its divalent and monovalent cationic species. Mercuric chloride ( $HgCl_2$ ) is also known as mercury chloride, mercury perchloride, and dichloromercury, corrosive

sublimate. It has a relative molecular mass of 271.52 g/mol, a melting point of 277 °C, and a boiling point of 302 °C. It occurs as white crystals, granules, or powder; rhombic crystals; or a crystalline solid. Mercuric chloride has a vapour pressure of 0.1 kPa at 136.2 °C and a water solubility of 28.6 g/L, which increases to 476 g/L in boiling water. It has a solubility in alcohol of 263 g/L.

Mercurous chloride ( $Hg_2Cl_2$ ) is also known as calomel, mild mercury chloride, mercury monochloride, mercury protochloride, and calogreen. It has a relative molecular mass of 472.09 g/mol and a boiling point of 384 °C, and it sublimes at 400-500 °C without melting. It occurs as a white heavy powder, rhombic crystals, or a crystalline powder. The solubility of mercurous chloride is 2 mg/L at 25 °C. It is insoluble in alcohol and ether.

Mercuric sulfide ( $HgS$ ) has a relative molecular mass of 232.68 g/mol and occurs as a heavy amorphous powder, as black cubic crystals (mercuric sulfide, black; metacinnabar) or a powder, as lumps, or as hexagonal crystals (mercuric sulfide, red; cinnabar). Red mercuric sulfide is also known as vermillion, Chinese red, Pigment Red 106, C.I.77766, quicksilver vermilion, Chinese vermilion, artificial cinnabar, and red mercury sulfuret.

Mercuric sulfide transitions from red to black at 386 °C. Black mercuric sulfide sublimes at 446 °C, and red mercuric sulfide at 583 °C. Black mercuric sulfide is insoluble in water, alcohol, and dilute mineral acids. Red mercuric sulfide is insoluble in water, but dissolves in *aqua regia* (with separation of sulfur) and warm hydriodic acid (with the evolution of hydrogen sulfide).

Mercuric acetate ( $HgC_4H_6O_4$ ) has a relative molecular mass of 318.70 g/mol. It is white in colour, and it occurs either as crystals or as a crystalline powder. It is soluble in water (250 g/L at 10 °C; 1000 g/L at 100 °C) and in alcohol or acetic acid. Mercuric acetate is also known as bis(acetoxymercury), diacetoxymercury, mercury diacetate, mercuriacetate, and mercury acetate.

### 3.3.1.3 Organic mercury

Organic mercury species are those mercury ions bonded to a carbon-based moiety. Common organic mercury species include methylmercury ( $CH_3Hg^+$ ), dimethylmercury ( $CH_3)_2Hg$ , or mercury bound to longer chained alkyl species. Other forms of organic mercury can exist in hydrocarbons, such as phenyl mercury  $Hg(C_6H_5)_2$ , and chloro-methylmercury  $ClHgCH_3$ .

Methylmercury is a positively charged ion that has a high binding affinity to sulfur-ligands, such as thiol groups found in amino acids and proteins. Methylmercury is the most bioaccumulative mercury species and is formed through biochemical processes of bacteria in environmental media. This process is governed by environmental conditions.

### 3.3.2 Mercury in the environment

Mercury is found ubiquitously in the environment. In fresh and marine waters, the dominant form of mercury is ionic (90%) with the remainder made up of elemental and methylated forms. Its concentration in environmental media is geographically variable. In Australia, background total mercury concentrations are typically low, being <0.02 mg/kg in sediments and <0.5 ng/L in marine waters (Cossa et al., 2011). These reflect inputs from a range of natural sources (Figure 3.7). However, these concentrations can become elevated from anthropogenic emissions, including mining, coal burning oil and gas production and manufacturing activities.

### 3.3.2.1 Natural sources

Natural sources of mercury are responsible for approximately 5000 t/y, with the majority coming from oceanic emissions, vegetation and soil emissions, and the burning of biomass (Pirrone et al., 2010; UNEP, 2013). The weathering of rocks and volcanic activity are other important sources (Figure 3.7). Natural sources of mercury typically cycle between volatilisation and deposition.

### 3.3.2.2 Anthropogenic sources

Anthropogenic activity is responsible for approximately 2,000 t/y of mercury emissions to the environment per year (Pirrone et al., 2010). The combustion of fossil fuels is the largest contributor, followed by the mining and smelting of metals and cement production (Nelson et al., 2012). Mercury has also been highly utilised in industries such as the electrical industry (switches, thermostats and batteries), dentistry (dental amalgams), and other industrial processes such as the production of caustic soda, nuclear reactors, as a solvent for metal, and as a preservative in pharmaceuticals (Tchounwou et al., 2003). Other anthropogenic sources of mercury include produced water discharges to the marine environment (Gallup and Strong, 2007). Industrial demand for mercury peaked in 1964 but has since been declining following bans and legislative measures on its use. Between 1990 and 2010, the anthropogenic emissions of mercury declined by 30% (Zhang et al., 2016).

Anthropogenic releases to aquatic environments is >1000 t/y, and the mercury content in the top 100 m of the world's oceans have doubled in the last 100 years (UNEP, 2013). Human inputs of mercury contamination into the marine environment can be direct from dredge disposal and industrial activities, or via diffuse pathways such as riverine and atmospheric inputs. However, the complexity and wide spatial and temporal variations in natural mercury processes make it difficult to confidently identify impacts of human-induced inputs.

## 3.3.3 Mercury in oil and gas basins

Mercury is well reported trace component in oil and gas reservoirs worldwide. There are competing theories attempting to explain how mercury concentrates in oil and gas basins. One possibility is that it originates from accumulated mercury in the organisms that lead to the formation of natural gas and hydrocarbons (Lang et al., 2012; Wilhelm, 2001a).

Others suggest that mercury migrates into the oil and gas reservoirs from source rocks and ore-fluids. This migration process may be encouraged by secondary processes such as geological and geothermal activities (Abai et al., 2015; Yan et al., 2017). Several authors believe that regions with higher tectonic and geological activity are responsible for higher mercury content within the natural gas reservoirs (Gallup, 2014; Lang et al., 2012; Yan et al., 2017). Consequently, interaction with metal-rich fluids, mineral matters, and formation waters also have a relationship with elevated mercury content (Filby, 1994).

No matter the origin, the mercury content of oil and gas basins and its global distribution may be affected by geographical location, geology, age of the basins, as well as tectonic and seismic activity of the region (Ryzhov et al., 2003; Wilhelm and Bloom, 2000). Levels of mercury reported by Ryzhov et al. (2003) in natural gases varied considerably over 20 years within the same location. The geographical variability of mercury in oil and gas basins is given in Table 3.3.

Mercury ranges from as low as 0.01 µg/m<sup>3</sup> in South America to as high as 1930 µg/m<sup>3</sup> in China (Ezzeldin et al., 2016; Lang et al., 2012; Liu, 2013; Mussig and Rothmann, 1997; Shafawi, 1999; Wilhelm et al., 2007; Woodside Energy Ltd, 2014; Yan et al., 2017).

The speciation of mercury in oil and gas basins may also be affected by basin geochemistry. Natural gas that contains elevated amounts of hydrogen sulfide (H<sub>2</sub>S) and carbon dioxide (CO<sub>2</sub>) is commonly termed as 'sour' gas. When these two sour gases are present at low concentrations, the gas is termed as 'sweet' gas. Authors have observed that in sour gas reservoirs, mercury is usually detected as its sulfur-bound species (HgS, HgS<sub>x</sub><sup>y-</sup>). On the other hand, mainly elemental mercury (Hg<sup>0</sup>) and several other species are usually detected in sweet gas reservoirs (Gallup, 2014).

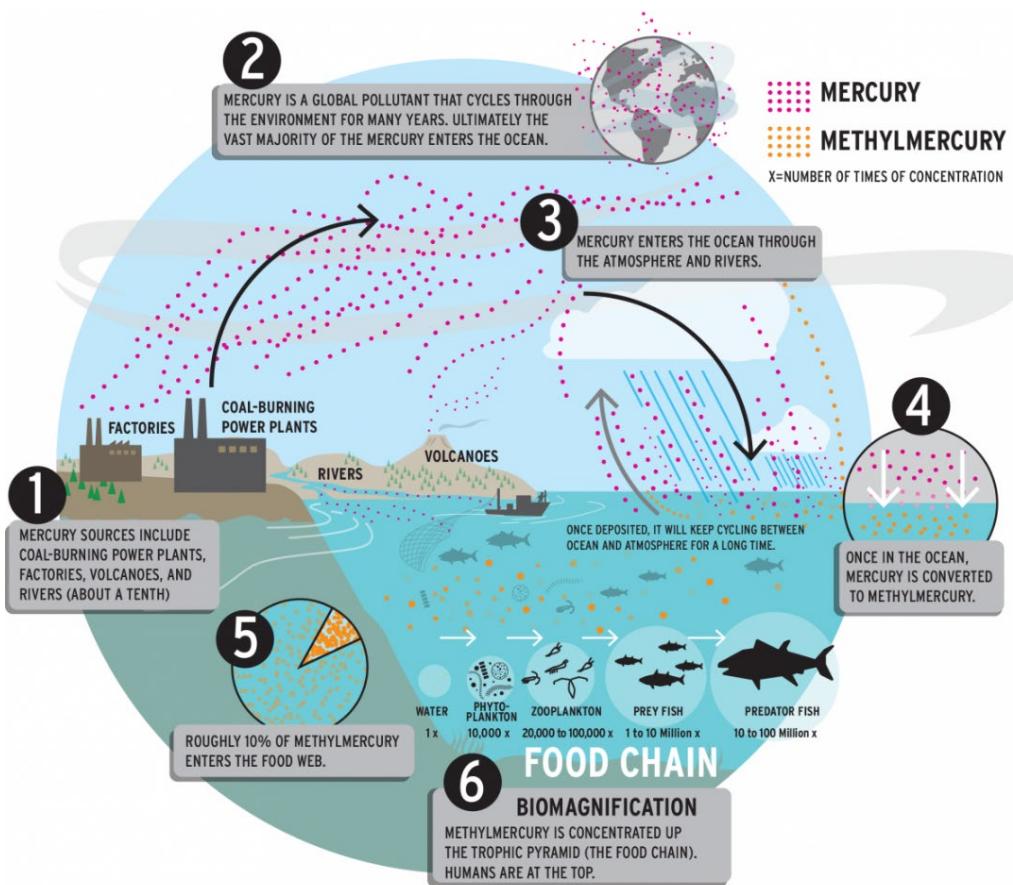


Figure 3.7 The natural and anthropogenic mercury cycle. Taken from Humphries (2018).

**Table 3.3 Mercury concentration within oil and gas reservoirs at different geological regions. Summarised from various sources including Shafawi (1999) and Ezzeldin et al. (2016).**

Region	Natural Gas ( $\mu\text{g}/\text{m}^3$ )	Condensate ( $\mu\text{g}/\text{kg}$ )	Crude Oil ( $\mu\text{g}/\text{kg}$ )
Europe	<0.1-450	-	3.6-19.5
Norway	0.5-30	-	19.5
Mexico	0.02-0.4	-	1.3
South America	0.01-120	-	5.3
Thailand	100-400	400-1200	593.1
Malaysia	1-200	10-100	-
Indonesia	0.1-300	10-500	-
Australia	38-83	0.035-0.041	0.8
China	0.015-1930	-	-
Middle East	1-9	-	0.8
Africa	1.25-200	20-1117	0.3-13.3

### 3.3.4 Mercury partitioning in production systems

Mercury has been reported to exist in multiple species of elemental, organic, and ionic forms (Wilhelm, 2001b; Wilhelm and Bloom, 2000). The behaviour of different mercury species in oil and gas processes differs according to their physical and chemical properties; including solubility, volatility, and binding affinity to different compounds. These differences lead to different partitioning in oil and gas phases or products in the production system (Wilhelm, 1999a). Figure 3.8 provides a schematic of the simplest form of an oil production system, a 3-phase separator connected to a single well, and defines the various partitioned mercury species which have been detected in all three phases (gas, water and condensate/oil phase).

Different mercury species have different solubilities in oil and gas phases (Edmonds et al., 1996), as shown in Table 3.4, and different volatilities as measured by their boiling point, Table 3.5. Different solubilities leads to preferential partitioning to the different production phases shown in Figure 3.8, and their different volatilities may lead to separation due to the different temperatures encountered along the extraction and processing system. For example, species with higher boiling points will be the first to condense from a gas to liquid when experiencing reduced temperatures.

During the extraction from the reservoir to the primary gas-liquid separator, the stream experiences a decrease in pressure and temperature which has been reported to cause condensation of small and large droplets of  $\text{Hg}^0$ . The smaller droplets have been found to remain suspended inside both the oil and water phase while the larger droplets tend to partition in the water phase because of different frictional forces experienced in fluids, a phenomenon described by Stoke's law (Salvá and Gallup, 2010). The condensation of mercury may lead to its physical incorporation into the other deposits including sludges in pipelines (Salvá and Gallup, 2010).

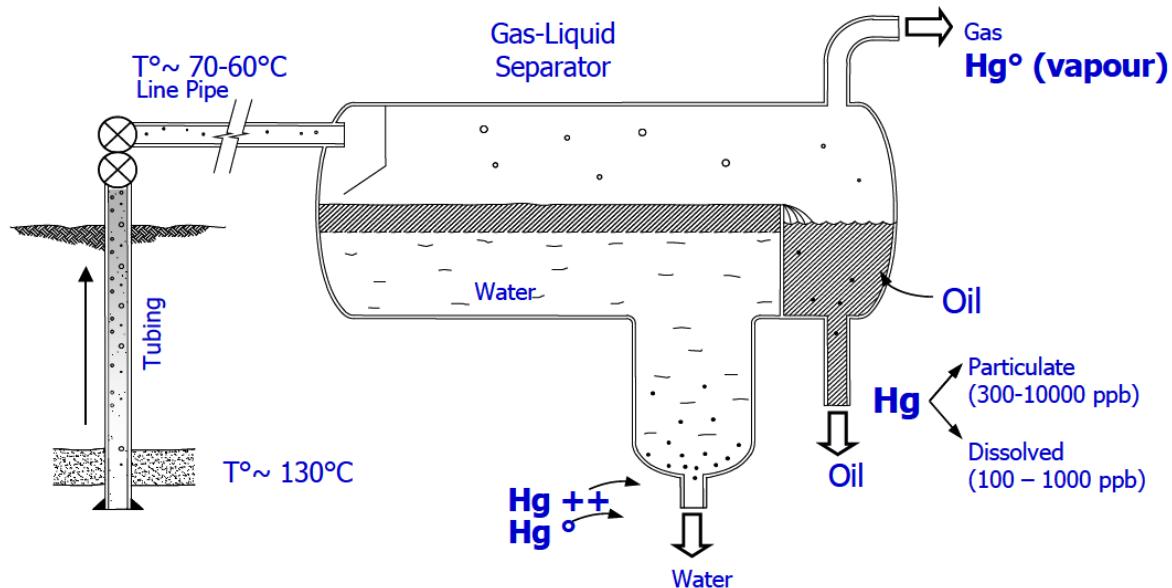


Figure 3.8 Mercury distribution in oil field separation system.  $\text{Hg}^0$  indicates elemental mercury,  $\text{Hg}^{++}$  indicates ionic mercury. Dissolved mercury may include organic mercury species. Taken from Salvá and Gallup (2010).

Table 3.4 The solubility of several mercury species in different oil and gas phases. (Wilhelm, 2001b).

Mercury Compound	Water (ppm)	Oil (ppm)	Glycol (ppm)
$\text{Hg}^0$	0.05	2	<1
$\text{HgCl}_2$	70,000	>10	>50
$\text{HgS}$	0.01	<0.01	<0.01
$\text{HgO}$	50	low	-
$\text{CH}_3\text{HgCl}$	>10,000	1,000	>1,000

Table 3.5 Boiling points of various mercury species (°C).

Mercury Species	Boiling Point (°C)
Elemental Mercury; $\text{Hg}^0$	356
Mercuric Chloride; $\text{HgCl}_2$	302
Di-isopropyl mercury; $\text{Hg}(\text{iC}_3\text{H}_7)_2$	170
Dipropyl mercury; $\text{Hg}(\text{C}_3\text{H}_7)_2$	190
Dimethyl mercury; $\text{Hg}(\text{CH}_3)_2$	93
Diethyl mercury; $\text{Hg}(\text{C}_2\text{H}_5)_2$	159
Dibutyl mercury; $\text{Hg}(\text{C}_4\text{H}_9)_2$	223
Diphenyl mercury; $\text{Hg}(\text{C}_6\text{H}_5)_2$	204

The speciation of mercury (and therefore its propensity to partition into different fluid phases or to different infrastructure) may also change within the oil and gas production systems. The oxidation of Hg<sup>0</sup> in crude oil leads to the formation of water-soluble ionic mercury compounds and complexes. Hg<sup>0</sup> and ionic mercury species are known to be very reactive with sulfur, forming the generally insoluble HgS species (Wilhelm, 2001b). Suspended HgS may be methylated by anaerobic bacteria into a water-soluble organic form (Chen et al., 2017; Gustafsson et al., 2007). However, dissolution rates are generally lower than 0.1%, and Jonsson et al. (2012) reports that metacinnabar had a methylation rate constant of ~0.001 d<sup>-1</sup>, up to two orders of magnitude lower than other geochemically-relevant mercury compounds.

### 3.3.5 Mercury speciation in oil and gas products

Production fluids contain several mercury species that co-exist within the multiphase nature of the fluids while being transported within the production tubing, subsea equipment and flowlines/risers prior to the upstream processing facility (Table 3.6). Mercury speciation that has been reported within the different phases have been reviewed and summarised in this section.

#### 3.3.5.1 Mercury Speciation in the Gas Phase

A range of mercury species has been detected in gas samples from different fields (Table 3.6). Measurements of mercury in gas products indicated presence of several mercury species; however, it is difficult to generalize their total concentrations and species distribution as they are highly dependent on geological location (Ryzhov et al., 2003). Zettlitzer et al. (1997) have reported total mercury concentration as high as 700-4400 µg/m<sup>3</sup> in the north German gas reservoirs, while Ezzeldin et al. (2016) reported a total mercury concentration of 1.25 µg/m<sup>3</sup> in the outlet gas of the primary separation unit in Egypt.

Elemental mercury, Hg<sup>0</sup>, is believed to be the dominant species in natural gas, making up of over 50% of total mercury detected. Wilhelm and Bloom (2000) reported dimethylmercury (Hg(CH<sub>3</sub>)<sub>2</sub>) in the gas phases in very small amounts (<1%). Several other organic mercury species would likely be present in the natural gas due to their volatile nature.

Ionic mercury is rarely detected in gas phases (Wilhelm, 2001b). Hg<sup>0</sup> is unstable and easily converted to other species through oxidation and reduction processes (Yan et al., 2017). In the presence of ions such as chlorides, sulfates and other sulfur-bound compounds, oxidized Hg<sup>0</sup> may react to form more stable compounds such as HgCl<sub>2</sub> and other chloride ligands (Kozin and Hansen, 2013; Ma et al., 2014) and other types of organic mercury with R-Hg-X structures (Zettlitzer et al., 1997). However, this can only occur when saline water is present with the gas.

#### 3.3.5.2 Mercury Speciation in Water Phase

Table 3.6 summarises the mercury species detected in produced water samples from different fields. Several authors have reported the presence of mercury in both the produced water stream from the primary separation unit and the wastewater stream (Corvini et al., 2016; Ezzeldin et al., 2016; Gallup, 2014; Gustafsson et al., 2007; Lothongkum et al., 2011). Typical mercury concentrations detected in produced waters from several well-known gas and oil fields are between <0.001-0.3 mg/L (Ahmadun et al., 2009; Gallup and Strong, 2007; Zettlitzer et al., 1997). Mercury occurs in the water phase in all forms, namely elemental, ionic, organic, and adsorbed to particulate matter (Gustafsson et al., 2007). Ionic species, such as HgCl<sub>2</sub> are very water soluble and have the highest abundance at 10-50% of

total mercury (Wilhelm, 2001b). However, not all ionic species are soluble, with HgS being insoluble in hydrocarbon and water phases at typical process conditions.

The presence of sulfur and its compounds in gas streams have resulted in the formation of mercury complexes such as C-S-Hg-S-C in produced waters. Table 3.6 shows the presence of HgCl<sub>2</sub> in most cases except for the samples analysed by Zettlitzer et al. (1997). The results of Zettlitzer et al. (1997), however, are only based on two samples, one in which all mercury species were below the instruments' detection limit of 0.1 mg/L.

### 3.3.5.3 Mercury Speciation in Gas Condensate/Crude Oil

All mercury forms are detected in gas condensates and crude oil samples. These are summarised in Table 3.6 for analysed samples from several different fields. Detected species include Hg<sup>0</sup>, methylmercury HgCH<sub>3</sub>, ethyl mercury HgC<sub>2</sub>H<sub>5</sub>, dimethylmercury Hg(CH<sub>3</sub>)<sub>2</sub>, diethyl mercury Hg(C<sub>2</sub>H<sub>5</sub>)<sub>2</sub>, phenyl mercury HgC<sub>6</sub>H<sub>5</sub>, diphenyl mercury Hg(C<sub>6</sub>H<sub>5</sub>)<sub>2</sub>, HgCl<sub>2</sub>, CH<sub>3</sub>HgCl, Hg<sub>2</sub>Cl<sub>2</sub>, and suspended mercury (as solid HgS).

Ionic mercury, such as HgCl<sub>2</sub>, is abundant dissolved in gas condensates and crude oil because of its solubility. Tao et al. (1998) detected the presence of HgCl<sub>2</sub> within the range of 53-97% of the total dissolved mercury in several condensate samples. Similarly, 58-85% of the total mercury in gas condensates detected by Zettlitzer et al. (1997) was found to be HgCl<sub>2</sub>.

Although present in multiple species, suspended mercury (HgS) was found to be present in both condensate and crude oil, making up ~60% of the total mercury detected by Zettlitzer et al. (1997). Avellan et al. (2018) and Ruhland et al. (2019) reported the presence of HgS nanoparticles of various sizes (mostly <100 nm but some larger) in the condensate.

The occurrence of other forms of organic mercury is still debated because of the challenges associated with sampling and preserving mercury species that are unstable outside oil and gas system conditions. For example, organic mercury tends to undergo species interconversion and photolysis to form Hg<sup>0</sup>, which might explain the abundance of Hg<sup>0</sup> in condensate (Bouyssiere et al., 2000; Dessy and Lee, 1960; McAuliffe, 1977).

Work by Schickling and Broekaert (1995) has also observed reactions between Hg(C<sub>6</sub>H<sub>5</sub>)<sub>2</sub> with HgCl<sub>2</sub> and CH<sub>3</sub>HgCl respectively to form other types of organic mercury. Furthermore, Gaulier et al. (2015) did not specify the actual non-ionic organic mercury detected in their samples. However, their analysis was based on the standards of Hg<sup>0</sup>, Hg(C<sub>6</sub>H<sub>5</sub>)<sub>2</sub>, HgCl<sub>2</sub>, mercury (II) pentyl, thiolate, octyl, and tetradecyl. This may indicate those species were present in the samples analysed.

The likelihood and abundance of organic mercury in condensate is higher due to the fact that it is more soluble in organic solutions compared to water (Haynes, 2017). Several authors have suggested that the partitioning of mercury into gas condensate is generally related to the boiling point of gas condensate itself (Edmonds et al., 1996; Shafawi, 1999). Table 3.5 presents possible mercury species that may be present based on their boiling points (Haynes, 2017). As gas condensates are made up of several straight chain alkanes (C<sub>2</sub>-C<sub>6</sub>+ ) with different boiling points, it is suspected that organic mercury will partition into those hydrocarbon phase as they condense from the gas phase.

**Table 3.6 Mercury species detected in different phases of oil and gas production fluids, including gas condensates and crude oil, water phases, and gas phases. Mercury species are described as elemental (solid or volatile), ionic, and organic.**

Elemental Hg	Ionic Hg	Organic Hg	Source
<b>Gas condensates and crude oil phases</b>			
Hg <sup>0</sup>	Ionic Hg (Hg <sup>+</sup> , Hg <sup>2+</sup> )	HgCH <sub>3</sub> , Hg(CH <sub>3</sub> ) <sub>2</sub> , HgC <sub>2</sub> H <sub>5</sub> , Hg(C <sub>2</sub> H <sub>5</sub> ) <sub>2</sub>	(Frech et al., 1996)
Hg <sup>0</sup>	HgCl <sub>2</sub>	Hg(CH <sub>3</sub> ) <sub>2</sub> , HgC <sub>3</sub> H <sub>8</sub> , Hg(C <sub>2</sub> H <sub>5</sub> ) <sub>2</sub> , Hg(C <sub>6</sub> H <sub>5</sub> ) <sub>2</sub>	(Tao et al., 1998)
Hg <sup>0</sup>	HgCl <sub>2</sub> , Hg <sub>2</sub> Cl <sub>2</sub> , HgS	CH <sub>3</sub> HgCl, HgCH <sub>3</sub> , Hg(CH <sub>3</sub> ) <sub>2</sub>	(Schickling and Broekaert, 1995)
Hg <sup>0</sup>	Hg <sup>2+</sup>	-	(Bouyssiere et al., 2002)
Hg <sup>0</sup>	Ionic Hg (not specified)	HgCH <sub>3</sub>	(Ezzeldin et al., 2016)
Hg <sup>0</sup>	HgCl <sub>2</sub> , Hg <sub>2</sub> Cl <sub>2</sub>	-	(Snell et al., 1996)
Volatile Hg	Ionic Hg (Hg <sup>+</sup> , Hg <sup>2+</sup> )	Non-ionic organic Hg (Not specified)	(Gaulier et al., 2015)
Hg <sup>0</sup>	Hg <sup>2+</sup> , HgS	HgCH <sub>3</sub> , HgC <sub>2</sub> H <sub>5</sub> , HgC <sub>6</sub> H <sub>5</sub>	(Zettlitzer et al., 1997)
Hg <sup>0</sup>	Ionic Hg	R-Hg ionic, R-Hg-R	(Nengkoda et al., 2009)
<b>Water phases</b>			
-	HgCl <sub>2</sub>	-	(Lothongkum et al., 2011)
Hg <sup>0</sup>	HgCl <sub>2</sub> , Hg <sub>2</sub> Cl <sub>2</sub> , HgS, HgO, C-S-Hg-S-C	Hg(CH <sub>3</sub> ) <sub>2</sub> , CH <sub>3</sub> HgCl	(Gallup, 2014)
-	HgCl <sub>2</sub>	HgCH <sub>3</sub>	(Zettlitzer et al., 1997)
Hg <sup>0</sup>	HgCl <sub>2</sub>	-	(Tao et al., 1998)
<b>Gas phases</b>			
Hg <sup>0</sup>	-	Hg(CH <sub>3</sub> ) <sub>2</sub>	(Wilhelm, 2001b)
Hg <sup>0</sup>	Ionic Hg	-	(Nengkoda et al., 2009)
Hg <sup>0</sup>	-	-	(Bingham, 1990)
Hg <sup>0</sup>	-	R-Hg-X (X= sulfate, halide or -S-R), R-Hg-R	(Zettlitzer et al., 1997)
Hg <sup>0</sup>	-	-	(Schickling and Broekaert, 1995)
Hg <sup>0</sup>	-	R-Hg-X and R-Hg-R	(Abbas et al., 2016)
Hg <sup>0</sup>	-	-	(Mussig and Rothmann, 1997)
Hg <sup>0</sup>	-	Hg(CH <sub>3</sub> ) <sub>2</sub>	(Wilhelm and Bloom, 2000)

### 3.3.6 Mercury contaminated products in oil and gas pipelines

Mercury partitions from production fluids in oil and gas infrastructure during production due to its chemical behaviour at different points of separation and extraction process. This partitioning leads to the potential for mercury-species specific accumulation in some infrastructure, such as in pipelines carrying raw well stream fluid, and single or dual phase streams downstream of separation systems.

Mercury accumulation within the oil and gas infrastructure is commonly identified by a delay in the detection of mercury in downstream oil and gas processing facilities. This is termed a ‘mercury lag’ and occurs because mercury accumulates in the pipelines system between the production wells and the downstream facilities. For example, Wilhelm and Nelson (2010) reports that for an offshore/onshore production system, operating for approximately 4 years, the mercury gas concentrations at the onshore facility measured less than 1 µg Hg/m<sup>3</sup> compared to the 200-300 µg Hg/m<sup>3</sup> at the wellhead. An estimate of the mercury lag time suggests that the pipeline has the capacity to retain 1.5-2 g/m<sup>2</sup> of pipe that transports both dry gas and hydrocarbon fluids (Table 3.7). Mercury accumulation in pipeline systems is further evidenced by published mercury concentrations. Zettlitz and Kleinitz (1997) reported total mercury concentrations of 1-80 mg/kg in the tubing and piping used for natural gas production system.

The accumulation of mercury is also known to depend on the pipeline material and any internal coatings, the pipeline temperature, and pressure conditions, and the mercury speciation. For example, pipelines coated with a flow coating layer may prevent the build-up of corrosion products that can accumulate mercury (Collet and Chizat, 2015).

Different mechanisms leading to mercury accumulation in production infrastructure include:

- Deposition of elemental mercury to pipeline surfaces by condensation (coated, corrosion-resistant, and uncoated)
- Adsorption to steel surfaces (in uncoated steel pipelines)
- Adsorption into corrosion products (in uncoated steel pipelines)
- Accumulation in sludges such as asphaltenes

Some mercury-contaminated products may be removed from infrastructure at the end of its productive life following routine cleaning including flushing and pigging. For example, loosely deposited mercury products such as elemental mercury deposits or some sludge products may be readily moved with production fluids in pipes or through pigging or flushing programs. There is a lack of available data to categorically state what types of mercury-contaminated products can be removed this way or the efficacy of these cleaning techniques, so all identified contaminated products are included for completeness.

#### 3.3.6.1 Adsorption and deposition of mercury on pipelines

Elemental mercury cannot penetrate (absorb) into the steel because of its large atomic size relative to the steel crystal lattice (Wilhelm and Nelson, 2010). Instead, Hg<sup>0</sup> is reported to adsorb to steel surfaces forming a monolayer (Linderoth and Morup, 1992; Roseborough et al., 2006), or as adsorbed Hg<sup>0</sup> droplets, approximately 1-10 µm in size (Chaiyosit et al., 2009). The process of elemental Hg<sup>0</sup> gas adsorption to pipe walls has been modelled by the firm Genesis (Crafts and Williams, 2020). They report that the process is reversible with a temperature dependency based on Fick’s first law of diffusion.

**Table 3.7 Estimation of mercury lag time in transport pipeline. (Wilhelm and Nelson, 2010).**

Capacity of Pipe Surface (g Hg/m <sup>2</sup> )	Time to Reach Shore (months)
1	28
2	56
5	140

\*Pipe length = 200 km, flow rate of 40,000 m<sup>3</sup>/h, mercury flow rate = 10 g/h.

Where H<sub>2</sub>S is present in the gas stream, the formation of particulate HgS may occur. This reaction is not favoured in the gas phase at process temperatures, so its reaction on surfaces or in solution following Hg oxidation is more likely (Wilhelm and Nelson, 2010). HgS may settle in pipelines if not carried by production fluids or adsorbed to steel, where it has been measured as a dominant mercury phase at concentrations of 0.05-0.1 g Hg/m<sup>2</sup> at a gas production unit (Wilhelm and Nelson, 2010).

Elemental mercury may also condense on surfaces when experiencing changing pressures and temperatures (Section 3.3.4). This may lead to the accumulation of elemental mercury depositions, particularly in low points of pipeline systems.

### 3.3.6.2 Adsorption in corrosion products

Corrosion products are typically oxidized steel components and may include iron oxides (Fe<sub>3</sub>O<sub>4</sub>, FeOOH) in carbon-steel pipes, oxides of chromium and nickel in corrosion resistance pipes, or iron sulfide (FeS, FeS<sub>2</sub>) where H<sub>2</sub>S is present. These form a corrosion layer sometimes referred to as a scale on the internal surface of pipes.

In uncoated pipelines, Hg<sup>0</sup> reacts with iron oxide and iron sulfide scales, which form a mercury-rich layer on the pipe walls (Wilhelm, 1999b). Mercury is adsorbed into the iron oxide scale via the formation of a surface complex while iron sulfide has a high affinity towards mercury where adsorption will take place (Gong et al., 2019). As a result, the major species reported in corrosion layers is HgS.

HgS in the form metacinnabar ( $\beta$ -HgS) made up ~25% of the compounds detected in a mercury-contaminated corrosion scale that also included FeOOH, FeCO<sub>3</sub> and Fe<sub>3</sub>O<sub>4</sub> (Wilhelm and Nelson, 2010). It is likely that adsorbed Hg<sup>0</sup> in the corrosion product reacts with H<sub>2</sub>S to form HgS rather than reacting in the gaseous phase which is not favoured (Hall et al., 1991). Other mercury species such as HgCl, HgCl<sub>2</sub>, Hg<sub>2</sub>Cl<sub>2</sub>, HgO<sub>2</sub> and HgO have also been reported within scale associated with gas transmission pipelines (Chanvanichskul et al., 2017).

The adsorption of mercury to corrosion products is superficial to the pipeline steel as mercury cannot penetrate the non-corroded steel. This has been confirmed by measuring mercury at different depths in the steel, which found <0.1% at 10 nm under the corrosion product layer (Chaiyasit et al., 2009). Similarly, Chanvanichskul et al. (2017) reported no mercury inside the pipe metal after removal of the corrosion product.

The characterization of a carbon steel pipe cross-section collected downstream of the primary separator in a gas production unit is shown in Figure 3.9 and Figure 3.10. Mercury was found both in the scale and on the surface of the metal as HgS and as Hg<sup>2+</sup> incorporated into the iron oxide scale with a concentration of <0.2 g of Hg/m<sup>2</sup> of pipe (Wilhelm and Nelson, 2010). The deposited HgS layer accounts for 0.05-0.1 g Hg/m<sup>2</sup> pipe. It can be seen from Figure 3.10 that mercury does not accumulate at the interface of both the scale and metal interface, suggesting that mercury did not form an amalgam with the steel itself.

Factors affecting the corrosion of stainless/carbon steel may increase the accumulation of mercury. For example, chloride ions and CO<sub>2</sub> are reported to accelerate steel corrosion (Pojtanabuntoeng et al., 2011). Flow-coated pipelines prevent the formation of these corrosion layers, meaning this type of mercury accumulation should be prevented (Bittrich et al., 2011). However, welded field joints are not flow-coated and so corrosion products would occur that have the potential to adsorb mercury.

### 3.3.6.3 Incorporation in sludges

Sludges are categorized as a heterogenous mixture of waste oils and paraffin/wax, resin, asphaltene, inorganic minerals, and other depositions that are not in scales. Hydrocarbon sludges are reported to have higher mercury content compared to the process fluid itself, with total mercury concentrations of 42.5–376.6 mg/kg reported (Camera et al., 2015).

Heavy hydrocarbons such as asphaltene and paraffin/wax are deposited on the pipeline due to surface adsorption or through precipitation and settling due to gravity. Mercury could accumulate in these sludges through adsorption (Wilhelm, 1999b). Hg<sup>0</sup> and organic mercury species may accumulate within the hydrocarbon sludges due to their higher solubility in this matrix and their higher molecular weight. Solid inorganic mercury such as HgS is also found to exist in these sludges.

Asphaltenes are present in crude oil dissolved or as suspended colloids. It has been found that mercury associated with asphaltenes accounts for a significant fraction, 10-50%, of the total mercury concentration within crude oils (Wilhelm et al., 2006). The particular mercury species associated with asphaltene has not been investigated further. Mercury associated with other depositions materials such as sand, paraffin/wax and oxides are usually bound by surface adsorption (Wilhelm, 1999b).

### 3.3.6.4 Other mercury-contaminated products

Sacrificial anode structures are used to prevent pipeline corrosion and contain traces of metals such as mercury, to protect the structure from corrosion during operational life (Oil & Gas UK, 2013). Oil & Gas UK (2013) notes that there is the potential for metal contamination of sediments occurring as the anodes degrade over time, although these impacts are considered to be minor in comparison to other forms of metal pollution from riverine sources. There is no further documented evidence of this.

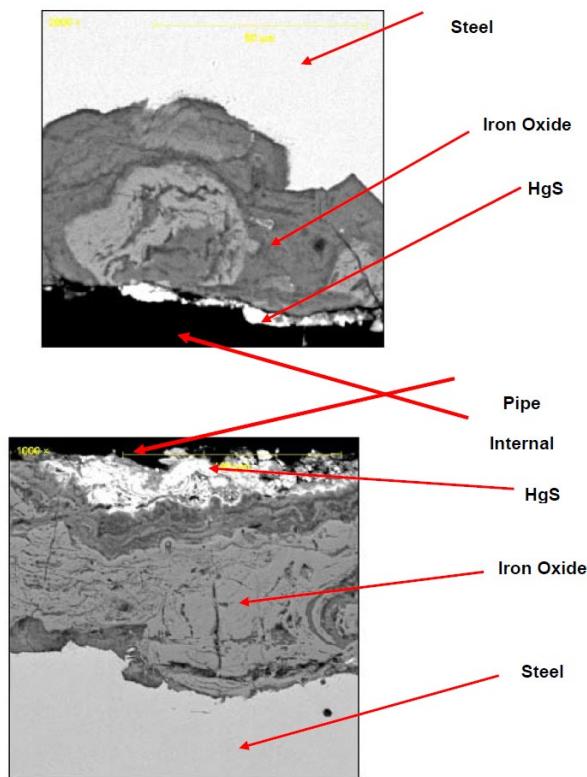


Figure 3.9 Two images of a cross-section of pipe scale contaminated with mercury. Taken from Wilhelm and Nelson (2010).

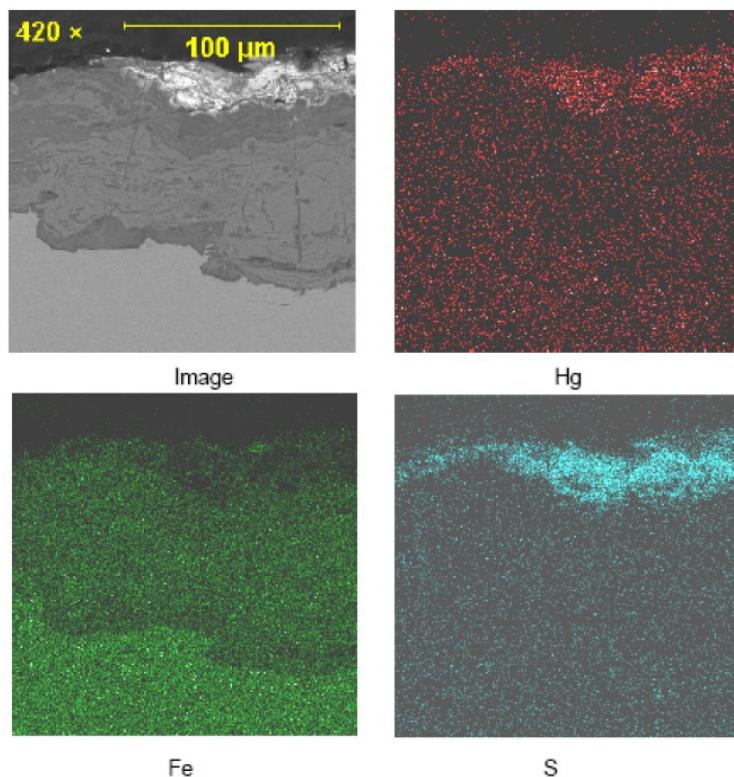


Figure 3.10 Atomic density and distribution of iron, mercury, and sulfur within the contaminated pipe. Taken from Wilhelm and Nelson (2010).

## 3.4 Behaviour of NORM in oil and gas systems

This section reviews the behaviour of mercury in oil and gas systems. An understanding of the contaminated products in oil and gas systems is important to understanding their risk to ecosystem receptors, how different decommissioning scenarios may impact risk, and how an assessment framework may be developed. In doing so, this section partly meets project objectives 3 and 4 and problem statement 2.

### Key findings:

- NORMs partition in oil and gas systems according to their solubilities in different production fluids. Radium follows water streams while radon will partition with gas and condensates, if separated.
- NORMs will form discrete contamination products based on their chemistries.
- Physico-chemical conditions control the formation of scale products including temperature and pressure changes, the mixing of incompatible liquids (i.e. ones high in group-2 metals and ones high in sulfate), and turbulence in pipelines.
- Radionuclide ingrowth contamination products will lead to temporal variability in activity levels and radionuclide composition.

### 3.4.1 Principles of radiation and NORM radionuclides

Some elements naturally emit ionising radiation, defined as the emission of particles or photons by the spontaneous atomic disintegration. These elements are termed radionuclides and by emitting radiation, they change the composition of their atomic nucleus leading to its transformation to a new element. This process, known as radioactive decay, occurs until a stable, non-radioactive isotope is formed. Radioactivity is measured in disintegrations per second, in the standard units of Becquerel (Bq). Different radionuclides have different rates of disintegration which are measured by their specific activity (Bq per mass of the radionuclide) or their half-life ( $t_{1/2}$ , the time it takes for the radioactivity of an element to decrease to half its original value).

Different NORM radionuclides emit different types of radioactivity termed alpha ( $\alpha$ ), beta ( $\beta$ ), and gamma ( $\gamma$ ) radiation. Each emission originates from the nucleus of a disintegrating atom and has a characteristic energy based on type of emission and the radionuclide it came from. However, different types of emissions have different properties and behaviours (Figure 3.11).

- Alpha particles contain two protons and two neutrons, but no electrons. They have a large mass and high energy compared to other types of radiation, with a range of 3-7 MeV (measure of radiation energy in electron volts). They only travel about 2 cm in air and will only penetrate 10  $\mu\text{m}$  or less in solid material.
- Beta particles are equivalent to electrons, but originate from the nucleus of an atom. They have a mass 1/2000 of a proton or neutron and have a wide range of energies – typically between 0.1 and 2 MeV. Depending on their energy, beta particles can travel up to 1 m in the air and can penetrate up to approximately 1 mm in solid material.
- Gamma photons are a type of electromagnetic radiation that accompanies the emission of a beta or alpha particle. They can easily penetrate biological tissue and protective equipment

and may only be shielded by dense materials such as lead (depending on the emission energy, typically 0.01 to 1 MeV).

NORMs are present in all environments at low background activities, depending on the geology of the local area. They naturally cycle and partition in the environment, reflecting different environmental processes and their chemistries. Because of these properties, they are often used to understand large-scale environmental processes such as particulate matter transport in oceans (Cresswell et al., 2020).

Three natural decay series, each supported by a long-lived head of chain radionuclide, may lead to elevated NORM activities in oil and gas basins: the uranium series ( $^{238}\text{U}$ ,  $t_{1/2}=4.47 \times 10^9$  y), thorium series ( $^{232}\text{Th}$ ,  $t_{1/2}=1.4 \times 10^{10}$  y), and the actinium series ( $^{235}\text{U}$ ,  $t_{1/2}=7.04 \times 10^8$  y) (Bourdon et al., 2003). Other types of NORM include the potassium decay series ( $^{40}\text{K}$ ,  $t_{1/2}=1.24 \times 10^9$  y) which is ubiquitously distributed in the environment and cosmogenic NORMs (such as  $^3\text{H}$  and  $^{14}\text{C}$ ) which are generated from the interaction of some atmospheric gases with cosmic rays.

### 3.4.2 Background and forms in oil and gas production systems

The  $^{238}\text{U}$ ,  $^{235}\text{U}$ , and  $^{232}\text{Th}$  natural decay series produce 41 isotopes of 12 elements in their decay chains, each with their own radioactive, chemical, and environmental behaviour (Nelson et al., 2015b). Of these, only a few radionuclides deposit in oil and gas infrastructure at detectable concentrations and may lead to environmental harm (Table 3.8). This is due to their natural abundance in the environment or chemical properties. For example, the actinium series head of chain,  $^{235}\text{U}$ , represents 0.72% of natural uranium which contributes minimal activity to total radioactivity levels (IOGP, 2016b). Therefore, the predominate source of NORM contamination in the oil and gas industry is the  $^{238}\text{U}$  and  $^{232}\text{Th}$  decay series (Figure 3.12).

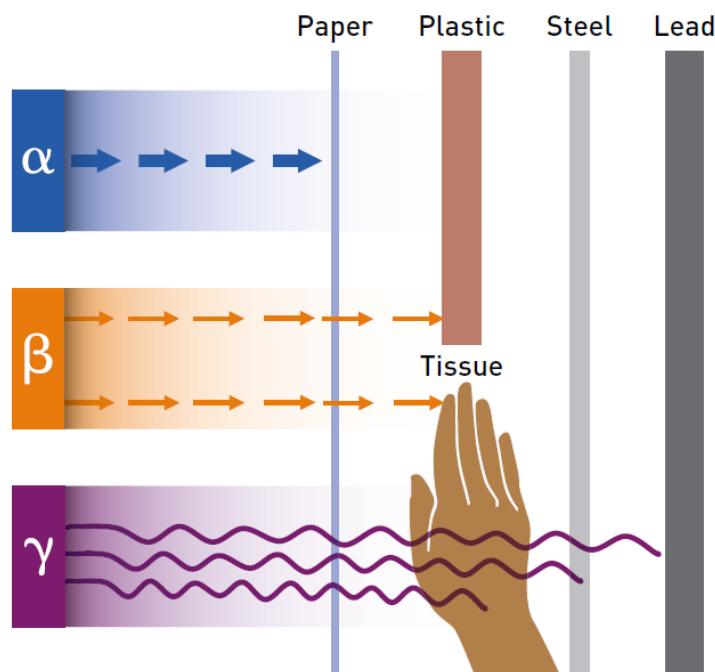


Figure 3.11. Penetrating power of different types of radiation. Taken from IOGP (2016a).

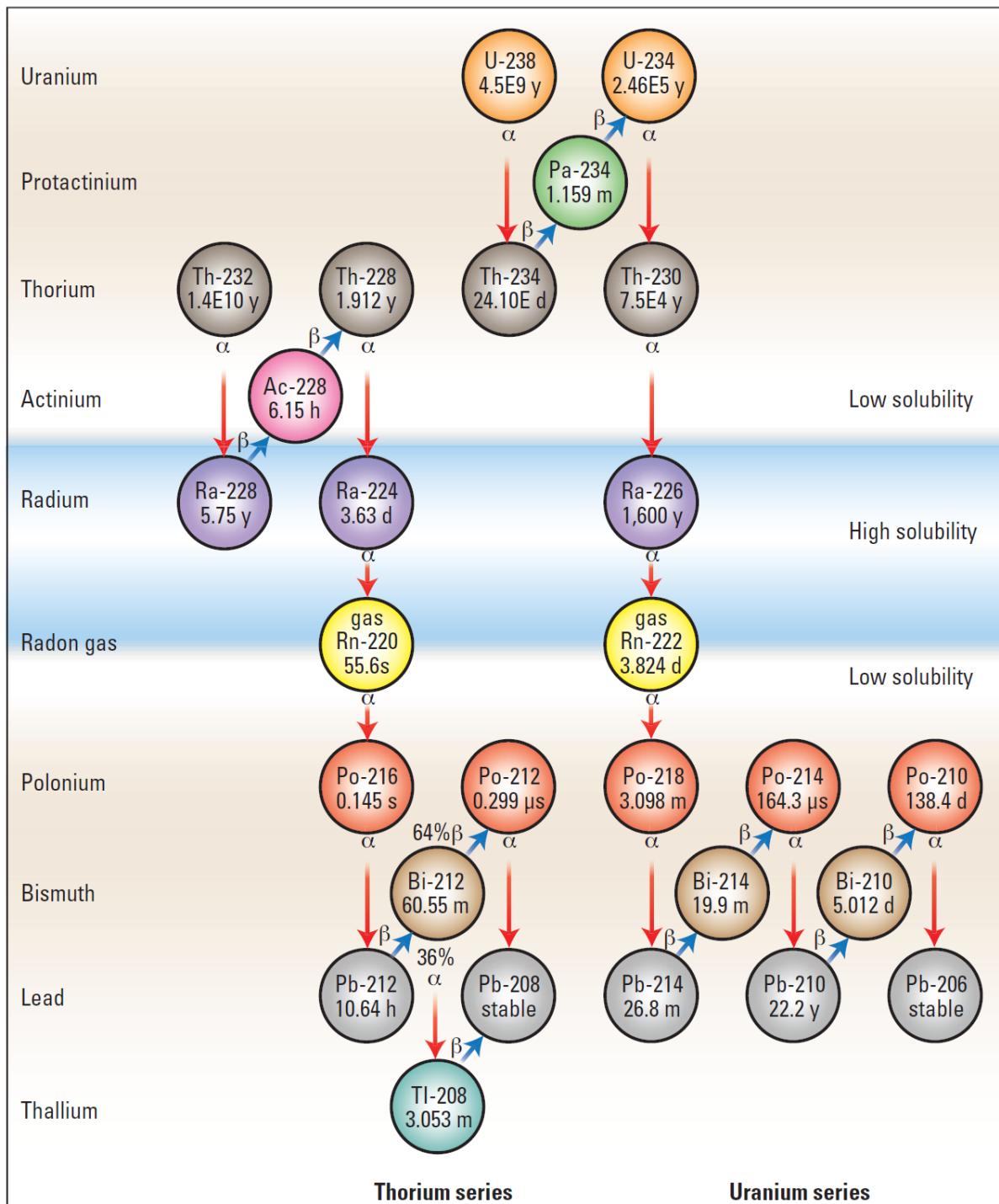


Figure 3.12 Thorium ( $^{232}\text{Th}$ ) and uranium ( $^{238}\text{U}$ ) decay chains showing radionuclide half-lives, decay pathways, and solubility in production fluids. Figure taken from Nelson et al. (2015a).

Uranium and thorium are largely insoluble in conditions typical of reservoirs, so are not often concentrated in production fluids. However, they may be present in particulate matter, such as sands moving through the formation fluids. Furthermore, daughter radionuclides including radium and radon are soluble and can be dissolved into production fluids prior to extraction (Figure 3.13, (Clever and Battino, 1979)). Radium radionuclides ( $^{224}\text{Ra}$   $t_{1/2} = 4$  d,  $^{226}\text{Ra}$   $t_{1/2} = 1600$  y, and  $^{228}\text{Ra}$   $t_{1/2} = 6$  y) are highly soluble in production fluids, as they often exist in their reduced form (e.g. RaS) and will partition to water streams. Radon is an unreactive gas that has a relatively short half-life ( $^{222}\text{Rn}$   $t_{1/2} = 3.8$  d), but a high mobility and solubility in both oil and water products. Changes in pressure and temperature will alter its partitioning in the dissolved or gaseous phase.

The process of oil and gas extraction, separation, and transport, partitions NORMs to different infrastructure. Once extracted, these NORMs are unsupported, meaning they will no longer be produced by their parent nuclide (i.e.  $^{238}\text{U}$ ,  $^{235}\text{U}$ , or  $^{232}\text{Th}$ ). This means smaller decay series headed by longer-lived decay products of the original series' will be established, such as  $^{226}\text{Ra}$ ,  $^{228}\text{Ra}$ , and  $^{210}\text{Pb}$  (Figure 3.13). These are the radionuclides likely to be found in oil and gas infrastructure (Smith, 2010).

### 3.4.3 NORM-contaminated products in oil and gas pipelines

Offshore production of oil and gas generates waste products that may contain NORM. These include scales, depositions, sludges, sands, and produced waters (Ali et al., 2019). Of these, scales, and depositions are known to contaminate pipeline infrastructure intended for decommissioning. The deposition of radionuclides in various offshore oil and gas waste products depend on the type of infrastructure and its location within the production system (i.e. up or downstream of production fluid processing), reflecting the separation of radionuclides based on their chemical properties (Table 3.9).

**Table 3.8 The range of radionuclide activity levels in representative oil and gas products. Taken from Schmidt (2000).**

Radionuclide	Crude oil (Bq/g)	Natural gas (Bq /m <sup>3</sup> )	Condensate (Bq/L)	Produced water (Bq/L)
<b><math>^{238}\text{U}</math> series</b>				
$^{238}\text{U}$	0.0000001-0.01	-	-	0.0003-0.1
$^{226}\text{Ra}$	0.0001-0.04	-	-	0.002-1200
$^{222}\text{Rn}$	0.002-0.02	5-200000	0.01-4200	-
$^{210}\text{Pb}$	-	0.005-0.02	0.3-230	0.05-190
$^{210}\text{Po}$	0-0.01	0.002-0.08	0.3-100	-
<b><math>^{232}\text{Th}</math> series</b>				
$^{232}\text{Th}$	0.00003-0.002	-	-	0.0003-0.001
$^{228}\text{Ra}$	-	-	-	0.3-180
$^{224}\text{Ra}$	-	-	-	0.5-40

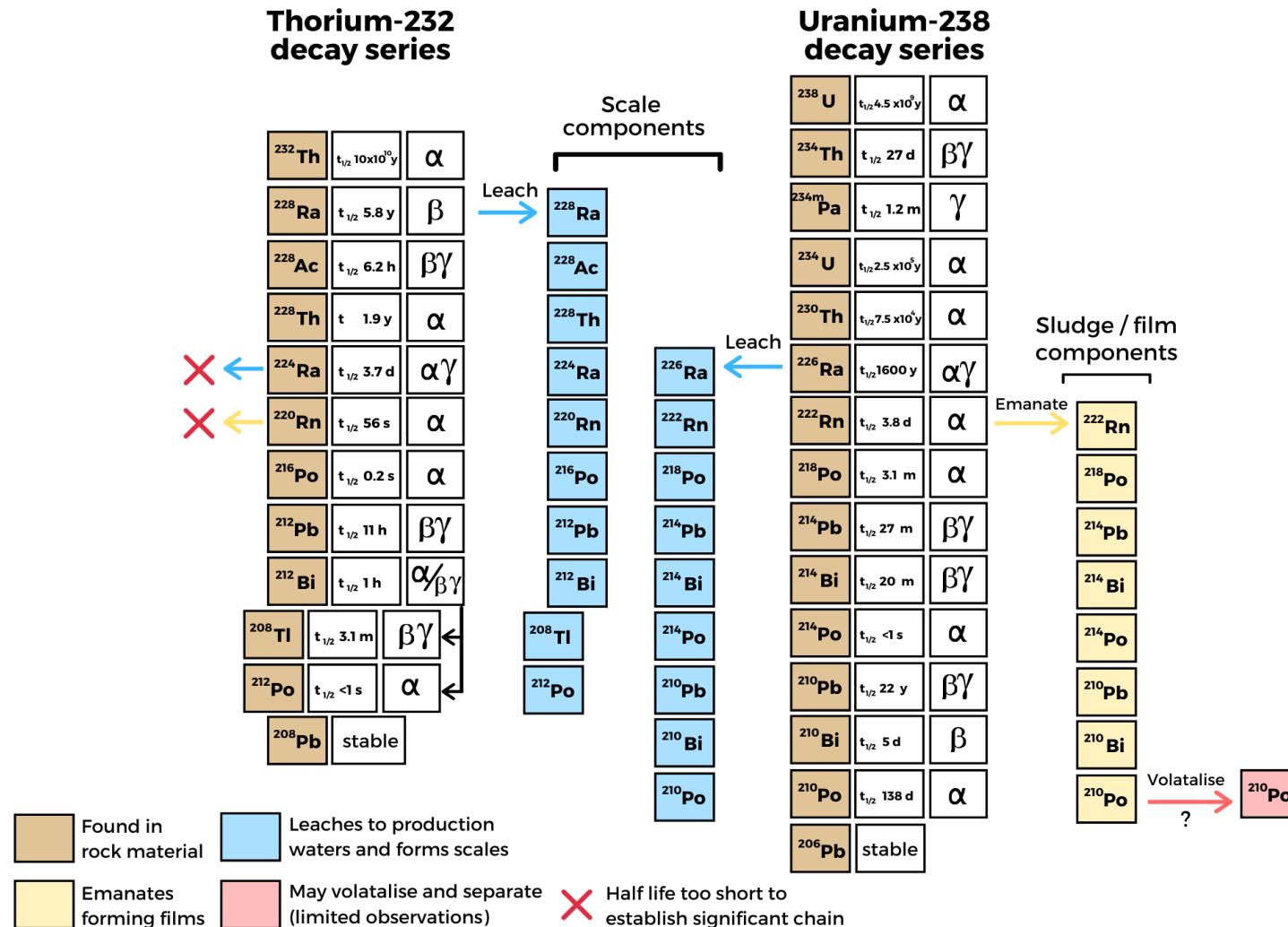


Figure 3.13 The behaviour and partitioning of <sup>232</sup>Th and <sup>238</sup>U decay series radionuclides in oil and gas production. Adapted from IOGP (2016a).

### 3.4.3.1 Scales

Scales are solid deposits of inorganic salts formed on the internal surfaces of oil and gas infrastructure. Scales typically have the highest radioactivity of any NORM-contaminated product in the oil and gas industry (Ali et al., 2019; IOGP, 2016b), with activities of up to 37,000 Bq/g total radioactivity reported (Smith, 2011). The major NORM component of scale is radium, which may be present at up to 15,000 Bq  $^{226}\text{Ra}$ /g and 2,800 Bq  $^{228}\text{Ra}$ /g, which over time will establish an equilibrium with their daughter products (Figure 3.12), by a process of radionuclide ingrowth (Ali et al., 2019).

Inorganic salts are the major matrix component of scales in pipelines. They are typically barium sulfate (barite,  $\text{BaSO}_4$ ) with minor contributions of strontium sulfate (celestite,  $\text{SrSO}_4$ ), calcium sulfate (gypsum,  $\text{CaSO}_4$ ), or calcium carbonate ( $\text{CaCO}_3$ ) and mixtures thereof. Scales may also include bacterial films, corrosion products, and metal sulfide depositions (Yang et al., 2020).

Production fluids from oil and gas basins have high concentrations of the group 2 metal cations calcium, strontium, and barium ( $\text{Ca}^{2+}$ ,  $\text{Sr}^{2+}$ , and  $\text{Ba}^{2+}$ ) and anions including chloride ( $\text{Cl}^-$ ) and carbonates ( $\text{CO}_3^{2-}$ ) (Heaton and Lambley, 1995). The chemical interactions within the production fluids form scale, when the internal pipe conditions favour precipitation (Heaton and Lambley, 1995; Li et al., 2017). A range of environmental and chemical factors may influence the potential for precipitation of scale (IOGP, 2016b), including:

- Changes in temperature and pressure from oil and gas basin to pipelines to other infrastructure
- Variation between laminar and turbulent flow, for example at bends in pipelines
- Pressure variation where pipeline diameter changes
- Nucleation sites within infrastructure, such as from rough surfaces or from seed minerals
- The mixing of incompatible fluids, such as seawater (rich in  $\text{SO}_4^{2-}$ ) and production fluids (rich in group-2 cations such as  $\text{Ba}^{2+}$ ,  $\text{Sr}^{2+}$ , or  $\text{Ca}^{2+}$ )

The mixing of incompatible fluids is the greatest contributor to scale in pipelines. In many oil fields, seawater is injected into reservoirs to maintain pressure or to improve oil mobility. The seawater eventually ‘breaks through’, the point where seawater mixes with production fluids thereby introducing high concentrations of sulfate to the group-2 cation-rich production fluid (Bahadori et al., 2013; Vetter et al., 1982). Similar processes may occur if sulfur-rich aquifer water mixes with production fluids and becomes oxidised (Miao et al., 2012). The formation of  $\text{BaSO}_4$  scale is highly favoured due to its low solubility product (Li et al., 2017). Its precipitation is also promoted by decreasing pressure and temperatures and increasing pH (Monnin, 1999), and the presence of nucleation sites (Zougari, 2010).

Scales become concentrated with radium in pipes because of the chemical similarity between radium and other group 2 metals (Ali et al., 2019; IAEA, 2014a). Of the group 2 metals, radium is most similar in chemical behaviour to barium, strontium, and calcium due to their 2+ oxidation state, similar electronegativities and hydrated atomic radii (3.98 Å, 4.04 Å, 4.12 Å, and 4.12 Å, for Ra, Ba, Sr, and Ca, respectively). Sulfate complexes of radium, barium, and strontium are also isostructural, meaning deposition may occur through either mineral precipitation or ionic exchange (Hedström et al., 2013).

While sulfate scales, such as barite, are regarded as the major inorganic scale component associated with radium deposition (Bahadori et al., 2013), silicate, iron oxides, and carbonate scales may also be important components of the overall scale matrix (Li et al., 2017). Calcium carbonate scales are highly sensitive to the physical-chemistry of the environment with increasing temperatures, decreasing

pressures, and increasing pH promoting the formation of scale (Khormali et al., 2016). In production fluids, calcium concentrations are naturally high and carbon dioxide maintains an equilibrium forming  $\text{HCO}_3^-$  or  $\text{CO}_3^{2-}$  depending on the pH. This means that the formation of calcium carbonate scales is less sensitive to the injection of incompatible waters than sulfate scales. However, calcium carbonate scales incorporate less radium in precipitates than barium sulfate scales (Curti, 1999).

Pipe scales are rarely homogenous, being mixtures of inorganic minerals and corrosion products with their deposition reflecting temporally variable pipe conditions and chemistries of production fluids (Yang et al., 2020). Scales may also be spatially variable due to varying environmental conditions (such as temperature and pressure), or the presence of turbulent sections such as from pipe bends or different interior pipe surfaces such the interlocking sections of flexible risers.

### 3.4.3.2 Lead films

Lead films of  $^{210}\text{Pb}$  are often reported oil and gas systems, but are particularly noted in gas production systems or gas export pipelines (Smith, 2010). Radon is an unreactive gas with the isotopes  $^{222}\text{Rn}$  and  $^{220}\text{Rn}$  produced by the  $^{238}\text{U}$  and  $^{232}\text{Th}$  decay series, respectively (Figure 3.12). The  $^{235}\text{U}$  series does produce the  $^{219}\text{Rn}$  radioisotope; however, the low abundance of  $^{235}\text{U}$  and short half-life of  $^{219}\text{Rn}$  (4 s) means that it is not a significant component of NORM contamination.

Of the radon isotopes,  $^{222}\text{Rn}$  from the  $^{238}\text{U}$  series will occur in a higher proportion due to its 3.8-d half-life compared to the 56-s half-life of the  $^{220}\text{Rn}$  isotope from the  $^{232}\text{Th}$  series. The radon may originate from the oil and gas basin, be produced from dissolved  $^{226}\text{Ra}$  in production fluids, or emanate from radium-contaminated scales in pipelines (Smith, 2010).  $^{222}\text{Rn}$  decays into a range of polonium, lead, and bismuth radioisotopes (Figure 3.13), with  $^{210}\text{Pb}$  having the longest half-life of 22.2 y.

Radionuclides in the decay chain between  $^{222}\text{Rn}$  and  $^{210}\text{Pb}$  are short lived (Figure 3.12) so  $^{210}\text{Pb}$  will accumulate and establish a decay series with  $^{210}\text{Bi}$  and  $^{210}\text{Po}$  before decaying to stable  $^{206}\text{Pb}$ . The produced  $^{210}\text{Pb}$  can be deposited directly from  $^{222}\text{Rn}$  decay or transported via dissolved material in condensate or water phases before precipitating out when pipeline conditions (such as temperature or pressure) changes (Schmidt, 2000).

Lead films are reported under various names but may be identified as sludge, soft or medium-hard scales, or pipeline depositions and are often detected in pigging dust or scrapings (Schmidt, 2000). It is unclear how these deposits accumulate and whether the flow of production fluids prevents their accumulation. Lead films may be easily removed through standard cleaning procedures such as pigging or flushing. However, no data on the effectiveness of these techniques exists.

### 3.4.3.3 Polonium films

$^{210}\text{Po}$  films may form in the absence of  $^{210}\text{Pb}$  in condensate or gas production systems (Schmidt, 2000; Smith, 2010). The mechanism behind the formation of these films is unclear, but is likely to be related to the volatility of polonium and its organic species.

Polonium is known to be volatile (Thakur and Ward, 2020) and may sublime at temperatures  $>25^\circ\text{C}$  (Eichler, 2002). Speciation is known to affect volatility, and organic-polonium species produced by environmental microbes are volatile at room temperatures (Hussain et al., 1995). Organic solvents may also complex with polonium, leading to increased volatility (Mabuchi, 1963). These mechanisms suggest that  $^{210}\text{Po}$  may be highly mobile in some gas production systems.

### 3.4.3.4 Sludges and sands

Sludge is a general term used to describe mixtures of sands, waste oils and waxes, inorganic minerals not in scales, and other depositions (Ali et al., 2019; IOGP, 2016b). A range of NORM products with elevated activity levels have been reported in sludges including  $^{226}\text{Ra}$ ,  $^{228}\text{Ra}$ ,  $^{210}\text{Pb}$ ,  $^{210}\text{Po}$ , and their progeny (Al Nabhani and Khan, 2020; Ali et al., 2019; Pillay et al., 2010). Other radionuclides including  $^{238}\text{U}$ ,  $^{235}\text{U}$ , and  $^{232}\text{Th}$  may be detected, but are typically reported at low activity levels representative of the parent rock material as they are usually transported with particulate matter. Sludges typically have lower activities than scales, but higher than produced waters and sands (Ali et al., 2019).

Sands are transported particulate matter from the oil and gas basin that are typically settled in first-stage separators and discarded. They contain low activities of radionuclides, typically reflecting background radionuclide concentrations in parent rock material (Ali et al., 2019). However, sands produced from a terrestrial Indonesian oil producing field contained some elevated radionuclides activities of  $^{228}\text{Th}$  (7.3 Bq/g),  $^{232}\text{Th}$  (6.5 Bq/g),  $^{238}\text{U}$  (4.8 Bq/g),  $^{226}\text{Ra}$  (4.7 Bq/g), and  $^{228}\text{Ra}$  (10.4 Bq/g) (Wisnubroto, 2005). While above exemption limits (IAEA, 2004a), activity levels were orders of magnitude lower than other NORM-contaminated products.

## 3.4.4 Spatial and temporal variability of NORM-contaminated products

### 3.4.4.1 Temporal variability of NORM

NORM contamination of oil and gas infrastructure is temporally variable because the chemical composition of production fluids may vary over the life of the extraction operation and radionuclide ingrowth will change the composition of NORM-contaminated products.

The formation of NORM-contaminated products in the oil and gas industry is controlled by physical conditions, particularly the presence of incompatible fluids, discussed in Section 3.4.3. A study by Yang et al. (2020) demonstrated that variability in well temperature, pressure, and the ionic composition of production fluid led to heterogeneous scale deposition in pipelines. A similar finding was reported by Nelson et al. (2016) for the NORM content of production fluids from a shale reserve.

Radionuclides decay into their daughter products based on well-established decay chains in a process termed ‘ingrowth’. As each radionuclide in the decay chain has its own half-life, the activity of parent radionuclides (which is a measure of the radionuclide decay rate) will control the activity of daughter radionuclides. This can be modelled using the Bateman Equation (Bateman, 1910) or online tools (for example <https://www.wise-uranium.org/rcc.html>).

If the parent radionuclide is long-lived and in a closed environment, secular equilibrium may be established – where all daughter radionuclides have the same activity as the parent radionuclide because their rate of formation equals the rate of decay of the parent. This occurs after about 6-10 half-lives of the progeny. The long-lived head of chain  $^{238}\text{U}$  and  $^{232}\text{Th}$  radionuclides will be in secular equilibrium within an undisturbed oil and gas basin, but partitioning will occur when NORM-contaminated products are formed in oil and gas extraction processes. This means truncated decay chains starting with the concentrated radionuclides will be established. This is most relevant to radium-contaminated scale.

Scales typically contain precipitated radium and barium from production fluids. Three radium isotopes may exist in this scale  $^{226}\text{Ra}$  ( $t_{1/2} = 1600$  y),  $^{228}\text{Ra}$  ( $t_{1/2} = 5.75$  y), and  $^{224}\text{Ra}$  ( $t_{1/2} = 3.6$  d).  $^{226}\text{Ra}$  is of greatest concern in ecological risk assessments because of its long half-life and decay products

present a persistent environmental risk ( $^{210}\text{Pb}$  and  $^{210}\text{Po}$ ). In a closed system, after approximately 20 d,  $^{226}\text{Ra}$  will be in secular equilibrium with ingrown  $^{222}\text{Rn}$ ,  $^{218}\text{Po}$ ,  $^{214}\text{Pb}$ ,  $^{214}\text{Bi}$ , and  $^{214}\text{Po}$  resulting in a 6-fold increase in total radioactivity. After approximately 120 y,  $^{210}\text{Pb}$  and  $^{210}\text{Po}$  will also be at secular equilibrium with  $^{226}\text{Ra}$  resulting in a total 8.5-fold increase in activity (which is the balance of ingrown progeny and the decay of  $^{226}\text{Ra}$ ). All radionuclides will then decay at a rate proportional to the half-life of  $^{226}\text{Ra}$ , meaning it will take approximately 5,000 y before radioactivity reduces to an equivalent activity level as the  $^{226}\text{Ra}$  when the scale was formed (Figure 3.15). The radiation risk is long lived, with the product's radioactivity reducing to 1% of its starting activity after approximately 16,000 y.

Two other radium isotopes from the  $^{232}\text{Th}$  series may contribute to total scale activity in the short to medium term, but will not pose a long-term risk.  $^{228}\text{Ra}$  with a half-life of 5.75 years will quickly form secular equilibrium, with  $^{228}\text{Ac}$  increasing activity levels by 2x. After approximately 5 years, secular equilibrium will be formed with the remaining progeny (Figure 3.12), leading to a 5x increase in activity level relative to the starting activity. Therein the total activity will decrease at a rate proportional to the half-life of  $^{228}\text{Ra}$ .  $^{224}\text{Ra}$  is a short-lived ( $t_{1/2} = 4$  d) daughter of the  $^{232}\text{Th}$  chain. It is likely to be present in the production fluids that are subject to the chemical reactions leading to scale formation.

Downstream gas pipelines may only contain radon decay products because of the short half-life of radon radioisotopes and its separation and transport with extracted natural gas. Two radon isotopes,  $^{219}\text{Rn}$  (from the  $^{235}\text{U}$  series) and  $^{220}\text{Rn}$  (from the  $^{232}\text{Th}$  series) have half-lives of <1 min and decay into radionuclides with half-lives <11 h meaning they will not contribute a significant long-term hazard.  $^{222}\text{Rn}$  from the  $^{238}\text{U}$  series has a 3.8-day half-life and decays into the longer-lived radioisotopes  $^{210}\text{Pb}$  and  $^{210}\text{Po}$  with half-lives of 22.2 years and 138 days, respectively. These are typically identified in the pigging dust of dry export gas lines (Nelson et al., 2015a).  $^{210}\text{Pb}$  rapidly forms an equilibrium with  $^{210}\text{Bi}$  in 30 days, increasing relative activity 2-fold. After approximately 2 years equilibrium will be formed with  $^{210}\text{Po}$ , increasing relative activity to a maximum 2.8-fold. From this point they will decay at a rate proportional to the  $^{210}\text{Pb}$  half-life taking approximately 35 years to return to its original activity level and 200 years to reduce to 1% of its starting activity.

Radionuclide ingrowth is an important consideration to ecological risk assessments because of the variable half-lives of a number of radionuclides that led to large increases in overall radioactivity occurring over time. Furthermore, reliance on the detection of single-radionuclides (such as from the use of gamma spectrometry) may disregard daughter or parent radionuclides and assumptions of secular equilibrium may not be applicable as ingrowth occurs from the point of deposition which will occur throughout the production lifespan and assumes no emanation or leaching.

#### 3.4.4.2 Spatial variability of NORM

NORM-contamination in oil and gas products are spatially variable across different basins. This is seen in the geographic variability of activity levels in reported NORM-contaminated products (Ali et al., 2019; McKay et al., 2020). This is likely a result of the different basin conditions and inputs that lead to oil and gas production (Nelson et al., 2015b). Variability of production fluid composition within a basin is also observed (Al-Masri, 2006; Gåfvert and Færevik, 2004). Changes to production fluid composition and chemistry may lead to variability in waste products, as has been shown for inorganic scale deposition in pipes (Yang et al., 2020).

The spatial variability of NORM-contaminated products could be affected by:

1. the physical separation of NORM from production fluids in downstream processing
2. spatially variable environmental conditions promoting product formation

3. the half-lives of the radionuclides
4. different radionuclide compositions of different oil and gas basins

Production fluids are a mixture of oil, gas, water, salt, and particulate matter. Down-stream separation of these components will also separate radionuclides based on their physical properties. For example, radon is a gas, radium is water soluble under certain conditions, and other radionuclides are generally adsorbed to particulate matter or mineralised. This means that downstream separation of gas from production fluids will selectively extract radon from the radionuclide mixtures (Figure 3.14).

Scale is an important NORM-bearing matrix because it is the most recalcitrant (i.e. difficult to dislodge from the pipe wall) NORM-contaminated product and generally contains the highest reported activity concentrations of radionuclides compared to other NORM-contaminated products. Factors promoting scale formation may vary spatially, including physical conditions such as temperature, pressure, pH, the presence of bends in pipes, surface coatings, surface roughness or articulations, or changes in production fluid chemistry such as the presence of incompatible liquids or fluid ionic strengths (discussed in Section 3.4.3.1). This therefore leads to spatial variability in the presence and magnitude of NORM in contamination products within a pipeline.

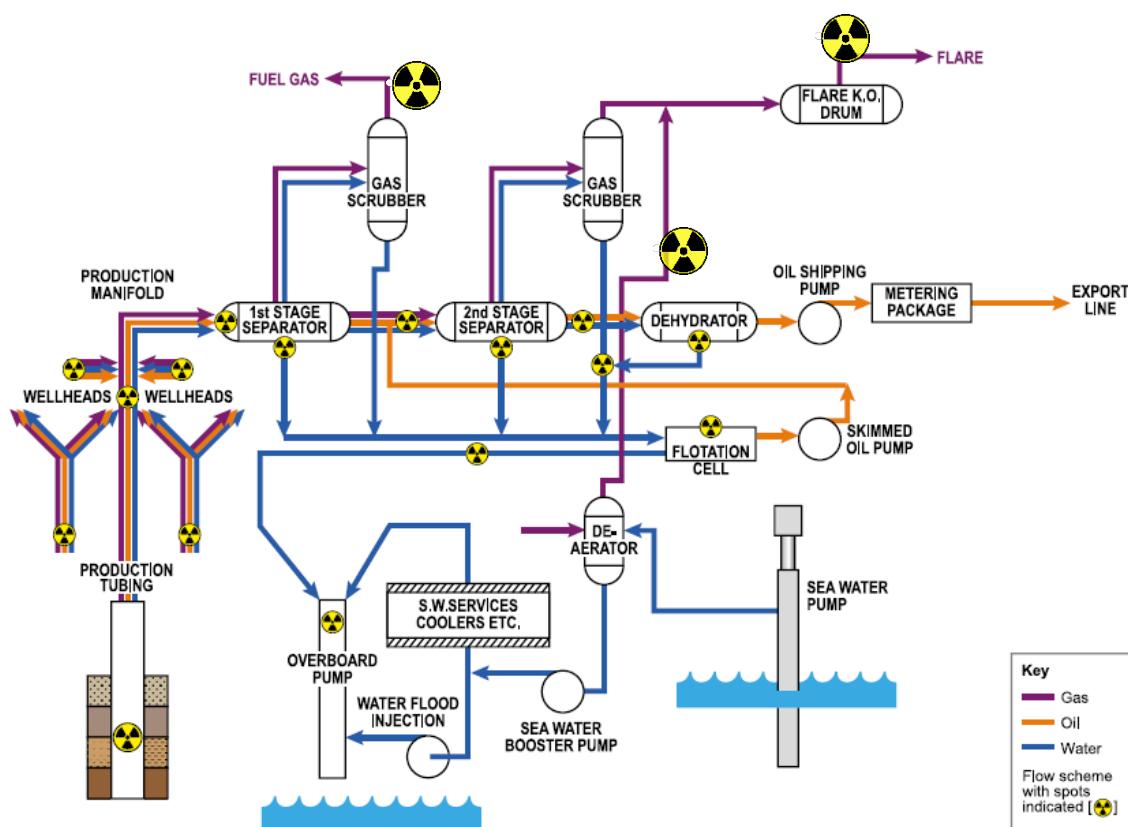


Figure 3.14 NORM extraction and separation in oil and gas infrastructure. Schematic represents an idealised oil and gas processing system. Radiation symbols indicate where NORM-contaminated products may be found. Figure taken from IOGP (2016a).

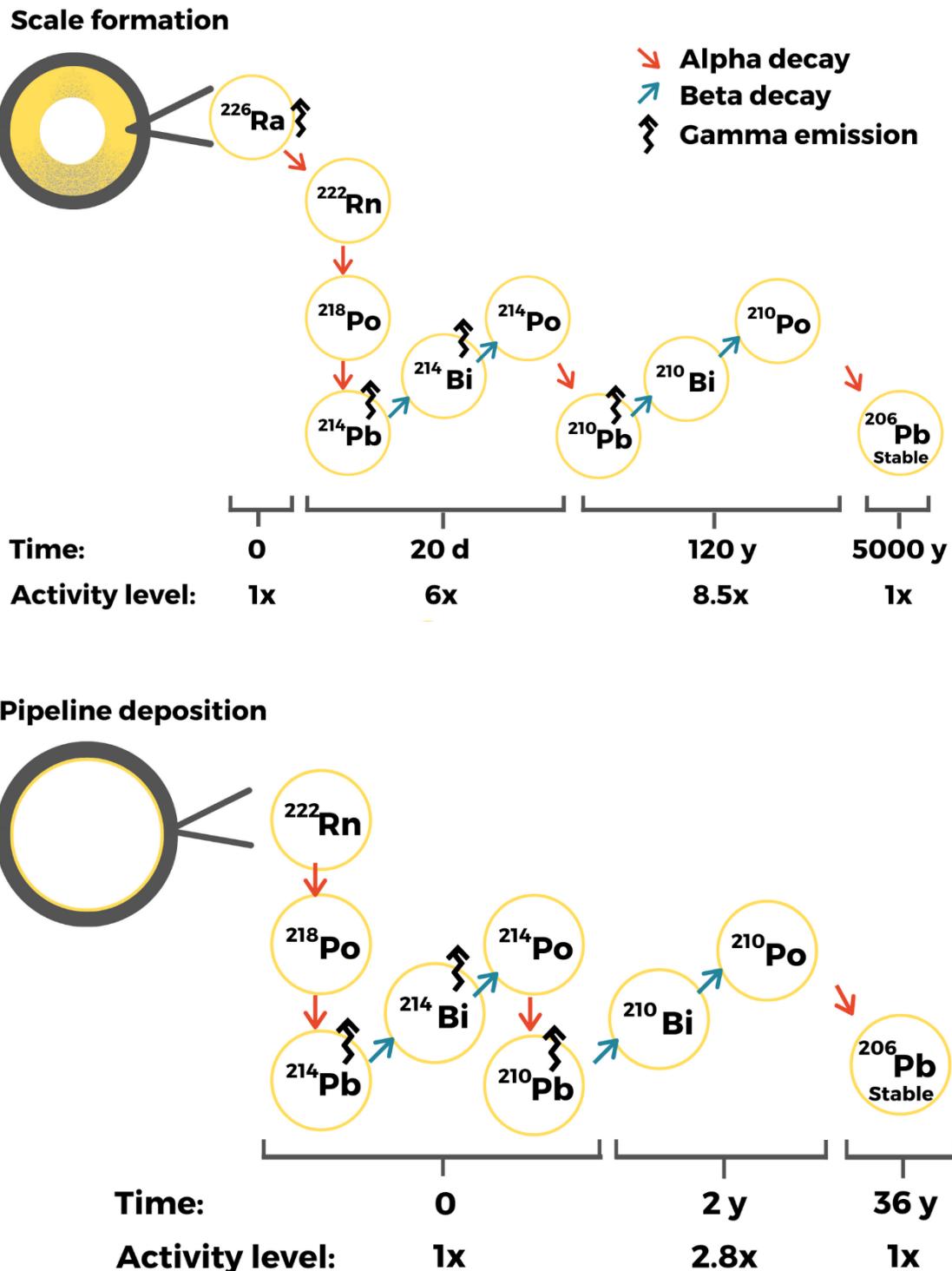


Figure 3.15 Radionuclide ingrowth in  $^{226}\text{Ra}$ -contaminated scale (top) and  $^{222}\text{Rn}$ -contaminated pipeline depositions (bottom) and the resulting change in radionuclide activity level. The time indicates the point at which indicated radionuclides are in equilibrium. For example, radioactivity increases to approximately 8.5x the starting activity level of  $^{226}\text{Ra}$  after 120 years, after which all radionuclides are in secular equilibrium with their progenitors. After this, radioactivity decreases at a rate proportional to the half-life of  $^{226}\text{Ra}$ , 1600 years. At 5000 years the activity level has decayed to that of the original  $^{226}\text{Ra}$  activity.

**Table 3.9 Summary of radionuclides in NORM decay chains and their properties. Table adapted from (Nelson et al., 2015b).**

<b>Element</b>	<b>Speciation in seawater</b>	<b>Average seawater concentration (a)</b>	<b>Elemental residence time (b)</b>	<b>Isotope</b>	<b>Emission α/β/γ (c)</b>	<b>Half-life</b>	<b>Representative seawater activity (d)</b>	<b>S/L/G (e)</b>
Uranium (U)	Dissolved as U(VI) carbonate mineral $\text{UO}_2(\text{CO}_3)_3^{4-}$ , precipitated as U(IV) in anoxic environments	3.3	$5 \times 10^4$ - $4 \times 10^5$ y	238	α	$4.468 \times 10^9$ y	0.044	S
				235	α, γ	$7.04 \times 10^8$ y		S
				234	α	$2.455 \times 10^5$ y	0.048	S
Protactinium (Pa)				234 <sup>m</sup>	γ	1.159 m		S
				234	β	6.7 h		S
Thorium (Th)	Th(IV) minerals are highly insoluble and available Th binds to particulate organic matter or mineral phases. Not redox reactive.	$2 \times 10^{-5}$	0.7 y – 25 d	234	β, γ	27.1 d		S
				232	α	$1.4 \times 10^{10}$ y		S
				230	α	75,400 y	$5 \times 10^{-5}$	S
				228	α	1.9116 y	$1 \times 10^{-4}$	S
Actinium (Ac)				228	β, γ	6.15 h		S/L
Radium (Ra)	Precipitated as $\text{RaSO}_4$ or co-precipitated with $\text{BaSO}_4$ . Possibly dissolved as $\text{Ra}^{2+}$ in anoxic sediments.	$1.3 \times 10^{-7}$	500-1000 y	228	β	5.75 d	0.004	S/L
				226	α, γ	1600 y	0.003	S/L
				224	α, γ	3.6319 d		S/L

Element	Speciation in seawater	Average seawater concentration (a)	Elemental residence time (b)	Isotope	Emission α/β/γ (c)	Half-life	Representative seawater activity (d)	S/L/G (e)
Radon (Rn)	In an elemental state gaseous or dissolved but unreacted.		10 – 22 d	222	α	3.8235 d	7 x10 <sup>-4</sup>	S/L/G
				220	α	55.6 s		S/L
Polonium (Po)	Po(IV) species adsorbed to particulate organic matter or mineral phases where present, otherwise significant proportions (>70%) in the dissolved phase as H <sub>2</sub> PoO <sub>3</sub> .	$1 \times 10^{-9}$	25 d – 0.6 y	218	α	3.098 m		S/L/G
				216	α	0.145 s		S/L
				214	α	164.3x10 <sup>-6</sup> s		S/L/G
				212	α	0.299 x 10 <sup>-6</sup> s		S/L
				210	α	138.376 d	0.004	S/L/G
Bismuth (Bi)	Bi(III) species adsorbed to particulate organic matter or mineral phases. Very limited dissolved component.	$3 \times 10^{-5}$		214	β, γ	19.9 m		S/L/G
				212	β, γ	60.55 m		S/L
				210	β	5.012 d		S/L/G
Lead (Pb)	Precipitated in carbonate minerals or adsorbed to particulate organic matter.	$2.7 \times 10^{-3}$	75 d to 1.5 y	214	β, γ	26.8 m		S/L/G
				212	β, γ	10.64 h		S/L
				210(d)	β, γ	22.2 y	0.005	S/L/G

a Values taken from White (2020) except for polonium which was from Hong et al. (1999). All values reported in µg/L and represent uncontaminated environments.

b Values from McDonald et al. (1996) and given as a range from coastal to open ocean residence times. Values are approximates and will vary with environmental conditions.

c Radiation types include alpha (α), beta (β), and gamma (γ).

d Activity levels in Bq/L taken from Valeur (2011) and references therein

e (S) expected to be present and/or generated in solid wastes (bit cuttings); (L) expected to be present and/or generated in liquid wastes (flow back/produced fluids); (G) expected to be present and/or generated gas streams (flared-waste gases and natural gas).

## 3.5 Measurement techniques

The identification of mercury and NORM in oil and gas infrastructure is an important step to understanding its overall risk. This section reviews the applicable techniques, standards, and guidelines for the identification and measurement of mercury and NORM types and quantities. In doing so, this section partly meets objective 2 and problem statement 3.

### Key findings:

- A wide range of laboratory-based measurement standards and techniques are applicable for mercury and NORM-contaminated products.
- Specific sampling, handling, and preservation techniques are required for different contaminants and different species of contaminants.
- No available *in-situ* techniques can reliably measure mercury or NORM contamination in pipelines. Technologies do exist for in-field measurements of contaminated products but have worse sensitivity and limited abilities to measure speciation compared to laboratory techniques.
- Radiation measurements from handheld devices can detect the presence of some radiation but cannot be used to assess the ecological risk of NORM-contamination because radionuclide composition and activity are not quantified. Additionally, <sup>210</sup>Pb films cannot be detected through pipe walls.
- Portable x-ray fluorescence (pXRF) techniques can measure mercury on pipelines in-field but are prone to underestimation because of shallow x-ray penetration of pipeline material and heterogenous contaminated products and surfaces.

### 3.5.1 Background to measurement techniques

The purpose of measurements is to define analyte concentration, composition, and speciation. To do this, an analysed sample is related back to the contaminated product from which it was taken.

Assurances that the sample taken reflects the contaminated product (or infrastructure the sample was intended to characterise) are critical to understanding contaminant risk. This assurance may be impacted by low sample numbers, sample heterogeneity and representativeness, transformations of the contaminated product prior to analysis during sampling, storage, and transportation steps, interferences relating to the sample matrix, and limitations of different analytical techniques. Many of these challenges are resolved through proper techniques which may be described in standards or by internal procedures. Accreditation of a laboratory by the National Association of Testing Authorities (NATA) in Australia indicates compliance with good laboratory practice and assurance of technical competence.

Common assurance measures include developing appropriate sampling strategies for the nature of the contaminated product being investigated, using certified standards to calibrate instruments, using certified reference materials to understand matrix interferences for the analytical method, adopting appropriate handling and preservation techniques for samples, and reporting the methods and quality control practices employed in the sample analyses.

Hundreds of standards exist describing sampling methods, approaches to calibration, sample preservation techniques, and analytical methods. These are produced and maintained by various international organisations. Relevant standard-producing organisations for the detection of mercury and radionuclides include the International Organisation for Standardisation (ISO), ASTM International, the American Water Works Association, the IAEA, and the United States Environmental Protection Authority (US EPA).

### 3.5.2 Mercury

This section reviews the measurement techniques available to quantify and speciate mercury in the various contaminated products identified in Section 3.3.6. The measurement and speciation of these products is also a function of the sampling method which can range from sampling scale from inside the pipe by pigging, extracting a removable coupon or pipeline spool, or cutting a sample of pipework. The testing methods can be classed as destructive or non-destructive, the former typically acid digestion and the latter x-ray fluorescence (XRF) techniques (Chanvanichskul et al., 2017; IPIECA, 2014).

Sampling techniques will be discussed in detail followed by in-field measurements and finally laboratory measurement techniques for the quantification of total mercury and its various species.

#### 3.5.2.1 Sampling techniques

Sampling techniques for scale deposits on the surface of pipelines are usually retrieved via swabbing, collection of displaced solid products from routine pigging campaigns or by placement of retrievable coupons connected to the pipelines within the facilities.

Collected pipe scale and metal coupons are commonly stored in sample containers or plastic bags to be transported to a laboratory for analysis. Due to the reactive nature of mercury and its tendency for species interconversion leading to sample loss, minute details such as selection of container materials itself is a challenge.

Mercury samples should never be stored in polyethylene containers (Feldman, 1974; Parker and Bloom, 2005; Yu and Yan, 2003). Mercury both in the sample and the atmosphere can permeate and interact with polyethylene functional groups in container walls and be adsorbed from solution. Other poorly performing containers include metal, polyvinyl chloride, and polypropylene (Bloom, 2000; Krivan and Haas, 1988; Leermakers et al., 1991; Rosain and Wai, 1973).

The use of Teflon has been demonstrated by several authors to have minimal reaction with mercury and is able to contain the sample for long period of time of a few months (Parker and Bloom, 2005; Yu and Yan, 2003). Glass containers have also been reported to perform well in the storage and handling of different mercury species (Feldman, 1974; Parker and Bloom, 2005).

Temperature is an important factor for sample preservation. Significant mercury loss may occur for samples stored at room temperature compared to refrigerated condition (Lansens et al., 1990). It is recommended that samples, especially non-aqueous be analysed immediately, otherwise they have to be stored in refrigeration (US EPA, 1998a).

Retrievable spool pieces are often large in size and need to be cut into smaller size coupons prior to conducting sample characterization and mercury measurement. Mercury contained within the corrosion product layer is readily vaporized into the atmosphere when hot works such as welding and torch cutting are carried out during inspection and maintenance with concentrations  $>50 \text{ mg Hg/m}^3$

have been reported (Wilhelm, 1999a). Cold-cutting technique is usually the common practise to machine coupon specimens or to cut samples of pipe to minimise loss of mercury through evaporation.

A recent pigging development by Chanvanichskul et al. (2017) proposed a design of specialized pigs that targets *in-situ* decontamination and sampling of mercury from pipelines. As shown in Figure 3.16 the design includes one chemical and two mechanical pigs. The first pig is designed to be used with chemicals as a pig train, while the second removes residual mercury mechanically. The specialized pigs allow for simultaneous sampling of the deposited pipe scale containing mercury during the decontamination process as samples can be stored inside the pig body (Figure 3.17) and collected for subsequent analysis in the lab for the composition of contaminants. Multiple samples could be obtained during multiple runs of the pigs to verify the amount of mercury within the pipeline prior to decommissioning.

### 3.5.2.2 In-field measurement techniques

No techniques currently exist for the robust detection of mercury in pipelines. In-field, portable, measurements are typically limited to pXRF and requires intrusive access contaminated surfaces. The use of a pXRF as a screening technique for determination of mercury surface contamination is growing in application as it is a quick method that provides real-time results and is a non-destructive analysis in comparison to conventional laboratory methods.

There are two XRF techniques that are commonly utilized, namely wavelength dispersive XRF and energy dispersive XRF (EDXRF) with the latter being more suitable for field applications due to its ease of use and portability (Kalmicky and Singhvi, 2001). XRF analysis utilises the use of an excitation source via a sealed radioisotope sources or x-ray tubes to irradiate a given sample to emit a characteristic fluorescent x-ray signals of a given atom for detection. Measurements are given in 'counts' per unit time which are compared to a known standard of that element and matrix to determine concentration. The signal amplitudes are used to determine the types of elements present.

A standard procedure for analysing various types of elements including mercury-containing solid samples using pXRF follows the SW-846 Test Method 6200 (US EPA, 2007a). pXRF detects mercury in a given detection window. This enables operators to identify mercury hotspots based on the formation of the contaminated product (Figure 3.18).

Although several studies have demonstrated the reliability of pXRF to accurately identify most metals, many reported poor quantitative accuracy, which leads to underestimation of mercury (Brent et al., 2017). It is therefore a common practise to send samples to a laboratory for confirmatory analysis utilising analysis instruments that are more selective with a lower limit of detection. There are several limitations of using pXRF in the determination of mercury in solid samples which are summarised here:

1. Speciation; pXRF is only able to measure total mercury and cannot be used to distinguish between various species of mercury which have been identified within the oil and gas production streams. For example, no difference in the sensitivity of pXRF has been reported for samples containing  $HgCl_2$  and  $Hg(C_6H_5)_2$  (Kibogy, 2010).
2. Profile of the solid surface to be measured should be as flat and smooth as possible. This poses a challenge as internal pipeline surface is often rough and pitted from corrosion and corrosion product topography.

3. Depth of x-ray penetration; pXRF is limited to measuring mercury at superficial depth. Sensitivity of the measurement is dependent on the density of the subject material (such as corrosion products or mercury sulfides scales) and thus will vary with both the depth and type of scale that may be present on the pipeline.
4. Limits of detection; pXRF has a much higher limit of detection in comparison to other conventional laboratory techniques for the analysis of mercury. Limits of detection of 7.4 mg Hg/kg (Brent et al., 2017) have been reported and the reliability of measurements is reported to be low at concentrations below 20 mg Hg/kg (Eckley et al., 2020).

An alternative *in-situ* method is SWASV (Square Wave Anodic Stripping Voltammetry) which can quantify total Hg measurement that is comparable to analysis using inductively coupled plasma-optical emission spectrometry (ICP-OES) (Silakorn et al., 2019). This method seems promising to be used as a screening option to analyse Hg contaminated samples as the device is portable in comparison to those used in traditional labs; however, it still requires an acid digestion step as the analysis requires liquid samples.

### 3.5.2.3 Laboratory measurement techniques

#### Total mercury in corrosion products and metal coupons

The total mercury content of solid samples is commonly determined using cold vapour atomic absorption spectrometry (CV-AAS) (US EPA, 2007b), cold vapour atomic fluorescence spectrometry (CV-AFS) (US EPA, 2007c), inductively coupled plasma-mass spectrometry (ICP-MS) (US EPA, 2014a), or ICP-OES (US EPA, 2018). Cold vapour techniques utilise the use of reductants such as  $\text{SnCl}_2$  (Clevenger et al., 1997) or  $\text{NaBH}_4$  (Schickling and Broekaert, 1995) to reduce the divalent mercury species in the solution into the more volatile  $\text{Hg}^0$ . The gaseous  $\text{Hg}^0$  is then introduced into either the AAS or AFS cell for detection.

Analysis using ICP-MS can be separated into four parts, namely sample introduction, the inductively coupled plasma torch, interface and finally mass spectrometer for detection (Bazilio and Weinrich, 2012). At the sample introduction phase, often samples for ICP-MS are in liquid phase; however, they must be converted into either gas or aerosol phase by means of a nebulizer and a spray chamber to be introduced into the plasma torch. The aerosol created is then injected into the torch where it will be atomized and ionized by the plasma which is generated when a continuous stream of argon gas is heated 6000 °C. The resulting ions from the plasma then enters the interface where it comprises of a sampler and skimmer cone. The interface allow rapid cooling and de-pressureising of the hot plasma gas to the appropriate condition before the gas is sent to the final stage for detection. Following the interface, the ions are focussed into a beam by a single lens for transmission into the MS. The MS is commonly in the form of a quadrupole (Skoog et al., 2007), consisting of 4 parallel rods that act as electrodes. The quadrupole separates the ions based on their mass to charge ratio prior to entering the detector where a measurable pulse is detected.

A summary of the standard procedures by U.S. EPA for analysis of mercury in solid are outlined in Table 3.10. Regardless of the methods employed, they require a sample pre-treatment step to convert mercury contained within the samples to its aqueous ionic form, mainly by performing an acid digestion step using *aqua regia* or a mixture of  $\text{H}_2\text{SO}_4/\text{HNO}_3/\text{KMnO}_4$  (US EPA, 1998b) at temperature >95 °C. As shown in Figure 3.19, ICP-OES have been employed to measure the occurrence of mercury within the bulk steel substrate by acid digestion of sliced metal layers (no mercury was detected in the bulk steel substrate by Chanvanichskul et al. (2017)).

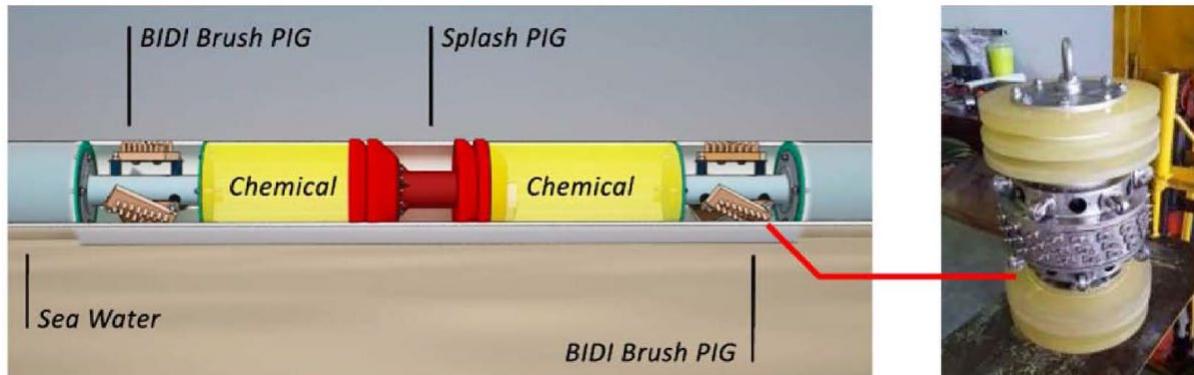


Figure 3.16 Pig train developed by PTTEP for simultaneous sampling and *in-situ* decontamination of pipeline (Chanvanichskul et al., 2017).



Figure 3.17 Sampling tool within the specialized pig design (Left - sample holder, Right - sample collected from contaminated pipe) (Silakorn et al., 2018).



Figure 3.18 pXRF measurement of a pipe section. Left: Measurement using sample grid. Right: High discrepancies of mercury concentration deposited ( $\mu\text{g Hg}/\text{cm}^2$  of pipe). Taken from Kibogy (2010).

The addition of a pre-concentration amalgamation step using a gold-coated silica trap which is heated to release the mercury atoms for detection at the spectrophotometer cell, further reduces the detection limit of the analysis technique. Alternatively, thermal decomposition of solid samples could be performed whereby the samples are heated in a furnace at  $\geq 750$  °C to release all the bound mercury before the pre-concentration step (US EPA, 1998a).

In comparison to the various methods shown in Table 3.11, AFS shows the highest sensitivity due to the low background of the fluorescence signals, however, preference using ICP-MS/ICP-OES are gaining popularity as they allow for the simultaneous analysis of multiple elements. On the other hand, it is known that ICP-MS/ICP-OES poses a memory effect issue (López-Antón et al., 2012) as mercury could be retained within the instrument as many components come into contact with the sample. Moreover, mercury is known to have a very high first ionization energy 10.44 eV (Linstrom and Mallard, 2005) and only around 4% of the samples could be ionized in the argon plasma resulting in suppressed signals when high concentrations are present in the sample matrix. Therefore, to some extent these factors may limit its uses.

#### **Speciation of Mercury in Corrosion Products and Metal Coupons**

Several analytical techniques have been employed in the attempt to identify the types of mercury species present within the corrosion products solids and scales on pipe surfaces such as x-ray diffraction (XRD), scanning electron microscopy-energy dispersive x-ray (SEM-EDX), x-ray absorption fine structure (XAES) and resolution flow discharge mass spectrometry (GDMS), with the most commonly used techniques combine XRD and SEM-EDX. The combination of these techniques provides information of element mapping along with their forms present (Chanvanichskul et al., 2017). Although SEM-EDX could only map out elements and not their chemical compound, one can elucidate elemental associations from the particle distribution in a given sample. In addition, X-ray photoelectron spectrometer (XPS) has also been utilized in combination of SEM-EDX to measure the distribution and depth of mercury penetration in pipelines (Chaiyasit et al., 2009). One major drawback of using these spectroscopic techniques is the detection limit that they offer. Several have reported for XRD analysis, the compound of interest has to make up at least 2% of the sample. Additionally, concentration of  $>3\%$  (Rompalski et al., 2019) and 1-100 mg Hg/kg (Chiarantini et al., 2017; Kim et al., 2004) within samples are needed for analysis using SEM-EDX and XAES respectively.

It is to be noted that these techniques are non-destructive and therefore only able to provide a qualitative analysis of the mercury species within a given specimen. All in all, these techniques provide valuable information regarding specific contaminants present to support remediation and decommissioning decisions.

Table 3.10 Standard sample preparation procedures of solid samples containing mercury.

Method	Hg Measured	Sample Preparation	Analysis Technique	Reference
7473	Total Hg and Hg species	None, or utilised in conjunction with other methods that first separate the individual mercury species	Thermal decomposition in a furnace ( $\geq 750^{\circ}\text{C}$ ), amalgamation and AAS	(US EPA, 1998a)
7471B	Total Hg	Acid digestion using <i>aqua regia</i> at $95^{\circ}\text{C}$ or $\text{H}_2\text{SO}_4/\text{HNO}_3/\text{KMnO}_4$	Cold Vapour Atomic Absorption (CV-AAS)	(US EPA, 2007b)
7474	Total Hg	Acid digestion using <i>aqua regia</i> at $190^{\circ}\text{C}$ (microwave assisted digestion)	CV-AFS	(US EPA, 2007c)
3200	Extractable Hg	2% (v/v) HCl and 10% (v/v) ethanol at $60\pm 2^{\circ}\text{C}$ (ultrasound assisted extraction)  4.0 M $\text{HNO}_3$ at $100^{\circ}\text{C}$ (microwave assisted extraction)	Not specified	(US EPA, 2014b)
	Non-extractable Hg (semi-mobile)	Residual solid from extractable Hg step: Water at $60\pm 2^{\circ}\text{C}$ to extract $\text{Cl}^-$ and discard; 1:2 $\text{HNO}_3$ at $95\pm 2^{\circ}\text{C}$		
	Non-extractable Hg (non-mobile)	Residual solid semi-mobile step: <i>aqua regia</i> at $95\pm 2^{\circ}\text{C}$		

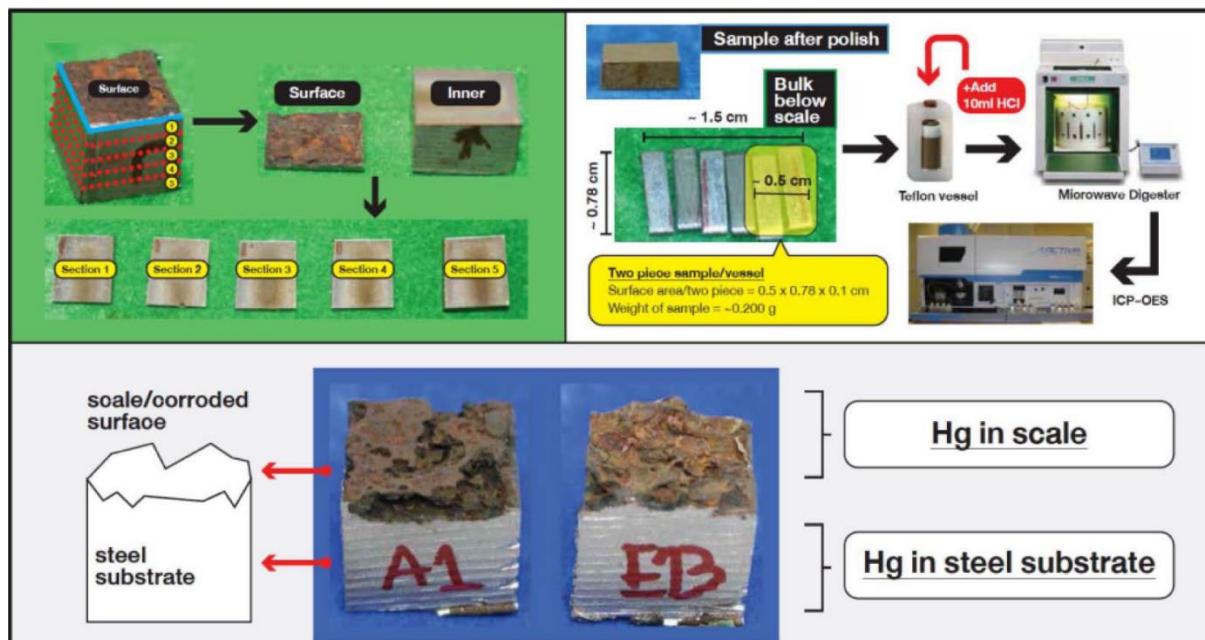


Figure 3.19 Sample preparation for analysis of mercury penetration depth in steel coupons using ICP-OES (Chanvanichskul et al., 2017). Note that no mercury was detected in the steel substrate.

An alternative method for quantification and speciation of mercury has also been developed by US EPA method 3200 (US EPA, 2014b) to allow for semi-qualitative analysis using sequential selective extractions (SSEs). As summarised in Table 3.10 and Table 3.12, mercury species could be separated into three main categories, namely extractable, non-extractable semi-mobile, and non-extractable non-mobile phase. Extractable phase constitutes extractable organic and ionic mercury species while both the non-extractable phases are mainly made up of ionic mercury with the non-mobile phase containing the more stable components. Extraction parameters use weak acids at lower temperatures to maintain the integrity of the organic and ionic species that might be present within the extractable phase. The presence of chloride ions could result in simultaneous extraction of the semi-mobile and non-mobile phase, hence should be discarded before the subsequent acid digestion step.

Analysis technique has not been specified, however, techniques involving chromatography such as gas chromatography-mass spectrometry (Nevado et al., 2011), high-performance liquid chromatography coupled with CV-AAS (Schickling and Broekaert, 1995) or ICP-MS (Bouyssiere et al., 2002) detection could be employed for sub-speciation of the extracted mercury species within the different fractions.

**Table 3.11 Comparison of the detection limit of techniques for total mercury analysis (Sainal et al., 2007).**

Methods	Detection Limit (ng)
X-ray fluorescence	10
Neutron activation	2
Gold film	0.5
Differential pulse voltammetry	0.04
Cold vapour atomic absorption spectrometry (CV-AAS)	0.01
Cold vapour atomic fluorescence spectrometry (CV-AFS)	0.0001
Inductively coupled plasma-mass spectrometry (ICP-MS)	0.001
Inductively coupled plasma-atomic emission spectrometry (ICP-OES)	50 ng/ml

**Table 3.12 Mercury species at different extractable and non-extractable fractions (US EPA, 2014b).**

	Operationally-defined total mercury fractions	Individual mercury species
Extractable mercury	Extractable organic mercury	CH <sub>3</sub> HgCl, CH <sub>3</sub> CH <sub>2</sub> HgCl
	Extractable ionic mercury	HgCl <sub>2</sub> , Hg(OH) <sub>2</sub> , Hg(NO <sub>2</sub> ) <sub>2</sub> , HgSO <sub>4</sub> , HgO, Hg <sup>2+</sup> complexes*
Non-extractable mercury	Semi-mobile mercury	Hg <sup>0</sup> , Hg <sup>0</sup> -M#, Hg <sup>2+</sup> complexes, Hg <sub>2</sub> Cl <sub>2</sub> (minor)
	Non-mobile mercury	Hg <sub>2</sub> Cl <sub>2</sub> (major), HgS, HgSe

\* Certain inorganic mercury complexes may be present in both the organic and ionic extractable fractions.

\* Amalgamated mercury species

### 3.5.3 NORM

The measurement of NORM relies on the detection of the radioactive emissions from a radioactive material (such as alpha or beta particles or gamma rays) or the detection of the radionuclide atoms themselves. The choice of method including any sample preparation is dependent on specific needs, detection limits, and the availability of samples.

A range of standards exist for laboratory methods for the measurement of NORM radionuclides. These include for sampling environmental samples containing radionuclides, radiochemical procedures for sample preparation (such as sample digestion, chemical separation, and radionuclide pre-concentration steps) as well as the analytical methods for detection. Some relevant standards are reported alongside the relevant detection methods below.

To analyse samples, different preparation steps are required because certain instruments require samples in a specific phase (such as solid, liquid, or gas), interferences such as the environmental media need to be removed, target analytes need to be concentrated to achieve detection limits, or samples need to be homogenised to reduce intra-sample variability (IAEA, 2019).

Sample preparation steps such as digestion, ashing, radiochemical separation by ion exchange resins and selective precipitation, and electro depositions are required depending on the instrument being used, the target radionuclide, and the detection method chosen. These preparations are generally well detailed in standardised methodologies. For example, the IAEA (2010) publication ‘Analytical Methodology for the Determination of Radium Isotopes in Environmental Samples’ details the analytical methods for the detection of radium in environmental samples.

#### 3.5.3.1 Detection methods

Four types of detection methods are commonly used to detect NORM, gas-filled detectors, scintillation detectors, solid-state detectors, and mass spectrometers (Schroeyers, 2017). However, a wide range of other techniques, sample preparation steps, and sampling techniques may also be used.

Gas-filled detectors work by detecting the electric charge that is generated when gas molecules become ionized due to radiation. These detections are recorded as an audible click and counted in a given time period (with a count per second, CPS, output). The Geiger-Muller tube (or G-M tube or Geiger counter) is a commonly used type of gas-filled detector for *in-situ* radiation detection. However, these detectors do not discriminate against radiation type, source, or energy.

Scintillation detectors use a scintillator material coupled to a photomultiplier tube to detect and measure radiation. Scintillating materials absorb ionizing radiation and re-emit the energy in the form of light, which is then passed to a photomultiplier tube for amplification and highly sensitive detection. For bulk radiation detection, organic phosphor is commonly used as a scintillation material. However, different scintillation material is able to detect specific types of radiation, for example, caesium iodide (CsI) and zinc sulfide (ZnS) can be used for alpha particles and sodium iodide (NaI) can be used for the detection of gamma radiation. Scintillator detectors can discriminate between radiation from individual radionuclides or measure bulk activity.

Liquid scintillation spectrometry is a type of scintillation technique which uses a scintillating liquid rather than a scintillating crystal. These are used in laboratory-based instruments where the radioactive substance is added to the scintillating liquid allowing for sensitive detection without

geometry issues. ASTM International (2017) standard D7283-17 details a standard test method for alpha and beta testing in water by liquid scintillation.

Solid-state detectors use semiconductor materials such as germanium in the place of scintillator materials. These have become the most common detector for gamma spectrometry. Gamma radiation interacting with a solid-state material generates an electronic signal that can be measured. Scintillators and solid-state detectors can measure the number of emissions as well as the energy of the radiation emissions (Section 3.4). This means they can be used for radioisotopic detection and quantification. ISO (2007) standard 10703 specifies a method for the simultaneous determination of various gamma-emitting radionuclides in water samples using a solid-state germanium detector.

NORMs may also be measured by ICP-MS. Unlike the other methods, ICP-MS detects the atomic nuclei rather than radioactive emissions. Samples are digested (often in strong acids under pressure) then aspirated into a plasma leading to their ionization into charged particles. This stream of particles is directed into a mass spectrometer where they are separated by their mass and charge using magnetic or electrostatic techniques. The filtered particles are then detected by an ion detector. ICP-MS has very low limits of detection, with fg/g levels reported for  $^{226}\text{Ra}$  in environmental samples (IAEA, 2010).

Where a method is used to specifically detect alpha, beta, or gamma emissions, they are referred to as alpha, beta, or gamma spectrometry. Only alpha and gamma spectrometry are undertaken for NORMs as most beta-emitting radionuclides also emit gamma radiation (one exception,  $^{90}\text{Sr}$  is a product of nuclear fission and therefore not relevant to NORMs). A list of common laboratory techniques for different radionuclides are given in Table 3.13.

A number of measurement interferences and uncertainties limits the utility of *in-situ* radionuclide detectors. That is, *in-situ* detectors typically measure total radioactivity, or a single-radionuclide's activity as counts per second or Bq/cm<sup>2</sup>. These measurements cannot be converted to specific activity because of unknown measurement geometries, sample densities, interferences. Ecological risk assessments typically require the specific activity of a contaminated material (i.e. Bq/g or Bq/L) and an understanding of the material's volume or mass.

### 3.5.3.2 Detection of radionuclides by ingrowth

Direct detection of all radionuclides is not possible due to the different types of radiation emissions, their different energies (Section 3.4), and the limitations of different detectors. Therefore, detection is often achieved via the ingrowth of a more detectable radioisotope. This process requires the isolation of the head of chain radionuclide to remove existing progeny. Once isolated, a new equilibrium can be formed and the quantity of the progeny will be directly relatable to the parent radionuclide.

For example,  $^{226}\text{Ra}$  is an alpha and gamma emitter so where alpha spectrometry is unavailable, it can be measured by gamma spectrometry (Table 3.13). However,  $^{226}\text{Ra}$  has a gamma peak at 186 keV with only a 3.6% emission probability (Figure 3.20). Its progeny,  $^{214}\text{Bi}$ , has an energy peak at 609 keV which is not subject to neighbouring interferences and has a far higher emission probability.

Therefore, quantification of  $^{226}\text{Ra}$  is often based on  $^{214}\text{Bi}$  signal. A sample containing  $^{226}\text{Ra}$  will be crushed and dried to volatilise metals including  $^{214}\text{Bi}$ . This also allows the deposition of a known mass of  $^{226}\text{Ra}$  onto a standard geometry that can be directly compared to a known standard. After approximately 4-6 weeks, sufficient ingrowth of  $^{214}\text{Bi}$  based on sufficient decay of  $^{226}\text{Ra}$  through ten half-lives of  $^{222}\text{Rn}$  (3.82 days half-life, Figure 3.12) will allow accurate detection.

**Table 3.13** NORM radioisotopes, their emissions, half-lives, and common detection methods. Methods include alpha spectrometry (AS), high purity germanium gamma-spectrometry (HPGe), and radon emanation (Emanation), and Inductively Coupled Plasma Mass Spectrometry (MS). Radionuclides in brackets next to HPGe indicate detection of the parent by the listed ingrown progeny. N/A indicate no available method. Table adapted from Nelson et al. (2015b).

Element	Isotope	Emission α/β/γ	Half-life	Detection method
Uranium (U)	238	α	4.468x10 <sup>9</sup> y	MS, AS, HPGe ( <sup>234</sup> Th, <sup>234m</sup> Pa)
	235	α, γ	7.04x10 <sup>8</sup> y	MS, AS, HPGe
	234	α	2.455x10 <sup>5</sup> y	MS, AS
Protactinium (Pa)	234 <sup>m</sup>	γ	1.159 m	HPGe
	234	β	6.7 h	HPGe ( <sup>234m</sup> Pa)
Thorium (Th)	234	β, γ	27.1 d	MS, HPGe
	232	α	1.4x10 <sup>10</sup> y	MS, AS
	230	α	75,400 y	MS, AS
	228	α	1.9116 y	AS
Actinium (Ac)	228	β, γ	6.15 h	HPGe
Radium (Ra)	228	β	5.75 d	HPGe ( <sup>228</sup> Ac)
	226	α, γ	1600 y	HPGe (direct or <sup>214</sup> Bi, <sup>214</sup> Pb)
	224	α, γ	3.6319 d	HPGe (direct or <sup>212</sup> Bi, <sup>212</sup> Pb)
Radon (Rn)	222	α	3.8235 d	Emanation
	220	α	55.6 s	Emanation
Polonium (Po)	218	α	3.098 m	N/A
	216	α	0.145 s	N/A
	214	α	164.3x10 <sup>-6</sup> s	N/A
	212	α	0.299 x 10 <sup>-6</sup> s	N/A
	210	α	138.376 d	AS
Bismuth (Bi)	214	β, γ	19.9 m	HPGe
	212	β, γ	60.55 m	HPGe, AS
	210	β	5.012 d	N/A
Lead (Pb)	214	β, γ	26.8 m	HPGe
	212	β, γ	10.64 h	HPGe
	210	β, γ	22.2 y	HPGe

### 3.5.3.3 Measurement uncertainty

Radiation measurements are susceptible to issues that increase uncertainty and reduce precision.

These affect field measurements more than laboratory measurements, where they are explicitly controlled in standardized methods.

#### Detection limits and count time

Different detectors will have different limits of detection, defined as the minimum sample signal able to be discriminated against background measurements. A related concept is the minimum detection limit, which describes the minimum number of counts an instrument must receive in a given count time and geometry to have 95% confidence that the count was not acquired from background or blank sources (Currie, 1968). Detection limits can be improved by reducing interference from background radiation sources. This is more easily achieved in laboratory settings where samples can be shielded with lead compared to field measurements. Alternatively, the time taken to record a measurement (count time) can be increased to increase the overall signal being detected.

#### Geometry

Radiation is emitted from a source with equal intensities in all directions. This means the position of the detector with respect to the radiation source may have significant implications for what is detected. Similarly, non-homogenous distribution of NORM in infrastructure or a sample can affect measurements. Furthermore, the amount of radiation in a given area decreases by a rate proportional to the inverse square of the distance from the radiation source ( $\frac{1}{r^2}$ ). This means the distance at which a measurement is taken to the radiation source can significantly affect the measurement value. These issues are controlled in laboratory settings but contribute large uncertainties to *in-situ* measurements (i.e. using a handheld detector) where they are less controlled. Where *in-situ* measurements are collected, detailed reporting of the location of the sample relative to the instrument, background radiation measurements and time for analysis (with associated 'radioactive counting statistics') should be recorded to allow for accurate interpretation of such results.

#### Shielding

The type of radioactive emission and its energy determines how far it can penetrate through different substances (Figure 3.11). The attenuation of radiation through different material is known as shielding. Alpha particles are the least penetrative, being able to travel 2-10 cm in air and being stopped even by a layer of dust, grease, or dead skin cells. Beta particles can travel 10s of cm in air to only a few mm in material. Gamma radiation is the only emission that can penetrate steel. A common measure of shielding efficacy is the thickness of a particular material required to reduce the amount of radiation to 10%. These values are given for NORM radionuclides and steel in Table 3.14. A common pipeline thickness range is 6.4-38.1 mm (i.e. API 5L) meaning only certain gamma emissions of some radionuclides may be detected outside the pipe, such as  $^{226}\text{Ra}$ ,  $^{210}\text{Po}$ , and  $^{228}\text{Th}$ . Practically, this means that  $^{226}\text{Ra}$  in equilibrium with its progeny ( $^{214}\text{Pb}$  and  $^{214}\text{Bi}$ ) may be detectable outside of a pipe, but  $^{210}\text{Pb}$  deposits from  $^{222}\text{Rn}$  decay will not be detectable (Krieger, 2005). Either way, not all radionuclide decays for these radionuclides produce a gamma emission, meaning the radiation signal will be severely attenuated and instrument limits of detection will be too high to give reliable measurements (see the incidence of gamma in Table 3.14).

**Table 3.14** The thickness of steel required to reduce the radioactivity of different NORM-related radionuclides to 10% of their original value (10<sup>th</sup> value thickness). The larger the 10<sup>th</sup> value thickness, the more penetrative the gamma emission. Radionuclide half-life and the incidence of gamma emissions is also reported.

Isotope & half-lives	Steel 10 <sup>th</sup> value thickness (mm)*	Incidence of gamma
<sup>226</sup> Ra (1.6 x10 <sup>3</sup> y)	30	3.60%
<sup>214</sup> Pb (26.8 m)	73	63.40%
<sup>214</sup> Bi (19.9 m)	101	100%
<sup>210</sup> Pb (22.3 y)	1	4.30%
<sup>210</sup> Po (138 d)	78	0.001%
<sup>228</sup> Ra (5.7 y)	<2	1.60%
<sup>228</sup> Th (1.9 y)	22	1.20%

### Interferences

NORM-contaminated products contain a mixture of radionuclides due to the ingrowth of progeny from their decay chains (Section 3.4.4). Each radionuclide of the progeny has a characteristic emission type and energy of emission. The gamma spectrum for <sup>226</sup>Ra and <sup>228</sup>Ra, characteristic of radium-contaminated scale, is shown in Figure 3.20. Some emissions with similar energies will have overlapping peaks. Successful detection and isolation of a specific peak requires its contribution to be discernible above that of other emissions. To eliminate or reduce these interferences, radium can be chemically separated from other radionuclides, typically through digestion and precipitation reactions.

Scale often contains high concentrations of calcium, strontium, and barium. These elements severely attenuate <sup>210</sup>Pb and <sup>226</sup>Ra gamma emissions through a process of self-absorption. This is reported to lead to <sup>210</sup>Pb underestimations of up to a factor of 4 (Landsberger et al., 2013).

ICP-MS is also susceptible to analytical interferences due to the formation of isobaric particles (particles with the same mass/charge ratio such as <sup>228</sup>Th and <sup>228</sup>Ra) and polyatomic interferences which are ionized molecules that appear like particles with higher mass/charge ratios (e.g. <sup>88</sup>Sr<sup>138</sup>Ba interfering with 226-mass particles). Non-spectral interferences may also influence the sensitivity of the method. For example, salt deposition from the matrix (e.g. seawater or digested marine sediments) can clog cone orifices and torches leading to signal suppression. Non-spectral interferences are normally accounted for by the use of standards, certified reference materials, or through sample preparation steps that remove the environmental matrix such as salts from seawater (Lariviere et al., 2006).

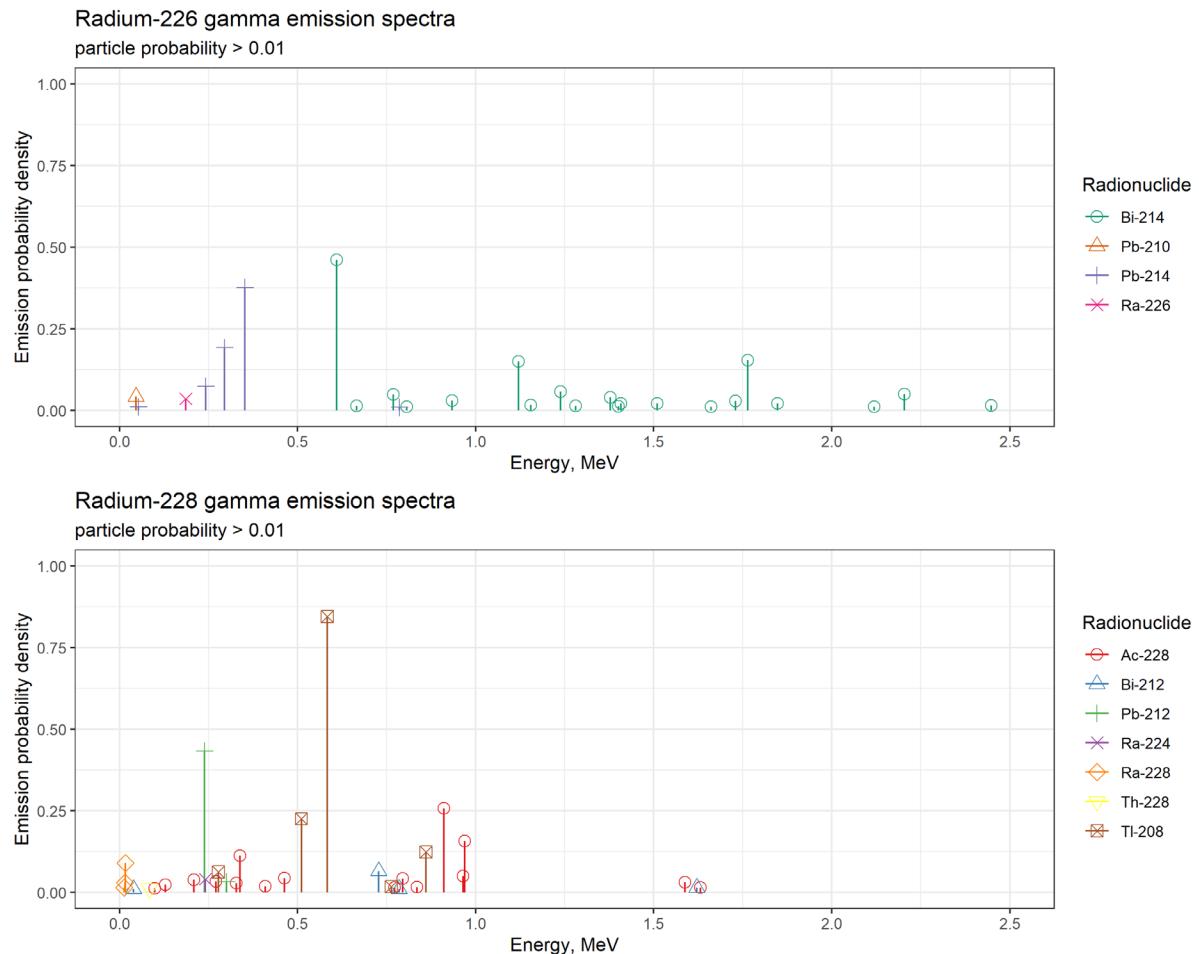


Figure 3.20 Gamma spectrum of  $^{226}\text{Ra}$  and  $^{228}\text{Ra}$  for emissions with a probability >0.01. Radium head of chain and their gamma-emitting progeny are shown. Data from the IAEA Nuclear Data Service.

## 3.6 Ecosystem receptors

Ecological risk assessments consider the exposure to, and hazard impacts on, important ecosystem receptors. This section reviews the ecosystem receptors relevant to an ecological risk assessment of mercury and NORM in subsea oil and gas infrastructure. It does this by considering the local environments and organisms of active basins and the legislative requirements for decommissioning considered in Section 3.2. In doing so, this section partly meets objective 1, 2, and 4, and problem statements 1 and 2.

### Key findings:

- Ecosystem receptors are too varied and regionally specific to be systematically considered. Ecological risk assessments should consider site-specific ecosystem receptors.
- Contaminants associated with subsea oil and gas pipelines pose the greatest threat to epibenthic and benthic pipeline communities due to their proximity to the pipelines. However, bioaccumulating contaminants may impact the wider food webs.
- Ecosystem receptors such as the recreational, commercial, and indigenous values for marine resources may be susceptible to impacts from contaminant bioaccumulation.

### 3.6.1 Overview of ecosystem receptors

Ecosystem receptors are environments, species, or values subject to consideration in an ecological risk assessment (Suter II, 2006). The *Environment Regulations* outlines the requirement for Environment Plans to contain a description of the existing environment that may be affected by the activity and details of particular relevant values and sensitivities where environment includes its social economic, and cultural values (*Environment Regulations r. 13-2*). These may include matters of national environmental significance (outlined in Section 3.2.4), but are not limited to them. For example, NOPSEMA refers to environmental receptors as including different plants, animals, ecological communities, and the social, economic and cultural features of those. This is a similar definition used in the Water Quality Management Framework which defines ecosystem receptors as any organism, habitat, or natural resource that could be negatively impacted by environmental contaminants (ANZG, 2018).

Some social, economic, and cultural features of an ecosystem will be common around Australia. Many of these values were recently compiled by Shaw et al. (2018) through extensive consultation across industry, academic, government, and community participants and included:

- Recreational values of an ecosystem (e.g. for fishing and swimming)
- Commercial values such as tourism and commercial fisheries and aquaculture
- Indigenous spiritual and cultural values of marine resource gathering and use

Habitat or organism receptors are too varied and regionally specific to be systematically considered here. Rather, a general description of the marine regions where the majority of offshore oil and gas extraction occurs and the specific ecosystems normally associated with offshore oil and gas

infrastructure is detailed in this section. Australia has a wide range of marine ecosystems that are grouped into six marine regions (Figure 3.21), defined as areas of similar biophysical and ecological characteristics (Waters et al., 2010). For each marine region, a marine bioregional plan has been developed to support environmental decision making by providing a consolidated understanding of different environments, their conservation value, regional priorities, and biodiversity objectives.

Additionally, Commonwealth Department of Agriculture, Water, and Environment has developed tools to assist operators identifying whether their activity overlaps areas of national environmental significance or key ecological features (important regional features of the Commonwealth marine environment). These include the Species Profile and Threats database (see <https://www.environment.gov.au/cgi-bin/sprat/public/sprat.pl>, date accessed 25 February 2021), the Protected Matters Search Tool (see <https://www.environment.gov.au/epbc/protected-matters-search-tool>, date accessed 25 February 2021), the National Conservation Values Atlas (see <https://www.environment.gov.au/marine/marine-bioregional-plans/conservation-values-atlas>, date 25 February 2021), and marine bioregional plans e.g. (Commonwealth of Australia, 2012a).

These sources may support operators in defining important ecosystem receptors (including ecosystem types, individual species, or other important values arising from the potentially impacted environment) subject to management considerations.

Marine ecosystems are spatially variable due to the habitat niches that form in response to local environmental conditions including temperature, depth, hard rocky or soft sedimentary substrate, sediment particle size and composition, nutrient availability, and the interconnectivity of surrounding environments through ocean currents (Underwood et al., 2020; Williams et al., 2010). This means that a single pipeline may be home to varied ecosystem patches along its length, as is often recorded by remote video footage (Bond et al., 2018b). In all cases the pipeline infrastructure will provide a hard substrate.

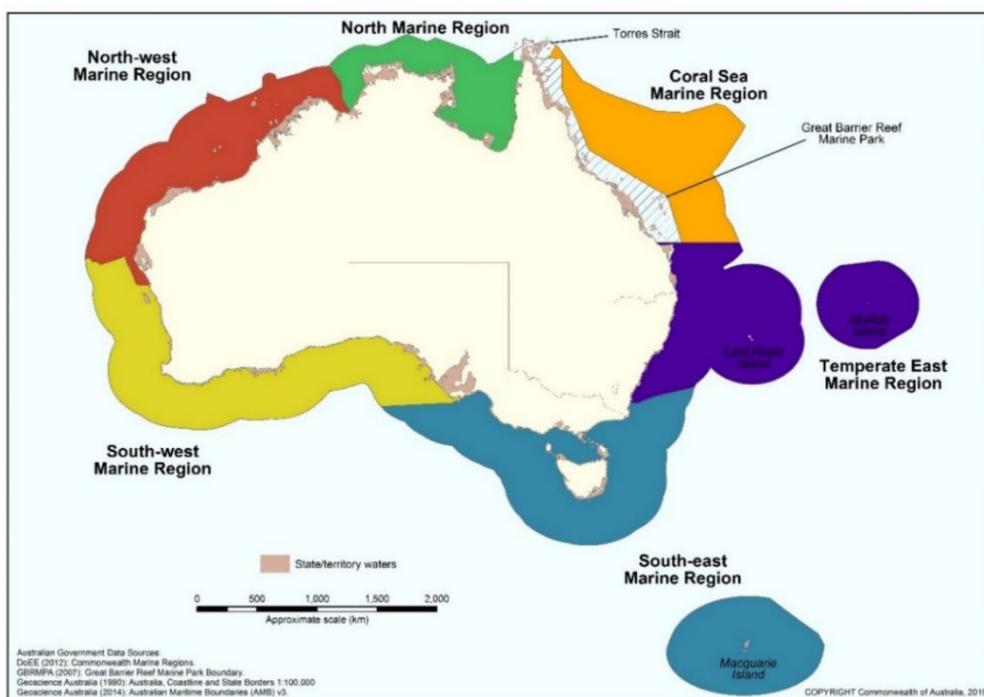


Figure 3.21 Marine regions in Australia. Taken from Commonwealth of Australia (2012a).

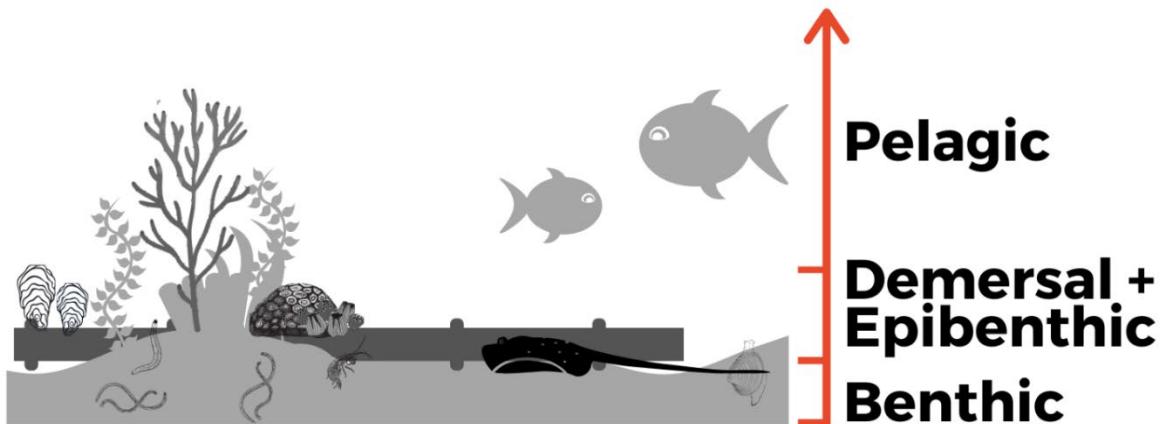


Figure 3.22 Different habitat types associated with pipeline infrastructure.

These habitat niches can be described as benthic (living in sediment), epibenthic (living on the sediment), demersal (living in the water close to the seabed) and pelagic (living within the entire water column) habitats (Figure 3.22). Pipeline infrastructure may exist across all these habitat types, for example, buried, on the seafloor, in free spans (Leckie et al., 2016) or in flexible or solid marine risers, and contaminants may partition across these habitats depending on their exposure pathways (discussed in Section 3.7). Different communities of organisms reside or associate with these habitats based on their ecology, life lifecycle, and feeding habits. These different habitats also have varying physical chemistries which will lead to different environmental transformations and exposure pathways for mercury and NORMs.

### 3.6.2 Marine regions

Nearly all offshore oil and gas activity in Australia occurs within three marine regions, the South-east, North-west, and North marine regions (Figure 3.21). Each marine region is characterized by different physical properties and ecological features. However, all marine regions contain threatened and endangered species, important migratory organisms, World Heritage Areas (excluding NT Marine Region), marine reserves, and other matters of national environmental significance.

All marine regions provide important ecosystem services that enable community values (which may also be referred to as socioeconomic values). For example, all marine regions include sites of important commercial and recreational fishing activities and significant tourism. In addition, sites that hold cultural and spiritual significance to Indigenous people who use marine resources for subsistence hunting and commercial activities.

The different properties of bioregions (i.e. ecosystems or physical and chemical parameters) means that different regions will be affected differently by pressures and stressors. These differences are accounted for by the incorporation of regional specific consideration of pressures and stressors into the water quality management framework (ANZG, 2018; Commonwealth of Australia, 2006).

#### 3.6.2.1 South-east marine region

The South-east marine region is large and diverse, incorporating the Bass Strait shelf and the Southern Ocean waters around Macquarie Island (Figure 3.21). The Bass Strait shelf is a vast shallow expanse with a maximum 90 m depth flanked by the Western Bass Strait Shelf and Southeast Shelf Transitions extending to 272 m and 359 m maximum depth. These three provincial regions encompass most oil and gas activity in the marine region (Commonwealth of Australia, 2015). Oil and

gas activity in this region is concentrated in the Gippsland Basin, occupying the east extent of the Bass Strait shelf, with the Otway and Sorrell Basins on the north-west and south-west of the shelf, respectively, representing the other oil and gas producing basins in the marine region.

The South-east marine region is a temperate environment affected by seasonal ocean currents. The shallow nature of the shelf means that seasonal temperature variability is more pronounced than other areas of this region. The western areas of the marine region receive cooler nutrient-rich waters owing to the Leewin and Zeehan Currents transporting Southern Ocean waters. The eastern areas of the marine region receive inputs from nutrient-poor Pacific sub-tropical waters transported by the East Australian Current in summer but not winter.

The Bass Strait shelf ecosystem includes a mosaic of rocky reefs and soft sediments habitats supporting diverse species from a broad range of taxonomic groups. The shelf transitions are sites of upwelling from ocean depths leading to greater marine productivity. For example, the Bass Strait Water Cascade is a feature along the shelf break east of the Bass Strait and is an area of high productivity owing to deep-ocean upwelling. Marine canyons also occupy the shelf transitions which form important habitat for sessile invertebrates such as corals and associated higher-order organisms. The biodiversity of molluscs, macroalgae, and echinoderms the South-east marine region is highly endemic, meaning many of these organisms are only found within this marine region.

Three key ecological features are defined for areas subject to oil and gas activities, the Bass Cascade and Upwelling east of Eden overlap the Gippsland Basin and the West Tasmania canyons overlap the Otway and Sorrell canyons. The Bass Cascade and Upwelling east of Eden features arise from nutrient mixing due to currents leading to high productivity supporting fisheries and diverse ecosystems. The West Tasmania canyons are physical features that act as sinks for organic sediments that create diverse sponge and fish habitat, particularly between 250 and 300 m depths.

The South-east marine region is the site of important commercial and recreational fishing activity. These include for scallops, rock lobsters, giant crabs, demersal and pelagic fish by line and trawls, and squid by jig. The coastal marine ecosystem also holds cultural and spiritual significance to Indigenous people who use marine resources for subsistence hunting and commercial activities.

### 3.6.2.2 North-west marine region

The North-west marine region includes the Western Australia–Northern Territory border in the north to Kalbarri, south of Shark Bay (Figure 3.21). The region comprises shallow coastal waters and extensive shelves, slopes, and plateaus and terraces. The ecosystem ranges from tropical in the north to sub-tropical at the southern extent (Commonwealth of Australia, 2012a).

Ecosystems in the North-west marine region are broadly classified as shallow-water tropical marine ecosystems with high species richness and low productivity subject to boom-bust seasonal cycles. Physical conditions and biodiversity in the North-west marine region are highly influenced by the Leewin current which transports tropical waters and associated larvae and organisms from Indonesia along the west coast of Australia. This dispersion and transport means that there are low rates of endemism in these ecosystems. Water temperatures range from 20-24 °C in winter to 24-28 °C in summer.

There are diverse habitat and ecosystem types within the region, including coralline algae and carbonate shoals in the north, coral reefs of the Ashmore, Hibernia, Scott, Seringapatam, Ningaloo

and the Rowley Shoals, and the muddy sediments containing mobile invertebrates and sessile filter-feeding organisms in the Joseph Bonaparte Gulf.

Thirteen key ecological features span the North-west marine region reflecting subsea features, plateaus, or areas of nutrient mixing. These overlap all areas of oil and gas activity, which extend along the entire marine region in the Carnarvon, Browse, and Bonaparte basins.

Oil and gas activities are listed as potential pressures to Seringapatam Reef and Commonwealth waters in the Scott Reef complex, Mermaid Reef and the Commonwealth waters surrounding Rowley Shoals, and Commonwealth waters adjacent to Ningaloo Reef key ecological features.

Parts of the North-west marine region may be considered degraded relative to their historical environmental condition. Japanese and Taiwanese fishing operations between 1959 and 1989 followed by a smaller domestic operation trawled areas of the North-west marine region (Sainsbury et al., 1997). This altered the epibenthic community away from its historical composition of sponges, alcyonarians, and gorgonians to more open sandy environments. This also changed the composition of fish species associated with the ecosystems.

### 3.6.2.3 North marine region

The North marine region extends from the Northern Territory–Western Australia border to the west Cape York Peninsula (Figure 3.21) and is characterised by a wide shelf typically no deeper than 70m but with a range from 10 to 357 m. The North marine region is susceptible to wind-driven mixing and strong tides which drive currents, rather than oceanographic currents which are more typical of other marine regions. The region also features high sediment inputs owing to monsoon and cyclonic weather activity on the northern Australia coast (Commonwealth of Australia, 2012b).

Three key ecological features based on unique seafloor structures may overlap oil and gas activities include the Pinnacles of the Bonaparte basin, eroded limestone remnants of underlying strata that provides vertical habitat creating high biodiversity aggregations; carbonate bank and terrace system of the Van Diemen rise, an area with variable water depth and substrate leading to unique seafloor ecosystems that provides ecosystem connectivity between the Joseph Bonaparte Gulf and the Timor Trough; and the shelf break and slope of the Arafura sea, a continental slope region of patchy reefs and hard substrate pinnacles.

Phytoplankton are important components of food webs in the North-marine region with a high diversity of large diatoms the primary producers for zooplankton. The region is also a site of important breeding and feeding grounds for a large number of migratory, threatened and endangered species including turtles, sea snakes, and cetaceans.

## 3.6.3 Habitat associated with subsea infrastructure

Contaminants associated with subsea oil and gas pipelines pose the greatest threat to epibenthic and benthic pipeline communities, due to their proximity to the pipelines. However, as these reef communities are foraged by pelagic organisms including fish and marine mammals, bioaccumulating contaminants may negatively affect the wider food webs. Given the complicated interactions defining the type and structure of pipeline ecosystems, risk assessments should consider them as generalised organism groupings, such as the standard growth types described by video analyses of pipeline communities (Bond et al., 2018a; Bond et al., 2018b; Bond et al., 2018c; McLean et al., 2017). If required, detailed risk assessments may consider site-specific ecosystem types relevant to individual

infrastructure being decommissioned, or exposure pathways to single species subject to special conservation requirements (such as threatened or endangered species).

For the purpose of ecological risk assessments, the epibenthic and benthic habitat created by, and immediately adjacent to, subsea pipeline infrastructure can be considered the most at-risk ecosystem from COPCs due to their proximity.

### 3.6.3.1 Epibenthic pipeline ecosystems

Subsea infrastructure, such as pipelines, rock dumps, and concrete blocks, provide a hard substrate that may be colonised by sessile organisms to form epibenthic communities (Fowler et al., 2018). These epibenthic communities become habitat for a range of organisms including invertebrates and fish, and provide foraging grounds for fish and marine mammals. For example, animal tracking studies have shown that Australian fur seals make use of ecosystems associated with anthropogenic structures in the Bass Strait (Arnould et al., 2015). Pipeline reefs also provide a commercial value by supporting higher biomasses of commercially-important fish stock than otherwise would exist in the region (Claisse et al., 2014; Rouse et al., 2018). Pipeline ecosystems have also been shown to support threatened species, such as those on the International Union for Conservation of Nature's red list species including whale sharks and manta rays (McLean et al., 2019). Over time, the benefits from these reefs may also extend to surrounding benthic ecosystems (Rouse et al., 2019).

Epibenthic communities on pipelines are often identified by autonomous video footage and are characterised based on a standard classification scheme (Althaus et al., 2015). Most commonly found on pipelines include encrusting growth (typically coralline microalgae), black and octocorals, cnidarians (Bond et al., 2018a), crinoids (Bond et al., 2018a), echinoderms (basket stars and urchins), molluscs (bivalves), and sponges (McLean et al., 2020). In more complex habitat structures, many of these growth types will coexist forming complex invertebrate communities, which are often correlated to increased fish abundance. The types of organisms that can be observed and identified by underwater video footage are limited and can often exclude smaller invertebrates and larvae (Macreadie et al., 2018). These may include zooplankton, polychaetes, and other benthic invertebrates. However, the known feeding habits of identified fish species confirm their presence.

A range of biotic and abiotic factors defines whether or not epibenthic reefs will form and their composition (Todd et al., 2018), including infrastructure depth, pipe position and type (McLean et al., 2017), ocean currents and scour (Gates et al., 2019), and surrounding ecosystem type (McLean et al., 2020). The size and composition of epibenthic communities may then have a controlling influence on the surrounding pelagic ecosystem, with the magnitude of fish associated with subsea pipeline infrastructure generally related to the complexity of epibenthic communities on the pipeline (Bond et al., 2018b; McLean et al., 2017). For these reasons the ecosystems or habitats can vary spatially along the transect and around the proximate radial area of the pipeline infrastructure (Todd et al., 2020). Pipeline ecosystems may also vary temporally due to the natural succession of colonising organisms and the feedback mechanisms between ecosystems and habitat (Todd et al., 2020). Older pipelines generally have a greater proportion of complex epibenthic communities relative to newly installed pipes which may first be colonised with simple encrusting organisms (Bond et al., 2018a; McLean et al., 2020). The combination of spatial, temporal, biotic, and abiotic determinants of epibenthic community structure makes predicting the types of communities that will colonise subsea infrastructure before it is placed or after it is decommissioned with any certainty impossible (Bond et al., 2018a; Macreadie et al., 2018).

Many of the biotic and abiotic factors that define the pipeline community will also affect contaminant risk. For example, sessile organisms that establish on a pipeline will have a greater exposure to radiation due to their proximity to the pipeline, compared to pelagic fish that only temporally graze or migrate around the pipeline. Pipeline sections dominated with filter-feeding organisms may be at a greater risk than sections dominated by macroalgae once contaminants are released due to the ingestion of particulate-bound  $^{210}\text{Po}$ ,  $^{210}\text{Pb}$ , or mercury (the likely speciation of those metals in oxic seawater). These are site-specific factors which should be considered on a case-by-case basis for contaminant exposure and release.

### 3.6.3.2 Benthic pipeline communities

The benthos is characterised by organisms living at or below the sediment-water interface of the seafloor. These ecosystems can be diverse and provide essential ecosystem services including breaking down organic matter settling from pelagic and epibenthic environments and releasing sediment-bound nutrients through burrowing activities or microbial biogeochemical reactions (Herbert, 1999; Levin et al., 2001). The organisms typically found may include amphipods, polychaetes, nematodes, molluscs, benthic and demersal fish, sea stars and a diverse prokaryotic and eukaryotic microbial community (Rouse et al., 2019).

Benthic marine sediments stratify into an oxic layer interfaced with overlying water, a suboxic layer where nitrate reduction occurs, and anoxic layers below where sulfate oxidation occurs (Glud, 2008). This reflects the limited penetration depths of oxygenated seawater into sediments. Specialised microbial communities reside in each layer and utilise alternative chemical compounds for their respiration where oxygen is limited. This leads to distinct chemical and habitat environments along the epibenthic – benthic depth gradient.

The benthic community is less characterised along oil and gas pipelines compared to pelagic communities because ecosystem monitoring is usually conducted with ROV video surveys (McLean et al., 2017). Benthic organisms may be microscopic (including prokaryotic and eukaryotic communities of bacteria, photosynthetic microalgae, and protists), residing on or under the sediment surface, or otherwise too small to identify. Nonetheless, subsea infrastructure is known to alter benthic communities, possibly through the formation of new epibenthic ecosystems, including those based on coral, sponge, mussels, or oyster communities (Manoukian et al., 2010) which in turn influence benthic community composition through nutrient inputs and benthic-pelagic interactions (Griffiths et al., 2017).

## 3.7 Exposure characterisation

The exposure characterisation of a risk assessment considers the movement of contaminants in an environment. This section addresses in part objective 3 and 4 and problem statement 2 by reviewing what is known about the speciation and environmental transformations of mercury and NORMs in the marine environment to infer likely exposure pathways linking contaminated product in subsea oil and gas infrastructure (defined in Sections 3.3.6 and 3.4.3) to ecosystem receptors (such as those defined in Section 3.6).

### Key findings:

- Limited information exists on the behaviour of likely mercury and NORM-contaminated products in pipelines, so exposure pathways are inferred from an understanding of the behaviour of the elements themselves.
- Key environmental transformations will govern the exposure pathways of mercury and radium-contaminated products.
- Various environmental variables such as temperature and the presence of organic matter affect these reactions and so will affect the ecological risk.
- The time between pipeline decommissioning and seawater breakthrough following degradation may allow for significant contaminated product speciation changes, radionuclide ingrowth or decay. These are uncertainties given the large range in degradation rates and unknown environmental conditions inside decommissioned pipes.

### 3.7.1 Contaminant exposure pathways in marine ecosystems

Exposure characterisation considers contaminants spatial and temporal distribution in environments and their exposure pathways to ecosystem receptors (Suter II, 2006). Exposure pathways are often represented in conceptual models describing environmental transformations, partitioning, and mobility of a contaminant source to an environmental receptor (Norton and Schofield, 2017). These models are recommended by the water quality management framework (ANZG, 2018). An example of a generic conceptual model is given in Figure 3.23 for contaminant exposure pathways to benthic and epibenthic communities (Simpson and Batley, 2016). An important component of an exposure characterisation is the understanding of contaminant speciation, partitioning, and transformations in the environment as these processes govern contaminant exposure pathways. Mercury and NORM-contaminated products are not well characterised and little to no research has been conducted on their behaviour in the marine environment (MacIntosh, 2020). Therefore, the general behaviour of mercury and NORMs are reviewed to infer likely transformations in the marine environment.

In the environment, metal contaminants may exist as different species including: dissolved unbound free ions, bound to simple inorganic ligands, or bound to dissolved organic ligands like fulvic acids or proteinaceous substances (Allen, 1993). Contaminants may also associate with the solid phase, precipitated as inorganic salts, adsorbed to particulate matter, or exist in their elemental form. Contaminants may fractionate over time to different binding phases based on their individual binding affinities and chemistries given a particular set of environmental conditions (Tessier and Campbell,

1987). This partitioning may change as environmental conditions change, especially the extent to which the environment is oxidising or reducing.

In sediments, contaminants are typically sequestered in solid phases (adsorbed to organic matter or particulate matter or precipitated) but may partition to aqueous (overlying and pore waters) phases by biogeochemical reactions, Figure 3.23. At the oxic sediment surface, metals are typically bound to amorphous iron and manganese oxides, particulate organic carbon, or are co-precipitated with carbonates. The subsurface of sediments may be anoxic, where metals form insoluble sulfide minerals or co-precipitates (Huerta-Diaz and Morse, 1992). This partitioning will depend on the physicochemical conditions of the environment, such as the presence of complexing organic matter and mineral phases, the pH, sediment texture, and redox potential. In turn, these factors will affect the extent to which contaminants may be sequestered in sediments or remain bioavailable to benthic organisms (Egginton and Thomas, 2004).

Alterations to the stratification of sediments can affect the bioavailability of metals in sediments. Bioturbation is the biological process of disturbing sediments and pore waters. This may be done to forage for food, to form a burrow or habitat, or irrigate a habitat. The process of bioturbation can radically change sediment and pore water geochemistry (Williamson et al., 1999) and has important implications for contaminant mobility (Amato et al., 2015; Remaili et al., 2016). For example, metal-sulfur complexes can be oxidised in anoxic-oxic sediment transitions leading to their mobilisation (De Jonge et al., 2012). However, this biological redistribution of contaminants is likely to be less significant compared to physical (e.g. current-driven) redistribution processes (Bradshaw et al., 2006).

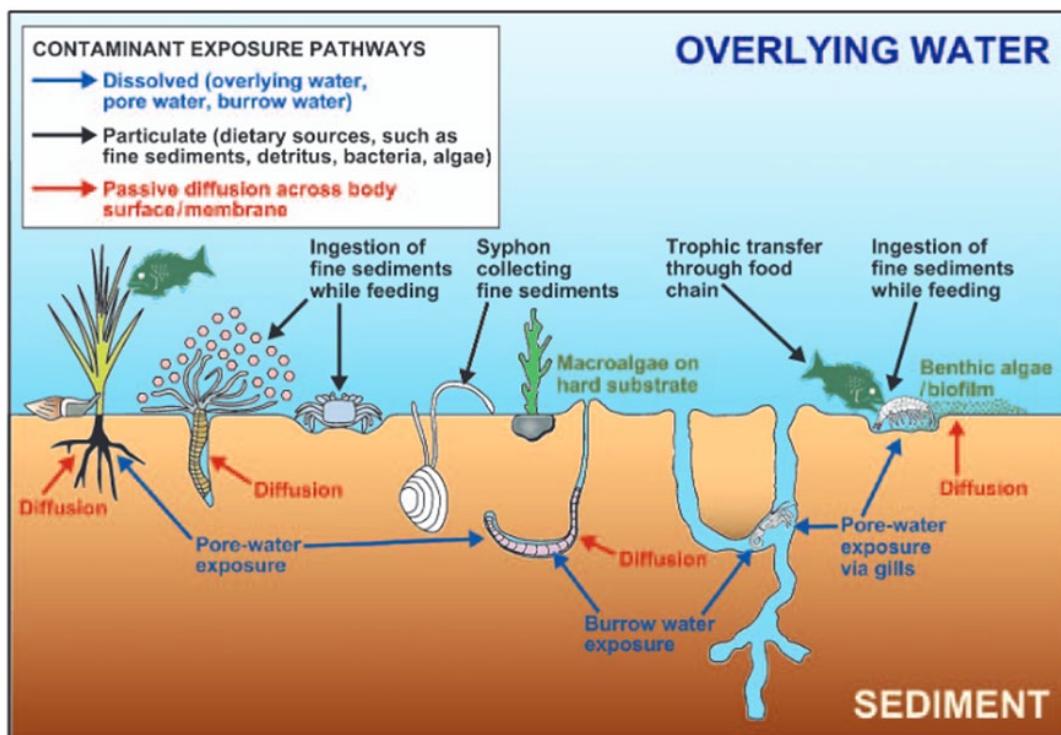


Figure 3.23 Conceptual model of some contaminant exposure pathways in marine environments. Taken from Simpson and Batley (2016).

Dietary exposure of contaminants may be an important exposure pathway to organisms, and will vary depending on their feeding behaviour (e.g. carnivore, herbivore, detritivore), rates (King et al., 2005; Luoma, 1989; Luoma and Fisher, 1997), and contaminant's speciation and partitioning in the marine ecosystem (Schneider et al., 2018). Some contaminants are also able to transfer effectively from organism to organism via dietary exposures which can lead to biomagnification, the progressive accumulation of contaminants in organisms up the food web.

A large number of factors affect contaminant exposure pathways in marine environments. Many of these are based on the relationship between contaminant speciation and physico-chemical parameters in the environment. These are discussed in more detail for mercury and NORM radionuclides below.

### 3.7.2 Mercury

The exposure pathway of mercury from the contaminated products to ecosystem receptors will be controlled by its chemical speciation which in turn is controlled by environmental conditions. Mercury-contaminated products occur in a variety of forms and species such as elemental or ionic mercury species adsorbed to steel surfaces, adsorbed to corrosion products, or entrained in sludges and asphaltenes (Section 3.3.6). Once exposed to the marine environment, mercury will undergo transformations through methylation, demethylation, reduction and oxidation process which will govern its mobility and bioavailability. These reactions are affected by environmental parameters such as temperature, pH, redox status, and the presence of anaerobic bacteria. Possible mercury environmental transformations and exposure pathways in the marine environment from contaminated infrastructures are discussed in the following sub-sections and shown in Figure 3.24.

There have been no studies investigating the basic properties of these likely contaminated products in the marine environment. It is also not known what speciation changes will occur during the period between pipeline decommissioning and the breakthrough of seawater due to corrosion. This will likely depend on the redox potential of the internal pipeline environment, which in turn will be determined by decisions to flush the pipelines, add biocidal and anti-corrosive chemicals, and leave it open or plugged.

The behaviour of mercury species and the important environmental transformations including oxidation and methylation reactions are reviewed here to infer likely exposure pathways of mercury in contaminated products.

#### 3.7.2.1 Exposure pathways of mercury-contaminated pipelines

The speciation of mercury in contaminated products and the environmental transformations that convert immobile species to mobile and bioavailable species will define its exposure pathway in marine ecosystems. At the time a pipeline is decommissioned, these contaminated products are contained inside the pipe and inaccessible to ecosystem receptors (Section 3.3.6).

Once the pipeline is corroded and seawater ingress occurs, ionic and elemental mercury species will remain generally insoluble and may disperse and mix into surrounding sediments. Over time, mercury will undergo transformations in the environment converting elemental or sulfide species to more bioavailable inorganic species through oxidation reactions with iron oxides or organic matter such as polysulfides and humic substances (Slowey and Brown, 2007).

Methylmercury is the mercury species of greatest concern because it is bioavailable. It can lead to significant impacts on marine life and humans following its accumulation in seafood (Bowles et al.,

2001; Scheuhammer, 2012). The degree to which, and rate at which, inorganic mercury in contaminated products is methylated will determine the risk it poses to the surrounding ecosystem (see discussion of factors affecting mercury methylation above). Different inorganic mercury species may have different bioavailabilities to microorganisms. For example, inorganic aqueous ions of mercury and mercury-organic matter species were shown to be more bioavailable to microbes and had greater methylation rates in estuarine sediments, when compared to HgS (Liem-Nguyen et al., 2016).

Once methylmercury is absorbed by organisms, it is not easily eliminated (Harding et al., 2018). The persistence of methylmercury in tissue results in biomagnification across the food chain (Driscoll et al., 2013; Gray, 2002). This particularly affects long-lived predatory fish, marine mammals, and seabirds and is discussed further in Section 3.8.2.2. The bioaccumulation of mercury means that the spatial range of impacts could extend to the whole food web. For example, Australian fur seals travelled 65–175 km to forage along sub sea oil and gas infrastructure (Arnould et al., 2015).

### **Mercury speciation in the marine environment**

In marine environments, mercury naturally exists as elemental species ( $Hg^0$ ), ionic species (including  $Hg^{2+}$  or various chloride species  $HgCl_3^-$ ,  $HgCl_2$ ), complexed with dissolved or particulate organic carbon, or as the organic species methylmercury ( $MeHg^+$ ) or dimethylmercury ( $DMeHg$ ). For the purposes of this review, methylmercury can be taken to mean both organic forms of mercury.

Background mercury concentrations in seawater are typically in the parts per trillion (ng/L)

When at equilibrium, mercury-contaminated products will partition to ionic and organic mercury species depending on environmental conditions (Morel et al., 1998). Organic mercury is much more soluble, but represents a smaller proportion of the total amount of mercury (Figure 3.25).

Transformations between elemental and ionic are known as oxidation and reduction reactions, respectively while transformations between ionic and methylated species are known as methylation and demethylation reactions, respectively. Organic forms of mercury pose the greatest risk to marine ecosystems because of their greater solubility and bioavailability (Harding et al., 2018).

Mercury methylation is a microbial-mediated process. The rate at which methylation occurs is highly dependent on environmental factors governing microbial activities (Ndungu et al., 2016), as well as the bioavailability of inorganic mercury to microorganisms (Ma et al., 2019). Methylation typically occurs in the upper layers of sediments under weakly reducing conditions; however, it may also occur in the atmosphere or water column on particulate organic matter. Anaerobic bacteria are largely responsible for mercury methylation and may include sulfate reducing, iron reducing, and methanogenic bacteria (Ma et al., 2019; Slowey and Brown, 2007). Anaerobic bacteria are usually found in anoxic sediments where they use oxidised sulfate, iron, or carbon species as electron acceptors for biochemical respiration reactions. These processes produce methylmercury as well as sulfide, iron(II), and methane, which have important implications for trace metal cycling in sediments generally (Allen et al., 1993).

### **Factors affecting mercury methylation**

Methylation rates vary widely between and within different environment types because of the range physico-chemical factors affecting the biochemical reactions. Factors including temperature, redox state, bioavailability of the inorganic mercury compound, and pH may affect these reactions (Heyes et al., 2006). A recent review reported a range of rates of 0–1.5 d<sup>-1</sup> across all environments and between 0–0.5 d<sup>-1</sup> for marine environments Figure 3.25, (Paranjape et al., 2017). Typical proportions of

methylmercury species are typically <15% in freshwater and <5% in seawater relative to total mercury concentrations (Figure 3.25).

Generally, factors increasing microbial activity in sediments and the degree of anoxia will increase methylation rates. For example, temperature and sediment organic matter content are positively correlated with methylation rates and will play dual roles in increasing microbial activity by increasing metabolic and reaction rates and providing an energy source and methyl groups for mercury, respectively (Dai et al., 2021). Total mercury content is important to methylation rates at low concentrations, but has an overall logarithmic relationship implying that microbial activity is rate-limiting at higher mercury concentrations (Dai et al., 2021; Paranjape et al., 2017).

As methylation is primarily a microbially-mediated process, chemical or environmental factors influencing mercury bioavailability will affect methylation rates. Methylating microorganism can only accumulate ionic forms of mercury (i.e.  $Hg^{2+}$  and its ionic species as reported in Table 3.6). This means that elemental mercury needs to be oxidised to be bioavailable. Different ionic mercury species themselves may have different bioavailabilities (O'Connor et al., 2019), and environmental factors including sediment pore-water sulfide content and the presence of the iron sulfide mineral mackinawite may reduce bioavailability (Johnson et al., 2015; Skyyberg et al., 2021). These factors lead to high spatial variability, even within single ecosystems.

Abiotic methylation reactions (i.e. methylation in the absence of a microbe) may occur in some conditions, such as in the presence of high organic matter and low sulfate concentrations, they are comparatively minor to total methylation rates and unlikely in seawater conditions (Dai et al., 2021).

Abiotic and biotic factors are also important determinants for the rate of mercury demethylation. Which may occur to a greater extent than methylation rates (Figure 3.25). Photodegradation is likely to be the primary abiotic demethylation mechanism (Black et al., 2012) while organomercury lyase from the MerB gene in bacteria is the primary mechanism for biotic demethylation (Schaefer et al., 2004).

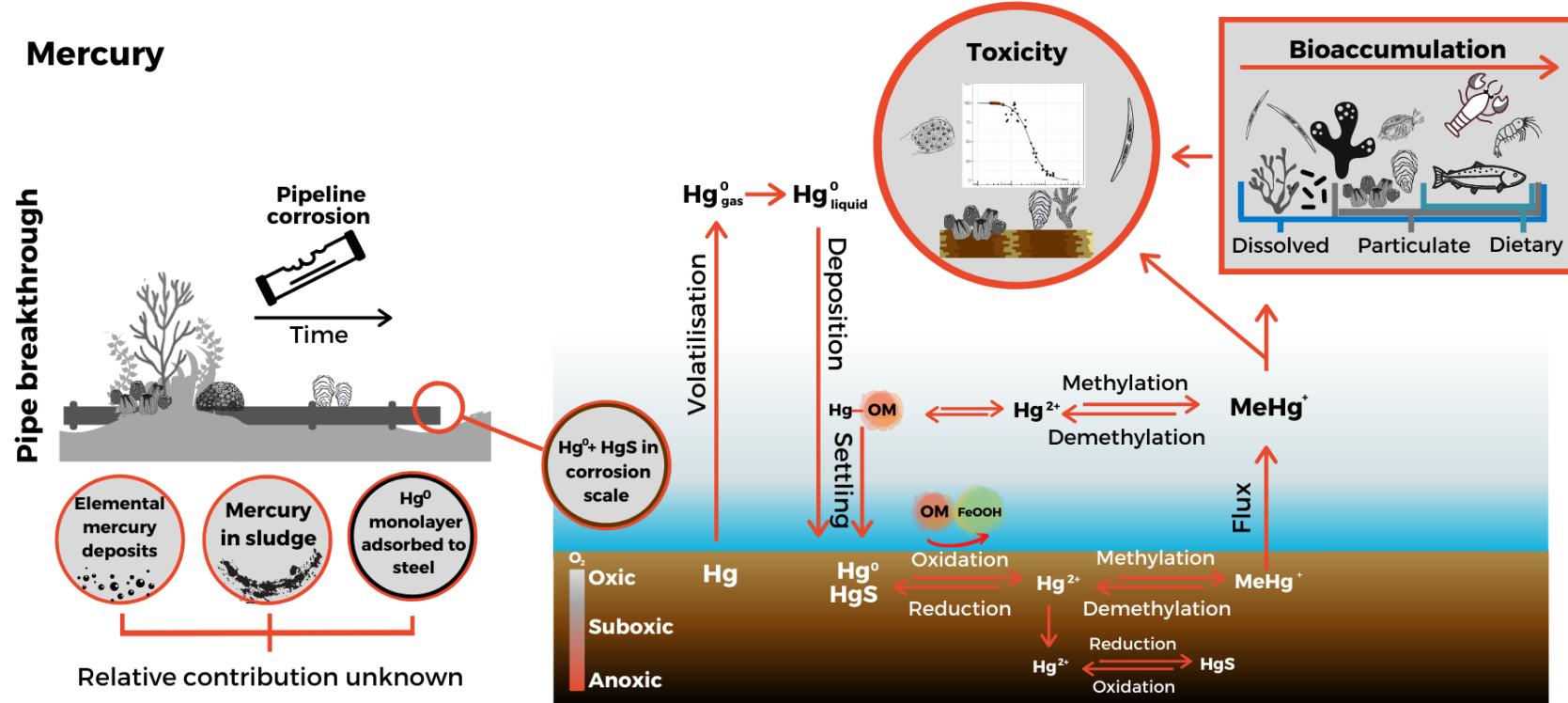
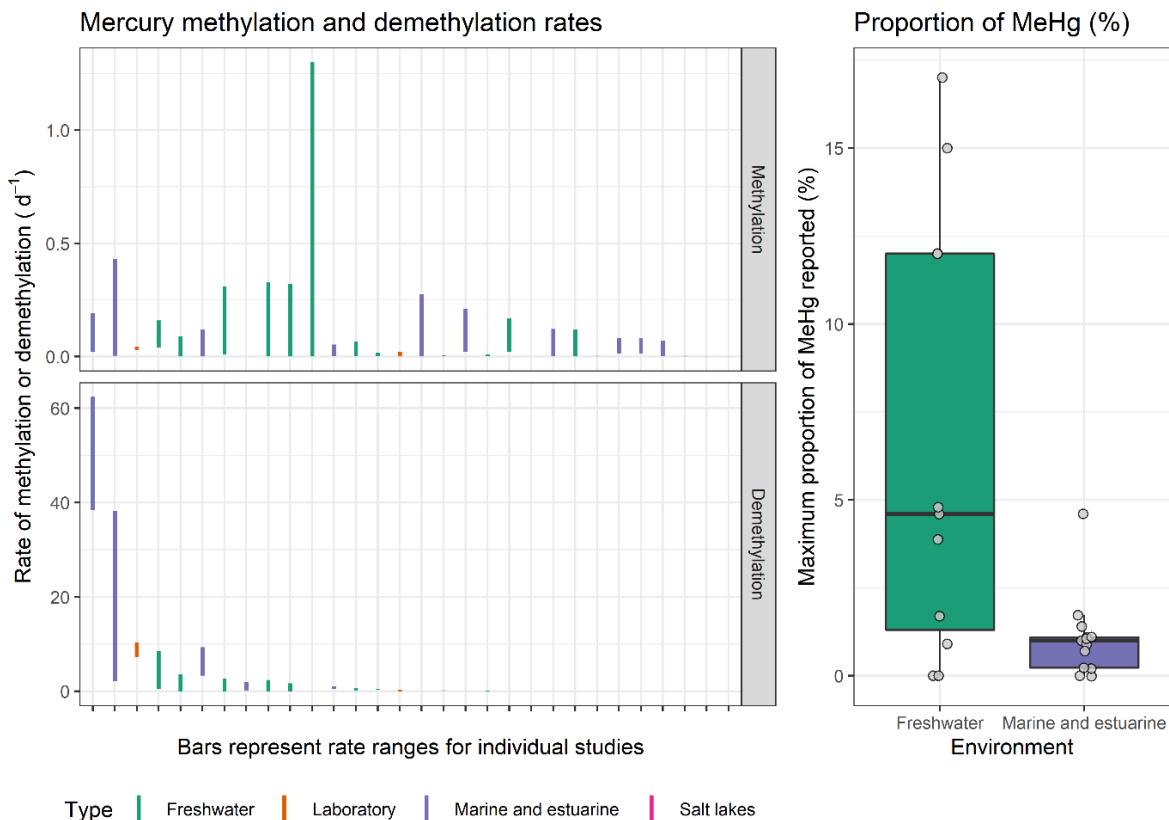


Figure 3.24 Mercury (Hg) exposure pathways in the marine environment including key environmental transformations such as oxidation and methylation reactions. The suffix OM refers to organic matter.



**Figure 3.25** Mercury methylation and demethylation rates, and the proportion of methylmercury (MeHg) found in different environments reported by Paranjape et al. (2017) and references therein. Individual bars and points represent the values for individual studies.

### 3.7.2.2 Mercury contamination in marine environments

Research into the environmental fate of mercury from two contamination events that may be somewhat analogous to the *in-situ* decommissioning of mercury-contaminated infrastructure reveals the importance of inorganic speciation and the controls of local environmental conditions for the overall risk of mercury in the marine environment.

Minamata Bay in Japan was subject to approximately 70-150 t of mercury discharge from industrial wastewater between 1932 and 1968. An estimated 0.6-6 t of this mercury was methylmercury. This resulted in total mercury concentrations in bottom sediments of 25 mg/kg which extended to depths of up to 4 m in sediment. A remediation effort removed the most contaminated sediments. However, natural sediment resuspension and deposition meant mercury concentrations of 6 mg/kg were persistent in surface sediments (Balogh et al., 2015). The bay sediments are high in organic matter (~7%), have a bottom seawater temperature range of 13-26 °C, and have methylmercury concentrations of  $1.7 \pm 1.0 \mu\text{g}/\text{kg}$ , 16-fold higher than nearby control sites but only reflecting a proportion of 0.06% of the total mercury content (Matsuyama et al., 2016). Fish caught around the bay have high methylmercury concentrations, at or near regulatory limits for human consumption in Japan of 0.4 mg/kg (Balogh et al., 2015), above concentrations found in fish caught at other sites in Japan.

The German submarine U-864 was sunk in WWII at a depth of 150 m carrying >60 t of elemental mercury. Surrounding sediments are contaminated with up to 108 g/kg mercury reducing to 1 mg/kg

at a distance of 100 m from the wreck. A study by Ndungu et al. (2016) found that the majority of the mercury was retained in the surface 10 cm of the sediments and a very small proportion, <0.05%, was in a methylated species. A mesocosm experiment was run with collected contaminated sediments over 6 months to determine to methylation rates, which found that a low sediment organic matter content (~1%) limited the methylation rate of mercury. There was a strong positive correlation between sediment organic matter and both organic and inorganic mercury flux to overlying waters - methylation rates increased from 0.01 to 0.59 ng g<sup>-1</sup> d<sup>-1</sup> in sediment amended with 1.18% organic carbon at 8–10 °C. However, in a field assessment of mercury bioaccumulation in caught fish and crabs, it was found that there was no significant bioaccumulation of mercury compared to fish and crabs from sites further away in the North Sea. The low methylation rate may also relate to the sediment temperatures. Temperature is related to microbial activity and is positively correlated to methylation rates (Matsuyama et al., 2018), though has less of an influence than sediment organic matter (Dai et al., 2021).

### 3.7.3 NORM

The pathways leading from a radionuclide source to organisms will be specific for the radionuclide and release scenario. The pathway will also be influenced by environmental conditions, radionuclide speciation and radionuclide composition which itself is variable due to radioactive decay and ingrowth (Section 3.4.4). Different NORM-contaminated products such as scales or sludges may have markedly different environmental chemistries and bioavailabilities.

Radionuclides may cause harm through external or internal irradiation, and so exposure pathways should consider both scenarios (Figure 3.26). General considerations include:

External irradiation:

- a) From the source material (i.e. scale or sludge in pipeline)
- b) Dissolved in the external seawater media
- c) Deposited to the sea-floor sediments
- d) Suspended in particulate matter
- e) Adsorbed to the surface of an organism

Internal irradiation

- a) By internalisation of the external media (e.g. ingestion of seawater, diffusion through gills, filtration of particulate matter)
- b) By consumption of radionuclide-contaminated food sources (i.e. other organisms in the trophic food chain)

Radionuclides in pipelines or in environmental media may contribute a radiation dose through external exposure in the absence of or in addition to an internal exposure. Generally, the contribution of external radiation doses to total doses are low in marine ecosystems. This is because of the shielding effect of water (which is approximately 1000x denser than air) (Goudriaan, 2019). The shielding effect of the pipe material should also be considered. As discussed in Section 3.5.3.3, very few radionuclides produce gamma radiation at meaningful emission rates that can penetrate steel thicknesses representative of pipelines. This means that the environmental partitioning and bioavailability of radionuclides will be significant predictors of total dose rate to organisms. These may be considered by the speciation of the different radionuclides in seawater conditions.

### 3.7.3.1 Radionuclide speciation in the marine environment

NORM occurs in the marine environment, with input from estuarine rivers or atmospheric deposition, *in-situ* production from radioactive decay, leaching from radionuclide-bearing solid phases like sediments and submarine groundwater discharge. Most studies investigating NORM radionuclides use them as radiotracers for oceanographic processes including particulate transport and to understand past environmental conditions. Data investigating the behaviour of NORM-contaminated products from oil and gas infrastructure in the marine environment is limited, with only 2 studies considering  $^{226}\text{Ra}$  and  $^{228}\text{Ra}$  (MacIntosh, 2020). Other studies investigating NORM are limited to terrestrial operations (Ali et al., 2019; Nelson et al., 2015b).

Radionuclides will undergo chemical or physical transformations leading to their partitioning in different phases. Each radionuclide present will have a unique chemistry and half-life that determines its behaviour in the environment. Of the radionuclides present in the  $^{232}\text{Th}$  and  $^{238}\text{U}$  decay chains, only radium, radon, bismuth, polonium, and lead are long-lived and reported in oil and gas pipelines (Section 3.4.1) and are reviewed below.

#### Radium (Ra)

Radium is the main NORM-component of contaminated scale in oil and gas infrastructure. Radium has two isotopes relevant to NORM contamination in oil and gas infrastructure:  $^{228}\text{Ra}$  and  $^{226}\text{Ra}$  are daughter products in the  $^{232}\text{Th}$  and  $^{238}\text{U}$  decay series.  $^{224}\text{Ra}$  ( $t_{1/2} = 3.6$  d) is a short-lived progeny of the  $^{232}\text{Th}$  decay series. Radium is highly soluble in production fluids, unlike its progenitors  $^{238}\text{U}$  and  $^{232}\text{Th}$  which remain in sediments or rocks. When mixed with high concentrations of sulfate anions (such as from seawater that is injected in oil and gas basins) and exposed to changes in pipeline pressure and temperature, alkaline earth metals including radium will rapidly precipitate with sulfate to form sulfate mineral scale (Ali et al., 2019). These may be specific minerals such as radium sulfate or co-precipitations with other metals including barium sulfate and strontium sulfate. Radium may also precipitate with carbonate, or co-precipitate with calcium in carbonate minerals (McDevitt et al., 2019). However, these are more common in freshwater and alkaline environments where sulfate concentrations are low (Arneodo et al., 2020).

In oxic environments typical of seawater, radium will have low solubility and remain in the solid phase in sulfate minerals. This has been observed with radium in produced waters disposed to seawater which rapidly form sulfate precipitates and sediment in surrounding areas (Dowdall and Lepland, 2012). Sulfate minerals are not stable in anoxic conditions and will be released to the aqueous phase by reductive dissolution (Huck et al., 1989). This process may be enhanced by microbial activity and environmental factors that increase microbial respiration rates, such as high organic matter contents (Falkner et al., 1993). This process may be related to the reduction of sulfate to sulfide species leading to an under saturation of sulfate (Torres et al., 1996). These fluxes are not expected to persist far into oxygenated seawater but the re-precipitated sulfate phases are likely to have very small particle sizes and may be more susceptible to current-driven dispersion or accumulation by filter-feeding organisms such as some corals, molluscs, and polychaetes (Lepland et al., 2000).

#### Radon (Rn)

Radon is a short-lived noble gas that is produced in the  $^{232}\text{Th}$ ,  $^{238}\text{U}$ , and  $^{235}\text{U}$  decay chains as the isotopes of  $^{220}\text{Rn}$ ,  $^{222}\text{Rn}$ , and  $^{219}\text{Rn}$ , respectively. The  $^{222}\text{Rn}$  isotope ( $t_{1/2}$  of 3.8 d) from the  $^{238}\text{U}$  decay chain is the most consequential for NORM-risk in the oil and gas industry because of its long-lived  $^{210}\text{Pb}$  ( $t_{1/2}$  of 22.3 y) and  $^{210}\text{Po}$  ( $t_{1/2}$  of 138 d) decay products.

Radon is transported as a gas or dissolved in production fluids. Its solubility is dependent on both salinity and temperature, being greatest at lower salinities and temperatures with a partition coefficient of up to 0.45 at 5 °C and 0 ‰ decreasing to less than 0.05 at >50 °C and >250 ‰ (Ye et al., 2019). Radon solubility is >200 times greater in organic liquids such as petroleum hydrocarbons than water (Clever and Battino, 1979). Taken together with its physical and chemical properties, this explains why radon is regularly associated with gas infrastructure downstream of separation.

### **Polonium (Po)**

Polonium displays similar chemical behaviour to sulfur, selenium, and tellurium. It is formed from the decay of  $^{222}\text{Rn}$  in the  $^{238}\text{U}$  series. Given their common source and known proportions in secular equilibrium,  $^{210}\text{Po}$ - $^{210}\text{Pb}$  ratios are commonly used as radiotracers in the marine environment to investigate particulate organic carbon export and remineralisation (Cresswell et al., 2020).

Polonium exists in a Po(IV) and Po(II) oxidation state in natural conditions with Po(IV) being dominant in all but the most acidic environments. Its main speciation in standard seawater conditions will be a  $\text{H}_2\text{PoO}_{3(\text{aq})}$  (Ram et al., 2019); however, adsorption to particulate organic matter and suspended solids, and incorporation into phytoplankton are more important controls on its mobility (Zhong et al., 2019). Because of its adsorption to suspended particles, the concentration of Po is depleted in surface waters and elevated in benthic environments (Jones et al., 2015). In anoxic sediments,  $\text{Po}_{(\text{s})}$  will be generated or in the presence of pyrite,  $\text{PoS}_{(\text{s})}$  (Ram et al., 2019). Polonium is resupplied to overlying waters, following the reductive dissolution of Fe and Mn oxyhydroxides (Benoit and Hemond, 1990), or through organic matter degradation (Ram et al., 2019). Polonium species remobilised from benthic sediments have been shown to be persistent in oxygenated waters (Jones et al., 2015), likely reflecting the formation of its  $\text{H}_2\text{PoO}_{3(\text{aq})}$  species in seawater and adsorption to particulate organic matter and other mineralised suspended solids (Bacon et al., 1988; Cherry and Heyraud, 1982). This has meant its concentration is elevated in marine bottom waters on particulate matter compared to Pb (Jones et al., 2015). Polonium speciation is expected to be dependent on temperature (as well as pH and Eh), but limited experimental data exists to confirm predictions (Ram et al., 2019).

### **Lead (Pb)**

Lead is only sparingly soluble in seawater and exists in the stable oxidation state Pb(II). Lead readily binds to organic matter and inorganic phases of iron and manganese hydroxides which reflects its dominant speciation in seawater (Bam et al., 2020). The minor component that may be dissolved will form carbonate or dissolved-organic species (Angel et al., 2015). Due to its propensity to bind to particulate matter and known relationship with  $^{210}\text{Po}$ ,  $^{210}\text{Pb}$  is used as a radiotracer to study oceanographic processes (Carvalho, 2011).

### **Bismuth (Bi)**

In the marine environment, bismuth exists in a Bi(III) oxidation state as the neutral hydrolysed  $\text{Bi}(\text{OH})_3^0$  species (Byrne, 2002). Three bismuth radioisotopes are produced in the  $^{238}\text{U}$  and  $^{232}\text{Th}$  decay chains,  $^{214}\text{Bi}$ ,  $^{212}\text{Bi}$ , and  $^{210}\text{Bi}$ , but are all short lived ( $t_{1/2} < 6$  d). Bismuth is rapidly adsorbed by particulate matter including organic detritus and iron or manganese oxides (Fowler et al., 2010). Given its short half-life and particle reactivity, bismuth is not a significant environmental risk at the concentrations expected from NORM-contaminated products. Rather, it is a short-term radionuclide in the decay chains that contributes beta and gamma radiation to the total activity levels. The gamma

emissions from  $^{214}\text{Bi}$  and  $^{212}\text{Bi}$  are used to detect their longer-lived parent radionuclides  $^{226}\text{Ra}$  and  $^{228}\text{Ra}$ , respectively.

### 3.7.3.2 Exposure pathway of radium-contaminated scale

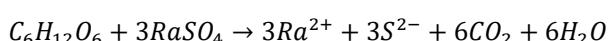
Radium-contaminated scales are insoluble in oxic marine conditions so exposure pathways will depend on environmental transformations such as sulfate reduction reactions or the leaching and emanation of its decay products.

The solubility of radium-contaminated scale is dependent on the solubility of the scale material (e.g.  $\text{BaSO}_4$ ,  $\text{SrSO}_4$ ,  $\text{CaCO}_3$ , etc.) or the radionuclide species itself (e.g.  $\text{RaSO}_4$ ,  $\text{RaCO}_3$ ,  $\text{PbCO}_3$ , etc.) which is in turn dependent on environmental conditions such as temperature and pressure (Monnin, 1999). Radium may exist as either radium sulfate or co-precipitated with barium sulfate which have similar solubilities (IAEA, 2014a; Matyskin, 2016). Carbonate and corrosion-based scales may also be a component of the overall scale matrix but will not have significant radium concentrations, relative to sulfate scales, given the lack of a mechanism to concentrate radium.

In pure water, the solubility products ( $\log K_{\text{sp}}$ ) for  $\text{RaSO}_4$  and  $\text{RaCO}_3$  are approximately -10.3 and -8.3 (Langmuir and Riese, 1985). That is, radium sulfate is 100-times less soluble than radium carbonate but neither are particularly soluble. These products are higher in seawater at approximately -8.5 and -5.5, but are still not particularly soluble (Matyskin, 2016). Background concentrations in seawater are much lower than these solubility limits. For example,  $^{226}\text{Ra}$  has been measured at dissolved concentrations ranging from  $\sim 3 \text{ fg/L}$  to  $\sim 80 \text{ fg/L}$  (IAEA, 2014a) across various ocean basins (Foster et al., 2004).

Leaching experiments of radium in sulfate scales with deionised water found that  $1.6 \pm 0.3\%$  of the total radium content was soluble (Ghose and Heaton, 2005). The leachable fraction was not affected by particle size, nor leachate pH across a range of 5-11. No publicly available studies have leached a barium/radium sulfate scale in seawater. However, based on the solubility products of radium described above, leached concentrations are expected to be orders of magnitude lower than in deionised water.

Sulfate-based scales will become soluble in anoxic sediments due to the microbial reduction of sulfate ( $\text{SO}_4^{2-}$ ) to sulfide ( $\text{S}^{2-}$ ) which will release radium to pore waters in a process termed dissolution (McCready et al., 1980; Phillips et al., 2001). The generalised chemical equation of this process is given in Equation 3.1 (Bowles et al., 2014).



Equation 3.1

This process has been well studied in radium and barium contamination from uranium mining or nuclear operations in terrestrial and freshwater environments but less so for marine sediments, where studies are limited to those investigating oceanographic processes or geological processes (Le Roy et al., 2018). Globally, rates of sulfate reduction in marine sediments are estimated to be  $10^{-5}$ - $10^{-1}$  mmol/cm $^2$ /y around Australia and will be influenced by the sediment organic matter content and the activity of the benthic microbial community. These factors are generally greatest in shallow inner-shelf environments and decrease along a depth gradient to the abyss (Bowles et al., 2014).

## Radium-contaminated scale

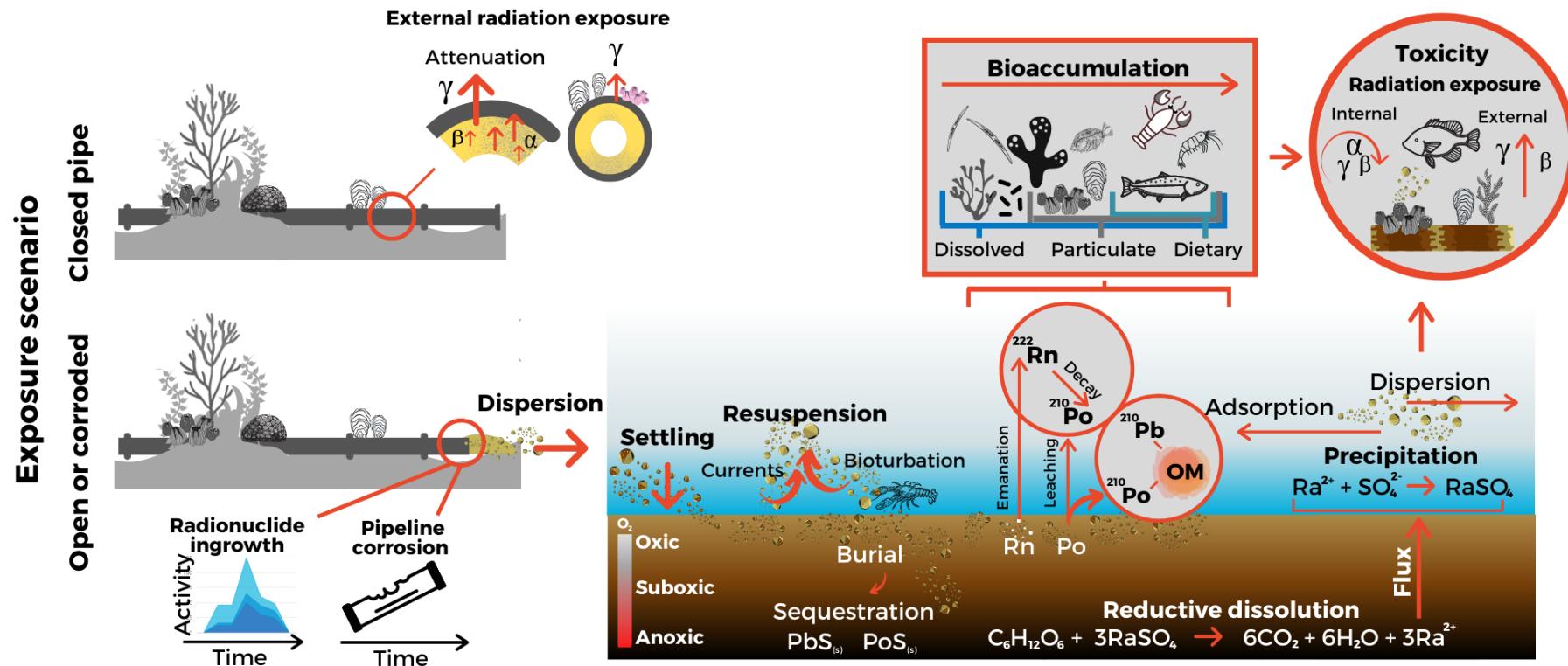


Figure 3.26 NORM exposure pathways and environmental transformations in the marine ecosystem. Two exposure scenarios are considered, where the pipe is closed and where the pipe is open to the surrounding environment. Only some radionuclides from the  $^{226}\text{Ra}$  and  $^{228}\text{Ra}$  (radium) decay series are reported as they are the mobile or long-lived decay products. They include Rn (radon), Pb (lead), and Po (polonium). Particulate organic matter (POM) represents sediment and marine-based organic matter. Radiation types are represented as their Greek character being alpha ( $\alpha$ ), beta ( $\beta$ ), and gamma ( $\gamma$ ).

Radium dissolution in anoxic sediments may lead to pore-water exposures of benthic organisms such as polychaetes and molluscs which reside below the sediment surface (Figure 3.26). Dissolved radium transported to overlying seawater will be rapidly oxidised and co/precipitated into a sulfate mineral. For example, studies investigating the disposal of produced waters containing radium and barium to seawater measured barite precipitates that had an average particle size of approximately 8 µm (Eriksen et al., 2006). The generation of these particles will be more mobile than hard scales and thus more accessible to dietary exposure paths.

Polonium is more soluble than radium in oxic marine waters so may be selectively leached from scales (Ram et al., 2019). This will likely be enhanced in environments with increased organic matter contents, as adsorption to particulate organic matter is a significant control on polonium mobility (Ram et al., 2019). Lead is generally insoluble in oxic marine water but may also form metal-organic complexes leading to some dissolution. Radon is an unreactive gas that is also soluble in seawater. White and Rood (2001) reported that up to 30% of radon emanating from radium-contaminated scale could escape the scale matrix. However, this reflects scale collected from land-based oil and gas pipelines. The emanation of radon from scale may lead to continuous flux of radon to pore and overlying waters potentially dispersing other decay products including  $^{210}\text{Po}$  (Figure 3.26).

### 3.7.3.3 Exposure pathways of lead and polonium films

Pipelines may be contaminated with  $^{222}\text{Rn}$  decay products in the absence of radium where pipeline conditions do not support the formation of radium-contaminated scales or where gas streams are separated from other phases, such as in downstream gas pipelines (Section 3.4.3). Of the  $^{222}\text{Rn}$  progeny  $^{210}\text{Pb}$  has the greatest half-life, which will form a secular equilibrium with the only other long-lived radionuclide  $^{210}\text{Po}$  (Figure 3.12). From the point operations cease,  $^{210}\text{Pb}$  will start to establish a new equilibrium increasing the product's radioactivity for 1.5 years to approximately 2.8-fold greater than its starting activity before decreasing at a rate proportional to its half-life. At 36-years after operations ceased radioactivity will have reduced to its starting activity and after 200 years will have reduced to 1% of its starting activity (Figure 3.15). This means that the temporal-extent of the  $^{210}\text{Pb}$  contamination hazard may be mitigated by the pipeline degradation times, depending on the time to pipeline breakthrough.

The speciation of  $^{210}\text{Pb}$  and  $^{210}\text{Po}$  in pipelines is not well understood.  $^{210}\text{Pb}$  may co-precipitate with elemental lead or lead sulfide in soft to medium-hard films or depositions (Schmidt, 2000). In oxic marine environments, lead is insoluble but may adsorb to particulate organic matter, inorganic iron and manganese oxides, or precipitated as solid lead carbonate minerals. In anoxic sediments, lead mobility will be controlled by reactive sulfide concentrations (Zhang et al., 2014). However, the mobility of lead in marine environments is low because of its poor solubility. For example, approximately 50% of lead was in a reducible phase and 40% in a residual (immobile) phase in contaminated marine sediments (Gao and Chen, 2012). However, resuspension of sediments may lead to short-term (~6 h) localised increases to dissolved lead concentrations followed by slower (~5 d) adsorption and ionic exchange processes reducing dissolved concentrations (Dang et al., 2020). This process was explained as the main mechanism that caused increased lead tissue concentrations in mussels around a contaminated harbour (Dang et al., 2015). This means that lead will only affect local organisms interacting with sediment surfaces or in pore waters.

Polonium behaviour in the marine environment is not well understood. It is known to bind to organic matter and may be efficiently accumulated by microorganisms (Heyraud and Cherry, 1979). Trophic transfer and potentially alkylation reactions leading to organo-polonium species may enhance

polonium mobility (Momoshima et al., 2001). This mobility may contribute to the reported bioavailability of  $^{210}\text{Po}$  which is accumulated to a much greater extent than  $^{210}\text{Pb}$  in marine ecosystems (Sirelkhatim et al., 2008).

The exposure pathways of lead and polonium originating from  $^{210}\text{Pb}$  films are equivalent to the pathways presented for lead and polonium in Figure 3.26 originating from  $^{226}\text{Ra}$ -contaminated scale.

### 3.7.4 Pipeline decommissioning options and degradation rates

Section 3.1.3 provided an overview of the potential decommissioning options which may enable the release of NORM and mercury contamination products to the marine environment or allow seawater or sediment pore water to ingress into the pipeline. The conceptual models illustrated in Figure 3.24 and Figure 3.26 are based on point source release in the benthic and epibenthic zones. They are not intended to address all the potential decommissioning option contaminant release scenarios, for example the exposure as a section of pipeline is cut and lifted through the water column to a construction vessel.

However, these conceptual models could be utilised for assessment of a release when a pipeline is cut on the seabed, or to assess the impact of an uncapped pipeline, or pipeline corrosion degradation along its length provided there were a means to quantify the location, extent and time scales of the seawater and pore water ingress.

#### 3.7.4.1 Seawater and sediment pore water breakthrough

Pipelines are the contaminant sources for COPCs considered in this review. Therefore, their degradation extent and time scales in the marine environment are an important consideration for COPC release and resulting exposure paths. This is either through the direct contact with marine environment from a breach in pipe wall (i.e. initial seawater or sediment pore water break through assuming pipeline is capped) or the eventual total breakdown of the pipe wall with the potential of releasing all contaminant products to the marine environment.

For NORM this is an important consideration in the exposure pathway because of the radioactive decay and radionuclide ingrowth of NORM-contaminated products (discussed in Section 3.4.4). All radionuclides decay at a rate proportional to their activity or half-life (Figure 3.12), this means the radiation risk will decrease after sufficient time. However, radionuclide ingrowth means that decay products of parent radionuclides will contribute to activity levels, and therefore the radiological risk, as discussed in Section 3.4.4. After operation cessation,  $^{226}\text{Ra}$ -contaminated scales will pose a long-term risk ( $\sim 5000$  y) with a peak in radiation occurring 120 y after scale formation while  $^{210}\text{Pb}$ -contaminated films will pose a shorter-term risk ( $\sim 36$  y) with a peak in radiation 2 y after film formation (Figure 3.15). This means that for seawater breakthrough times  $> 100$  years, the activity levels  $^{228}\text{Ra}$  and  $^{210}\text{Pb}$  contaminated products will be reduced significantly, while  $^{226}\text{Ra}$  contaminated products will be near the peak of its radiation ingrowth and therefore activity level.

From a mercury perspective the assumption is the contaminated products within the pipeline will always be present at the levels defined at the commencement of decommissioned life (i.e. post cleaning and capping) and so the exposure pathway is triggered at the point of seawater or sediment pore water breakthrough.

### 3.7.4.2 Pipeline degradation timescales

Flowlines and pipelines used in subsea production and offshore export systems are designed to prevent hydrocarbon release to the environment over their defined operating envelop (i.e. maximum pressure and temperature, hydrocarbon compositions etc.) and design life (i.e. typically 25 years). The subsea pipeline design standards are well established (DNV-GL, 2017), a key requirement of which is an assessment of internal and external corrosion rates which drives material (e.g. carbon steel, corrosion resistant alloy) and coating systems selection and the cathodic system design.

Importantly for the current study the assessment of degradation rate after the cessation of production is not a current design consideration and is therefore uncertain. While this has been identified as an important consideration for pipeline decommissioning from a liability or design criteria perspective (Chandler et al., 2017), there are no current scientific publications quantifying decommissioned pipeline degradation rates over the timescales associated with NORM or mercury exposure to the environment.

Rigid pipelines can be segregated by material type, either carbon steel or a Corrosion Resistant Alloy (CRA) for example 13% Cr, 22% Cr (Duplex) or 25% Cr (Super Duplex) with a range of coating systems. The other main category is flexible unbonded pipe which comprises multiple steel and thermoplastics layers, the inner carcass of which is typically CRA. As an illustration of degradation performance for these pipeline system post decommissioning, we will refer to two recently tabled decommissioning environmental plans (Santos Limited, 2020; Woodside Energy Ltd, 2020a).

Figure 3.27 presents the degradation timescales of a CRA rigid pipeline from a gas condensate service suggesting a range for seawater or sediment pore water breakthrough of hundreds to over a thousand years but this is very difficult to determine precisely from the figure. What appears to be primary degradation drivers are the extent of burial and the coating system breakdown but the basis, assumptions and details of this model are not available to verify the time scales quoted. It might also be reasonable to assume, given it is a 13% Cr pipeline, that flexible unbonded pipe would exhibit a similar or better performance.

From a  $^{210}\text{Pb}$  from  $^{222}\text{Rn}$  progeny perspective, these timescales would potentially mitigate radiological risk. However, the risk from  $^{226}\text{Ra}$  may still require assessment if scale cannot be discounted for a raw well stream gas service.

Figure 3.28 is an example of a carbon steel rigid pipeline from a dehydrated gas export service condition. While we do not have access to the modelling basis and key assumptions, it would appear that seawater and sediment pore water break though is likely to occur at field joints (i.e. where there is no internal flow coating making it susceptible to internal corrosion and the external coating is more likely to breakdown) and at a time scale of less than 100 years to potentially hundreds of years. Burial along with cathodic protection depletion is the initial driver for degradation followed by internal and external coating breakdown. Similarly with the 13% Cr pipeline, it is very difficult to precisely determine the time at which seawater or sediment pore water would breach the carbon steel pipe wall.

While in this instance (i.e. dry gas service) the risk from  $^{228}\text{Ra}$  and  $^{210}\text{Pb}$  contaminated products will be reduced, if this were a carbon steel pipeline in a service susceptible to scale the exposure to  $^{226}\text{Ra}$  contaminated products might be material. For both these examples there is insufficient information to form a definitive position but illustrates a knowledge gap in the peer reviewed literature.

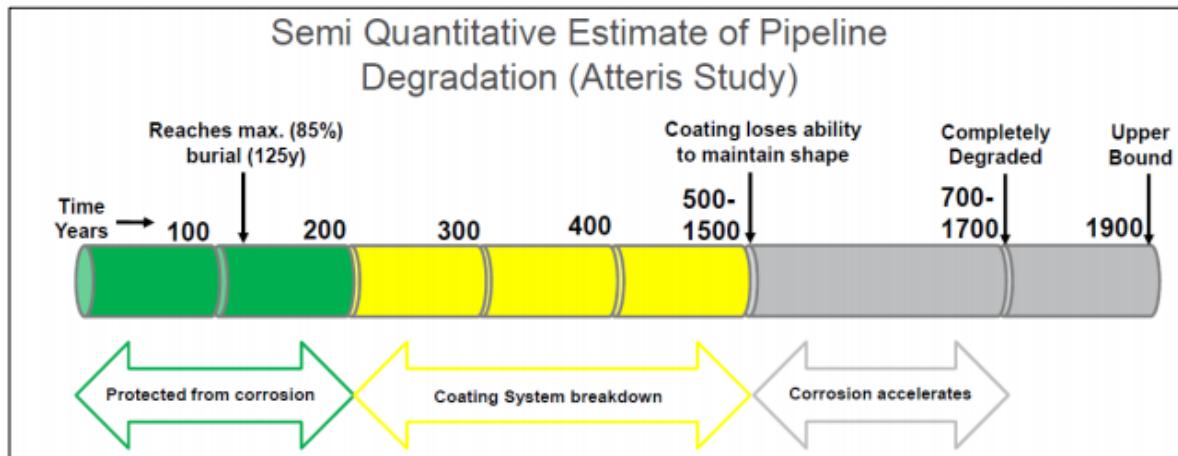


Figure 3.27 Echo Yodel pipeline degradation timeline estimate - left *in situ* post decommissioning. Pipeline is diameter 324 mm x 16.9 mm wall thickness x 23 km, 13% Cr, cathodic protection, capped and filled with seawater mixed with biocide, oxygen scavenger and corrosion inhibitor, multiphase gas service connecting subsea wells to offshore facility in the Carnarvon basin (Woodside Energy Ltd, 2020a).

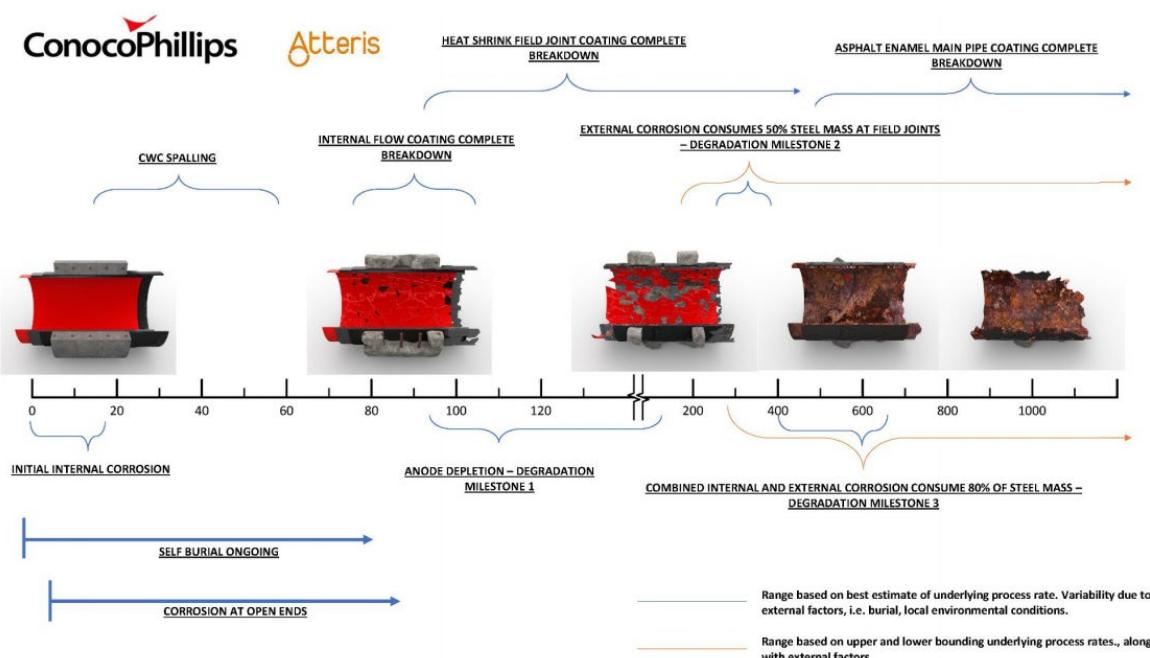


Figure 3.28 Bayu-Undan export pipeline degradation timeline - left *in situ* post decommissioning. Pipeline is diameter 660 mm x 20.1 mm wall thickness x 346 km, Carbon X65, internal epoxy flow coat 75 µm, 3 layer polypropylene/5 mm asphalt enamel external coating, external concrete weight coating 40 to 110 mm, assumed up capped and filled with seawater, dry export gas service from offshore facility to onshore LNG plant in the Bonaparte basin operating since 2005 (Santos Limited, 2020).

## 3.8 Hazard characterization

Hazard characterization is a step in an ecological risk assessment that identifies the contamination levels at which harm to ecological receptors begin to be observed and may also be called effects analysis, impact assessment, or effects characterization (Suter II, 2006). This section consolidates what is known about the impacts of mercury and NORM to exposed ecosystem receptors and corresponding threshold levels for toxicity, biodiversity, and bioaccumulation lines of evidence. In doing so, this section partly meets objective 1 and 4 and problem statement 1.

### Key findings:

- The assessment of multiple impacts (i.e. toxicity, impacts to biodiversity, and bioaccumulation) from a hazard is considered best practice in ecological risk assessments.
- Radiation hazards need to be assessed by their absorbed dose rate to organisms. Radiation risk assessment tools exist to relate radionuclide specific activity to dose rates. More research is needed to investigate the applicability of these tools to Australian marine conditions and organisms.
- Environmental reference levels are used to understand how contaminant concentrations relate to potential hazard impacts. There are water quality guidelines and sediment quality guidelines for mercury and lead, and dose-rate reference levels to assess toxicity impacts.
- Mercury and polonium bioaccumulate and potentially biomagnify in food webs. A generalised environmental reference level for bioaccumulation exist for mercury, but not for polonium.
- Assessing hazard impacts to biodiversity requires the development of site-specific environmental reference levels.

### 3.8.1 Hazard impacts in ecological risk assessments

A hazard may have multiple potential impacts to an ecosystem receptor (Suter II et al., 2017). In an ecological risk assessment, all these hazards should be assessed to give a holistic understanding of potential impacts. Hazard impacts often include:

- Biodiversity – measured changes to populations and communities of organisms
- Biomarkers –such as the bioaccumulation of a contaminant in a representative organism
- Toxicity – measured response of an organism to a contaminant (where causality is defined)

Hazard impacts to ecosystem receptors are assessed using lines of evidence, which are measured by defined indicators relating measurements of physical, chemical, biological or socio-economic parameters to impact. Lines of evidence are often accompanied by environmental reference levels (which may be numerical guidelines, a reference condition, or a predetermined qualitative scale). These lines of evidence (represented by indicators) are typically integrated into a weight of evidence framework with an understanding of the contaminant exposure to determine risk (Chapman et al., 2002). The risks can then be considered against the management objectives (and legislative requirements) to assess whether they have been met.

Environmental reference levels are developed to simplify the hazard impact analysis step in an ecological risk assessment (Section 3.8.3). They typically describe the contaminant concentration in an environmental medium below which limited to no impact is expected to occur (Warne et al., 2018). Where reference levels are not available, such as where not enough ecotoxicological data for a contaminant exists to derive one or where the impact to an ecosystem receptor is based on comparisons to reference or background environmental qualities then a site-specific guideline can be derived (examples are discussed for bioaccumulation reference levels in Section 3.8.3).

Exceedance of an environmental reference level for one measurement of a hazard impact does not necessarily indicate unacceptability of the hazard impact. Rather, should trigger a more detailed assessment of the hazard impact. Default environmental reference levels are typically overly conservative and are not intended to consider the site-specific physical chemistry that is known to modify contaminant bioavailability and risk. This approach forms the basis of tiered assessments for hazards.

Common lines of evidence for contaminant impacts in the marine ecosystem are discussed further below.

### 3.8.2 Lines of evidence

#### 3.8.2.1 Chemistry line of evidence

Chemistry lines of evidence are used to relate the measured presence of a contaminant in an environmental media or contaminated product to expected harm using known concentration-response relationships. Ecotoxicological tests are typically conducted in laboratory settings to control or investigate environmental parameters that may modify toxicity. These tests expose increasing concentrations of a known contaminant to an organism and measures the physiological response (Depew et al., 2012). The concentration where significant toxicity is observed is a threshold concentration. If enough threshold concentrations can be derived for representative organisms (ideally greater than 8 across 4 taxonomic groups) then a sediment or water quality guideline can be derived using a probabilistic model such as a species sensitivity distribution (Warne et al., 2014). This differs to toxicity lines of evidence where the contaminated environmental media is used in an ecotoxicological assay to directly investigate toxicity.

Chemistry lines of evidence relate measured contaminant concentrations to known concentration-response relationships which are typically based on laboratory exposures. This means that environmental factors that may influence toxicity are not investigated. These may include adsorbents such as particulate matter, water chemistry that can change contaminant speciation and thus bioavailability, and so on. This means that different measures of the contaminant (such as the total fraction, the fraction extractable using weak acids, or the bioavailable fractions only) or environmental parameters can be used in a tiered assessment framework to progressively investigate the most relevant fraction (at increasing effort).

For a chemistry line of evidence, radionuclides are considered differently. Toxicity arising from radiation is measured against the absorbed radiation dose, expressed as a dose rate, to an organism. This is calculated using a range of partitioning and dosimetry models which is discussed in more detail below. However, once a dose rate has been calculated, dose-rate reference levels can be used as environmental reference levels in the same way that water and sediment quality guidelines are used.

## Mercury

Mercury is highly toxic to marine organisms and the magnitude of effect on marine ecosystems depends on the mercury speciation, concentration, and exposure duration (Wiener, 2013). Mercury's mechanism of toxicity in higher-order taxonomic groups is through neurotoxic effects. Observations of marine medaka (*Orizys melastigma*) exposed to mercury chloride indicate that inorganic mercury may cause neurotoxicity by inducing oxidative stress, cytoskeletal assembly dysfunction and metabolic disorders (Wang et al., 2015). Mercury may also induce morphological changes in the brain including the number and volume of neurons and glial cells, which in turn may induce changes in swimming behaviour; and long-term neurological effects (Wang et al., 2015).

Mercury toxicity is life-stage dependent with embryonic and larval stages more sensitive to mercury than adult stages. For example, toxicity tests on red sea bream *Pagrus major* suggested mercury concentrations exceeding 20 µg/L can reduce hatching success, increase mortality, and induce teratogenicity (a disruption of development) in both embryo and larvae (Huang et al., 2011). Other impacts resulting from mercury exposure relate to malformations to the spine, bladder, head and fins with these deformities leading to reduced chances of survival (Dong et al., 2016). Example toxicity thresholds (summarised for individual species at the phylum level) are given in Table 3.15.

In humans, mercury can have long term health implications including neurological damage, reproductive effects and increased cardiovascular risks. Humans are at risk from dietary exposure to bioaccumulated mercury in seafood, with methylmercury being the most bioavailable mercury species. Typical symptoms of methylmercury poisoning in humans (known as Minamata disease) included sensory disturbances, ataxia, dysarthria, constriction of the visual field, auditory disturbances and tremor were also documented. Foetuses can also be poisoned by methylmercury when mothers ingest contaminated marine life (Sakamoto et al., 2018). The US Natural Resources Defense states that fish with a mercury level of 0.3 to 0.49 mg/kg can be safely eaten 3 times a month without adverse impacts, but not during pregnancy.

Environmental reference levels have been developed for mercury in marine and sediment ecosystems. These combine individual toxicity thresholds for a range of species into a single value below which harmful ecosystem impacts are not expected (ANZG, 2018). Reported environmental references levels for mercury are given in Section 3.8.3.

**Table 3.15 No observed effect concentrations (NOEC, the maximum concentration where no toxicity is observed) of mercury for different taxonomic groups aggregated at the phylum level.**

Taxonomic Group	NOEC (µg/L)
Fish	7.4 to 160
Crustaceans	0.8 to 10
Echinoderms	4
Molluscs	0.12 to 10.14
Annelids	3.4 to 18
Algae	0.9 to 88

## NORM

The toxicity from radiation exposure can be evaluated by calculating the absorbed radiation (known as dose) from all radiation sources to an organism and comparing it to radiation toxicity studies or environmental reference levels (ICRP, 2007; UNSCEAR, 2008). Relating activity levels in contaminated products to absorbed doses in representative organisms can be data intensive and requires a good understanding of each radionuclides environmental partitioning and bioaccumulation potential. Dispersion, partitioning, wildlife transfer, and dosimetry models and databases have been developed by the ICRP (2008) and others to formalise this process. It has widely been adopted and implemented by the IAEA (2014c) and its member states including Australia (ARPANSA, 2015b). Figure 3.29 shows these steps and their accompanying models.

Given the limited environmental partitioning and transfer data for most radionuclides, organisms, and environment types, a number of assumptions and idealised parameters have been developed for these models. These make radiation hazard assessments possible with limited input data (e.g. environmental or contaminated product specific activities). However, introduces significant uncertainties (Brown et al., 2013).

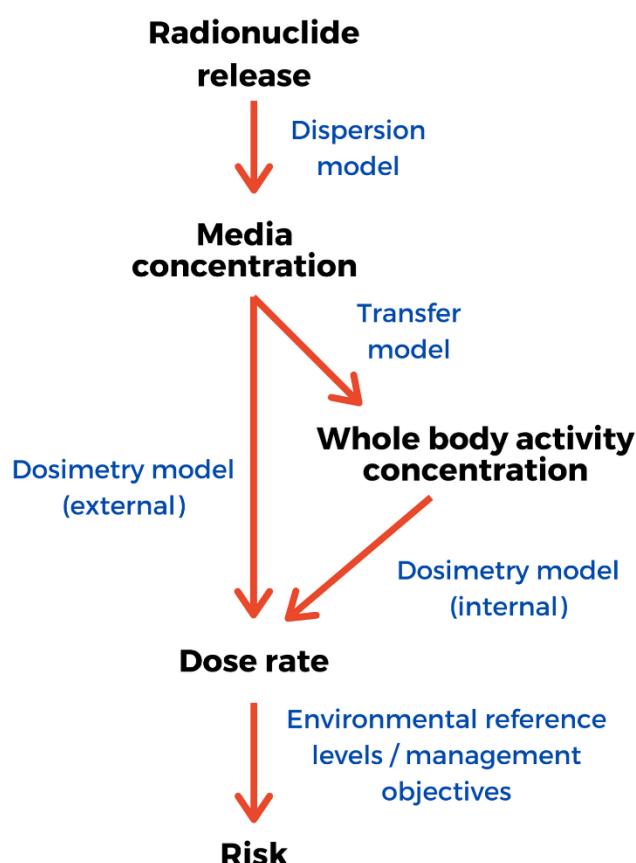


Figure 3.29 Components of a typical radiological risk assessment process. Adapted from Dr Nick Beresford, University of Salford.

The Environmental Risk from Ionising Contaminants: Assessment and Management (ERICA) integrated approach (Brown et al., 2008), developed through European Commission funded projects, is one method that has become widely adopted. Other tools exist, mostly developed by national jurisdictions but are based on the same assessment approach and produce generally equivalent assessments of risk based on intercomparison studies (Beresford et al., 2008; Johansen et al., 2012; Vives i Batlle et al., 2011).

#### [Environmental partitioning and media concentrations](#)

Environmental partition describes the equilibrium state of radionuclide distribution in environmental media. The environmental chemistry of some radionuclides in the marine environment has been discussed in Section 3.7.3. Given the complexities and limited studies explaining radionuclide behaviour in the environment, simplifications are used to relate the sediment to water radionuclide concentrations. In radiological assessment tools, the relationship between sediment and water radionuclide activity concentrations is described by a partitioning coefficient ( $K_d$ ) value, calculated by Equation 3.2.

$$K_d \left( \frac{L}{kg} \right) = \frac{\text{Sediment Concentration } \left( \frac{Bq}{kg} \right)}{\text{Water Concentration } \left( \frac{Bq}{L} \right)}$$
Equation 3.2

A range of  $K_d$  databases have been compiled for use in radiological risk assessments, including from the IAEA and ICRP (IAEA, 2004b; ICRP, 2009). These values introduce uncertainty into radiological risk assessments because they do not take into account different sediment conditions or the different sources of radionuclides to the environment.

#### [Wildlife transfer](#)

Radionuclide concentrations in organisms are required to calculate internal dose rates for radiological hazard assessments. Radionuclides in the environment accumulate to different extents in different organisms based on the organism's biology such as diet and habitat, and the chemical properties of the radionuclides that govern their exposure pathway and transfer in the environment and food webs (Whicker and Schultz, 1982). To simplify the various exposure pathways and radionuclide-environmental interactions governing transfer to different organisms (Salbu and Skipperud, 2009), whole-body concentration ratios (CR) of radionuclides can be calculated for different organisms (Equation 3.3).

$$CR = \frac{\text{Activity concentration in biota } \left( \frac{Bq}{kg} \text{ fresh weight} \right)}{\text{Activity concentration in media } \left( \frac{Bq}{L} \text{ or } \frac{Bq}{kg} \text{ dry weight} \right)}$$
Equation 3.3

CRs are used to relate radionuclide activity levels in the environment to whole-body activity levels in organisms. This allows for the modelling of radiation hazard where site-specific data are not available. This is the case for all proposed decommissioning scenarios where the release of radionuclides has not yet occurred. The Wildlife Transfer Database (Coppleson et al., 2013) has been developed of CR values that may be used to parameterise radiological assessment tools. However, these values may not reflect site-specific conditions or the unique biologies of local organisms (Hirth et al., 2017).

Furthermore, not all radionuclides have accompanying CR values for all organisms, which means other approaches or analogues must be used (Hosseini et al., 2008).

#### Dosimetry

The amount of radiation that an organism will absorb depends on the activity concentration of the radionuclide in the organism and its surrounding media, the size of the organism, the type of radiation being emitted, the energy of the radiation being emitted, and the distribution of the radionuclide within the organism. To undertake these calculations, biota is expressed by simple geometries and internal radionuclides are assumed to be homogenously distributed. The ICRP has developed 12 reference animals and plants models (ICRP, 2008), while the ERICA project has developed 36 reference organisms (Brown et al., 2008). These idealised geometries do not reflect any specific animal or plant but provide a mechanism that allows for the calculation of radiation dose rates in different organisms given their different sizes and their occupancy in different environment types (e.g. on soil, under water, in sediment, and so on). Recent research has investigated the use of voxelised phantom models based on 3-dimensional CT or MRI scans of organisms (Higley et al., 2015). These allow for heterogenous radionuclide distribution (reflecting that some radionuclides are accumulated to in specific organs) and so may reduce uncertainty but are not well adopted for routine ecological risk assessments.

A dose coefficient (DC), also known as a dose conversion coefficient (DCC) or dose conversion factor (DCF) (Hutchins and Bruland, 1998), is used to relate external or internal radionuclide activity concentrations to absorbed dose for a given reference organism geometry and size (ICRP, 2008). DCs have the units of Gy/h per Bq/kg (or Bq/L for liquids or Bq/m<sup>3</sup> for air). A DC accounts for the effects of radionuclide-specific radiation type and energy and organism size. Some DCs also include the contribution for short-lived radionuclide progeny. For example, the DCCs used by the ERICA tool include the contribution from progeny with half-lives <10 days long (Brown et al., 2008); the DCFs used by the US Department of Energy's RESRAD-Biota tool include progeny with half-lives <180 days long (US DOE, 2004); and the ICRP have recently published a list of DCs that do not include any radionuclide progeny, which will be adopted in future version of the ERICA tool (ICRP, 2017). The decision to use one DC value or another can lead to variation in the predicted dose rates and should be justified in any dosimetry model (Psaltaki et al., 2013).

#### Relating absorbed dose to harm

The harmful effects of radiation to non-human biota is studied by exposing organisms to increasing doses of radiation and measuring the resulting effects. Alternatively, where radionuclides have been released into the environment, such as the Chernobyl and Fukushima nuclear accidents (Fisher et al., 2013; Hinton et al., 2007), effects can be monitored in local organisms. These two approaches are methodologically difficult and limited in the ability to test the range of radionuclides that may be released to the environment. To aid environmental radiological risk assessment, a database of radiation effects data, FREDERICA, was developed (Copplestone et al., 2008). This remains the most up-to-date source of radiation effects data following updates from an IAEA project group (IAEA, 2014b). Similarly, a summary of radiation effects has been published by UNSCEAR (2008) which details the dose rates which lead to observed harm for broad taxonomic groups. For aquatic organisms these are 40–400 µGy/h.

Toxicity from radiation is dependent on the overall absorbed radiation dose, and so is not radionuclide specific. Some radionuclides may also exhibit chemical toxicity, but these data are limited and it is difficult to disentangle the toxicity from chemical interactions and toxicity from the radiation.

Three studies have investigated the toxicity of radium:

- Bioaccumulation and lack of oxidative stress response in the ragworm *Hediste diversicolor* following exposure to  $^{226}\text{Ra}$  in sediment (Grung et al., 2009).
- There were low measureable impacts to Atlantic cod (*Gadus morhua*) embryonic cells exposed to environmentally relevant doses of  $^{226}\text{Ra}$  (Olsvik et al., 2012).
- Individual and molecular level effects to nauplii and adult females of *Calanus finmarchicus* were observed in response to produced water contaminants, including  $^{226}\text{Ra}$  (Jensen et al., 2016).

These studies largely conclude that  $^{226}\text{Ra}$  is not toxic at environmentally relevant concentrations. However, toxicity may be conflated with additives such as scale inhibitors, or from co-precipitated metals such as barium (Spangenberg and Cherr, 1996).

#### [Radionuclide transfer in the environment](#)

Concentration ratios (e.g. environmental medium to organism ratios) are compiled in the Wildlife Transfer Database (Coppleson et al., 2013; Howard et al., 2013) that underpin radiological assessment tools such as ERICA and ICRP recommendations (Clement et al., 2009). The range of published CR values for NORM radionuclides such as U, Th, Ra, Po, and Pb is incomplete (Figure 3.30). There is limited data for polychaetes, crustaceans, sea anemones and true corals, and marine mammals, and limited data generally for the radionuclides U, Th, and Ra. A large number of CR values exist for the radionuclides Po and Pb and most organisms which likely reflects their use as a natural radiotracer, as discussed in the Section 3.7.3. However, these studies typically investigate background concentrations of radionuclides. Bioaccumulation may not be linear with respect to exposure concentrations.

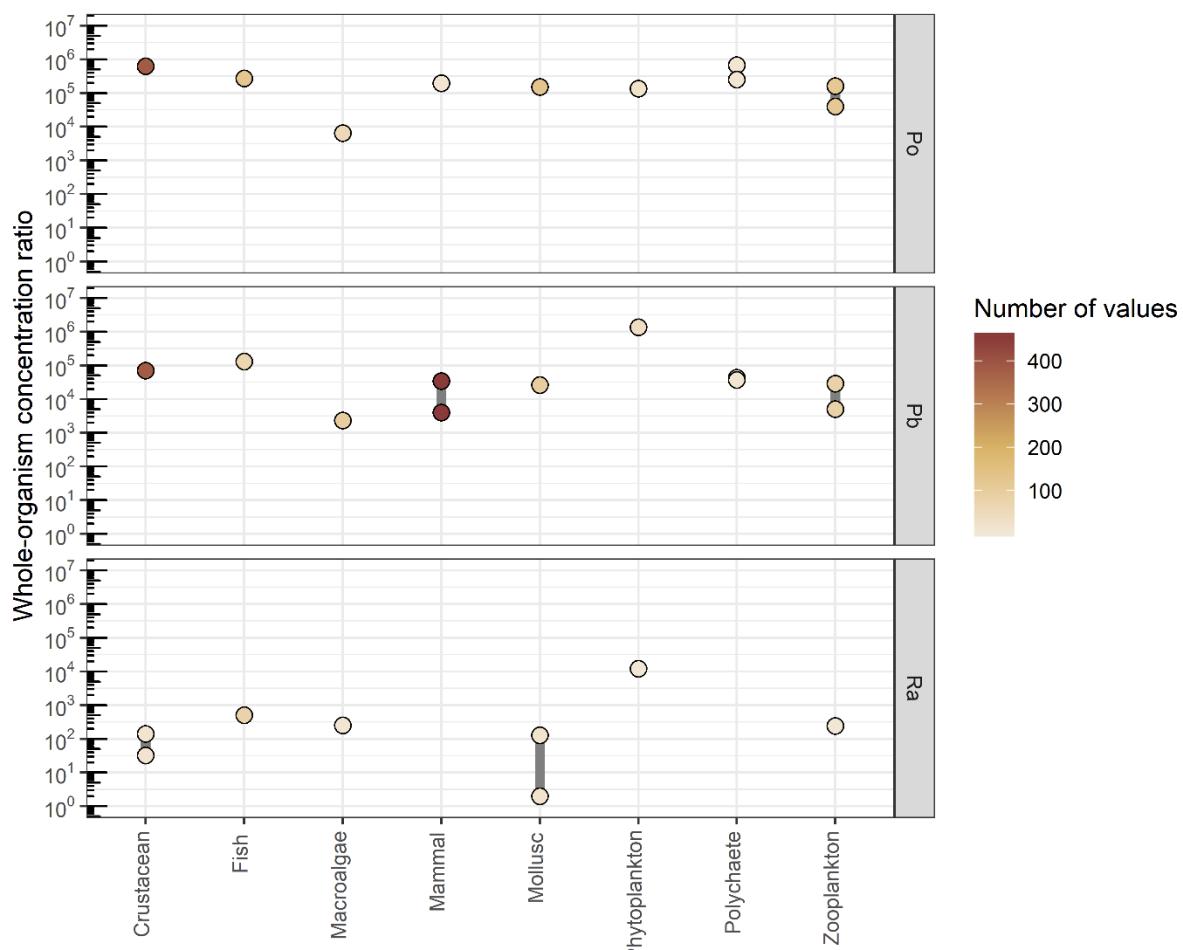
Different assessment tools are used to address missing data in different ways. In the ERICA tool, data may be derived by a range of methods including using CR values for similar organisms, using the CR value of a radionuclide with similar geochemistry, various modelling approaches, use of the highest available CR, and so on (Brown et al., 2013). These approaches add uncertainty to the overall risk assessment.

The Wildlife Transfer Database (WTD) contains marine CR values that have predominately been collected in Northern Hemisphere environments. The assumption that these CR values reflect Australian marine organisms is not well founded and has been tested in terrestrial environments. CR values from the WTD were found to be underestimations of Australian CR for the U-decay series in Australian organisms (Doering et al., 2019). It has also been reported that CR values were the input parameter contributing the greatest variability exposure models (Beresford et al., 2008; Johansen et al., 2012). For this reason, much of the terrestrial radiological risk assessment research has focused on developing Australian-relevant CR values. However, this is limited to Australian terrestrial ecosystems including the subtropical region around Ranger Uranium mine in the Northern Territory and the arid/semi-arid region around the Flinders Ranges in South Australia.

#### Radionuclide partitioning in the environment

Partitioning coefficients used in radioecology and dispersion modelling are based on a compilation of  $K_d$  values from IAEA (2004b), Figure 3.31. These may not reflect the real solubility and speciation for NORM products generated in the oil and gas industry, as NORM products are not defined as sediments or waters and at an equilibrium state. For example, a report investigating the amount of  $^{226}\text{Ra}$  and  $^{210}\text{Po}$  that can leach to seawater from NORM-contaminated products found that  $^{210}\text{Po}$  was much more soluble and radium was much less soluble than default  $K_d$  values suggest (Confidential Industry Report).

Different  $K_d$  values will significantly impact resulting radiation doses to organisms where site-specific measurements do not exist. Furthermore, the  $K_d$  values will change with the physicochemistry of different environments, including salinity, temperature, pressure, particle size, redox conditions, or the presence of complexing ions such as organic matter. The influence of these factors on radionuclide partitioning should also be investigated given the range of environments relevant to pipeline decommissioning activities.



**Figure 3.30** The whole-body concentration ratios of naturally occurring radioactive materials for between marine organisms and marine water as published in the Wildlife Transfer Database (Coppleson et al., 2013). Colour fill reflects the number of observations for each organism and radionuclide pair in the database and points are the arithmetic mean plus and minus one standard deviation. Where only one point is visible there was either too few data points to derive a standard deviation or the deviation is too small to be perceived on the chart.

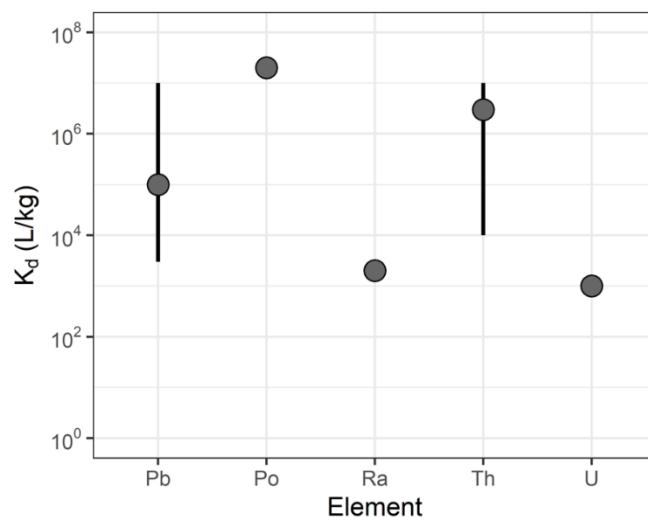
### Limitations of NORM hazard assessments

Radiological impact assessments for non-human biota rely on databases of parameter values that are skewed to northern hemisphere organisms and ecosystems and are incomplete for some organisms and radionuclides. Some parameter values simply don't exist for all radionuclides. For example, the  $K_d$  value for polonium, on the order of  $10^7$  (Figure 3.31), is based on the  $K_d$  value of a 'periodically adjacent elements' (IAEA, 2004b). Ram et al. (2019) report a likely  $K_d$  value on the order of  $10^5$ , which would suggest polonium is 100-fold more soluble than is currently being considered. A summary of radiation partitioning, transfer, and dosimetry parameters and the data gaps and limitations for NORMs and marine ecosystems are given in the following subsections and in Table 3.16.

#### Radiation Effects Data

Data on the effects of radiological impacts to non-human biota is limited. A recent assessment of the FREDERICA database, the most up to date source of radiation effects data to non-human biota (Coppleson et al., 2008), found that only 36% of the data reflect chronic toxicity studies (the highest quality data for guideline development), and that only one or two studies exist for each of the broad taxa of crustacea, zooplankton, and bacteria (Real and Garnier-Laplace, 2020). For ecotoxicological studies specific to NORM-contaminated products, only three exist for radium-contaminated scale (Grung et al., 2009; Jensen et al., 2016; MacIntosh, 2020; Olsvik et al., 2012).

More toxicity tests using representative local species would reduce the uncertainty in the derivation of environmental reference limits and may allow the derivation of site-specific guideline values. This is recommended under the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZG, 2018) and an approach adopted by other extractive industries, such as for uranium in the freshwater ecosystem (van Dam et al., 2017), radiation in the soil ecosystem (Doering and Bollhöfer, 2016) of the Kakadu National Park and for nickel in tropical fresh and marine ecosystems (Gissi et al., 2016; Stauber et al., 2020).



**Figure 3.31**  $K_d$  values for NORM elements reported in IAEA (2004b). Grey points represent the recommended  $K_d$  value, line segments represent other reported  $K_d$  ranges. Note that the value for polonium is based on an unspecified periodically adjacent element. The greater the  $K_d$  value the more partitioned the nuclide will be to sediments.

**Table 3.16 Model parameters used in radioecology and dosimetry models and their limitations.**

Model parameter	Use	Data source	Limitation	Reference
Dispersion models	Determine the spread of radionuclides in the environment from a point source	IAEA SRS-19 dispersion models	<ul style="list-style-type: none"> <li>• Not developed for ocean contexts, just a point source release from a coastline</li> <li>• Does not account for chemical transformations in the environment</li> <li>• Assumes non-changing water and sediment geometries and flow characteristics over time and space.</li> <li>• Assumes equilibrium between sediment and water partitioning</li> </ul>	(IAEA, 2001)
Partitioning coefficients	Relate radionuclide activity concentration in one environmental medium to another environmental medium	Published list of radionuclide partitioning coefficients	<ul style="list-style-type: none"> <li>• Assumes equilibrium between sediment and water partitioning</li> <li>• Does not account for chemical transformations in the environment and the influence of site-specific physicochemical characteristics</li> </ul>	(IAEA, 2004b)
Concentration Ratios	Summarise the range of exposure pathways and site-specific factors affecting radionuclide uptake into a value that relates environmental activity concentration to whole organism activity concentrations.	Wildlife Transfer Database – developed in support of the IAEA and ICRP and maintained by an international collaboration	<ul style="list-style-type: none"> <li>• High variability of CR values within radionuclides and species</li> <li>• Values are aggregated at a high taxonomic level so may not be relevant for the site</li> <li>• Does not account for local chemical, physical, and ecological processes that affect radionuclide bioavailability</li> <li>• Limited data or missing values for some marine species and radionuclides</li> </ul>	(Coppelstone et al., 2013; IAEA, 2014b)

<b>Model parameter</b>	<b>Use</b>	<b>Data source</b>	<b>Limitation</b>	<b>Reference</b>
Dose Coefficients	Relates internal and external radionuclide activity coefficients to the absorbed dose in a representative organism model	Use reference organisms or reference animals and plants numerical models to calculate the amount of radiation absorbed from each radionuclide given external and internal activity concentrations	<ul style="list-style-type: none"> <li>• Dose absorption based on homogenous geometrical shapes</li> <li>• Organ-specific accumulation not accounted in dosimetry models</li> <li>• Different dosimetry models incorporate different progeny of parent radionuclides</li> </ul>	(Brown et al., 2008; ICRP, 2017)
Toxicity data	Relates absorbed dose to the expected harm to an organism	UNSCEAR reports summarising published literature ecotoxicological studies.	<ul style="list-style-type: none"> <li>• Very few primary sources for radiation effects data in controlled exposure scenarios</li> <li>• Wide variability of effects in different species and taxa meaning high uncertainty in the effects range</li> <li>• Species not relevant to local ecosystems</li> </ul>	(UNSCEAR, 1994; UNSCEAR, 2008)

### 3.8.2.2 Bioaccumulation line of evidence

Contaminants that bioaccumulate pose a particular threat because low environmental media concentrations may accumulate in organisms to levels that can cause harm. Furthermore, contaminants that are able to be transferred efficiently from prey to consuming organism may cause harm through a process of bioconcentration in the food web. For these reasons, the bioaccumulation line of evidence serves to protect multiple community values including biodiversity, commercial fishing interests, human health, and indigenous spiritual values for use of aquatic resources (Moggridge et al., 2019).

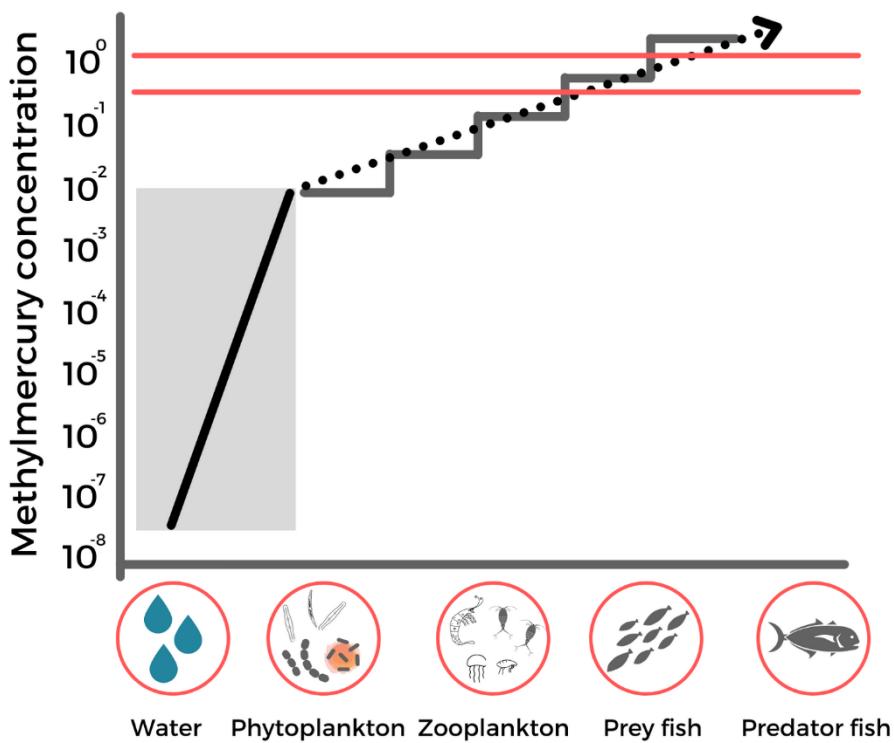
Bioaccumulation may be measured directly in the tissue content of exposed organisms or through surrogate techniques including passive samplers and bioluminescent bacteria. Passive sampler techniques, such as diffusive gradients in thin-films, use chemical absorbents to accumulate target analytes in a manner analogous to bioaccumulation processes. This also allows for easier and more sensitive analysis. For example, MnO<sub>2</sub> has been used as an absorbent for <sup>226</sup>Ra and <sup>228</sup>Ra in seawater from produced water discharged in the North Sea (Eriksen et al., 2006), (Bodrogi et al., 2005). They have also been used to measure methylmercury production rates and fluxes in sediments (Clarisso et al., 2011). Bioluminescent bacteria can be used as a whole-cell biosensor for target analytes such as mercury. The fluorescence emitted by these bacteria is proportional to bioavailable mercury concentrations (Dahl et al., 2011). The bacteria themselves can be sourced from the local environment which can increase the site-specificity of the measurement (Din et al., 2019).

The remainder of this section considers the bioaccumulation potential of mercury and NORM, and the environmental variables that influence their bioavailability.

#### Mercury bioaccumulation

Methylmercury is the bioavailable species of mercury and is readily absorbed by organisms to tissues, where it is not easily eliminated (Harding et al., 2018). The persistence of methylmercury in tissues, results in biomagnification across the food web (Bowles et al., 2001), particularly in large, long-lived organisms occupying predator trophic positions (Wu et al., 2019). A recent meta-analysis found that the bioconcentration of methylmercury from water to plankton was the key driver and predictor of fish methylmercury content, rather than aqueous methylmercury concentrations, which is shown in Figure 3.32 (Wu et al., 2019). This step concentrates aqueous methylmercury by three to six orders of magnitude to plankton. This step is the most variable component of mercury biomagnification, being affected by environmental parameters including dissolved organic carbon concentrations and pH (Schartup et al., 2018).

Trophic position and dietary preference are also important determinants of mercury accumulation. For example, Gilmour et al. (2019) investigated mercury concentrations from blood samples from two seabird species that exhibit different foraging strategies in Western Australia: great-winged petrels *Pterodroma macroptera* which feed on pelagic squid and flesh-footed shearwaters *Ardenna carneipes* that forage close to the coast. Mercury was found to be six times higher in great-winged petrels ( $3.360 \pm 0.180 \mu\text{g/g ww}$ ) than flesh-footed shearwaters ( $0.554 \pm 0.109 \mu\text{g/g ww}$ ). A range of predictive equations, usually incorporating indicators of trophic level such as stable nitrogen isotope ratios, have been developed to predict mercury biomagnification in food webs (Lavoie et al., 2013; Wu et al., 2019).



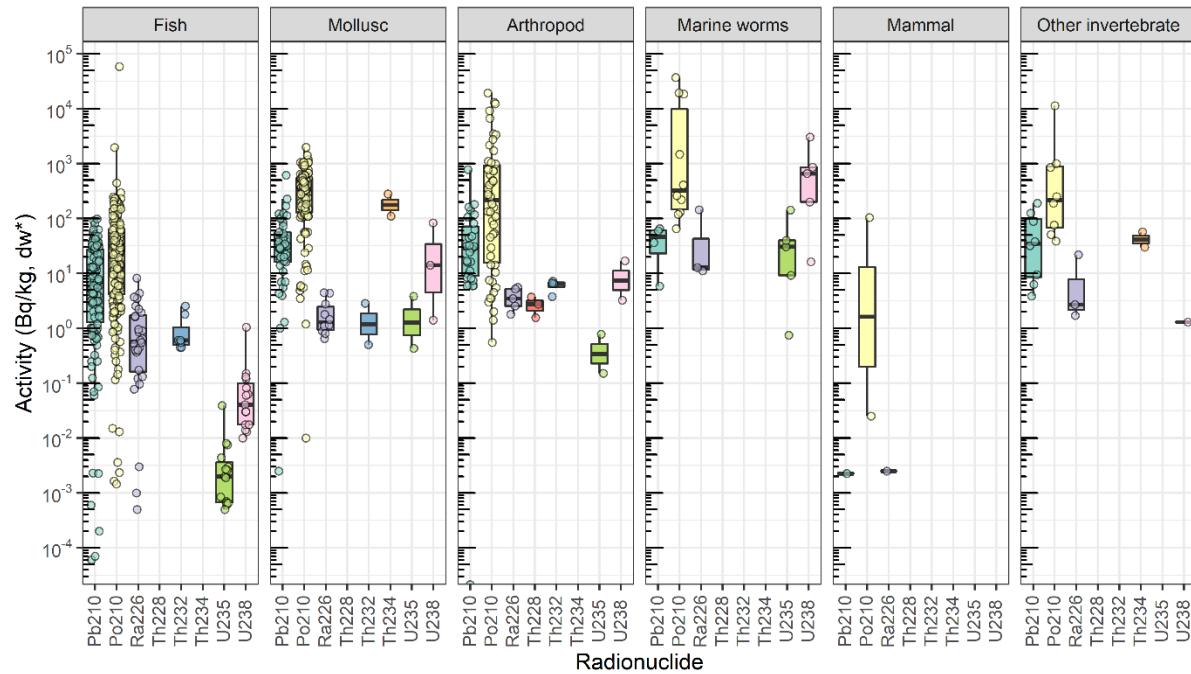
**Figure 3.32 Conceptual model of mercury bioaccumulation showing that bioconcentration to phytoplankton (inclusive of particulate organic matter) accounts for the greatest single biomagnification step. Transfer up trophic levels occurs in a stepwise manner which can be summarised by predictive equation represented as the dotted black line. The two red lines represent the FSANZ (2017) mercury limits for fish known to accumulate mercury (1 mg/kg) and all other seafood (0.5 mg/kg).**

### NORM bioaccumulation

Many studies have investigated NORM bioaccumulation in the context of understanding oceanographic processes, rather than understanding uptake from contaminated sites. Nonetheless they demonstrate radionuclide-specific and organism-specific bioaccumulation trends (Figure 3.33). For example that  $^{210}\text{Po}$  and  $^{210}\text{Pb}$  are able to accumulate in fish, molluscs, and crustacea at concentrations greater than other NORM radionuclides (Figure 3.33). The activity levels of  $^{40}\text{K}$  for all marine taxonomic groups are between  $10^2\text{-}10^3 \text{ Bq/kg}$ . These activity levels are similar to background seawater concentrations and likely reflect the well regulated nature of  $^{40}\text{K}$  in all organisms. Few data exists for polychaetes, marine mammals and reptiles, and sharks and rays to draw strong conclusions; however, polychaetes may have the greatest NORM concentration factors of any marine organism.

### Polonium

In the marine environment,  $^{210}\text{Po}$  contributes the greatest dose fraction of all NORM to marine organisms (Aarkrog et al., 1997; Jia et al., 2020), even when compared to anthropogenic contamination such as  $^{137}\text{Cs}$  following the Fukushima nuclear accident (Fisher et al., 2013). This is likely a result of its ability to bioaccumulate in organisms and transfer efficiently in the food web and through the higher order trophic levels.



**Figure 3.33 Reported radionuclide activity concentrations measured in whole organisms or tissue samples from marine organisms.** Where activity was provided as a wet weight, a conversion factor of 0.2 was used. Note that reported values typically reflect radionuclide exposure in uncontaminated environments. Data taken from MacIntosh (2020). Data are aggregated approximately at a phylum level with the following exceptions: marine worms include polychaeta, chaetognatha, and sipuncula; arthropods include crustacean and barnacles; other invertebrates include cnidaria, tunicata, and echinoderma; and fish (including ray-finned, sharks, and rays) and marine mammals are shown separately.

Accumulation of polonium predominately occurs through gut absorption from feeding, with only minor contributions from dissolved-phase absorption (Carvalho, 2011). Concentration factors of  $^{210}\text{Po}$  from seawater to whole animals have been reported to be around  $10^4$  for phytoplankton (Aarkrog et al., 1997; Stewart and Fisher, 2003; Uddin et al., 2018b) and fish (Aoun et al., 2015),  $10^5$  for copepods (Uddin et al., 2018a) and up to  $10^6$  in the shrimp *Sergestes* sp. (Heyraud and Cherry, 1979). This indicates the potential for trophic biomagnification of  $^{210}\text{Po}$  through the food chain, particularly for lower organisms at the base of the food chain (Heyraud and Cherry, 1979; Mathews and Fisher, 2008). Polonium accumulates to soft tissues of organisms like muscle and internal organs, rather than the shell or exoskeleton, which could explain its bioaccumulation and biotransfer (Deiaa et al., 2020). Transfer factors may be up to 0.7 from plankton to copepods and amphipods, but generally decreases up the food chain with transfer factors of 0.1 from crustaceans or cephalopods to large fish (Carvalho, 2011). The reason behind this drop in transfer factor is not well established.

Diet type is an important determinant of  $^{210}\text{Po}$  accumulation, with differences in  $^{210}\text{Po}$  activity levels in fish attributed to their feeding behaviour. For example, activity concentrations of up to 250-500 Bq/kg (dry weight) in anchovies and red mullet, which are benthic feeders (Mat Çatal et al., 2012). Filter-feeding mussels are commonly used as biomonitoring species and have been reported to accumulate 2000-3000 Bq/kg (dry weight of soft-tissue) in environments contaminated by fertiliser and coal combustion industries (McDonald et al., 1996; Uğur et al., 2011).

Environmental factors that influence the mobility of Po in seawater will influence Po bioaccumulation. For example, deep-sea shrimp species accumulate greater concentrations of  $^{210}\text{Po}$  than shrimp from coastal or estuarine environments (Cherry and Heyraud, 1981). This likely reflects the remobilisation

of Po from anoxic sediments in benthic environments and subsequent adsorption to particulate matter (Bacon et al., 1988; Jones et al., 2015).

#### Lead

Lead may bind to particulate organic matter which contributes to its moderate bioaccumulation in organisms (Stewart et al., 2008). However, lead does not biotransfer to any great extent in the marine food web. This could be because lead is mostly accumulated to the digestive tracts or shell material (exoskeleton) which may not be ingested by or be in a bioavailable form for consuming organisms (Deiaa et al., 2020; Fisher et al., 1996). Surveys have shown lead bioaccumulation is highest in filter feeding organisms such as mussels, with limited concentrations in fish and cephalopods (Deiaa et al., 2020). Therefore, concentration factors of  $^{210}\text{Pb}$  from seawater to organisms are generally of the order of  $10^2$  (Heyraud and Cherry, 1979).

Many of the bioaccumulation studies using  $^{210}\text{Pb}$  investigate accumulation from uncontaminated seawater. In contaminated environments, the accumulation of lead may be higher. For example,  $^{210}\text{Pb}$  activities in the herbivorous fish *Siganus rivulatus* were 99 Bq/kg (wet weight). This resulted from an environment contaminated by the discharge phosphate fertiliser production facility (a NORM-producing industry), compared to activity levels in clean reference sites which resulted in fish activity levels  $\leq 5$  Bq/kg (Aoun et al., 2015).

#### Other radionuclides

Other radionuclides are not known to bioaccumulate but may be particle reactive, meaning that they could adsorb to microorganisms such as phytoplankton leading to relatively high accumulation and transfer factors. These may not reflect incorporation of the radionuclides into the organisms, but still poses a source of internal radiation exposure.

Radium has been shown to accumulate and associate with particulate matter including phytoplankton with concentration factors of  $<10^3$  (Stewart et al., 2008). In a study using juvenile Atlantic cod,  $^{226}\text{Ra}$  was found to accumulate through a dietary exposure, but whole-body activity levels reduced to pre-exposure levels after 4 days without exposure (Eriksen et al., 2006).

Bismuth is rapidly adsorbed by particulate matter including organic ligands, and so bioaccumulate in some marine organisms including phytoplankton (concentration factors of  $10^5\text{-}10^7$ ) and macroalgae (concentration factors of  $10^2\text{-}10^3$ ) (Fowler et al., 2010; Kearns and Turner, 2016). However, transfer factors from microalgae to zooplankton were low and there's no indication that bismuth can bioconcentrate in the food web.

Thorium is transported in the particulate phase and so may be accumulated by filter or benthic feeders to activity levels of  $70 \pm 15$  Bq/kg ( $^{232}\text{Th}$ , dry weight) in a mussel or sea urchin in uncontaminated areas (Hurtado-Bermúdez et al., 2019). However, there is no indication it may significantly bioaccumulate or biotransfer in the marine ecosystem.

#### 3.8.2.3 Toxicity line of evidence

A toxicity line of evidence measures hazard impacts from exposure to a contaminated media. This is normally conducted on contaminated sediments or waters collected from the field which are then exposed to laboratory-cultured or field-collected organisms to investigate toxicity effects (Regoli et al., 2019).

Toxicity is a description of harm to organisms attributable to a contaminant. This may be from a disruption of cellular processes, the generation of harmful by-products such as reactive oxygen species, the utilization of cellular resources for detoxification, or from changes to cell morphology (Jakimska, 2011). A range of standard ecotoxicological tests exist that can be used to investigate one or more of these toxic effects, for example, growth inhibition to cyanobacteria and microalgae (OECD, 2011). A significant change in one or more measures of the organism's health compared to the response in control conditions (such a media from a clean reference site) is used as an indication of toxicity.

The toxic response is the net result of all interactions with the environmental media so causality with a single contaminant, such as with the laboratory investigations behind chemistry lines of evidence, is usually not possible. However, the toxicity accounts for all interactions with the environmental media, such as factors that may modify toxicity, so it may be a more representative test for environmental impacts.

#### 3.8.2.4 Biodiversity line of evidence

Biodiversity is a measure of the species composition and richness of an ecosystem. Conservation of biodiversity is usually an explicit management goal. For example, in the *UN Convention on Biological Diversity* and Australia's Strategy for Nature (Commonwealth of Australia, 2019), the *EPBC Act*, and the National Water Quality Management Strategy (Commonwealth of Australia, 2018). The biodiversity line of evidence measures a change in a specific population or community of organisms in the environment, usually compared to a baseline or reference condition of the environment. Unlike the toxicity line of evidence, there may not be a specific causal relationship between increasing contaminant concentration and biodiversity change as complicated interconnections may mask toxic effects. For example, sensitive organisms may die leading to an increase in the proportion of tolerant species.

The *EPBC Act* defines a provision for the development of bioregional plans for each marine region with the goal of identifying and monitoring biodiversity. These have been developed for all Commonwealth marine regions (discussed in Section 3.6) and identifies components of biodiversity including species, habitats, ecological communities, genes, ecosystems and ecological processes.

Measuring impacts to biodiversity typically requires field studies and taxonomic assessments of communities of organisms (Ling et al., 2018). Recent advances allow for the extraction of environmental DNA (eDNA) to assess community composition at depth and resolutions unobtainable using traditional taxonomic assessments (Chariton et al., 2015; Simpson and Batley, 2016). Either approach can be used in laboratory mesocosm studies or *in situ* to assess contaminant impacts to biodiversity. However, these approaches require access to representative contaminated products or contaminated field sites.

#### 3.8.3 Environmental reference levels

Environmental reference levels exist to support ecological risk assessments by providing a basis upon which to link contaminant concentration to harm. They are tools that support assessment against management objectives, rather than regulatory limits themselves. Reference levels should be applied to characterise a hazard line of evidence against a management objective for an environment subject to protection (Commonwealth of Australia, 2018).

Environmental reference levels are referred to by many names including guideline values, environmental quality standards, benchmark values, threshold values, and environmental consideration levels. For the purpose of this review they are defined as the concentration of a contaminant in environmental media below which there is a low risk of a defined hazard impact occurring, and are used interchangeably with the term guideline values (ANZG, 2018). Different reference levels have different purposes or provide different levels of protection or consideration.

The level of protection afforded to an environment depends on its baseline or desired condition. The level of protection will typically be decided in consultation with stakeholders, including the regulator, and should consider community values associated with the ecosystem. Three ecosystem condition categories are recognised in the Water Quality Management Framework:

- High conservation or ecological value
- Slightly to moderately disturbed
- Highly disturbed

The water quality guidelines provide descriptions of appropriate levels of protection for indicator types in each category. For example, for chemistry lines of evidence, a contaminant concentration that protects 99% of species is used for pristine ecosystems versus 80% species protection for highly disturbed ecosystems.

### 3.8.3.1 Radiation reference levels

Radiation reference levels are based on dose rates that have the potential to cause harm to a population of non-human biota. A number of dose-rate reference levels reported from various international authorities may be used to assess potential radiation impacts to ecosystem receptors (Table 3.17). A 10 µGy/h dose-rate reference level is recommended by ARPANSA for conservative initial assessments of radionuclide risk to non-human biota in all environments (i.e. terrestrial, marine, freshwater) (ARPANSA, 2015b; ARPANSA, 2017b). This is based on a probabilistic species sensitivity distribution model using dose-rates shown to cause harm to organisms in scientific studies (Garnier-Laplace et al., 2010) and is also adopted by the ERICA tool. In a tiered risk assessment, exceedance of this reference level requires further assessment of a radiation risk.

Dose-rate reference levels based on the dose rates below which negative effects are not expected to occur in specific taxonomic groups have also been developed. For populations of aquatic organisms it is 400 µGy/h (UNSCEAR, 2008), or for marine reference animals and plants they range between 400-4000 µGy/h for a reference crab, and 40-400 µGy/h for a reference flatfish and brown seaweed (ICRP, 2008). These reference levels form the basis for most national radiation environmental reference levels, Table 3.17 (Real and Garnier-Laplace, 2020).

Using conservative modelling and exposure assumptions, dose rates may be converted into specific activities for radionuclides. The ERICA tool adopts this approach for its first-tier risk assessment (Brown et al., 2008). The tool uses a database of representative taxa-specific radionuclide transfer, bioaccumulation, and dosimetry parameters (described in Section 3.8.2.3) to calculate the highest activity level of a radionuclide that would comply with an environmental reference level. The resulting Environmental Media Concentration Limits (EMCLs) for NORM radionuclides are given in Figure 3.34 for a screening level of 10 µGy/h. Where multiple radionuclides are present in the environment, their individual activities are divided by their EMCL and summed as a risk quotient to assess the total exposure to an organism. This approach will likely overestimate risk given the conservative parameterisation used.

**Table 3.17 Select environmental reference levels for radiological protection of the environment.**

Organisation	Environmental reference level ( $\mu\text{Gy/h}$ )	Taxa protected	Reference
ICRP Derived	4.2-42	Deer and Rat, Duck, Pine tree	(ICRP, 2008)
Consideration Reference Levels (DCRL)	42-420 420-4200	Frog, Fish, Grass and Seaweed Earthworm, Bee, Crab	
United Nations Scientific Committee on the Effects of Atomic Radiation	100  400	Terrestrial organisms  Aquatic organisms	(UNSCEAR, 1994; UNSCEAR, 2008)
United States Department of Energy	42  420  420	Terrestrial animals  Terrestrial plants  Aquatic animals	(US DOE, 2019)
European Union through the ERICA project	10	Ecosystem screening level	(Garnier-Laplace and Gilbin, 2006)

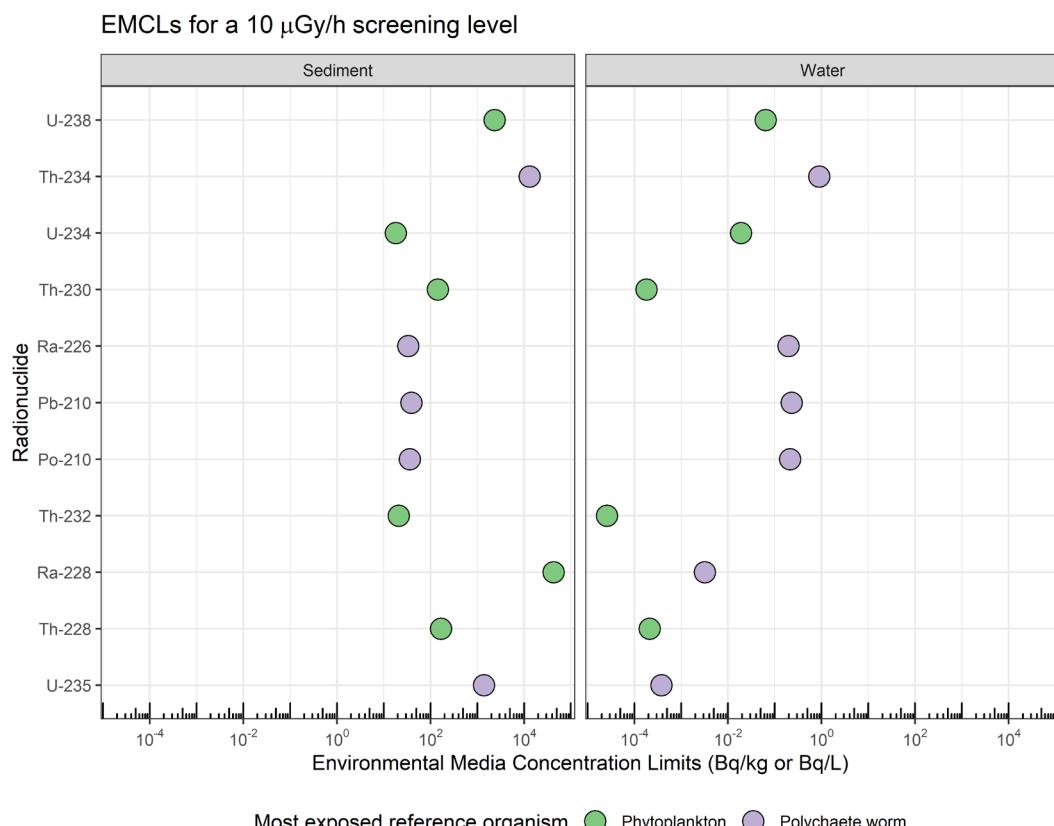


Figure 3.34 Environmental Media Concentration Limits (EMCL) for NORM radionuclides and a 10  $\mu\text{Gy/h}$  screening level and the corresponding most exposed reference organism. EMCLs for sediments are given in Bq/kg and for water in Bq/L. Note that the EMCLs presented are for single radionuclide exposure scenarios.

Various other regulatory limits exist, mostly designed to protect human health from likely exposure pathways or to define substances subject to regulatory control. However, these are not designed to protect ecological receptors so should not be used for such purposes. These may include exclusion criteria, criteria for dredging activities, or criteria for drinking water quality:

- Exemption or exclusion criteria (discussed in Section 3.2.4.3)
- National Assessment Guidelines for Dredging: 35 Bq/g (sum of gross alpha and beta) (Commonwealth of Australia, 2009)
- Drinking Water Guideline screening values: 0.5 Bq/g for  $^{226}\text{Ra}$  and  $^{228}\text{Ra}$  (NHMRC and NRMMC, 2011)

### 3.8.3.2 Chemical reference levels

The Water Quality Management Framework is accompanied by default water and sediment-quality guidelines (ANZG, 2018). These are derived using toxicological data for individual contaminants to organisms in different taxonomic groups. Default water-quality guidelines are accompanied by a defined level of protection which specifies the degree of environmental protection expected to be afforded by the guideline value (e.g. the default water quality guideline is specific for 95% species). The water and sediment environmental reference levels are derived by different methods.

The environmental reference levels for chemicals in waters are derived by applying species sensitivity distributions (a type of probabilistic model) to threshold concentrations (i.e. the concentration of a contaminant where a biological response is observed) for a range of individual species. Using species sensitivity distributions to derive guideline values is data intensive (Warne et al., 2014), and guidelines for a number of contaminants are yet to be developed (Table 3.18). There are fewer toxicological data for sediments compared to waters so environmental reference levels are based on ranking of biological effects (empirical guidelines) or based on equilibrium partitioning (mechanistic guidelines) (Simpson and Batley, 2016). Default sediment quality guideline values are given as a low (SQG low) and high value (SGQ high) concentrations, between which toxic effects may start to be observed to sensitive organisms (Table 3.18).

Guideline values should not be the only tool by which water and sediment quality is assessed. Default guideline values are considered a useful starting point but may not be relevant to all local conditions. To account for local environmental factors which may modify contaminant risk, and to ensure protection of important species within the local ecosystem, the water quality management framework recommends the derivation of site-specific guidelines (van Dam et al., 2019). A recommended process to derive site-specific guidelines is outlined in the Water Quality Management Framework (ANZG, 2018).

Where information does not exist, specific guidelines for specific hazard impacts can be developed. For example, a risk of mercury is dietary exposure to other organisms, typically in the methylmercury species. No guidelines exist for dietary exposure and no guidelines exist for environmental media concentrations of methylmercury species. However, Depew et al. (2012) was able to calculate dietary toxicity thresholds, which were 0.5 mg/kg (wet weight of diet source) for behavioural, growth, and lethal impacts and 0.04 mg/kg for reproductive and biochemical effects.

### 3.8.3.3 Bioaccumulation reference levels

All organisms accumulate metals to some extent, but these are typically well regulated through cellular uptake, homeostasis, and detoxification mechanisms. Some contaminants, however, may not

be able to be eliminated from an organism at a rate equal or greater than their accumulation. This can lead to an elevation of the contaminant well above concentrations observed in environmental media.

Bioaccumulation is defined as the accumulation of chemicals by any route, including dietary, respiratory, or direct contact with environmental media. Bioconcentration is defined as the accumulation of chemicals only through the surrounding environment (i.e. excluding dietary pathways). Biomagnification is the process where chemicals accumulated in organisms are transferred through the food web. The proportion of chemical transferred from the prey to the consuming organism is called the transfer factor.

In the Australian water quality management framework, Simpson and Batley (2016) have defined a ranking scheme for bioaccumulation as a line of evidence in water and sediment quality assessments:

1. Not significantly different from controls
2. Significantly different ( $P < 0.05$ ) and  $\leq 3$  control value
3. Significant different ( $P < 0.05$ ) and  $> 3$  control value

These values are arbitrary and may not be protective unless background contaminant concentrations and the toxicity of the contaminant are known, but provide a starting point to assess whether bioaccumulation may be of concern to environmental receptors.

Other approaches do not necessarily consider a control value, instead bioaccumulation is defined empirically using bioaccumulation factors (BAF). Bioaccumulation factors are calculated as the concentration of a chemical in an organism divided by the concentration of the chemical in its environment.

For example, US EPA (2001) defines bioaccumulation potential as:

- Low (BAF <250)
- Medium (BAF 250-1000)
- High (BAF 1000-5000)
- Very high (BAF >5000)

An important class of bioaccumulation reference levels exist for food. Maximum permissible levels for some chemicals are defined for some seafood taxa for the purpose of protecting human health. In the context of decommissioning offshore oil and gas infrastructure, these levels may be considered also relevant to protecting community and economic values relating to commercial and recreational fishing. In Australia the Food Standards Australia and New Zealand (FSANZ) administers the Food Standards Code which defines maximum levels of some contaminants in seafood. For mercury, the maximum levels are 1 mg/kg for species of fish that are known to contain high levels of mercury (swordfish, southern bluefin tuna, barramundi, ling, orange roughy, rays and sharks), and 0.5 mg /kg for all other species of fish, crustaceans and molluscs. For lead, the maximum level is 0.5 and 2 mg/kg for fish and molluscs, respectively (FSANZ, 2017).

### 3.8.3.4 Toxicity reference levels

A toxicity line of evidence can be investigated by assessing the response of representative organisms to a contaminant, or more commonly contaminated environmental media such as sediments or waters. This has been applied in the oil and gas industry to typically assess the toxicity of cuttings piles or sediments surrounding platforms which contain a mixture of contaminants (Gillett et al., 2020; Regoli et al., 2019). The reference level is often defined as a significant response in the exposed

organism/s compared to the response of the organism/s in 'control' or uncontaminated media. The types of toxic effects that can be measured in organisms are varied and may include chronic effects such as population growth rates over multiple lifecycles or reproduction success, or chronic effects such as changes to mass, changes in biochemical processes such as enzyme activity, organism mobility, or organism mortality.

### 3.8.3.5 Development of site-specific reference levels

Site-specific environmental reference levels may be developed for lines of evidence if done transparently and with suitably conservative parameter assumptions (ANZG, 2018). These may be based on:

- Reference-site data - such as background concentrations of mercury and NORMs in organisms or environmental media
- Laboratory-effects data - measuring the toxicity of a contaminated product to a representative organism
- Field-effects data – measuring the response of an ecosystem to a contaminated product *in situ*.
- Multiple lines of evidence based on two or more of the above

The reference site approach may be appropriate for contaminants that do not have sediment quality guidelines. For example, the ANZG (2018) recommends the use of the 80<sup>th</sup> percentile of reference-site concentrations as a quality guideline where none exist.

**Table 3.18 Environmental reference levels for mercury and NORM elements relevant to oil and gas infrastructure decommissioning.** Default water quality guidelines (WQG) and default sediment quality guidelines (SQG) are used to assess toxicity impacts, and food standards maximum concentration levels and a classification of bioaccumulation can be used to assess bioaccumulation impacts. All water and sediment guidelines values taken from ANZG (2018), food standard levels from FSANZ (2017), and the potential to bioaccumulate as defined by US EPA (2001). Note that only elements with environmental reference levels are tabled and missing values indicate that no guideline exists. However, this should not be taken to mean no hazard exists.

Element	Potential to bioaccumulate	Food standards code maximum levels in seafood (mg/kg)	WQG for 95% species protection (µg/L)	WQG for 99% species protection (µg/L)	SQG low (mg/kg)	SQG high (mg/kg)
Uranium	Low		0.5 <sup>a</sup>	0.5 <sup>a</sup>		
Radium	Medium-high, BAFs 10 <sup>3</sup>					
Polonium	Very high, BAFs 10 <sup>4</sup> -10 <sup>6</sup>					
Lead	Medium, BAFs ~10 <sup>2</sup>	0.5 for fish 2 for mollusc	4.4	2.2	50	220
Mercury	Very >high, 10 <sup>6</sup>	1 for fish known to accumulate mercury 0.5 for all other seafood	0.4 <sup>b</sup>	0.1 <sup>b,c</sup>	0.15	1

<sup>a</sup> No marine guideline exists. Freshwater default water quality guideline (WQG) recommended as a low reliability value. No level of species protection provided.

<sup>b</sup> For inorganic or total mercury species

<sup>c</sup> To account for the bioaccumulative nature of mercury, the 99% species protection level WGV is recommended by ANZG (2018).

## 3.9 Risk analysis

A risk analysis considers the exposure of a contaminant to an ecosystem receptor and its resulting hazard impacts against the management goals for the environment. This process can be achieved in a number of ways, the preferred method as adopted by Commonwealth and state governments in Australia is reviewed in this section. In doing so, it partly meets the objectives 3 and 4 and problem statement 1.

### Key findings:

- The *Environment Regulations* do not prescribe what risk assessment approach needs to be used.
- Risk assessments considering multiple lines of evidence of hazard impacts to ecosystem receptors in a weight of evidence framework are considered best practice.
- Mercury and NORM contaminants in oil and gas infrastructure have multiple exposure pathways which may lead to exposure concentrations that can cause harm to ecosystem receptors. Significant uncertainty exists in all steps of a risk assessment because of the various factors affecting exposure pathways and hazard impacts.
- A tiered risk assessment balances conservatism with data requirements and is recommended by the National Water Quality Management Framework and Guide for Radiation Protection in the Environment.
- Multiple contaminants may lead to cumulative impacts, even if individual contaminants are all below environmental reference levels.

### 3.9.1 Overview of a risk analyses

A risk analysis considers the outcomes of exposure pathways to parameterize hazard exposure-response models for given management objectives, contaminants, and ecosystem receptors, as shown in Figure 3.35 (Suter II, 2006). Different approaches developed use different language and may include contaminant-specific or environment-specific considerations, however, will include these steps. For example, the water quality management framework includes 10-steps which includes integrated planning and adaptive management steps and water and sediment guidelines to support hazard characterisations (ANZG, 2018). For radiological risk assessments, the Guide for Radiation Protection of the Environment details dosimetry considerations to undertake hazard characterisations and emphasizes radiation-specific exposure descriptions (ARPANSA, 2015b). It should be noted that the environmental risk of radiological hazards can be integrated to the environmental risk frameworks for other types of stressors (Rhodes, Bréchignac et al. 2020). In Australia this would be the Water Quality Management Framework and associated guidelines (ANZG, 2018).

The risk analysis step is where multiple lines of evidence detailing hazard impacts to ecosystem receptors are integrated into an assessment against management objectives can be undertaken by quantitative, qualitative, approaches or combinations therein. Common methodologies include: weight of evidence (Suter II et al., 2017), bow-tie methodologies (Jones and Israni, 2012), and consequence-liability matrices (Beer, 2006).

Operators are required to undertake and report on their ecological risk assessments (Section 3.2), which largely corresponds to the process described in Figure 3.35. The *Environment Regulations r. 13*, specifies that Environment Plans contain:

- Description of the existing environment
- Details of environmental impacts and risks, an evaluation of these impacts and risks, and details of control measures for these impacts and risks
- Consideration for impacts and risks arising directly or indirectly arising from the activities

The *Environment Regulations* do not, however, prescribe what assessment approach needs to be used. This review has detailed that NORM and mercury contaminants in oil and gas infrastructure have multiple exposure pathways (Section 3.7) which may lead to exposure concentrations that can cause negative impact to marine receptors (Section 3.8). Significant uncertainty exists in all steps of a risk assessment because of the multitude of factors contributing to contaminant bioavailability and toxicity. For example, some of these discussed throughout this review are reported in Table 3.19.

To simplify the risk assessment process, conservatism may be traded for contaminant and exposure path specific data requirements. That is, a range of conservative assumptions may be made that reduce the need to understand every possible environmental transformation, the entire speciation range of contaminated products, the individual contaminants toxicity to each organism, and so on. This is termed a tiered assessment in ecological risk assessments and are recommended in the Water Quality Management Framework (ANZG, 2018) and in the Guide for Radiation Protection in the Environment (ARPANS, 2017a). Application of default environmental reference levels is an example of this trade-off.

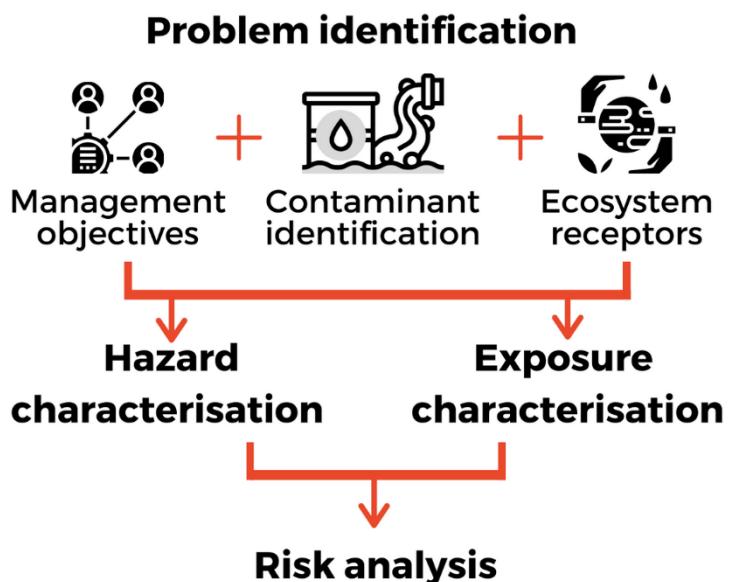


Figure 3.35 Basic components of all ecological risk assessments (Suter II, 2006).

### 3.9.2 Tiered assessments

Tiered assessment frameworks use increasingly data-intensive parameterizations of exposure and hazard models to reduce uncertainty. This could include measuring the speciation and environmental transformations of contaminants in the environment subject to management, developing dispersion models to understand the resulting contaminant concentration in environmental media, using locally important species for toxicity and bioaccumulation assessments, undertaking field studies of impact to biodiversity, and so on.

Environmental reference levels are important components of a tiered assessment. The guide for radiation protection of the environment and the sediment and water quality guidelines typically use comparisons of specific radioactivity or total contaminant concentrations against environmental reference levels as first tier levels. Progressive steps require more considered evaluations should be considered (ANZG, 2018; ARPANS, 2015b). For example, in the sediment quality guidelines, a tiered risk assessment process focuses on identifying contaminant bioavailability in sediments. A conservative initial assessment may use total concentrations (i.e. all contaminant species). If this concentration is above the guideline value then assessment of bioavailable concentrations (Jiang et al., 2021) and the environmental factors modifying toxicity (Atkinson et al., 2007) can be undertaken. If these concentrations are still above the sediment quality guidelines, then direct toxicity testing is recommended. (Figure 3.36, Simpson and Batley (2016)). For water quality guidelines dissolved contaminant measurements are recommended. If above the guideline value, then labile or bioavailable species can be measured (Koppel et al., 2019). For radiological hazards, the use of increasingly site-specific data for the parameterization of dispersion, wildlife transfer, and dosimetry models reduces conservatism. For example, only media activity levels for radionuclides are required at the first tier (Figure 3.34).

### 3.9.3 Cumulative impacts

Cumulative impacts describe the interaction of multiple contaminants to an ecosystem receptor or the combination of stressors from multiple sources on a basin scale.

The presence of multiple contaminants may lead to cumulative impacts, even if individual contaminants are all below environmental reference levels (Nys et al., 2018). There are a number of approaches to assessing cumulative impacts, with a common approach being to sum the individual contributions of contaminants in a multi substance potentially affected fraction (msPAF) (de Zwart and Posthuma, 2005), divide contaminant concentrations against guideline values and sum them as a hazard quotient (Amato et al., 2014). This also applies to radiological risks where the contribution of all radionuclides can be assessed against reference levels or their individual contribution to the overall absorbed dose rate is summed.

An alternative to reliance on reference levels is to undertake direct toxicity assessments, where a contaminated product is directly exposed to an environmental receptor to understand the overall toxicity without necessarily disentangling the individual components or relative contributions of contaminants within the product (Carlsson et al., 2014).

Additionally, the combination of multiple stressors on an ecosystem receptor should be considered. This may include stressors originating from external sources such as other decommissioned assets within a relevant spatial footprint (Chariton et al., 2016).

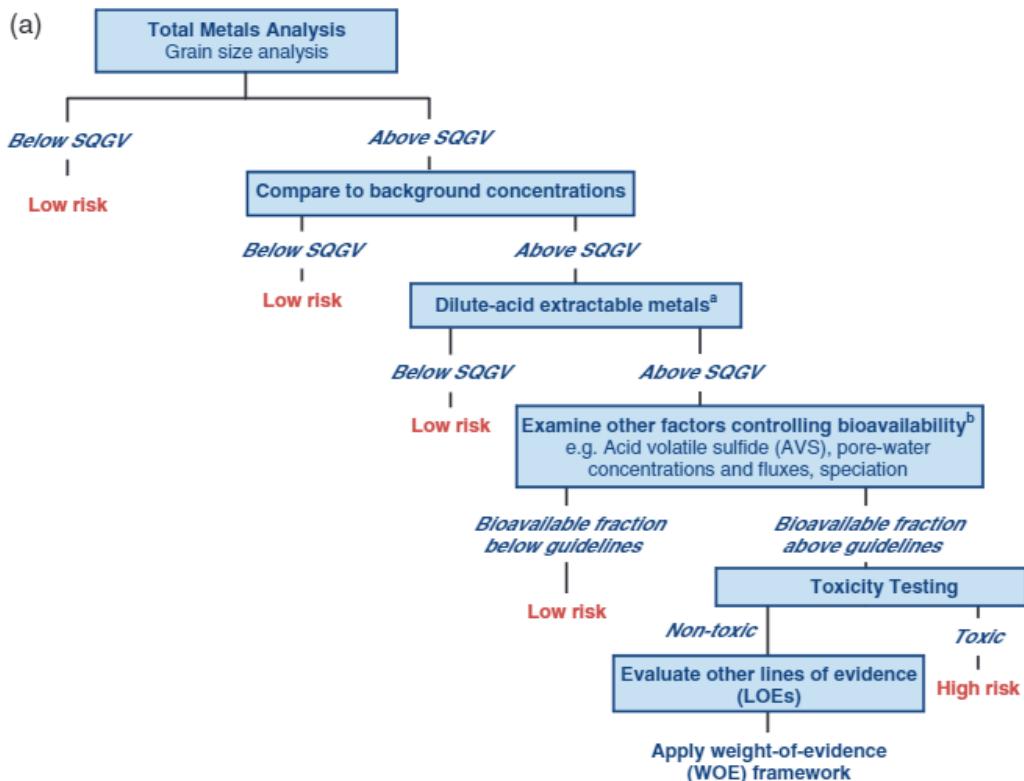


Figure 3.36 Step-wise approach to assessing metal contaminants against the sediment quality guidelines (SQGV). The assessment framework increasingly considers the bioavailability of a contaminant and the environmental factors that may affect exposure paths or toxicity. Figure taken from Simpson and Batley (2016).

**Table 3.19 Factors that affect the risk of mercury (Hg) and NORMs including polonium (Po), lead (Pb), radium (Ra), and radon (Rn) in subsea oil and gas infrastructure.**

Type	Factor	Why	Examples
Physical	Temperature	Temperature solubility relationships	Calcite solubility decreases with increasing temperature Microbial reaction rates increase with increasing temperature
	Pressure	Pressure solubility relationships	Barite solubility increases with increasing pressure
	Sediment texture	Binding capacity increases with decreasing particle size	Clay/silt sediments have greater binding capacities than sandy sediments
		Sediment texture influences sediment redox states	Clay/silt sediments promote reducing conditions by minimising oxygen penetration below the sediment-water interface
		Sediment texture may influence gas permeability	Rn as a decay product of Ra scale may be more mobile in sandy sediments than silty/clayey sediments with implications for accumulation of its daughter nuclides Po and Pb
	Organic matter	Organic matter is a strong binding phase for Hg and NORM	Hg, Po, and Pb readily bind to organic matter and may form aggregates
Chemical	Currents	Dispersion of NORM nuclides / sediment mixing	Areas experiencing greater currents may have greater particle dispersion reducing the risk of localised exposures
	Sediment redox potential	Redox reactions can increase or decrease mobility of contaminants in pore and overlying waters and sediments	Anoxic sediments can bind metals as sulfide minerals (e.g. Pb and Hg) Anoxic sediments can release insoluble minerals to pore waters (i.e. Ra) Oxidation of sulfides in sediments can create acidic conditions and release bound contaminants Reductive dissolution of Fe and Mn particles can release adsorbed contaminants
	Chemical speciation	Solubility	Different elements and species of elements have different solubilities. Po speciation leads to greater dissolved phase partitioning than Pb and especially Ra in seawater. MeHg much more soluble than inorganic Hg species.
		Bioavailability	Different elements and species of elements have different bioavailabilities. Hg and Po more bioavailable than Pb and Ra and MeHg more bioavailable than inorganic Hg.

Type	Factor	Why	Examples
	Saturation state	Marine Ba, Sr, and Ca concentrations control precipitate solubility	Ba concentrations are lower in surface waters which can increase the solubility of barite scale
	Binding affinity	Different nuclides will bind to different environmental receptors with consequences for toxicity and biotransfer.	Hg and Po have a high binding affinity for S-based ligands meaning it is incorporated into soft tissues to a greater extent than nuclides which have a stronger binding affinity for O-based ligands which accumulate in bone and shell (like Pb)
	Specific activity / half-life	Longer-lived radionuclides have a greater persistent hazard	$^{226}\text{Ra}$ has a half-life of 1600 y and concentrates to barium scales while $^{210}\text{Pb}$ and $^{210}\text{Po}$ are shorter lived (22 y and 138 d, respectively), meaning that they pose less of a radiological hazard over long timescales if unsupported.
Biological	Dietary pathway	Dietary vs particulate vs dissolved exposure	Particulate-bound contaminants (Hg, Pb, and Po) may cause a higher risk to filter feeders / detritus consumers than pelagic consumers.
	Sediment-interactions	Bioturbating organisms alter redox processes and can resuspend sediments	Bioirrigation of burrows introduces oxygen, bioturbating by demersal fauna (e.g. fish, stingrays) resuspends sediments, which may make particulate matter available to filter feeders and epibenthic ecosystems.
	Habitat preference	NORM-contaminated infrastructure may serve as a habitat for some organisms increasing lifetime exposure period	Sessile invertebrates, macroalgae, and hydroids have greater exposure periods owing to their fixed locations and proximity to contamination sources. Burrowing organisms like polychaetes and some crustaceans may have increased exposure risk due to contaminant release in pore waters NORM radionuclides may substitute for elements like Sr and Ca in shells and exoskeletons (e.g. corals, barnacles, diatoms, radiolarians)

## 4 Current practice

The purpose of this section is to review industry's practices and approaches to the ecological risk assessment of NORM and mercury contaminated subsea pipelines subject to decommissioning decisions. These practices are then compared with the review of current knowledge described in Section 3 to identify knowledge gaps and insights.

To recap, three problem statements were defined in Section 1.2 focussing on the need to: 1) better understand the threshold levels at which NORM and mercury in subsea production infrastructure became a concern to the receiving environment, 2) better understand the processes and exposure pathways of individual contaminants to receptors post decommissioning and 3) improve the tools and procedures for measuring NORM and mercury for decommissioning decisions and long-term monitoring.

The basis for assessing the current practice in respect of these problem statements has centred around the NORM and mercury contamination data and reports supplied by the Industry Participants and the interpretation of a small subset of that data located in a number of the publicly available decommissioning Environmental Plans.

Industry participants have provided the Research Team a range of non-attributed NORM and mercury contamination data from well tubing, casing, trees, various onshore pipelines, offshore topsides piping, topsides spools and subsea export pipelines in the North West Shelf and Bass Strait. This data was limited in detail containing only a testing method title, a contamination range and associated measurement units. In addition, contamination test reports for flexible flowlines and risers of two Carnarvon Basin assets were separately provided with a more detailed definition of testing methodology and scope.

In terms of production measurements, mercury concentration data was made available for various processing streams of two non-attributed offshore gas production assets, both feeding onshore LNG plants. For one of these assets a number of detailed mercury and NORM contaminant characterisation and assessment reports were also made available containing state of the art measurement and assessment techniques.

While the majority of this data provides evidence of contamination type and level, it does not indicate how the data is interpreted and applied to inform risk and acceptability positions. From the Research Team's perspective this was deemed important to understand, so the problem statements could be appropriately contextualised and addressed. To gain this understanding, a range of publicly available Environmental Plans were reviewed with the findings captured throughout this section.

Section 4 starts by addressing, at a high level, current approaches to ecological risk assessments. This is informed by published Environmental Plans and seeks to understand the applied definition of acceptability, the spatial and temporal extent of management, choice of environmental receptors, the hazard impacts considered and the environmental reference levels applied. The current practice of NORM and mercury measurement techniques against the requirements from Section 3 is also addressed followed by a visualisation and quantification of measured contamination inventories. This is compared with published values to gauge the contamination levels on a global basis. Finally, a summary is provided of the contamination mitigation measures currently employed in preparation for the decommissioning phase of the asset to reduce the ecological risk to an acceptable level.

## 4.1 Approach to ecological risk assessments

Oil and gas operators' approach to ecological risk assessments are reviewed against the approach presented throughout Section 3. This section provides an understanding of how ecological risk assessment frameworks, protective standards, and guidelines are currently being applied to understand mercury and NORM risk in the marine environment. In doing so this section partly meets objectives 1 and 4.

### Key findings:

- Best practice ecological risk assessments are given by the Water Quality Management Framework or APRANSA's Guide for the Protection of the Environment. Currently, these frameworks are only used selectively.
- Only a toxicity hazard impact is typically considered in ecological risk assessments by the direct comparison of contaminated product concentrations to a water or sediment quality guideline value.
- The robustness of ecological risk assessment for mercury and NORM-contaminated product varies widely across Environment Plans.
- Ecological risk assessment approaches for contaminants in produced water (such as direct toxicity testing and dispersion modelling), can be applied to contaminated products in pipelines.

### 4.1.1 Overview of current approaches to ecological risk assessments

The *Environment Regulations* requires operators to undertake ecological risk assessments for the impacts of hazards and report them in Environment Plans (discussed in Section 3.2). The *Environment Regulations r. 13(1-7)* specifies the required contents of an Environment Plan, which has been adopted as their *de facto* template. The approach to identifying and evaluating impacts and risks (*Environment Regulations rr. 13(5 and 6)*) are not prescribed by the regulations or NOPSEMA and are conducted using internal organizational processes. Typically, titleholders use a risk assessment process that aligns with AS/NZS ISO 31000 (Risk management – Principles and Guidelines), which follows a qualitative likelihood-consequence-risk matrix approach, and AS/NZS ISO 14001 (Environmental management systems – Requirements with Guidance for Use). For more critical or difficult risk based decisions the framework defined in Oil & Gas UK (2014) can also be applied for the justification of risk tolerability (i.e. below the upper tolerability limit and ALARP) and includes both safety and environmental incidents.

There are very few examples of Environment Plans considering the ecological risk of NORMs or mercury in pipelines for the purpose of decommissioning (PTTEP AA, 2014; Santos Limited, 2020; Woodside Energy Ltd, 2020a). Mercury and NORMs as contaminants of potential concern are often reported in Environment Plans relating to well plugging and abandonment or the cessation of operations activities (BHP Billiton Petroleum (Australia) Pty Ltd, 2017; BHP Billiton Petroleum (Australia) Pty Ltd, 2018; ConocoPhillips Pipeline Australia Pty Ltd, 2019; Eni Australia Limited, 2013; Esso Australia Resources Pty Ltd, 2018). These Environment Plans do not necessarily consider the ecological risk of mercury and NORM due to their confinement in pipelines and the expectation that they will be subject to future decommissioning Environment Plans. Mercury and NORM are also

reported in Environment Plans for operational activities (Esso Australia Resources Pty Ltd, 2019a; Shell Australia Pty Ltd, 2016).

Two environment plans currently under assessment that consider decommissioning pipelines *in situ* are discussed in more detail here as they are the most relevant to this review.

#### 4.1.1.1 Bayu-Undan

Santos Limited (2020) considered the risk of NORMs and mercury to various ecosystem receptors from their release during the decomposition of the pipeline structure over the degradation timeframes. The stated management objective for the contaminant risk assessment was to ensure that impacts from long-term degradation of the *in situ* pipeline are minimal and acceptable. (Un)acceptability is defined in the Santos environmental management process for different ecosystem receptors as:

- Substantial change in the air, water, or sediment that may adversely impact biodiversity, ecological integrity, social amenity or human health are unacceptable.
- Adverse interactions with other marine users are unacceptable for socio-economic receptors
- Substantial change that may modify, destroy, fragment, isolate or disturb an area of habitat such that an adverse impact on marine ecosystem functioning or integrity are unacceptable
- Substantial impacts to threatened and migratory marine fauna are unacceptable
- Substantial impacts to the Oceanic Shoals Marine Park and KEFs are unacceptable

Additional sub-criteria are given to describe types of significant impacts to threaten and migratory marine fauna and Oceanic Shoals Marine Park and KEFs. For example, these include measurable statements that define substantial impacts as changes that may:

- Lead to a long-term decrease in the size of a population
- Disrupt the breeding cycle of a population
- Displace threatened and migratory marine fauna from habitat critical areas

These are referenced to recommendations in marine regional plans (an output of the *EPBC Act*, described in Section 3.6), species-specific conservation and recovery plans, and Marine Parks Network Management Plan.

The ‘Description of the Source of the Risk’ section fulfils the function of the problem identification step in an ecological risk assessment (i.e. provides a description of the NORM and mercury contaminated product). The ‘Potential Impacts’ section then discusses impacts to ecosystem receptors. This includes the results of an exposure characterisation (such as dispersion modelling and consideration of environmental transformations), and hazard characterisation (from the comparison to guideline values) to each ecosystem receptor. These potential impacts are informed by the outcome of bespoke research projects, such as from ANSTO.

Based on the potential impact description and further controls and mitigation measures (which is acknowledged as being consistent with the precautionary principle), a rating is given to consequence and likelihood and an assessment of risk is given.

#### 4.1.1.2 Echo Yodel

The Woodside environmental management process defines 6 ecosystem receptors potentially impacted by petroleum activities which are assessed: marine sediments, water quality, air quality, ecosystem/habitats, species, and socio-economic. Environmental impacts and risks are informed by environmental hazard identification workshops with relevant internal participants and project advisors.

Consequences are defined by a 5-level rating based on temporal extents of impacts (e.g. short-term impacts lasting less than one year versus long-term impacts lasting more than 50 years) as well as a description of the importance of the environment attributes being impacted (i.e. impact not significant to environmental receptors or impacts to highly valued ecosystems, species, habitat or biological attributes). Similar descriptions are given to social and cultural consequences.

Likelihood descriptions for the consequence are defined by frequency (ranging from 1 in 100,000 to 1,000,000 y to >1 in 10 y) and experience (from unheard of in industry to has occurred frequently at the location or is expected to occur) on a 5-level rating.

Risks are then determined by a consequence-liability table, with decision types explicitly defined by the Oil & Gas UK (2014) framework. Any risk rating may be considered acceptable if they are below defined acceptable levels for that impact or risk and meet the *Environment Regulations* requirements of ALARP, consider ecologically sustainable development, the internal and external context, and other requirements.

The 'Impact Assessment' section discusses the evidence behind consequence and likelihood decisions. For the Echo Yodel pipeline, this included a mean surface radiation measurement of 0.11 µSv/h from a hand held gamma dose rate meter at an unspecified point of two subsea pipeline spools recovered to surface (exterior or interior not defined). This method does not account for the presence (or absence) of any beta or alpha emitting radionuclides. The EP then relates this human dose rate to an ecological dose rate (measured in Gray, Gy) to ecological dose rate reference levels of 10 µGy/h and 40 µGy/h discussed in Section 3.8.3. Mercury testing occurred on a recovered pipeline section, which was used to determine mercury content in the pipeline of 0.003 µg/kg. This value was compared to representative background and sediment quality guideline values for mercury.

While the Bayu-Undan and Echo Yodel Environment Plans are a limited sample size, they are considered to be representative of current practice adopted for risk assessments of contaminants. They have highlighted limitations in the current approach because of knowledge gaps around measurement technologies, contaminant exposure pathways, and hazard impacts.

#### 4.1.2 Defining management objectives

Management objectives are the objectives required to meet legislative requirements for a decommissioning decision that accounts for the environment and community context of a contaminated product (Section 3.2.2). These are typically well defined (in various terms) in Environment Plans.

In Environment Plans, the primary focus is on ecosystem receptors that are matters of national environmental significance rather than consideration to the conservation of biological diversity and ecological integrity (*EPBC Act s. 3A*). This is understandable given the specific requirement of the *Environment Regulations* (*Environment Regulations rr. 13(3)*), defined nature of migratory species, threatened species, and key ecological features, and helpful tools maintained by DAWE to identify them within an operational footprint.

Ecosystems, however, are maintained by a wide range of species performing unique functional roles that service and maintain the broader ecosystem. By limiting the consideration of environmental impacts to matters of national environmental significance, impacts to species performing critical roles in different ecosystems may be missed. For example, impacts to species not specifically listed as

threatened or migratory may cause negative flow-on impacts to the broader ecosystem, including those listed as being of national environmental significance.

For planned contaminant releases, consideration of environmental receptors reflecting habitats such as 'benthic' or 'epibenthic' communities can meet this need in ecological risk assessments. The environmental baseline study undertaken by ConocoPhillips for the Bayu-Undan pipeline is a best-practice example of how this can be achieved because it characterises the type of ecosystems, the key indicators of biodiversity, and the connectivity of the reef systems (Heyward et al., 2017).

Other community values may also be important considerations and can relate to commercial, recreational, and cultural or spiritual uses of an ecosystem, and vary by region (Tung, 2020).

Stakeholder engagement by Shaw et al. (2018) have identified many examples of these priorities.

Management objectives should define the spatial scale of management relevant to the hazard impacts that may occur. This is important because it constrains or expands contaminant dispersion or food-web transfer considerations. Given that:

- (i) the baseline decommissioning option is complete removal unless an alternative option delivers equal or better environmental outcomes
- (ii) the ecological benefit of *in-situ* decommissioning options arises from the development of epibenthic reef structures on the pipeline (Bell and Smith, 1999; Pradella et al., 2014), see Section 3.4
- (iii) the reef structure provides a feeding ground to pelagic organisms which may be affected by bioaccumulative contaminants (Bond et al., 2018a)

The toxicity impacts from pipeline-associated contaminants should be considered within the pipeline footprint to epibenthic and benthic communities, while bioaccumulation indicators should be assessed relative to the broader food web and trophic levels.

#### 4.1.3 Consequence and hazard impacts

An ecological risk assessment should identify potential hazard impacts (i.e. lines of evidence, Section 3.8) to ecosystem receptors. These are typically identified in a conceptual model, such as those developed for mercury and NORM (Figure 3.23, Figure 3.26). Multiple lines of evidence are recommended to understand hazard impacts; however, only a toxicity line of evidence is usually considered against environmental reference levels. This may be because the Water Quality Management Framework defines sediment and water quality guideline values for toxicity endpoints.

Environment Plans identify consequences by qualitative statements of ecosystem harm defined by the temporal extent of the impact and the value of the impacted receptor. Where available, environmental reference levels are applied against expected environmental concentrations. However, their application is often inconsistent with their specified purpose. For example, the incorrect reference level may be applied or the incorrect measurement is used. In the absence of multiple lines of evidence and defined exposure pathways linking contaminated product to ecosystem receptors environmental reference levels may be the single quantitative of regulatory acceptance, rather than their intended purpose of interpreting an aspect of a contaminant's impact to an ecosystem receptor. In some instances, only best professional judgement seems to be used.

For NORMs specifically, some operators have compared individual radionuclide activities or surface radioactivity measurements against exemption criteria for drinking water quality guidelines or the transportation code to determine consequence. To assess risk, the specific activity concentration of

radionuclides in a contaminated product is required to undertake the modelling needed to assess dose rates to representative organisms. The quickest way to achieve this would be to compare activity levels to the EMCLs, Figure 3.34 (equivalent to a first tier ERICA assessment), using transparent dispersion and partitioning assumptions. This is appropriate because they are based on a dose-rate reference level of 10  $\mu\text{Gy/h}$  and use conservative  $K_d$  and CR values.

Surface activities (i.e. dose rates determined from the surface of a piece of recovered infrastructure) cannot be used to infer ecological risk and comparison against the transportation code is inappropriate to assess radiation hazard. See Section 3.5.3.3 for a more detailed discussion about the issues relating surface activities activity concentrations to dose rates. Similarly, comparison against the Drinking Water Quality Guidelines (which include a screening value of 0.5 Bq/L for both  $^{226}\text{Ra}$  and  $^{228}\text{Ra}$  (NHMRC and NRMMC, 2011)) cannot be used to infer ecological risk.

Oil and gas operators routinely apply environmental reference levels to assess the risk of metal contaminants in marine sediments and waters. This is generally achieved from direct comparison to the default sediment or water quality guideline values (ANZG, 2018). Typically, measurements or summary statistics are not reported. Instead, are reported as below guideline values. Where possible, the number of samples or measurements with their uncertainty should be reported to give confidence in the sampling design and conclusions of risk reached. No Environment Plan assessed considered the development of site-specific guideline values or used environmental reference levels for non-toxicity lines of evidence.

Background measurements of contaminants, however, can be used to develop site-specific guideline values and assess impact from predicted contaminant dispersion, discussed in Section 3.8.3 (ANZG, 2018). Mercury and NORM concentrations in seawater and sediments have been measured by some operators prior to an activity. For example, ConocoPhillips Australia Exploration Pty Ltd (2017) reported one water-column measurement was above the limit of reporting at 0.068 Bq/L while sediment concentrations were low, typically of the range 0.005 – 0.04 Bq/g for  $^{226}\text{Ra}$ ,  $^{228}\text{Ra}$ , and  $^{228}\text{Th}$  in an environment where a pipeline was to be installed.

Assumptions and parameters underlying environmental partitioning, dispersion, wildlife transfer, and dosimetry also need to be transparent. PTTEP AA (2014) reported the identified radiation was at a level that would result in minimal impact to individual organisms, species, and biodiversity without aforementioned transparency. Approximately 50 km of flow lines were decommissioned *in situ* which were reported to contain some ‘produced water scale’. The Environment Plan did not outline how the radiation was measured, what reference levels the radiation was assessed against, and what assumptions were made in undertaking this assessment. As measurement technologies are limited for *in-situ* radiation assessments, it is unclear how the ecological risk was quantified or verified.

In general, bioaccumulation lines of evidence are not explicitly considered, despite mercury and polonium being known to bioaccumulate (Section 3.8.2.2). The default Water Quality Guidelines for mercury suggest the use of guideline values one level of protection higher than the designated ecosystem condition category (Section 3.8.3). Generally, this means 99% species protection levels compared to the default 95% species protection level. However, which assumptions are used to predict water concentrations from contaminated product concentrations (i.e. via the dissolution of mercury from a contaminated solid into surrounding seawater) need to be accounted for. The risk arising from the methylation of insoluble inorganic and element mercury species to the bioavailable methylmercury species is critical to understand this but generally not reported.

#### 4.1.4 Likelihood and exposure characterization

Likelihood is a concept that is not directly comparable to the exposure characterization component of an ecological risk assessment. Statements describing discrete, event-based probabilities are not relevant to contaminant impacts. Reframing the likelihood to consider the likelihood of harm arising from the defined exposure pathway for a contaminant to an exposure pathway would be an alternative approach. However, Environment Plans generally do not consider exposure pathways to ecosystem receptors to support the hazard assessment to a management objective. Once exposure pathways have been developed for contaminants to ecosystem receptors, the spatial and temporal considerations can be made. For example, if a contaminant disperses it is likely to dilute. What constitutes a dilution that does not cause harm (or harms an acceptable spatial range) depends on the management objectives.

An example of a comprehensive exposure assessment of NORM contaminants in offshore oil and gas productions is given by Akarsu et al. (2001) who considered conservative exposure paths for NORM-contaminated products arising from oil and gas operations in the Gippsland Basin and calculated expected dose rates to humans and other ecosystem receptors – all of which were below the then accepted international limits.

##### 4.1.4.1 Spatial scale

The spatial extent of environmental management is not well defined by the *OPGGS Act*, *Environment Regulations*, NOPSEMA, or Environment Plans. Were they linked to defined management objectives, they could be more explicitly defined. For example, to deliver equal or better biodiversity outcomes compared to the baseline environment condition may require only a narrow spatial consideration of toxicity and population changes along the epibenthic and benthic communities around a pipeline. However, ensuring that there is no risk to the commercial, recreational, and indigenous use of marine resources may require consideration of bioaccumulation within the broader food web if exposure pathways show a bioaccumulation potential (discussed in Section 3.8).

Operators have a well-established capability to assess the spatial dispersion of contaminants. For example, dispersion modelling of contaminant plumes (including hydrocarbons, treated seawater, produced water, and sewage) are routinely conducted. Similar studies have been applied to COPCs. For example, the risk of produced formation water to marine organisms has been assessed with dispersion modelling compared against environmental reference levels or direct toxicity assessments in a number of Environment Plans (Esso Australia Resources Pty Ltd, 2019a; Shell Australia Pty Ltd, 2016; Woodside Energy Ltd, 2019).

To make decisions about the acceptability of decommissioning contaminated pipelines, contaminant concentrations and inventories of products need to be converted into expected concentrations in environmental media using dispersion models and accounting for environmental transformations. For example, this was achieved in one Environment Plan using a conservative assumption that the pipeline will mix with sediment at a ratio of 1:1. Direct comparison of the contaminated product with environmental reference levels also occurs (Woodside Energy Ltd, 2020a). This approach effectively assumes no dispersion or mixing and is the most conservative assessment that can be made.

Dispersion and environmental transformation leading to partitioning have also been assessed for NORMs. To undertake dosimetry modelling, predicted activity levels of radionuclides in seawater and sediment media are required. Santos Limited (2020) achieved this using a combination of dispersion modelling from an open pipe point source coupled to dosimetric modelling. The dosimetry modelling

may have used default parameter values for partitioning coefficients and organism concentration ratios, which is appropriate for these types of assessments, to determine the distance at which dose rates would be higher than the appropriate environmental reference level. This method can be considered best practice; however, the use of default parameters introduces high uncertainties into risk assessments (discussed in Section 3.8.2.3).

#### 4.1.4.2 Temporal scale

The temporal extent of environmental management is well defined. Ecologically sustainable development (one of three environmental management goals of the *Environment Regulations*, Section 3.2.2) is defined by 5 principles, two of which are relevant to temporal scales of management (*EPBC Act ss. 3A a and c*): decision-making processes should effectively integrate both long-term and short-term economic, environmental, social and equitable considerations; and, the principle of inter-generational equity—that the present generation should ensure that the health, diversity and productivity of the environment is maintained or enhanced for the benefit of future generations. ARPANSA similarly address the temporal scale of management as an objective in ARPANSA (2020) by stating: people and the environment, present and future, must be protected against radiation risks. These suggest that the whole life of contaminated products needs to be considered. This may be 10-100s of years for  $^{210}\text{Pb}$ , 1000s of years for  $^{226}\text{Ra}$ , and indefinitely for mercury.

Environment Plans typically consider temporal factors by the degradation rates of infrastructure containing contaminants. This is sometimes related to contaminant leaching rates but, temporal considerations of environmental transformations (such as methylation rates for mercury) or radionuclide decay are typically not considered. Temporal scales are particularly important for NORMs because of radioactive decay and ingrowth. This can lead to increased risk or reduced risk, depending on the radionuclide and timeframe (Figure 3.15). For example, Santos Limited (2020) considered  $^{210}\text{Pb}$  and  $^{210}\text{Po}$  contaminants in a capped pipeline. The radionuclides' decay rates were compared to pipeline corrosion times and it was concluded that the radiation risk will be negligible by the time the pipelines corrode.

#### 4.1.5 Cumulative impacts

Contaminated products may contain multiple contaminants, where each will contribute to the overall risk. Therefore, the cumulative risk of all components should be considered. Section 3.8.3 discussed some options used to consider cumulative impacts. Operators have detailed some approaches to considering cumulative impacts in their Environment Plans, typically through wet testing / direct toxicity testing of contaminated products. These are typically done for produced formation waters disposed to seawater during operations. For example, produced waters from the OKHA FPSO are collected and used to create a dilution series used in direct toxicity assessments through ecotoxicological testing (Woodside Energy Ltd, 2019). These data are used to model the safe dilution required to meet a defined level of ecosystem protection for the management objectives. A similar approach has been used to assess produced formation water discharge in the Gippsland Basin.

#### 4.1.6 Approaches to ecological risk assessments in other industries

Approaches to the management of COPCs in other Australian industries have focused on improving available data that can be used to parameterise risk models, such as the ERICA tool for NORM. This has largely meant collecting more concentration ratios for organisms relevant to exposed sites.

#### 4.1.6.1 Ranger Uranium Mine

Data limitations hampered ecological risk assessments for radionuclides arising from uranium mining in the Alligators River Region of the Kakadu National Park (Hirth, 2014). It was identified that more site-specific data was required to parameterise risk assessment tools, reduce conservatism, and ensure local biodiversity and ecological integrity is preserved (Copplestone et al., 2020). These data gaps were addressed by field studies which collected representative flora and fauna and measured more than 10,000 concentration ratios. These were used to develop a site-specific radiological quality guideline values (Doering and Bollhöfer, 2016; Doering et al., 2019). This approach used the UNSCEAR benchmark limits of 100 µGy/h for soil and 400 µGy/h for aquatic environments (UNSCEAR, 1994; UNSCEAR, 2008). The most exposed organism was identified and the maximum concentration of radionuclides that could be present in the respective environment without exceeding the benchmark limit was calculated (this is equivalent to the EMCLs presented in Figure 3.34; however, they were derived using site-specific species data). Uncertainty was accounted for by using the 95% percentile value. Site-specific water quality guideline values assessing the chemical toxicity of uranium and magnesium, were also developed for the local creek system (van Dam et al., 2017; van Dam et al., 2010).

#### 4.1.6.2 Uranium mining in Arid and Semi-arid Regions

Mining in arid and semi-arid regions of Australia may produce NORM-contaminated by-products that pose a risk to local ecosystems. The NERA project 'Improving environmental impact assessments for life of mine activities' (<https://www.nera.org.au/NERA-projects/Radiological-Risk-Assessments>) adopted a similar approach as was taken with the Ranger Uranium Mine to improve radiation risk assessment by improving the parameters that can be used in assessment tools. Soil, flora, and opportunistic fauna samples were collected to determine CR ratios, some of which were higher than reported CR ratios in existing international databases. The team also found seasonally-dependent uptake of radionuclides that were not radionuclide or species specific.

#### 4.1.6.3 Nickel mining risk to tropical marine ecosystems

Nickel mining in tropical regions may pose a risk to marine ecosystems through tailings discharge. Water quality guidelines existed, but represented mostly temperate organisms rather than important local representative species. Toxicity testing was conducted using tropical marine organisms which were then used to develop a new water quality guideline value specific to tropical marine ecosystems. Chemical characterization of mine tailings to determine environmental transformations and exposure pathways were also undertaken (Gissi et al., 2020).

## 4.2 Inventories

The inventory of mercury and NORM in subsea oil and gas infrastructure subject to decommissioning decisions is reviewed in this section. Inventory here means concentration, mass, speciation, and distribution. This section contributes to fulfilment of objective 3.

### Key findings:

- Each of the major oil and gas-producing basins (Gippsland, Otway, Carnarvon, Browse and Bonaparte) have the potential to produce mercury and NORM-contaminated products in Australia.
- The presence and concentration of mercury and NORM in contamination products generally falls within concentrations reported in public literature from around the world.
- The inventory (including concentration, mass, speciation, and distribution) of contaminated products is not well characterised in infrastructure. In particular:
  - Contaminated-product mass and concentration are rarely quantified
  - The spatial distribution in infrastructure is not known
  - The speciation of mercury and NORM is not well known

### 4.2.1 Introduction

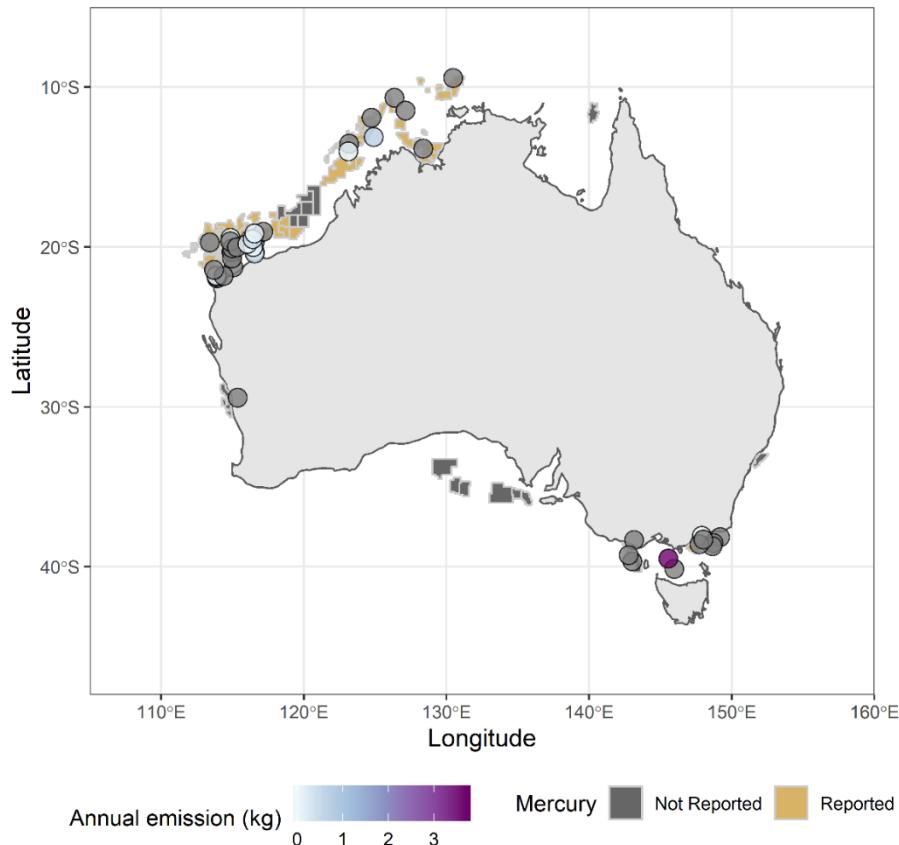
The speciation, magnitude, and spatial extent of mercury and NORM contamination in oil and gas pipeline infrastructure are not well defined. Sections 3.3.6 and 3.4.3 identified likely contaminated products based on the chemical behaviour of COPCs in oil and gas systems, as well as reported contamination of COPCs from publicly available sources. This section supplements this by presenting the industry-provided contamination data to determine the extent to which the inventory definition might be improved or whether, based on discrete contamination measurements, the inventory has been formally quantified.

### 4.2.2 Mercury

This review has found that all of Australia's oil and gas basins have the potential to produce mercury contaminated products. The concentrations of contaminated products are within global ranges for similar contaminated products. Regional variation of mercury contamination may occur either with or between the basins of interest with exploration and appraisal fluid samples indicating a potential trend of higher concentrations with the Bonaparte basin. However, limited data prevents any specific conclusions about regional variability being drawn.

#### 4.2.2.1 Geographical distribution

Interrogation of EPs and the National Pollutant Inventory for the year 2017-2018 showed that mercury was present in all oil and gas-producing basins (Figure 4.1). The annual emissions to water (possibly via produced water over boarding) were generally <0.7 kg, with one operation in the Otway Basin emitting 3.7 kg of mercury.



**Figure 4.1** Mercury identified in offshore oil and gas operations. Oil and gas basins are coloured according to whether mercury contamination was reported in any Environment Plans within that basin. Points are oil and gas projects and are coloured on a purple scale by the annual emission of mercury to the aquatic environment reported in the National Pollutant Inventory for 2018/2019. Grey points and basins represent no reported mercury emissions Coloured basins should not be taken to mean all titleholders with activities in the basin have reported mercury.

Very few Environment Plans specifically assessed the risk of mercury in the marine environment. This may be because discharges to the marine environment is limited during the operational phase, or because mercury is considered under an aggregated category of ‘metal’ or ‘pollution’ risks. Nonetheless, all major oil and gas producing basins had at least one operation reporting mercury contamination in either an Environment Plan or the National Pollutant Inventory.

#### 4.2.2.2 National and global reported concentrations

The concentration of mercury in Australian well production and reservoir fluids is comparable to reported values around the world (Figure 4.2 and Table 3.3); however, no measurements were provided for crude oil products and the mercury content of Australian condensate was up to  $10^6$  µg/kg. These condensate values are 2-3 orders of magnitude higher than what is reported in literature. This could reflect a general lack of reporting in publicly available literature.

#### 4.2.2.3 Inventory quantification

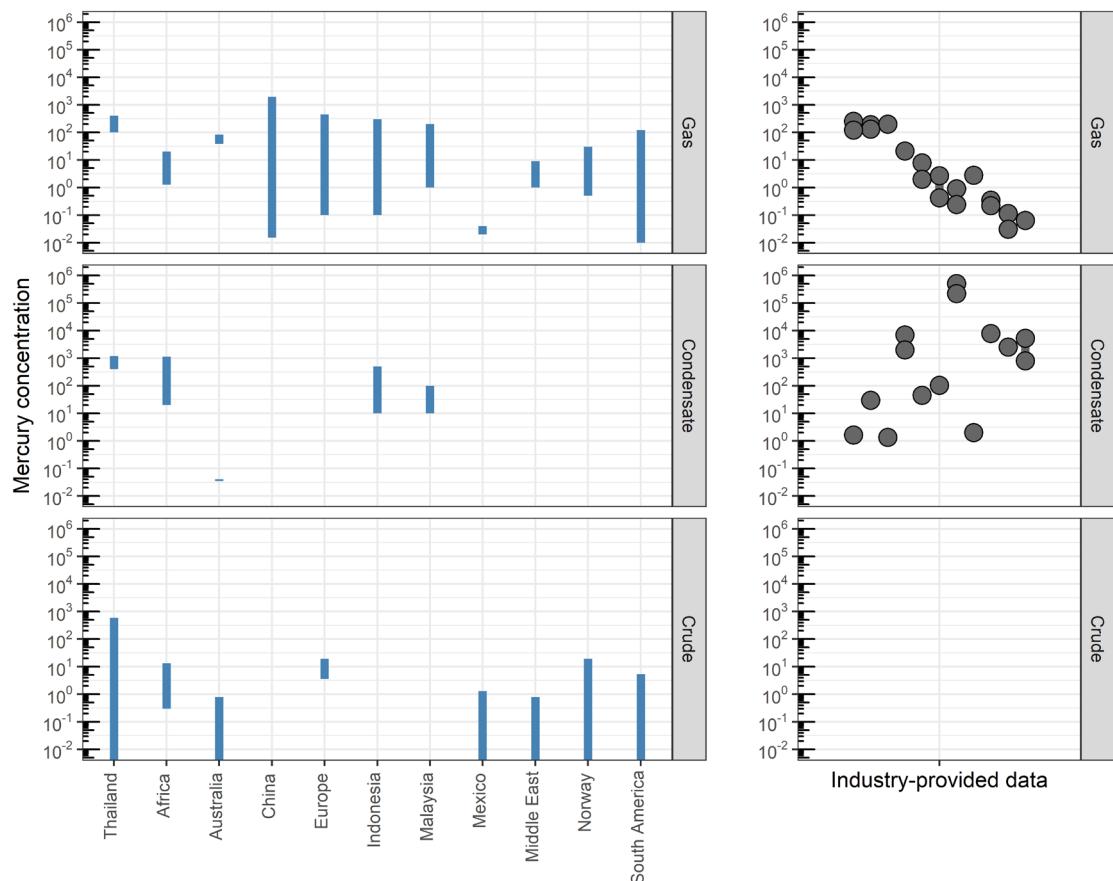
Industry-supplied mercury measurements from infrastructure or contaminated products has been compiled and summarised in Figure 4.3. These data are too limited to represent an inventory for any given asset or reflect the variability from Australian assets. Greatest detail was provided for pigging dust. These were collected from flow-coated pipes with un-coated field weld joints representing

approximately 1% of the total pipeline length. These data were compared to limited publicly available data for comparable products or pipelines in the USA (Wilhelm and Nelson, 2010) and Thailand (Silakorn et al., 2019) and an undisclosed source (O'Rear et al., 2016). Where suitable data on pipeline length or dimensions were provided, these data were converted to approximate mercury concentrations per meter of contaminated pipeline.

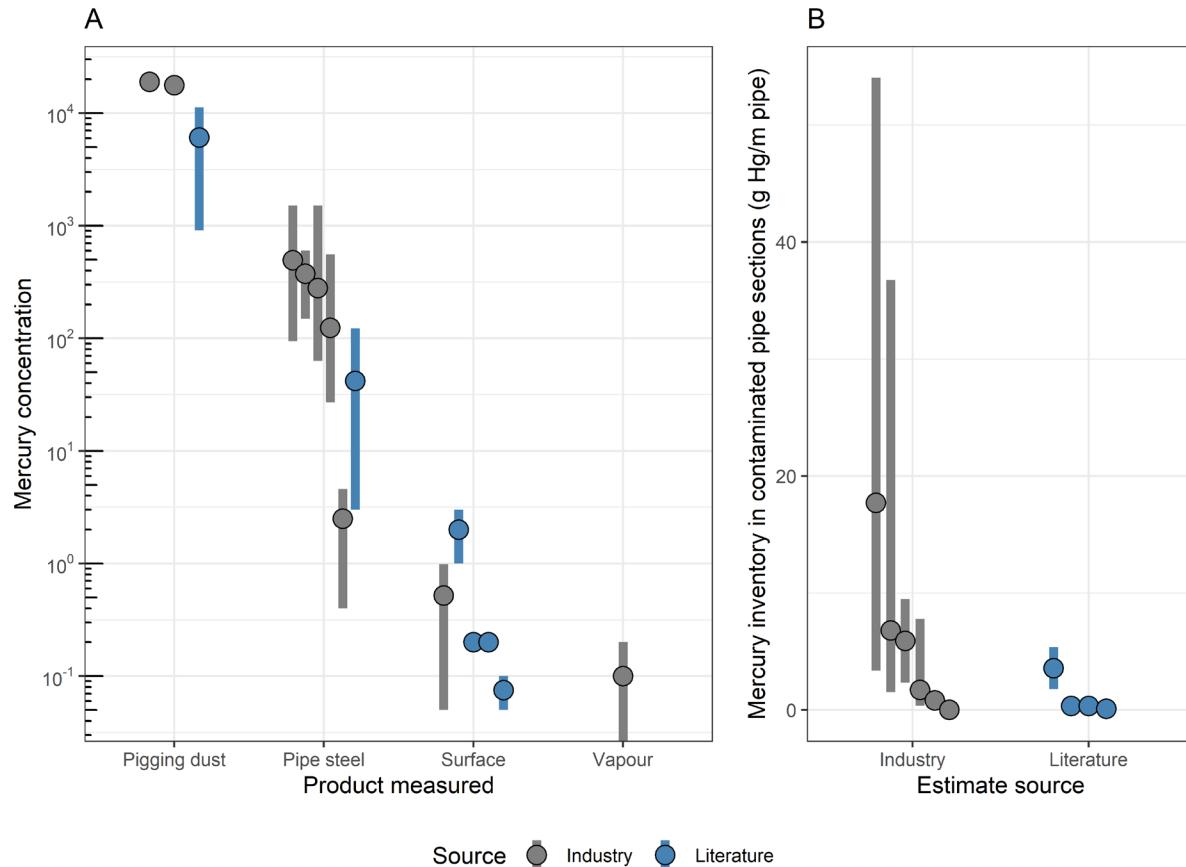
Pigging dust measurements were higher than literature-reported values while surface measurements were generally comparable. When compared on a mercury mass by length, some measurements were comparable to values reported internationally but some were up to 10-times higher. It is not clear what differences lead to this variability but it could reflect the presence of mercury adsorbed to steel corrosion products or entrained in a sludge product compared to mercury adsorbed to bare steel surfaces.

#### 4.2.2.4 Inventory uncertainties

Uncertainties regarding the inventory of mercury contaminated products arise from lack of *in-situ* measurement options, the measurement techniques currently used, unknown spatial distribution of mercury, the unknown interaction of mercury and different infrastructure material or corrosion products, and the inability to forecast contamination.



**Figure 4.2 Comparison of the mercury content in oil and gas products from different countries and those supplied by industry partners for this review. Country-specific data and sources are provided in Table 3.3. Units are  $\mu\text{g}/\text{m}^3$  for gas and  $\mu\text{g}/\text{kg}$  for condensate and crude products. Line segments represent concentration ranges while points represent high and low values of those ranges. No industry data was provided for crude oil products.**



**Figure 4.3 (A)** Industry-provided mercury measurements for infrastructure and contaminated products (grey) compared to values reported in public literature (blue, Wilhelm and Nelson (2010), Silakorn et al. (2019), and O'Rear et al. (2016)). A total of 9 measurements were provided. Points represent average concentration (if provided) or the midpoints of the range. Concentrations are measured in units of mg/kg for pigging dust and pipe steel,  $\mu\text{g}/\text{cm}^2$  for surface, and  $\mu\text{g}/\text{cm}^3$  for vapour. (B) Measurements were converted to concentrations by pipe length (in g Hg/m). Note the different scales (log and linear) for figures A and B, respectively.

## 4.2.3 NORM

This review has found that all of Australia's oil and gas basins have the potential to produce NORM-contaminated products (Figure 4.4). The activity levels of contaminated products are within global ranges for similar contaminated products. Regional variation of NORM contamination may occur, as has previously been suggested in a study of produced waters from the Gippsland Basin (Akarsu et al., 2001). However, limited industry-provided data prevented any specific conclusions about regional variability being drawn.

### 4.2.3.1 Geographical distribution

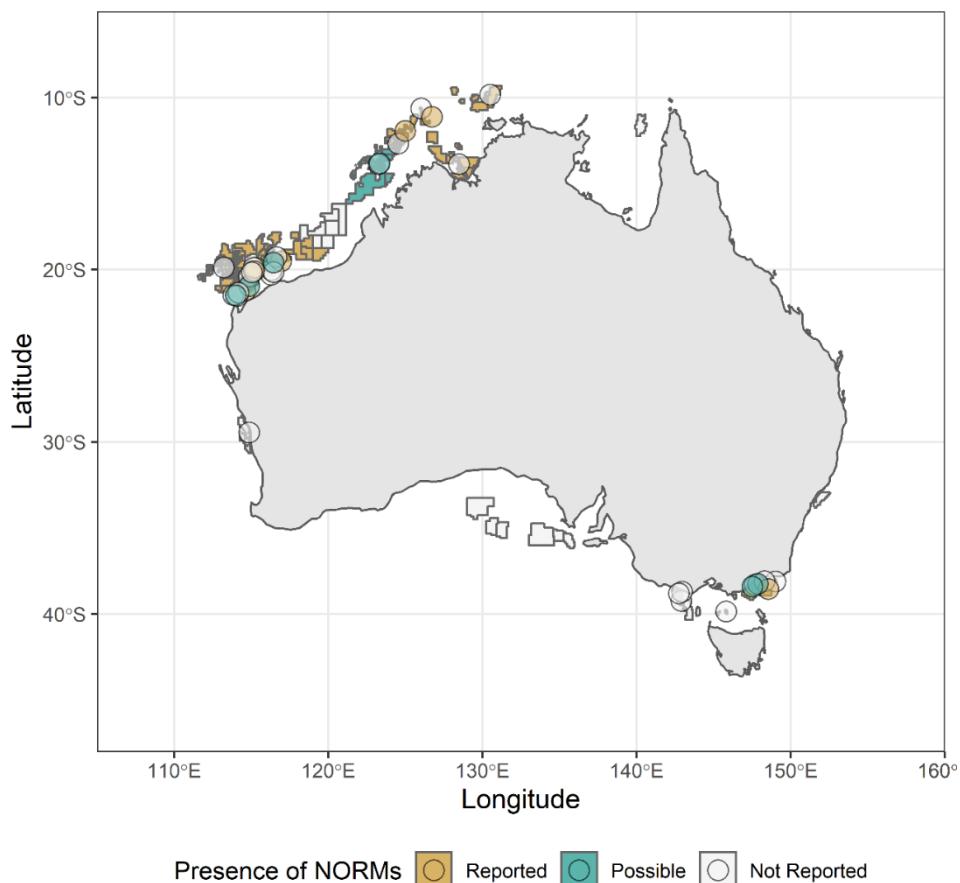
NORM-contaminated products have been identified by operators in the Carnarvon, Gippsland, and Bonaparte basins. Furthermore, Environmental Plans submitted to NOPSEMA have indicated an expectation that NORMs may be produced during operations in the Browse Basin. This suggests a ubiquitous geographical distribution of NORM-contaminated products in nearly all offshore oil and gas producing basins, Figure 4.4. This assessment was based on reported NORMs in environment plans for decommissioning, operations, cessation, plug and abandon, or infrastructure modification activities, so may not have captured all reported incidences of NORM.

Reasons why NORM contaminants may not have been reported by some operators in basins that are known to produce NORM contaminants include:

- Difficulty in measuring NORM *in situ*
- Different extraction and pipeline conditions
- Use of mitigation techniques such as scale inhibitors
- Age of activity or infrastructure

#### 4.2.3.2 Reported NORM-contaminated products and activities

Industry-provided data included 8 measurements of NORM-contaminated products (specific activities in Bq/g, Figure 4.5) and 16 surface activity levels (from handheld devices, in Bq/cm<sup>2</sup>) for this project (Figure 4.6). Measurements provided included specific activities of radionuclides in hard mineral scale and pigging dust, and the surface activity (i.e. radiation measured with a handheld device) from the interior and exterior of different retrieved infrastructure. The industry provided data supports the geochemical theory behind the formation of NORM-contamination products (Section 3.4.3). I.e. shows that in pipeline scales <sup>226</sup>Ra and <sup>228</sup>Ra are precipitated in the absence of their long-lived parent radionuclides <sup>238</sup>U and <sup>232</sup>Th and that <sup>222</sup>Rn partitioning leads to the accumulation of unsupported <sup>210</sup>Pb and <sup>210</sup>Po (i.e. <sup>210</sup>Pb in the absence of <sup>226</sup>Ra).



**Figure 4.4 The presence of NORM reported in Environment Plans.** Coloured points indicate the presence of NORMs by oil and gas project. Where NORM contamination was identified the corresponding basin was coloured accordingly. Coloured basins should not be taken to mean all titleholders with activities in the basin have reported NORM.

Radium-contaminated scale measurements were reported for two pipelines from different fields. In all measurements,  $^{226}\text{Ra}$  was  $> ^{228}\text{Ra}$  but there were differences in the ratio of  $^{210}\text{Pb}$  to  $^{226}\text{Ra}$ . At equilibrium  $^{210}\text{Pb}$  and  $^{226}\text{Ra}$  will be in secular equilibrium and have equal specific activities. Scale from one pipeline had a ratio of  $^{210}\text{Pb}$  to  $^{226}\text{Ra}$  was 0.34, indicating that secular equilibrium had not been established. Based on the activity of  $^{210}\text{Pb}$  (and assuming that  $^{210}\text{Pb}$  was only produced from  $^{226}\text{Ra}$  decay), the radium scale could be aged at approximately 13 years old. This is much younger than age of the pipeline (10-20 years old) and time since production ceased (5-10 years). This would indicate that radium scale was only formed part-way through the production life of the pipeline, possibly reflecting the ‘breakthrough’ point of seawater in the production fluids.

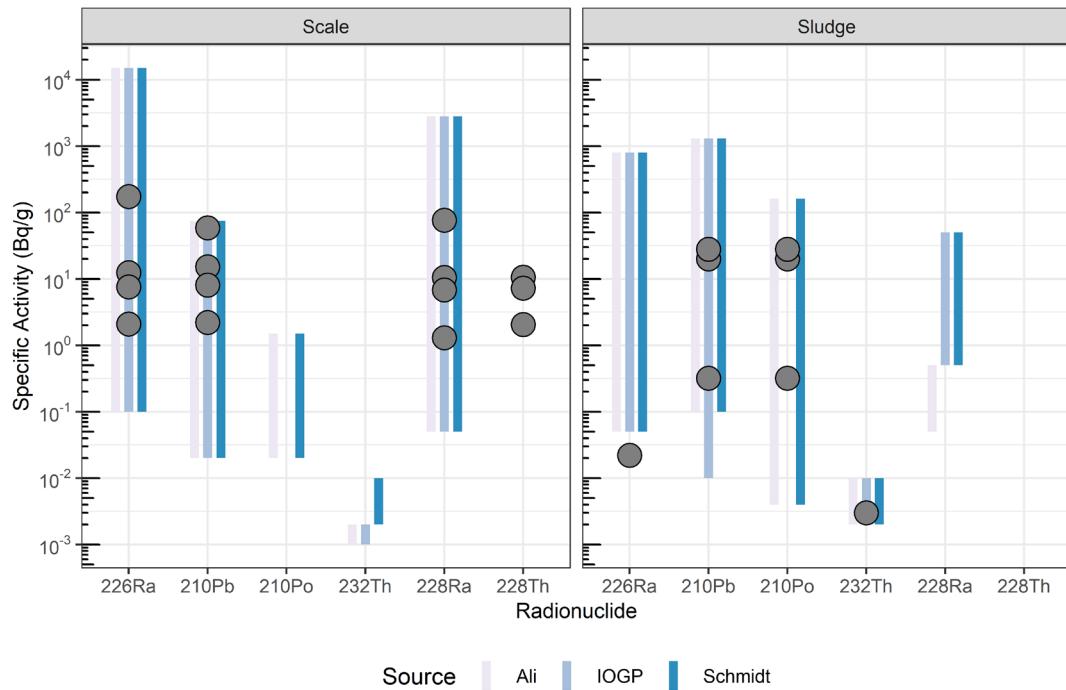
A different result was found for the hard scale in the other pipeline, which had a  $^{210}\text{Pb}$  to  $^{226}\text{Ra}$  of  $1.1 \pm 0.1$ . This indicates enrichment of  $^{210}\text{Pb}$  relative to  $^{226}\text{Ra}$  as a ratio of 1 would indicate secular equilibrium. This could only arise from the removal of  $^{226}\text{Ra}$  or the deposition of  $^{210}\text{Pb}$ . The deposition of  $^{210}\text{Pb}$  is the more likely hypothesis because even if scale is inhibited,  $^{222}\text{Rn}$  will be produced and can deposit  $^{210}\text{Pb}$ . A mass of 30 kg was recovered from an oil-system flexible riser which we have estimated to be 130 m to 500 m in length based on the depth of the well. This is the only ‘inventory’ of NORM that was provided and key information about the spatial distribution of NORM within the pipeline and the length of the pipeline limit an understanding of the overall NORM inventory.

Pigging dust measurements from a dry gas export pipeline reported  $^{210}\text{Pb}$  and  $^{210}\text{Po}$  contamination in secular equilibrium with one another. This demonstrates a  $^{222}\text{Rn}$ -only source of radionuclides that had deposited in a manner that could be collected by a pigging operation. The radionuclides were in a matrix of iron oxide indicating that it was collected alongside a corrosion product. Unlike mercury, however, it is unclear whether there is a relationship between corrosion products and  $^{210}\text{Pb}$  adsorption. The efficiency of the pigging operation and the pipe dimensions are unknown so an inventory could not be determined.

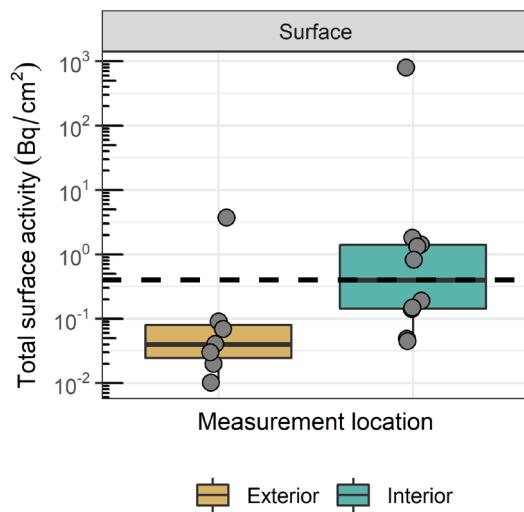
The specific activity measurements of NORM-contaminated products support radium-contaminated scale and  $^{210}\text{Pb}$  films as the two major types of NORM contamination that will be found in pipelines subject to decommissioning decisions. The specific activity measurements were compared against literature-reported ranges for equivalent contaminated products, reported by Ali et al. (2019), IOGP (2016b), and Schmidt (2000). The surface measurements, without context provided by the provenance of the infrastructure and corresponding measurements of the specific activity of the NORM-contamination product, provide little information beyond indicating the presence of radiation.

These radionuclide specific activity ranges broadly corroborate the review undertaken by Smith (2011) which found that worldwide NORM activity levels in contaminated products ranged from 0-37,000 Bq/g. The Australian median activity level for contaminated products was also similar to the worldwide median, at approximately 30 Bq/g. Data provided by industry for this project was within ranges published in literature ranges except for a  $^{226}\text{Ra}$  measurement in a ‘sludge’ product. This is likely due to the wide definitions of sludge products used in literature, which may include pipeline depositions, scrapings, surface dust, corrosion products (such as those collected from pigging operations) and sludge as defined in Section 3.4.3 (Schmidt, 2000).

Publicly available measurements of NORM-contaminated products are rare, with many studies, reviews, and industry reports detailing activity ranges for contaminated products. However, it is clear that Australian NORM-contaminated products are similar to those reported elsewhere, and that environmental conditions and operator-specific extraction processes and infrastructure play important roles in the development of NORM-contaminated products.



**Figure 4.5** Measured radionuclide activities in NORM contaminated product provided by industry participants to the NDRI project. Grey circles indicate individual measured values. Blue bars represent radionuclide activity ranges reported in literature. Publicly available NORM measurements were sourced from Ali et al. (2019); IOGP (2016a); Schmidt (2000).



**Figure 4.6** Surface measurements of total NORM radioactivity provided by industry participants to the NDRI project. Grey circles indicate individual measured values. Box plots show summary statistics of the measurements including the median (horizontal black line), 25% and 75% quartiles (lower and upper box extents, respectively) and whiskers representing 1.5x the interquartile range. The dotted horizontal line indicates a surface activity of 0.4 Bq/cm<sup>2</sup> which is the clearance criteria for the transportation of radioactive substances. This is of limited to no value in the determination of ecological risk.

## 4.3 Measurement techniques

This section reviews sampling and measurement techniques utilised by industry for NORM and mercury contaminant quantification in subsea pipeline infrastructure to provide insights as to what can be practically executed today. It also informs the extent to which the contaminants can be appropriately quantified for subsequent decommissioning risk assessment purposes (i.e. if information of NORM and mercury speciation, activity levels and concentrations, and spatial distributions are known). In doing so this section contributes to meeting objective 2.

### Key findings:

- There are no commercially-available *in-situ* measurement techniques for mercury and NORM in subsea pipelines.
- Handheld radiation detectors cannot be used to assess ecological risk. If coupled to ROV to assess pipelines it can provide information about spatial distribution and relative activity differences.
- Specific sampling, handling, preservation, and measurement techniques may be required for contaminant speciation measurements.
- It is not well established what measurement detail (i.e. concentration, mass, speciation, or spatial distribution) is necessary for a robust ecological risk assessments.
- There are limited measurement techniques to verify the efficacy of mitigation options.

### 4.3.1 Overview of measurement techniques

Common challenges for both mercury and NORM measurements relate to the inaccessibility of contaminated products and lack of *in-situ* measurement technologies. Industry-provided measurements include products collected from cleaning activities, such as of pigging dust, or measurements of topside products. It is unclear the extent to which these measurements are representative of the contamination in infrastructure being decommissioned. Additionally, no examples were found or received describing the spatial distribution of mercury or NORM contamination in pipelines. This is relevant to understand the spatial variability in contamination concentration or inventory. A diversity of measurement types for both mercury and NORM were also provided. It is understood that these reflect the current range of measurements, which are often used for purposes other than ecological risk assessments (i.e. monitoring for occupational exposure).

Different measurement of products provide different indications of spatial distribution. For example, pigging dust may be used to measure mercury and NORMs. Pigging programs collect dust over large pipeline distances so spatial variability cannot be assessed. Furthermore, it is unclear whether concentrations or specific activities can be related back to total inventories in pipelines. Personal communication with ARPANSA highlights this as a key question for risk assessments. No reported technique exists to assess hard scales *in situ* so spatial distribution cannot be determined for these products.

### 4.3.2 Mercury

Measurements conducted by industry participants suggest similar if not the same methods as outlined and discussed in Section 3.5. Measurement techniques behind industry-provided data are summarized in Table 4.2 and are supplemented with relevant mercury-contaminated pipeline measurements published in the publicly-available literature. Section 3.3.6 identified a range of processes and outcomes associated with mercury-contaminated products which are expected to exist in oil and gas pipelines (i.e. adsorption and deposition of mercury on pipelines, adsorption in corrosion products, or ionic, organic and elemental mercury in sludge). The measurement techniques adopted by industry-participants are herein assessed in the context of these products to gauge whether there is the potential to over or under report different mercury species.

#### 4.3.2.1 Total mercury content

Industry-provided data of total mercury content was measured using strong-acid digestion followed by detection with instruments such as ICP-MS/OES or CV-AAS, or *in-situ* pXRF techniques.

Acid digestion followed by instrument detection was used on pipe coupons and dust collected from a pigging operation. This measurement approach can be expected to detect all types of mercury contaminated products present. Concentrated acid digestions are common sample preparation steps because they dissolve the host material and any recalcitrant product. However, the concentrated acids also transform all mercury species into the  $\text{Hg}^{2+}$  species, meaning that mercury speciation cannot be determined.

Pigging dust was analysed after total acid digestion, however, the mercury content in the coupon was measured after dissolving the mercury deposited only on the inner surface of the pipe. The amount of mercury detected in the acid-digest solution was converted to mg Hg/kg steel pipe from the volume of acid used and the original mass of the steel coupon before digestion. Acid digestion were performed twice to quantify residual mercury associated to incomplete initial digestion.

Total mercury content measured using pXRF *in situ* was also provided to the Research Team. Like acid digestion followed by instrument detection, pXRF measures total mercury which will be inclusive of all mercury-contaminated products but does not differentiate mercury species. pXRF measurements are given in counts per second (cps, discussed in Section 3.5.2.2) which can be converted to concentrations of Hg in  $\mu\text{g}/\text{cm}^2$ .

The mercury measurement executed on pipelines found in the literature along with the industry provided data are mostly limited to gas export pipelines and minimal to no information are available for pipelines transporting crude oil.

#### 4.3.2.2 Speciation measurements

Speciation of mercury-contaminated products in pipe scale and pigging dust was assessed by techniques such as XRD, SEM-EDX, and  $\mu$ -EDXRF. These methods are used along with total mercury measurements to identify inorganic species and provide elemental mapping within the sample substrate.

Pigging dust samples were analysed by x-ray absorption spectrometry and x-ray diffraction. These techniques identified that the pigging dust was predominately an iron oxide matrix (i.e. a corrosion product) and the mercury was only inorganic HgS.

#### 4.3.2.3 Measurement uncertainties

Pigging dust samples may have been subject to speciation transformations following collection. Samples were collected in 2014 and analysed >5 years later. The samples only showed inorganic mercury species, which could indicate that no organic mercury was present, or any organic mercury was oxidised during storage. As discussed in Section 3.5.2.1, samples are prone to mercury loss if not subjected to proper preservation and storage methods.

Pipe surfaces covered with a residual hydrocarbon layer could potentially absorb  $Hg^0$  and organic mercury by dissolution (Wilhelm and Nelson, 2010). As the hydrocarbon layer evaporates when a pipe is opened, organic mercury species may be lost by surface evaporation as these species are well-known to be more volatile compared to inorganic mercury as seen in Table 4.1. For example, dimethylmercury has been detected in the hydrocarbon phase (Wilhelm et al., 2006) and has a vapour pressure multiple orders of magnitude greater than the other mercury species. This means that sampling strategies will have to take into account these sample transformations during collection and storage.

Measurements by pXRF are susceptible to measurement uncertainties and require proper calibration. This is difficult given the lack of reference standards for mercury scale on pipe steel. Most mercury will commonly exist in the scale layer rather than residing inside the bulk of the steel pipe. Therefore, the dilution factor contributed from the ‘less contaminated’ bulk of the pipe needs to be taken into consideration to get a representative mercury concentration within a given pipe section. This is possible if the steel pipe thickness and the thickness of the scale layer is known. However, pXRF x-ray penetration depth depends on the density of the sample and varies with the depth and type of scale present on the pipe. The mercury scale thickness is usually reported as approximately 0.01-20  $\mu m$  in steel (Chaiyosit et al., 2009; Chanvanichskul et al., 2017). However, uncertainties remain, with scale thickness reported to vary from several microns to several inches. Furthermore, its distribution unlikely to be homogenous due to the turbulent flow of the fluids during service. This means that there may be uncertainty converting the mercury content of the surface scale to a whole-steel mercury concentration.

Table 4.1 Vapour pressure of various mercury species at 20 °C.

Mercury Species	Vapour Pressure (Pa)	Reference
$Hg^0$	0.171	(Huber et al., 2006)
$(CH_3)_2Hg$	$8.8 \times 10^3$	(ICSC, 2016)
$CH_3HgCl$	1.13	(NCBI, 2021)
$HgCl_2$	0.0128	(Bernard et al., 1997)

**Table 4.2 Current practice of mercury measurement in pig dust and decommissioned pipeline. Techniques include x-ray absorption spectrometry (XAS), inductively-coupled plasma mass spectrometry (ICP-MS) or atomic emission spectrometry (ICP-AES), portable x-ray fluorescence (pXRF), atomic absorption spectrometry (AAS), scanning electron microscopy (SEM) with energy dispersive x-ray (EDX), x-ray diffraction (XRD), energy dispersive x-ray fluorescence (EDXRF), glow discharge mass spectrometry (GDMS), and square wave anodic stripping voltammetry (SWASV).**

Source	Species	Technique (sampling/pre-treatment/analysis)	Specimen	Lab/in situ	Purpose of measurement
Industry Participant supplied pigging dust assessment report for a dry gas export pipeline.	Total Hg.	<ul style="list-style-type: none"> <li>Acid digestion using <i>aqua regia</i> + microwave decomposition.</li> <li>Successive analysis using ICP-MS and ICP-OES.</li> <li>Pre-concentration of cellulose.</li> </ul>	Pig dust.	Lab	
	HgS (alpha and beta polymorph).	<ul style="list-style-type: none"> <li>Successive analysis using XAS (x-ray absorption spectrometry).</li> <li>Extracted using dilute HCl (1-h contact time) and filtered (0.45 µm).</li> <li>Successive analysis using ICP-MS.</li> </ul>	Pig dust + cellulose (1:10 ratio).	Lab	Pigging dust characterization.
	Extractable by dilute acid.	<ul style="list-style-type: none"> <li>Incorporate crushed pig dust in food pellets for feeding of sea organisms.</li> <li>Dissolved Hg analysed using ICP-MS.</li> </ul>	Pig dust.	Lab	
	Bioaccumulation test.		Seawater.	Lab	Leaching of Hg from the food pellets into the water.
Industry Participant supplied - Carbon Steel Spool Sampling and Mercury Absorption into Pipeline Coating Study.	Total Hg at surface - Uncoated pipe, field joints (scale deposit, internal surface).	Surface measurement: pXRF (US EPA 6200 method, modified).		<i>In situ</i>	Determination of Hg contamination within the pipeline.
	Total Hg in coupon - Uncoated pipe, field joints (scale deposit and internal surface).	Acid digestion and AAS (US EPA 7473 method).	Spool pieces retrieved from offshore installation at Bayu-Undan.	Lab	
	Total extractable Hg by seawater - Uncoated pipe.	Extracted using seawater for 18 h (AS4439.9 method). Filtered solids determined by acid digestion-AAS (US EPA 7473 method).	Upstream of gas export pipeline, cut into smaller coupons using cold-cutting technique.	Lab	Potential leaching of Hg into the sea from contaminated pipeline.
	Adsorbed Hg - exposed to 90 µg/m <sup>3</sup> Hg <sup>0</sup> in N <sub>2</sub> at 90 bar, 53 °C, 13-d exposure (static test).	Acid digestion-AAS (EPA 7473)	APL 5L X65.	Lab	Adsorption of Hg <sup>0</sup> onto pipe surfaces (coated with flow coating and uncoated specimen).
Chanvanichskul et al. (2017) Public domain.	Speciation of Hg on pipe surface.	XRD		Lab	
	Total Hg at different depth profile within the scale and steel substrate.	<ul style="list-style-type: none"> <li>SEM-EDX.</li> <li>µ-EDXRF.</li> <li>Acid digestion with 10ml HCl (concentration not specified).</li> <li>Successive analysis using ICP-MS and GDMS.</li> </ul>	Steel pipe (top of line corroded).	Lab	Identification and mapping of Hg species of scale on pipe surface.
	Total Hg on pipe surface (top of line corroded pipe).	<ul style="list-style-type: none"> <li>Retrieval of pipe scale using 'Sampling pig'.</li> <li>Successive analysis in the lab using ICP-MS or AA.</li> </ul>	Scale deposited on internal pipe surface.	Lab	

Source	Species	Technique (sampling/pre-treatment/analysis)	Specimen	Lab/in situ	Purpose of measurement
Silakorn et al. (2019) Public domain.	Total Hg - ionic form (acid digested with <i>aqua regia</i> at 90 °C for 3 minutes until clear.)	SWASV	Pig dust.	<i>In situ</i> /Lab	Hg measurement in pigging dust.
Wilhelm and Nelson (2010) Public domain.	<ul style="list-style-type: none"> <li>• Mercury species in pipe scale and interior of pipe.</li> <li>• Steel pipe (ASME A106 grade B).</li> <li>• Location: Downstream of primary separator.</li> <li>• Exposure condition: 50 °C, wet gas, 5 mol% CO<sub>2</sub>, 4-5 ppm H<sub>2</sub>S, hydrocarbon mostly methane balance, 500-800 µg/Sm<sup>3</sup> Hg gas.</li> </ul>	SEM-EDX  XRD	Steel pipe  Pipe surface scale - scraped from pipe coupons.	Lab  Lab	Interaction of steel surfaces with Hg <sup>0</sup> .  Interaction of steel surfaces with Hg <sup>0</sup> (Presence of H <sub>2</sub> S and CO <sub>2</sub> ).
Kibogy (2010) Public domain.	Total Hg on surface of pipe.	pXRF	Steel pipe obtained from refinery torch cut then cold cut into sample panels.	Lab	Development of Hg standards for field calibration.

### 4.3.3 NORM

The primary purpose of NORM measurement data provided to the research team were to assess clearance criteria for the transportation and cleaning of infrastructure and to ensure occupational health and safety. The exceptions were measurements of radium-contaminated scale in two pipelines and measurements of pigging dust which were all measured with gamma spectrometry. Pigging dust samples were also analysed by alpha spectrometry to determine  $^{210}\text{Po}$  concentrations and XRD to identify the speciation of the dust matrix.

Not enough data was provided by industry participants to answer the following questions related to sampling:

- Whether topside detection of NORMs can be used to infer pipeline contamination
- The spatial distribution of contamination within pipeline systems
- Whether external measurements can be related to internal measurements

#### 4.3.3.1 Current *in-situ* topsides techniques

The mechanisms behind scale formation and deposition suggest that radium-contaminated scale will be spatially variable within oil and gas infrastructure (Section 3.4.4). However, topside infrastructure are accessible parts of the oil & gas production system and NORM contamination there may pose a risk to occupational health and safety, thereby requiring its monitoring.

Current measurement methods used for topside infrastructure include:

- Physical sampling of solid material inside vessels and pipework for laboratory analysis
- Surface measurements

NORM scale measurements provided by industry participants were collected from fragments following pressure-cleaned infrastructure. This provides a good sample to measure activity and radionuclide composition, but means that the scale thickness, distribution within the pipeline, and sometimes total mass are not reported.

Pigging dust from gas pipelines were measured for  $^{210}\text{Pb}$  and  $^{210}\text{Po}$ . Pigging dust provides a sample that can be measured by gamma or alpha spectrometry and reflects an averaged concentration across the length of pipe pigged. However, the spatial distribution of NORMs within a pipeline is unknown there is no way to relate pigging dust concentrations with absolute quantities of contaminated product because of the uncertainties around pigging efficacy to remove scale.

#### 4.3.3.2 Current *in-situ* subsea techniques

Options are limited for *in-situ* NORM detection because destructive entry is generally not possible and external detection methods have high uncertainties and interferences. For example, due to variable geometry between source and detector, attenuation from pipeline material and the seawater or sediment, and interferences from background radionuclides in sediment and seawater (Section 3.5.3.3).

*In-situ* detection of gamma radiation can be conducted using NORM detectors mounted on subsea ROVs (Nancekievill et al., 2018; Oceaneering, 2017). This approach can provide valuable qualitative data regarding the presence and spatial distribution of radium-contaminated scale in pipeline infrastructure. However, like any surface measurement technique, it cannot detect  $^{210}\text{Pb}$  and  $^{210}\text{Po}$  deposits in pipelines (which can be present in the absence of  $^{226}\text{Ra}$ , Section 3.4.3) because their

alpha and beta radiation is attenuated by the pipe material (see Section 3.5.3.3). This approach cannot be used to understand contaminant risk to marine ecosystems because measurements of counts per second cannot be converted to dose rates which are required to understand risk (Krieger, 2005).

#### 4.3.3.3 Surface contamination measurements

Surface measurements using handheld Geiger Muller or scintillator detectors provide an indication of radiation but do not inform on specific activity, radionuclide composition, and are susceptible to a range of interferences that cannot easily be controlled in the field. Section 3.5.3.3 outlined the challenges recording consistent and accurate surface measurements. Furthermore, the absence of detectable external surface radiation does not prove the absence of NORMs due to the different decay modes of radionuclides and the attenuation of radiation through scale and pipeline material (IOGP, 2016b). For example,  $^{226}\text{Ra}$  may be detectable but  $^{210}\text{Pb}$  will not be (Krieger, 2005).

Several meters on the market include the ability to report surface contamination in  $\text{Bq}/\text{cm}^2$ . For example, industry-provided data reported the use of Tracero NORM meters. These types of detectors are issued by the manufacturer calibrated to a  $^{226}\text{Ra}$  source. However, this type of detector does not consider measurement uncertainties including differences in measurement geometry, shielding of material (particularly for alpha and beta emissions, Section 3.4.1) or the contribution of mixtures of radionuclides. Therefore, the values reported are likely to severely underestimate the amount of radiation and cannot be used to infer radionuclide composition or the specific activity of the contaminated product.

## 4.4 Risk mitigation

The identification and assessment of risk provides the framework to also identify and apply controls for the mitigation or reduction of environmental impact severity. There is also an ordered hierarchy to those applied controls with elimination of the hazard the priority followed by substitution, engineering, and administrative actions. This hierarchy guides the judgement as to the effectiveness of the controls and therefore the final quantification of residual risk with the controls in place. Mitigation options currently applied by operators are assessed in this section, which in part meets objectives 3 and 4.

### Key findings:

- A range of mitigation options currently exist for mercury and NORM-contaminated pipelines. These follow one of three approaches:
  - Prevent the formation of contaminated products by using anti-scaling chemicals or flow coatings
  - Reducing the mass of contaminated product in pipelines subject to decommissioning by flushing/cleaning.
  - NORM contaminants may be isolated through capping to allow for the decay of contaminated products with shorter-lived head of chain (i.e.  $^{210}\text{Pb}$ ) radionuclides.
- Mitigation is a well-adopted risk-reduction approach in the absence of a full understanding of ecological risk, in line with the precautionary principle.
- The efficacy of different mitigation options is not well established for all contaminated products and infrastructure types.

### 4.4.1 Introduction to mitigation options

A range of control or mitigation measures have been employed to reduce the risk of NORM and mercury products in pipelines subject to decommissioning decisions. There are also engineering design decisions resulting in an unintended impact on the COPC risk reduction, the learnings of which could be utilised for future pipeline design.

In total, the current mitigation measures include chemical inhibition of the production stream, injected either in or close to the well bore, pipeline materials and internal coatings, seawater flushing or pigging of the pipeline, chemical cleaning, mechanical abrasive cleaning, and capping. While not all these mitigation measures have been applied to primarily address COPC risk they can influence and reduce the impact.

By way of example Figure 4.7 captures a range of inherent design attributes of a pipeline system exposed to a production stream of an operating asset(s) which contain mercury. The Research Team does not have any of the underlying data or assessment work to underpin the various decision points within this decision tree. However, the mercury concentrations in steel support the independent findings in Figure 4.3 and highlights a number of the parameters governing whether a pipeline can be cleaned of mercury deposits by pigging or flushing providing there is no mercury adsorption onto the steel surface. Mitigations allowing this include a cascade of: multiphase service, low levels of  $\text{H}_2\text{S}$ , CRA surface or a flow coating has been applied to the internal surface of the pipeline.

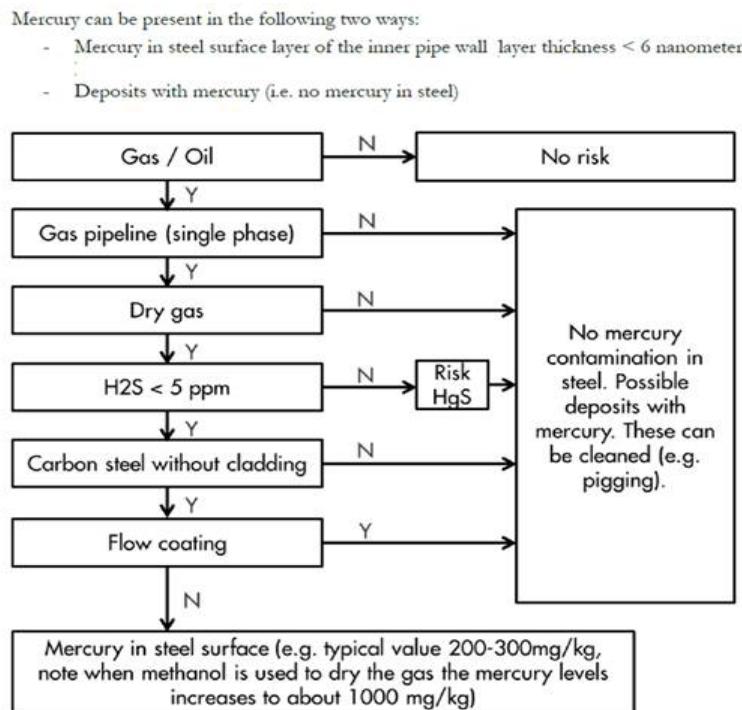


Figure 4.7 Cleaning regime of mercury contaminated steel (personal communication, Shell 2020).

#### 4.4.2 Chemical inhibitors

##### 4.4.2.1 Scale

The common method for preventing mineral scale formation in either well production tubing, production flowlines or topsides processing facilities is through the use of scale inhibitors during the producing life of the asset. The primary purpose of which is flow assurance to reduce flow restrictions which may reduce the production flowrates through physical restriction or, in the worst case, prevent production. From a NORM perspective this inhibition will also reduce the deposition and accumulation of BaSO<sub>4</sub> in any of the production system exposed to raw well fluid. A number of operators have reported scale inhibitor use as a mitigation measure of NORM contamination (Eni Australia Limited, 2013; Santos Limited, 2018; Woodside Energy Ltd, 2020b).

In general scale inhibitors work on an adsorption process suppressing the growth of the mineral crystal, altering its shape and adhesion properties (Khormali et al., 2016). They are injected in the most upstream location where the scale is likely to form and typically within the production tubing inhibiting the production stream to the primary separator. Their effectiveness or efficiency is variable and typically less than 90% for studies investigating both BaSO<sub>4</sub> and CaCO<sub>3</sub> scales (Boak et al., 1999; Khormali et al., 2016). Given this level of performance the use of scale inhibitors cannot completely mitigate the NORM risk and would need to be verified on a case by case basis.

##### 4.4.2.2 Asphaltene and waxes

For similar flow assurance drivers to scale inhibitors, asphaltenes and waxes are also typically managed in hydrocarbon production systems by chemical inhibition during the producing life of the asset. There was no industry based publication or data to confirm the use of wax or asphaltene inhibitors as a mitigation measure for NORM or mercury contamination.

#### 4.4.3 Flushing and pigging

Before decommissioning pipelines are depressurised and hydrocarbons within the structure are removed in line with relevant regional regulations by flushing and pigging. Pipelines are thoroughly cleaned whether being removed from the seabed or left *in situ*. In some cases, for pipelines with a small diameter or an unfavourable architecture, pigging cannot be carried out and the pipeline can only be flushed. Typically the pipeline is flushed with treated seawater until the oil content is reduced to an acceptable level.

A study commissioned by Bureau of Safety and Environmental Affairs suggests that the current industry practice for the flushing of out-of-service pipelines removes hydrocarbon based contaminants (i.e. oil in water) from the pipelines to the same level or lower than that of seawater. Flushing volumes of 1.5-2 times the pipeline volume at a rate of 100 gpm is effective (Manouchehri, 2017). However, in some cases deposits may not be fully removed and leaving the pipeline *in situ* to degrade may lead to release of contaminants such as mercury and NORM into the water column (Oil & Gas UK, 2013).

Seawater flushing and pigging is likely to remove contaminants that are relatively mobile (e.g. liquid hydrocarbons, loose corrosion products, sand, soft waxes or sludge) and is reported as a mitigation measure (Woodside Energy Ltd, 2020b). However, it is not clear the extent to which flushing and pigging can remove Hg<sup>0</sup> adsorbed to the steel pipeline wall, rigid corrosion products, mineral scales containing HgS and <sup>226</sup>Ra and films comprising <sup>210</sup>Po and <sup>210</sup>Pb.

#### 4.4.4 Internal pipeline materials and coatings

It has not been possible to find a definitive study or a series of published works correlating the various contamination products defined in Sections 3.3.6 and 3.4.3 and a quantification of their occurrence as a function of pipeline material type (i.e. carbon steel, CRA) or the internal coating system (i.e. epoxy coating or other). There are a sample of references which provide evidence to make statements of a general nature only.

The key differentiation for pipeline materials relates to the likelihood to form corrosion products which, for a carbon steel, may promote the accumulation of HgS, <sup>226</sup>Ra, <sup>210</sup>Pb and <sup>210</sup>Po as opposed to the characteristics of a CRA material. However it is not clear whether there is any differentiation of the Hg<sup>0</sup> adsorption to a carbon steel pipeline compared with a CRA pipeline excluding the impact from scale.

In terms of internal coatings (e.g. epoxy flow coat) there are a number of industry examples where the adhesion or accumulation properties might be similar to CRA. An example is illustrated in Figure 4.7 that is provided by one of the industry participants for cleaning mercury in contaminated steel. The information is consistent with the findings within Section 3.3.6 where majority of mercury in a flow-coated pipe will be present in the form of loose-adhering Hg<sup>0</sup> and HgS, which can be removed by pigging (Collet and Chizat, 2015).

Santos Limited (2020), in support of their Environmental Plan for the Bayu-Undan pipeline, have also conducted laboratory analysis on topsides dry gas export pipeline spools and test coupons with epoxy flow coating exposed to mercury contamination. This analysis was undertaken to compare the contamination levels in bare carbon steel field joint sections of the pipeline (which represent approximately 1% of the total surface area) with flow coated sections. The limited duration flow coating adsorption trials delivered mercury contamination below the detection limit. However, in the

case of non-flow-coated sections at the pipeline field joints, mercury was adsorbed in the steel surface and/or incorporated into the pipe scale.

#### 4.4.5 Chemical cleaning

In cases where it is known that high levels of residual mercury exist, for example in the Gulf of Thailand, it is common to employ an additional chemical cleaning step using lixivants to reduce any potential long-term risk of methylation that could arise from residual insoluble mercury. Lixivants refers to liquid chemicals that can be used to scavenge any residual mercury from the pipe after flushing and pigging. Majority of these lixivants are either acidic, strong oxidants and/or a complexing agent which removes mercury from the pipelines via oxidation to its more soluble forms. Acidic solutions are able to dissolve mercury alongside the iron oxide and iron sulfide but not HgS that are present within the pipe scale.

Similarly, in mercury contaminated pipelines that consist of primarily HgS, inorganic sulfide/polysulfide solutions are commonly employed due to their ability to dissolve both HgS and Hg<sup>0</sup> to yield soluble HgS<sub>2</sub>H<sup>+</sup> and HgS<sub>2</sub><sup>2-</sup> complexes as HgS is not soluble even in high concentrations of acids. A 1 wt% sodium sulfide solution was able to dissolve HgS to yield a 1663 µg/L mercury in solution (O'Rear et al., 2016). The use of another lixiviant comprising of 0.2 M Iodine (I<sub>2</sub>)/2 M potassium iodide (KI) has been reported to successfully remove adsorbed Hg<sup>0</sup> from contaminated carbon steel (Chaiyasit et al., 2009). Mercury that are present in the contaminated pipes in other forms such as HgS and HgO are able to be dissolved by the I<sub>2</sub>/KI lixiviant following an oxidation and complex formation mechanism, forming soluble HgI<sub>4</sub><sup>2-</sup> (Ebadian, 2001).

There is a commercial flushing chemical lixiviant known as MerCure that has been used in tests on contaminated sections of pipe from the Bayu-Undan basin (Santos Limited, 2020). The chemical composition has not been disclosed, but testings have shown removal efficiency of 98%. However, the basis for verification of its efficacy along the entire length of the pipeline in question is not determinable based on the information available.

In terms of scale removal (e.g. BaSO<sub>4</sub>), published in-service industry practice for decontamination of pipelines could not be identified but a number of acid and chelating agents have been used with production tubing (Kamal et al., 2018). It is not clear whether the application of these methods are effective and viable for subsea pipelines.

#### 4.4.6 Mechanical abrasive cleaning

Section 3.5.2.1 introduced a combined chemical and mechanical pipeline pig train cleaning system for the removal of scale from a carbon steel pipeline. The scale was found to contain HgS and HgCl (Chanvanichskul et al., 2017). There are two mechanical scale removal prototypes under development, one for combinational use with a chemical and the other for separate residual scale removal follow-up operation. No quantification of its efficacy has been provided, nor the pipeline diameter and length, other than the statement 'the pig can significantly remove scale'. No further publications were identified capturing further qualification or field trial testing by the operator (i.e. PTTEP) but it is understood the application was underpinning the leave *in-situ* position for a mercury contaminated pipeline in the Gulf of Thailand.

#### 4.4.7 Pipeline capping

As described in Section 3.7.4 the point at which seawater or sediment pore water can enter the pipeline increases the exposure and risk to the marine environment. This was described in context of pipeline degradation but is also applicable during subsea decommissioning operations. For the latter it is unlikely this exposure can be eradicated entirely due to construction practicalities and pipeline access requirements. Capping as a mitigation strategy has been referred to in various Environment Plans as a way of minimising NORM exposure to the environment (BHP Billiton Petroleum (Australia) Pty Ltd, 2017; BHP Billiton Petroleum (Australia) Pty Ltd, 2018; Esso Australia Resources Pty Ltd, 2019b; Santos Limited, 2020). Capping normally occurs shortly (within a week) of pipe disconnection. However, Esso Australia Resources Pty Ltd (2018) plan to only cap pipes within one year of disconnection and assessed NORM impacts to marine organisms in their accepted Environment Plan.

Based on the pending pipeline decommissioning plans, Santos Limited (2020) and Woodside Energy Ltd (2020a) both have opted to install caps on the end of the respective pipelines and in the case of the Bayu-Undan pipeline subsea operations are driving an expected open pipe duration of 7 days. The quantification of risk reduction as a result of capping has not been quantified in either report other than characterising it as a precautionary measure.

#### 4.4.8 Verification and monitoring

What is currently not clear across all mitigation measures introduced above, is the quantification of effectiveness and what specifically has been addressed in terms of verification and monitoring of the residual NORM and mercury contaminant levels along the full length of the pipeline.

## 5 Knowledge gaps and recommendations

This project aims to enhance the understanding of the risks of NORM and mercury in the marine environment, from decommissioned oil and gas infrastructure, to support science-based decommissioning decision-making. This literature review provides the background information required to frame and answer this project's objectives.

Section 3 has outlined the components of an ecological risk assessment and outlined the role of each component (problem identification, exposure characterisation, hazard characterisation, and risk analysis) in contributing to an understanding of risk. Section 4 considers the current application of risk assessments from industry's understanding of NORM and mercury in their systems and environmental context. This section highlights the knowledge gaps and provides recommendations that are structured along the problem statements provided by industry partners.

The recommendations build towards the development of an ecological risk assessment that can be generalised to all contexts but provide the flexibility to tailor the assessment and risk quantification to local contexts. These recommendations are tabled without a detailed assessment of effort level, resourcing and cost or a desk top only constraint. They are also assuming sufficient data (i.e. quality and quantity) is available to execute a fit for purpose research plan.

### 5.1 Understanding thresholds of NORM and mercury in the environment

*The Australian oil and gas industry seeks to understand the threshold levels at which NORM in subsea production equipment and mercury in subsea gas transmission systems in the offshore environment become a concern during and after decommissioning, relative to the receiving environment, specifically: levels/dosage/concentration, form/chemical species and likely behaviour (how are they likely to leach over time and what are the likely pathways) if NORM and mercury is present in O&G infrastructure.*

This problem statement touches on all aspects of an ecological risk assessment for the development of threshold levels. The Research Team believe 'threshold levels' to mean exclusion criteria, here defined as the concentration of a contaminated product or pipeline below which there is robust evidence that it does not pose an unacceptable ecological risk. This is aligned with the definition provided by the IAEA and ARPANSA for material requiring regulatory concern (Section 3.2.4). An equivalent criteria does not exist for metal contaminants under the Water Quality Management Framework. The use of 'thresholds' is distinct from environmental reference levels which are used at the hazard characterisation step of an ecological risk assessment to understand the impact of a contaminant concentrations exposed to an ecosystem receptor (discussed in Section 3.8.3).

#### Key knowledge gaps:

1. How ecological risk assessments are applied to meet legislative requirements
2. Whether exclusion-type criteria ('thresholds') can be derived for mercury and NORM for decisions around subsea oil and gas pipeline decommissioning
3. What is the inventory of contaminated products – including the contaminated products' form, species, mass, and distribution in pipelines

It is not clear whether or not exclusion-type criteria can be developed given the uncertainties in risk-model parameters, the limited available inventory data, and the variability of each operator's context. However, this review has defined the process which can be undertaken to develop them. These components and the knowledge gaps related to them are:

- **Problem identification:** The extent and magnitude of contamination is not well defined, nor are the objectives of the ecological risk assessment process. Key limitations around the problem identification step include:
  - Management objectives are not well defined and should be developed with the participation of all stakeholders, including regulators.
  - The understanding of contaminated product types was derived from literature and the chemical and physical properties of mercury and NORMs (Sections 3.3.6 and 3.4.3), and corroborated by limited measurements provided by industry partners to the Research Team (Section 4.2). However, uncertainties remain around concentrations, speciation, mass, and distribution in pipelines.
  - The mechanisms behind the formation of contaminated products is uncertain. This includes understanding the extent to which radium-contaminated scale forms in the absence of seawater injection, whether loose depositions (such as of elemental or ionic mercury, or  $^{210}\text{Pb}$  deposits from  $^{222}\text{Rn}$  decay) can accumulate in pipelines carrying liquids such as condensates, water, or multiphase fluids, whether the mercury species detected in oil and gas fluids (i.e. as reported in Table 3.6) are present in mercury-contaminated products (Section 3.3.6), how different pipe types (e.g. carbon steel, CRA, flow-coated) affect mercury and NORM accumulation, and the physical and chemical variables that affect the accumulation of mercury and NORM.
- **Exposure characterisation:** The exposure pathways (including likely environmental transformations of contaminants, Section 3.7), the rates at which these occur, and how these are affected by environmental parameters (i.e. as presented in Table 3.19) need to be better understood.
  - Significant gaps in knowledge prevent our understanding of likely environmental transformations of contaminated products in marine systems. For example, the proportion of radium, radon, lead, and polonium that can leach from NORM-contaminated products into seawater, the methylation potential of different mercury-species, and how contaminated products will behave in anoxic conditions such as if buried in sediment.
  - Pipeline degradation mechanisms and timeframes are not defined, particularly for the range of *in-situ* pipeline systems (rigid carbon steel, rigid CRA, flexible unbonded pipe and the various internal and external coating systems).
- **Hazard characterisation:** The impact of mercury and NORM to ecosystem receptors is defined for toxicity but not bioaccumulation or biodiversity lines of evidence. How different environmental parameters modify these hazard impacts is also not well understood (i.e. as presented in Table 3.19).
  - Environmental reference levels for toxicity, biodiversity, and bioaccumulation impacts do not exist for all elements (Table 3.18). It is unclear whether site-specific levels can be derived with existing data.

- Environmental reference levels are based on data that may not be protective/may be overprotective of species and ecosystems relevant to subsea oil and gas decommissioning decisions. This is particularly the case for radiation dose-rate reference levels (such as the DCRLs, Section 3.8.2.3) which have a very limited basis of evidence relevant to Australian marine conditions. Similar limitations exist for other parameters (Table 3.16).
- **Risk analysis:** Using multiple hazard impacts in a weight of evidence assessment to quantify risk against management objectives is best practice (Section 3.9).
  - The need for multiple lines of evidence and their use meeting management objectives for legislative requirements can only be determined if the management objectives are defined. These are lacking so an understanding of risk is limited.
  - Cumulative impacts of multiple contaminants or the combined pressure from multiple infrastructure is not currently considered or known.
  - The efficacy of mitigation options (Section 4.4), and verification thereof, are not well understood

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## Recommendation

### Problem identification

- Develop a NERA-issued report (or peer-reviewed journal article) that summarises best practice assessment methodologies for NORM and mercury risk in marine ecosystems for management objectives required to meet legislative requirements of decommissioning. These reports should include input from NOPSEMA, ARPANSA, and DAWE to ensure that the developed methodologies harmonise across all legislative requirements.
- Develop case scenarios that reflect representative contamination inventories that also capture the range of environmental variables that can influence contaminant risk relevant to the ecosystems of Australian oil and gas producing basins.
- Identify opportunities to investigate the spatial extent and inventory of mercury and NORM contamination in infrastructure, e.g. through modelling approaches or greater sampling.
- Investigate the possibility of linking production history and mercury and NORM concentrations in production fluids to forecast contaminated product inventory.

### Hazard characterisation

- Undertake direct toxicity assessment of different mercury and or NORM contaminated products with representative species to inform on the joint hazard. In particular the joint presence of  $^{210}\text{Pb}$ ,  $^{210}\text{Po}$ , and mercury.
- Identify research options to develop more radiation effects and mercury toxicity data for marine organisms relevant to ecosystems where pipelines may be decommissioned *in situ*.
- Investigate whether environmental reference levels can be developed for bioaccumulation or biodiversity lines of evidence. For example, through research projects with direct toxicity assessments, the use of bioavailability-based measurements such as passive samplers and bioluminescent bacteria, field studies, or the use of baseline/background data.

### Exposure characterisation

- Knowledge gaps considered in Problem Statement 2 (Section 5.2).

### Risk analysis

- Consolidate all components of an ecological risk assessment and define the parameters and their uncertainties to identify whether ‘thresholds’ (exemption-based criteria for contaminants) can be developed.
- Identify the most important factors controlling exposure pathways and hazard impacts. This will guide future research objectives.
- Develop a tiered approach to radiological risk assessment. For example, using three tiers that increasingly require site-specific data to offset decreasingly conservative assumptions and reference levels. Such approaches are applied in the sediment quality guidelines for mercury (Simpson and Batley, 2016), the drinking water quality guidelines (NHMRC and NRMMC, 2011), and recommended by ARPANSA (2015b).

## 5.2 Understanding the fate of NORMs and mercury

*The Australian oil and gas industry seeks to understand the processes and exposure pathways whereby individual contaminants might (or might not) reach receptors after decommissioning.*

This problem statement asks for an understanding of the behaviour of mercury and NORM-contaminated products in the marine environment. This understanding underpins an assessment of likely exposure pathways from contaminated product to ecosystem receptors.

### Key knowledge gaps:

4. How the speciation and distribution of contaminated products in pipelines affects important environmental transformations controlling exposure pathways
5. How different environmental conditions affect environment transformations of contaminated products and their resulting mobility and bioavailability

Information exists on the behaviour of individual elements (including mercury, radium, polonium, and lead) in the marine environment. However, these data are based on the elements' speciation and phase at equilibrium in the environment (i.e. the behaviour of their likely speciation and form in the marine environment). This is not reflective of mercury and NORM-contaminated products, the speciation and form of which are ill defined (Section 4.2). Thus, significant knowledge gaps exist around the behaviour of the mercury and NORM-contaminated products in the marine environment. This review used inferences of likely mercury and NORM forms and species reported in literature (corroborated by industry-supplied data) to identify likely fates and exposure pathways. Key ecosystem receptors were identified including the epibenthic and benthic pipeline communities, and for the bioaccumulative elements mercury and polonium, the food webs of ecosystems (Section 3.6).

Conceptual models of possible exposure pathways were thus defined for mercury and NORM in the marine environment (Figure 3.24 and Figure 3.26). Important environmental transformations were identified as governing exposure pathways and thereby potential impacts. For mercury, this was the methylation of inorganic to organic forms (Section 3.7.1). For radium-contaminated scale this was the leaching of radium's decay products (particularly polonium), emanation of radon, or reductive dissolution of barium or radium sulfate (Section 3.7.3).

### Key uncertainties include:

- The inventory of contaminated products in subsea pipelines is not well characterised.
- Pipeline degradation timeframes to the point of seawater breakthrough are not well defined.

Uncertainties about the contaminant-specific behaviours and exposure pathways include:

#### Mercury:

- Environmental factors that affect the methylation and demethylation of mercury
- The propensity of different mercury species to undergo methylation
- Likely rates of methylation in different environmental conditions
- The influence of organic matter in facilitating leaching and methylation from contaminated products

## NORM:

- The extent of the external radiation risk to epibenthic and benthic communities
- The solubility of the decay products of radium in contaminated barite scale (particularly polonium)
- The amount of radon that can emanate from contaminated scale and its resulting dispersion
- The biomagnification potential of polonium in commercially-relevant seafood
- The dispersion re-oxidised radium from pore-waters to overlying waters
- The influence of organic matter in facilitating leaching and methylation from contaminated products

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## Recommendation

- Consider the influence of various pipe decommissioning and degradation scenarios in ecological risk assessments. For example, in the context of speciation changes or for radionuclide decay/ingrowth before corrosive breakthrough.
- Consolidate environmental factors that influence exposure pathways and apply case scenarios to examine whether literature-reported ranges can be used to constrain the uncertainty of these processes and their resulting influence on risk (e.g. Table 3.19).
- Assess how well existing parameters describing radionuclide transfer (such as published  $K_d$  and CR values, (Section 3.8.2.3 and Table 3.16) reflect the Australian marine context.
- Undertake experimental work to improve the environmental relevance of parameters used in risk assessment tools:
  - Partitioning coefficients for radionuclides in sediment and water conditions reflective of Australian marine environments
  - Methylmercury flux from sediments to waters (i.e. the net rate of methylmercury production and release)
  - Bioaccumulation potentials (particularly of mercury,  $^{210}\text{Po}$ ,  $^{210}\text{Pb}$ , and  $^{226}\text{Ra}$ )

## 5.3 Understanding of tools and procedures

*The Australian oil and gas industry seeks to improve its tools and procedures for measuring mercury and NORM associated with equipment/infrastructure in situ to inform decommissioning decisions and long term monitoring objectives.*

This problem statement deals with measurement technology for mercury and NORM. The options for subsea *in-situ* measurements were very limited and largely still in the conceptual or prototype phase. These included scraper/grinding or XRF-mounted pigs, or radiation detector-mounted ROVs.

### Key knowledge gaps:

6. **What measurements are required for ecological risk assessments**
7. **What are the current and future technology options for measuring subsea *in-situ* mercury and NORM contamination that can meet ecological risk assessment requirements**

To understand the risk of mercury or NORM-contaminated products in the marine ecosystem the concentration (or specific activity), mass, speciation, and composition of contaminated products may be required. An understanding of the spatial distribution of contaminated products in subsea pipeline infrastructure will also be required if decommissioning decisions about sections of a pipeline are to be made (i.e. remove contaminated infrastructure and decommission remaining infrastructure *in-situ*).

Measuring capability for mercury and NORMs in contaminated pipelines is primarily limited by access to pipelines *in situ*. Samples for analysis need to be representative of the contaminated product in the infrastructure. This is challenging because samples are difficult and costly to obtain and to analyse, and are likely to be spatially heterogeneous in infrastructure. Useful laboratory measurements of mercury and NORM contaminated products are well established and include ICP MS, XRF, XRD, gamma and alpha spectrometry, etc. (Section 3.5). However, sample preservation and handling considerations could be considered by operators in greater detail.

Key uncertainties around measurement capability in general include:

- How available NORM and mercury samples (including spool, coupon, pigging dust, and scale measurements) reflect actual mercury and NORM-contaminated product inventories
- Whether provided measurements were appropriately sampled, preserved, and stored for the measurement of different mercury and NORM species.

The technological development of *in-situ* subsea mercury and NORM measurement technology is potentially being delayed due to a lack of joint industry leadership and definition of need. The key requirements for these technologies and their potential limitations need to be clarified to ensure their development aligns with ecological risk assessment need.

Current limitations of measurement technologies for *in-situ* mercury and NORM detection include:

- How the shielding and variable geometry issues of subsea external NORM measurements relate to internal NORM-contaminated products (particularly the predominately alpha and beta radiation emitting  $^{210}\text{Pb}$  and  $^{210}\text{Po}$ ).
- The spatial resolution of XRF-mounted pigs given the required XRF measurement times and pig movement.

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## Recommendation

- Define the measurement requirements for ecological risk assessments of mercury and NORM in subsea oil and gas pipelines.
  - Ensure any developed environmental reference levels or indicators for hazards are coupled to measurements using available techniques.
  - Investigate how presently-available measurements (e.g. pigging dust, spools, coupons) can be used to infer the spatial distribution of mercury and NORM in pipelines.
  - Develop guidelines outlining the quality assurance requirements for in-field and laboratory measurements of mercury and NORM contaminated products.
  - For NORM contamination, investigate whether the absence of detection from external measurements is sufficient to inform an ecological risk assessment based on the different emission types of NORM-contaminated products, their decay rates, and the adoption of mitigation technologies such as pigging, flushing, and capping.
  - Investigate the potential of using models to forecast the inventory of contaminated products from concentrations in production fluids.
  - Conduct a mercury and NORM measurement technology review, identifying all options and their readiness level, determining which opportunities could potentially be utilised for *in-situ* measurement.
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## 6 References

- Aarkrog, A., et al., 1997. A comparison of doses from  $^{137}\text{Cs}$  and  $^{210}\text{Po}$  in marine food: A major international study. *Journal of Environmental Radioactivity*. 34, 69-90.
- Abai, M., et al., 2015. An ionic liquid process for mercury removal from natural gas. *Dalton Transactions*. 44, 8617-8624.
- Abbas, T., et al., 2016. Mercury capture from natural gas by carbon supported ionic liquids: Synthesis, evaluation and molecular mechanism. *Fuel*. 177, 296-303.
- Advisian Pty Ltd., Offshore Oil and Gas Decommissioning Liability (Australia): Executive Summary. National Energy Resources Australia, Perth, Australia, 2020, pp. 9.
- Ahmadun, F. I.-R., et al., 2009. Review of technologies for oil and gas produced water treatment. *Journal of Hazardous Materials*. 170, 530-551.
- Akarsu, E., et al., 2001. NORM in the oil and gas industry - exposure assessment and management plan. *The APPEA Journal*. 41, 737-747.
- Al-Masri, M. S., 2006. Spatial and monthly variations of radium isotopes in produced water during oil production. *Applied Radiation and Isotopes*. 64, 615-623.
- Al Nabhani, K., Khan, F. I., 2020. Nuclear radioactive materials in the oil and gas industry. Elsevier.
- Ali, M. M. M., et al., 2019. Concentrations of TENORMs in the petroleum industry and their environmental and health effects. *RSC Advances*. 9, 39201-39229.
- Allen, H. E., 1993. The significance of trace metal speciation for water, sediment and soil quality criteria and standards. *Science of The Total Environment*. 134, 23-45.
- Allen, H. E., et al., 1993. Analysis of acid-volatile sulfide (AVS) and simultaneously extracted metals (SEM) for the estimation of potential toxicity in aquatic sediments. *Environmental Toxicology and Chemistry*. 12, 1441-1453.
- Althaus, F., et al., 2015. A Standardised Vocabulary for Identifying Benthic Biota and Substrata from Underwater Imagery: The CATAMI Classification Scheme. *PLOS ONE*. 10, e0141039.
- Amato, E. D., et al., 2015. Metal fluxes from porewaters and labile sediment phases for predicting metal exposure and bioaccumulation in benthic invertebrates. *Environmental Science & Technology*. 49, 14204-14212.
- Amato, E. D., et al., 2014. Diffusive gradients in thin films technique provide robust prediction of metal bioavailability and toxicity in estuarine sediments. *Environmental Science and Technology*. 48, 4485-4494.
- Angel, B. M., et al., 2015. Lead solubility in seawater: an experimental study. *Environmental Chemistry*. 13, 489-495.
- ANZG, Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Australian and New Zealand Governments and Australian state and territory governments. Canberra, ACT, Australia. Available at [www.waterquality.gov.au/anz-guidelines](http://www.waterquality.gov.au/anz-guidelines), 2018.
- Aoun, M., et al., 2015. Assessment of committed effective dose due to the ingestion of  $^{210}\text{Po}$  and  $^{210}\text{Pb}$  in consumed Lebanese fish affected by a phosphate fertilizer plant. *Journal of Environmental Radioactivity*. 140, 25-29.
- APPEA, Scientific Literature Review: Environmental Impacts of Decommissioning Options. Advisian, Perth, 2017, pp. 167.
- Arneodo, F., et al., 2020. Characterization of Naturally Occurring Radioactive Material (NORM) Generated from the Lower Cretaceous Carbonate Formations in the Arabian Peninsula and Gulf. *BHM Berg- und Hüttenmännische Monatshefte*. 165, 353-363.

- Arnould, J. P. Y., et al., 2015. Use of Anthropogenic Sea Floor Structures by Australian Fur Seals: Potential Positive Ecological Impacts of Marine Industrial Development? *PLOS ONE*. 10, e0130581.
- ARPANSA, Code of Practice and Safety Guide for Radiation Protection and Radioactive Waste Management in Mining and Mineral Processing. Australian Radiation Protection and Nuclear Safety Agency, Yallambie, 2005, pp. 63.
- ARPANSA, Code of Practice for the Safe Transport of Radioactive Material In: A. R. P. a. N. S. Agency, (Ed.). Commonwealth of Australia, Yallambie, 2008a, pp. 189.
- ARPANSA, Safety Guide for the Management of Naturally Occurring Radioactive Material (NORM). Australian Radiation Protection and Nuclear Safety Agency, Yallambie, 2008b, pp. 127.
- ARPANSA, Fundamentals for Protection Against Ionising Radiation. Australian Radiation Protection and Nuclear Safety Agency, Yallambie, 2014, pp. 36.
- ARPANSA, Background Radioactivity in Northern Australian Seafood. In: David Urban, et al., Eds.). Australian Radiation Protection and Nuclear Safey Agency, Yallambie, Victoria, 2015a, pp. 26.
- ARPANSA, Guide for Radiation Protection of the Environment. Australian Radiation Protection and Nuclear Safety Agency, Yallambie, 2015b, pp. 46.
- ARPANSA, Guide for Radiation Protection in Existing Exposure Situations. Australian Radiation Protection and Nuclear Safety Agency, Yallambie, 2017a, pp. 46.
- ARPANSA, National Directory for Radiation Protection. Radiation Protection Series. Commonwealth of Australia, Yallambie, Victoria, 2017b, pp. 80.
- ARPANSA, Code for Radiation Protection in Planned Exposure Situations Radiation Protection Series C-1. Australian Radiation Protection and Nuclear Safety Agency, Yallambie, Victoria, 2020, pp. 40.
- ASTM International, Standard Test Method for Alpha and Beta Activity in Water By Liquid Scintillation Counting. Vol. D7283-17. ASTM International, West Conshohocken, PA, 2017.
- Atkinson, C. A., et al., 2007. Effect of overlying water pH, dissolved oxygen, salinity and sediment disturbances on metal release and sequestration from metal contaminated marine sediments. *Chemosphere*. 69, 1428-1437.
- Australian Government, Discussion Paper – Decommissioning Offshore Petroleum Infrastructure in Commonwealth Waters. In: I. a. S. Department of Industry, (Ed.), 2018.
- Avellan, A., et al., 2018. Speciation of Mercury in Selected Areas of the Petroleum Value Chain. *Environmental Science & Technology*. 52, 1655-1664.
- Bacon, M. P., et al., 1988. Lead-210 and polonium-210 in ocean water profiles of the continental shelf and slope south of New England. *Continental Shelf Research*. 8, 841-853.
- Bahadori, A., et al., 2013. Estimation of potential barium sulfate (barite) precipitation in oilfield brines using a simple predictive tool. *Environmental Progress & Sustainable Energy*. 32, 860-865.
- Balogh, S. J., et al., 2015. Tracking the Fate of Mercury in the Fish and Bottom Sediments of Minamata Bay, Japan, Using Stable Mercury Isotopes. *Environmental Science & Technology*. 49, 5399-5406.
- Bam, W., et al., 2020. Variability in 210Pb and 210Po partition coefficients (Kd) along the US GEOTRACES Arctic transect. *Marine Chemistry*. 219, 103749.
- Bateman, H., 1910. Solution of a system of differential equations occurring in the theory of radioactive transformations. *Proceedings of the Cambridge Philosophical Society*. 15, 423-427.
- Bazilio, A., Weinrich, J., Inductively Coupled Plasma - Mass Spectrometry (ICP-MS). College of Engineering, University of Massachusetts Amherst, 2012.

- Beer, T., 2006. Ecological Risk Assessment and Quantitative Consequence Analysis. *Human and Ecological Risk Assessment: An International Journal.* 12, 51-65.
- Bell, N., Smith, J., 1999. Coral growing on North Sea oil rigs. *Nature.* 402, 601-601.
- Benoit, G., Hemond, H. F., 1990. Polonium-210 and lead-210 remobilization from lake sediments in relation to iron and manganese cycling. *Environmental Science & Technology.* 24, 1224-1234.
- Beresford, N. A., et al., 2008. An international comparison of models and approaches for the estimation of the radiological exposure of non-human biota. *Applied Radiation and Isotopes.* 66, 1745-1749.
- Bernard, L., et al., 1997. D'etermination de la pression de vapeur de HgCl<sub>2</sub> par la m'ethode d'effusion de Knudsen. *Journal de Physique III* 7, 311-319.
- BHP Billiton Petroleum (Australia) Pty Ltd, Stybarrow operation cessation environment plan summary. BHP Billiton Petroleum (Australia) Pty Ltd, 2017, pp. 22.
- BHP Billiton Petroleum (Australia) Pty Ltd, Griffin Development Cessation. 2018, pp. 40.
- Bills, K., 2018. Building a world-class Australian decommissioning industry. *The APPEA Journal.* 58, 690-694.
- Bingham, M. D., 1990. Field Detection and Implications of Mercury in Natural Gas. *Society of Petroleum Engineers Production Engineering.* 120-124.
- Bittrich, D. R., et al., 2011. Speciation of Mercury (II) and Methylmercury in Cloud and Fog Water. *Aerosol and Air Quality Research.* 11, 161-169.
- Black, F. J., et al., 2012. Factors controlling the abiotic photo-degradation of monomethylmercury in surface waters. *Geochimica et Cosmochimica Acta.* 84, 492-507.
- Bloom, N. S., 2000. Analysis and stability of mercury speciation in petroleum hydrocarbons. *Fresenius J Anal Chem.* 366, 438-443.
- Boak, L. S., et al., The Influence of Divalent Cations on the Performance of BaSO Scale Inhibitor Species. *SPE International Symposium on Oilfield Chemistry, Vol. All Days,* 1999.
- Bodrogi, E., et al., 2005. Application of MnO<sub>2</sub>-coated discs in the case of the measurement of 226Ra with alpha-spectrometric method. *Radioprotection.* 40, S833-S837.
- Bond, T., et al., 2018a. Diel shifts and habitat associations of fish assemblages on a subsea pipeline. *Fisheries Research.* 206, 220-234.
- Bond, T., et al., 2018b. The influence of depth and a subsea pipeline on fish assemblages and commercially fished species. *PLOS ONE.* 13, e0207703.
- Bond, T., et al., 2018c. Fish associated with a subsea pipeline and adjacent seafloor of the North West Shelf of Western Australia. *Marine Environmental Research.* 141, 53-65.
- Bourdon, B., et al., 2003. Introduction to U-series Geochemistry. *Reviews in Mineralogy and Geochemistry.* 52, 1-21.
- Bouyssiere, B., et al., 2000. Analytical methods for speciation of mercury in gas condensates. *Oil & Gas Science and Technology.* 55, 639-648.
- Bouyssiere, B., et al., 2002. Speciation analysis for mercury in gas condensates by capillary gas chromatography with inductively coupled plasma mass spectrometric detection. *Journal of Chromatography A.* 976, 431-439.
- Bowles, K. C., et al., 2001. Bioaccumulation and biomagnification of mercury in Lake Murray, Papua New Guinea. *Canadian Journal of Fisheries and Aquatic Sciences.* 58, 888-897.
- Bowles, M. W., et al., 2014. Global rates of marine sulfate reduction and implications for sub-sea-floor metabolic activities. *Science.* 344, 889-891.

- Bradshaw, C., et al., 2006. The use of tracers to evaluate the importance of bioturbation in remobilising contaminants in Baltic sediments. *Estuarine, Coastal and Shelf Science*. 66, 123-134.
- Brent, R. N., et al., 2017. Validation of handheld X-ray fluorescence for in situ measurement of mercury in soils. *Journal of Environmental Chemical Engineering*. 5, 768-776.
- Brown, J. E., et al., 2008. The ERICA Tool. *Journal of Environmental Radioactivity*. 99, 1371-1383.
- Brown, J. E., et al., 2013. Approaches to providing missing transfer parameter values in the ERICA Tool – How well do they work? *Journal of Environmental Radioactivity*. 126, 399-411.
- Byrne, R. H., 2002. Inorganic speciation of dissolved elements in seawater: the influence of pH on concentration ratios. *Geochemical Transactions*. 3, 11.
- Camera, A. S., et al., 2015. Total Mercury Determination in Petroleum Green Coke and Oily Sludge Samples by Cold Vapor Atomic Fluorescence Spectrometry. *Journal of the Brazilian Chemical Society*. 26, 2116-2124.
- Carlsson, G., et al., 2014. Toxicity Screening of Produced Water Extracts in a Zebrafish Embryo Assay. *Journal of toxicology and environmental health*. 77, 600-615.
- Carvalho, F. P., 2011. Polonium (210Po) and lead (210Pb) in marine organisms and their transfer in marine food chains. *Journal of Environmental Radioactivity*. 102, 462-472.
- Chaiyasit, N., et al., 2009. Decontamination of mercury contaminated steel of API 5L-X52 using iodine and iodide lexicant. *Modern Applied Science*. 4.
- Chandler, J., et al., 2017. Engineering and legal considerations for decommissioning of offshore oil and gas infrastructure in Australia. *Ocean Engineering*. 131, 338-347.
- Chanvanichskul, C., et al., In Situ Mercury Decontamination for Pipeline Decommissioning in the Gulf of Thailand. Abu Dhabi International Petroleum Exhibition & Conference, Vol. Day 3 Wed, November 15, 2017, 2017.
- Chapman, P. M., et al., 2002. Weight-of-Evidence Issues and Frameworks for Sediment Quality (And Other) Assessments. *Human and Ecological Risk Assessment: An International Journal*. 8, 1489-1515.
- Chariton, A. A., et al., 2015. Metabarcoding of benthic eukaryote communities predicts the ecological condition of estuaries. *Environmental Pollution*. 203, 165-174.
- Chariton, A. A., et al., 2016. Emergent technologies and analytical approaches for understanding the effects of multiple stressors in aquatic environments. *Marine and Freshwater Research*. 67, 414-428.
- Chen, Y., et al., 2017. Analytical methods, formation, and dissolution of cinnabar and its impact on environmental cycle of mercury. *Critical Reviews in Environmental Science and Technology*. 47, 2415-2447.
- Cherry, R. D., Heyraud, M., 1981. Polonium-210 content of marine shrimp: Variation with biological and environmental factors. *Marine Biology*. 65, 165-175.
- Cherry, R. D., Heyraud, M., 1982. Evidence of high natural radiation doses in certain mid-water oceanic organisms. *Science*. 218, 54-56.
- Chiariantini, L., et al., 2017. Mercury speciation in *Pinus nigra* barks from Monte Amiata (Italy): An X-ray absorption spectroscopy study. *Environmental Pollution*. 227, 83-88.
- Claisse, J. T., et al., 2014. Oil platforms off California are among the most productive marine fish habitats globally. *Proceedings of the National Academy of Sciences*. 111, 15462-15467.
- Clarisse, O., et al., 2011. Predicting Net Mercury Methylation in Sediments Using Diffusive Gradient in Thin Films Measurements. *Environmental Science & Technology*. 45, 1506-1512.
- Clement, C. H., et al., 2009. Environmental Protection: Transfer Parameters for Reference Animals and Plants. *Annals of the ICRP*. 39, 1-111.

- Clevenger, W. L., et al., 1997. Trace Determination of Mercury : A Review. *Critical Reviews in Analytical Chemistry*. 27, 1-26.
- Clever, H. L., Battino, R., 1979. Krypton, xenon, and radon: gas solubilities. Pergamon Press, Oxford ; New York.
- Collet, P., Chizat, B., The Proven Efficiency of Epoxy Flow Coats for The Protection of Gas Transmission Pipelines. Pipeline Technology Conference 2015, Berlin, Germany, 2015, pp. 9.
- Commonwealth of Australia, A guide to the integrated marine and coastal regionalisation of Australia. In: D. o. t. E. a. Heritage, (Ed.), Vol. IMCRA version 4.0 Commonwealth of Australia, Canberra, 2006, pp. 16.
- Commonwealth of Australia, National Assessment Guidelines for Dredging. Canberra, 2009, pp. 92.
- Commonwealth of Australia, Marine bioregional plan for the North-west marine region. In: E. Department of Sustainability, Water, Population and Communities, (Ed.). Australian Government, Canberra, 2012a.
- Commonwealth of Australia, Marine bioregional plan for the North Marine Region. In: E. department of Sustainability, Water, Population and Communities, (Ed.). Australian Government, Canberra, 2012b, pp. 200.
- Commonwealth of Australia, South-east marine region profile: A description of the ecosystems, conservation values and uses of the South-east Marine Region. In: D. o. t. Environment, (Ed.), Canberra, 2015, pp. 94.
- Commonwealth of Australia, Charter: National Water Quality Management Strategy. In: D. o. A. a. W. Resources, (Ed.). Australian Government, Canberra, 2018, pp. 24.
- Commonwealth of Australia, Australia's Strategy for Nature 2019–2030. In: B. W. Group, (Ed.). Commonwealth of Australia, 2019.
- ConocoPhillips Australia Exploration Pty Ltd, Barossa area development: offshore project proposal. ConocoPhillips Australia Exploration Pty Ltd., 2017, pp. 495.
- ConocoPhillips Pipeline Australia Pty Ltd, Bayu-Undan to Darwin Gas Export Pipeline Production Cessation Environment Plan. Santos NA Darwin Pipeline Pty Ltd, 2019, pp. 311.
- Copplestone, D., et al., 2013. An international database of radionuclide concentration ratios for wildlife: development and uses. *Journal of Environmental Radioactivity*. 126, 288-298.
- Copplestone, D., et al., 2008. The development and purpose of the FREDERICA radiation effects database. *Journal of Environmental Radioactivity*. 99, 1456-1463.
- Copplestone, D., et al., 2020. Protection of the environment. *Annals of the ICRP*. 014664532094429.
- Corvini, G., et al., Mercury removal from natural gas and liquid streams. UOP A Honeywell Company, Houston, Texas, USA, 2016.
- Cossa, D., et al., 2011. Mercury in the Southern Ocean. *Geochimica et Cosmochimica Acta*. 75, 4037-4052.
- Crafts, P., Williams, M., 2020. Mercury partitioning in oil and gas production systems - design optimisation and risk mitigation through advanced simulation. *The APPEA Journal*. 60, 97-109.
- Cresswell, T., et al., 2020. Exploring New Frontiers in Marine Radioisotope Tracing – Adapting to New Opportunities and Challenges. *Frontiers in Marine Science*. 7.
- Currie, L. A., 1968. Limits for qualitative detection and quantitative determination. Application to radiochemistry. *Analytical Chemistry*. 40, 586-593.
- Curti, E., 1999. Coprecipitation of radionuclides with calcite: estimation of partition coefficients based on a review of laboratory investigations and geochemical data. *Applied Geochemistry*. 14, 433-445.

- Dahl, A. L., et al., 2011. Bacterial bioreporter detects mercury in the presence of excess EDTA. *Environmental Chemistry*. 8, 552-560.
- Dai, S.-S., et al., 2021. Global distribution and environmental drivers of methylmercury production in sediments. *Journal of Hazardous Materials*. 407, 124700.
- Dang, D. H., et al., 2020. Kinetic processes of copper and lead remobilization during sediment resuspension of marine polluted sediments. *Science of The Total Environment*. 698, 134120.
- Dang, D. H., et al., 2015. Evidencing the Impact of Coastal Contaminated Sediments on Mussels Through Pb Stable Isotopes Composition. *Environmental Science & Technology*. 49, 11438-11448.
- DAWE, Ratification of the Minamata Convention on Mercury: Final Regulation Impact Statement. In: W. a. t. E. Department of Agriculture, (Ed.). Australian Government, Canberra, 2020, pp. 81.
- De Jonge, M., et al., 2012. The impact of increased oxygen conditions on metal-contaminated sediments part I: Effects on redox status, sediment geochemistry and metal bioavailability. *Water Research*. 46, 2205-2214.
- de Zwart, D., Posthuma, L., 2005. Complex mixture toxicity for single and multiple species: Proposed methodologies. *Environmental Toxicology and Chemistry*. 24, 2665-2676.
- Deiaa, E., et al., 2020. The assessment of radiological risks associated with the ingestion of 210Po, 210Pb, and 137Cs in marine organisms collected from the local fish market of Kenitra, Morocco. *Journal of Radioanalytical and Nuclear Chemistry*. 324, 1069-1076.
- Depew, D. C., et al., 2012. Toxicity of dietary methylmercury to fish: Derivation of ecologically meaningful threshold concentrations. *Environmental Toxicology and Chemistry*. 31, 1536-1547.
- Dessy, R. E., Lee, Y. K., 1960. The Mechanism of the Reaction of Mercuric Halides with Dialkyl and Diarylmercury Compounds. *Journal of the American Chemical Society*. 82, 689-693.
- Din, G., et al., 2019. Engineering a bioluminescent bioreporter from an environmentally sourced mercury-resistant *Enterobacter cloacae* strain for the detection of bioavailable mercury. *Journal of Applied Microbiology*. 127, 1125-1134.
- DISER, Offshore Petroleum Decommissioning Guideline. Department of Industry, Science, Energy, and Resources, Canberra, 2018, pp. 21.
- DNV-GL, Submarine pipeline systems. 2017.
- Doering, C., Bollhöfer, A., 2016. A soil radiological quality guideline value for wildlife-based protection in uranium mine rehabilitation. *Journal of Environmental Radioactivity*. 151, 522-529.
- Doering, C., et al., 2019. Whole organism concentration ratios in freshwater wildlife from an Australian tropical U mining environment and the derivation of a water radiological quality guideline value. *Journal of Environmental Radioactivity*. 198, 27-35.
- Dong, W., et al., 2016. Developmental toxicity from exposure to various forms of mercury compounds in medaka fish (*Oryzias latipes*) embryos. *PeerJ*. 4, e2282-e2282.
- Dowdall, M., Lepland, A., 2012. Elevated levels of radium-226 and radium-228 in marine sediments of the Norwegian Trench ("Norskrenna") and Skagerrak. *Marine Pollution Bulletin*. 64, 2069-2076.
- Driscoll, C. T., et al., 2013. Mercury as a Global Pollutant: Sources, Pathways, and Effects. *Environmental Science & Technology*. 47, 4967-4983.
- Ebadian, M. A., Mercury contaminated material decontamination methods: investigation and assessment. National Energy Technology Laboratory (NETL), Pittsburgh, PA, Morgantown, WV, and Albany, OR (United States), 2001, pp. 73.
- Eckley, C. S., et al., 2020. The assessment and remediation of mercury contaminated sites: A review of current approaches. *Science of The Total Environment*. 707, 136031.

- Edmonds, B., et al., Mercury Partitioning in Natural Gases and Condensates. GPA European Chapter Meeting, London, 1996.
- Eggleton, J., Thomas, K. V., 2004. A review of factors affecting the release and bioavailability of contaminants during sediment disturbance events. *Environment International*. 30, 973-980.
- Eichler, B., Volatility Properties of Polonium. Switzerland, 2002, pp. 52.
- Eni Australia Limited, Woollybutt Field Management, Plug and Abandonment and Equipment Removal. Eni Australia Ltd, 2013, pp. 22.
- Eriksen, D. Ø., et al., 2006. Radionuclides in produced water from Norwegian oil and gas installations — concentrations and bioavailability. *Czechoslovak Journal of Physics*. 56, D43-D48.
- Esso Australia Resources Pty Ltd, Blackback Plugging and Abandonment Environment Plan. Esso Australia Resources Pty Ltd, 2018, pp. 299.
- Esso Australia Resources Pty Ltd, Bass Strait Operations Environment Plan. Esso Australia Resources Pty Ltd, 2019a.
- Esso Australia Resources Pty Ltd, Whiting Plug and Abandonment Environment Plan. 2019b, pp. 1172.
- Ezzeldin, M. F., et al., 2016. Mercury Speciation and Distribution in an Egyptian Natural Gas Processing Plant. *Energy & Fuels*. 30, 10236-10243.
- Falkner, K. K., et al., 1993. The behavior of barium in anoxic marine waters. *Geochimica et Cosmochimica Acta*. 57, 537-554.
- Feldman, C., 1974. Preservation of Dilute Mercury Solutions. *Analytical Chemistry*. 46, 99-102.
- Filby, R. H., 1994. Origin and nature of trace element species in crude oils, bitumens and kerogens: implications for correlation and other geochemical studies. Geological Society, London, Special Publications. 78, 203-219.
- Fisher, N. S., et al., 2013. Evaluation of radiation doses and associated risk from the Fukushima nuclear accident to marine biota and human consumers of seafood. *Proceedings of the National Academy of Sciences*. 110, 10670-10675.
- Fisher, N. S., et al., 1996. Accumulation and Retention of Metals in Mussels from Food and Water: A Comparison under Field and Laboratory Conditions. *Environmental Science & Technology*. 30, 3232-3242.
- Foster, D. A., et al., 2004. 226Ra and Ba concentrations in the Ross Sea measured with multicollector ICP mass spectrometry. *Marine Chemistry*. 87, 59-71.
- Fowler, A. M., et al., 2018. Environmental benefits of leaving offshore infrastructure in the ocean. *Frontiers in Ecology and the Environment*. 16, 571-578.
- Fowler, S. W., et al., 2010. Scavenging and retention of bismuth by marine plankton and biogenic particles. *Limnology and Oceanography*. 55, 1093-1104.
- Frech, W., et al., 1996. Determination and Speciation of Mercury in Natural Gases and Gas Condensates. *Analytical Communications*. 33, 7H-9H.
- FSANZ, Australian and New Zealand Food Standards Code - Schedule 19. In: Food Standards Australia New Zealand, (Ed.), Vol. F2017C00333. Commonwealth of Australia Canberra, 2017.
- Gäfvert, T., Færevik, I., Natural Radioactivity in Produced Water from the Norwegian Oil and Gas Industry in 2003. Norwegian Radiation Protection Authority, Østerås, 2004.
- Gallup, D. L., Removal of mercury from water in the petroleum industry. 21st International Petroleum Environmental Conference 2014.
- Gallup, D. L., Strong, J. B., Removal of Mercury and Arsenic from Produced Water. Chevron Corporation. Chevron Corporation, 2007.

- Gao, X., Chen, C.-T. A., 2012. Heavy metal pollution status in surface sediments of the coastal Bohai Bay. *Water Research.* 46, 1901-1911.
- Garnier-Laplace, J., et al., 2010. A multi-criteria weight of evidence approach for deriving ecological benchmarks for radioactive substances. *Journal of Radiological Protection.* 30, 215-233.
- Gates, A. R., et al., 2019. Ecological Role of an Offshore Industry Artificial Structure. *Frontiers in Marine Science.* 6.
- Gaulier, F., et al., 2015. Mercury speciation in liquid petroleum product : comparison between on-site approach and lab measurement using size exclusion chromatography with high resolution inductively coupled plasma mass spectrometric detection (SEC-ICP-HR MS). *Fuel Processing Technology.* 131, 254-261.
- Ghose, S., Heaton, B., The release of radium from scales produced in the North Sea oil fields. *Radioactivity in the Environment.* Elsevier, 2005, pp. 1081-1089.
- Gillett, D. J., et al., 2020. Benthic habitat condition of the continental shelf surrounding oil and gas platforms in the Santa Barbara Channel, Southern California. *Marine Pollution Bulletin.* 160, 111662.
- Gilmour, M. E., et al., 2019. Mercury as an indicator of foraging ecology but not the breeding hormone prolactin in seabirds. *Ecological Indicators.* 103, 248-259.
- Gissi, F., et al., 2016. A review of nickel toxicity to marine and estuarine tropical biota with particular reference to the South East Asian and Melanesian region. *Environmental Pollution.* 218, 1308-1323.
- Gissi, F., et al., 2020. Deriving a Chronic Guideline Value for Nickel in Tropical and Temperate Marine Waters. *Environmental Toxicology and Chemistry.* 39, 2540-2551.
- Glud, R. N., 2008. Oxygen dynamics of marine sediments. *Marine Biology Research.* 4, 243-289.
- Gong, Y., et al., 2019. Application of Iron-Based Materials for Remediation of Mercury in Water and Soil. *Bulletin of Environmental Contamination and Toxicology.* 102, 721-729.
- Goudriaan, J., 2019. Too much fear for radioactive contamination of seawater. *Europhysics News.* 50, 19-22.
- Gray, J. S., 2002. Biomagnification in marine systems: the perspective of an ecologist. *Marine Pollution Bulletin.* 45, 46-52.
- Griffiths, J. R., et al., 2017. The importance of benthic–pelagic coupling for marine ecosystem functioning in a changing world. *Global Change Biology.* 23, 2179-2196.
- Grung, M., et al., 2009. Bioaccumulation and lack of oxidative stress response in the ragworm *H. diversicolor* following exposure to 226Ra in sediment. *Journal of Environmental Radioactivity.* 100, 429-434.
- Gustafsson, M. B., et al., Wastewater mercury removal process. In: E. R. a. E. Co, (Ed.), Vol. US 8,034,246 B2, 2007.
- Hall, B., et al., 1991. Chemical Reactions of Mercury in Combustion Flue Gases. *Water, Air, and Soil Pollution.* 56, 3-14.
- Harding, G., et al., 2018. Bioaccumulation of methylmercury within the marine food web of the outer Bay of Fundy, Gulf of Maine. *PloS one.* 13, e0197220-e0197220.
- Haynes, W. M., CRC Handbook of Chemistry and Physics. CRC Press/Taylor & Francis, Boca Raton, FL, 2017.
- Heaton, B., Lambley, J., 1995. TENORM in the oil, gas and mineral mining industry. *Applied Radiation and Isotopes.* 46, 577-581.
- Hedström, H., et al., 2013. Characterization of Radium Sulphate. *Journal of Nuclear Chemistry.*

- Herbert, R., 1999. Nitrogen cycling in coastal marine ecosystems. *FEMS microbiology reviews*. 23, 563-590.
- Heyes, A., et al., 2006. Mercury methylation in estuaries: Insights from using measuring rates using stable mercury isotopes. *Marine Chemistry*. 102, 134-147.
- Heyraud, M., Cherry, R. D., 1979. Polonium-210 and lead-210 in marine food chains. *Marine Biology*. 52, 227-236.
- Heyward, A., et al., Barossa Environmental Baseline Study, Regional Shoals and Shelf Assessment 2015 Final Report. A report for ConocoPhillips Australia Exploration Pty Ltd by the Australian Institute of Marine Science. Australian Institute of Marine Science, Perth, 2017, pp. 143.
- Higley, K., et al., 2015. Creation and application of voxelised dosimetric models, and a comparison with the current methodology as used for the International Commission on Radiological Protection's Reference Animals and Plants. *Annals of the ICRP*. 44, 313-330.
- Hinton, T. G., et al., 2007. Radiation-induced effects on plants and animals: findings of the United Nations Chernobyl Forum. *Health Physics*. 93, 427-440.
- Hirth, G., A review of existing Australian radionuclide activity concentration data in non-human biota inhabiting uranium mining environments. Technical Report Series. ARPANSA, Yallambie, 2014, pp. 40.
- Hirth, G. A., et al., 2017. Whole-organism concentration ratios in wildlife inhabiting Australian uranium mining environments. *Journal of Environmental Radioactivity*. 178-179, 385-393.
- Hong, G.-H., et al., 1999. Lead-210 and polonium-210 in the winter well-mixed turbid waters in the mouth of the Yellow Sea. *Continental Shelf Research*. 19, 1049-1064.
- Hosseini, A., et al., 2008. Transfer of radionuclides in aquatic ecosystems – Default concentration ratios for aquatic biota in the Erica Tool. *Journal of Environmental Radioactivity*. 99, 1408-1429.
- Howard, B. J., et al., 2013. The IAEA handbook on radionuclide transfer to wildlife. *Journal of Environmental Radioactivity*. 121, 55-74.
- Huang, W., et al., 2011. Toxicity Testing of Waterborne Mercury with Red Sea Bream (*Pagrus major*) Embryos and Larvae. *Bulletin of Environmental Contamination and Toxicology*. 86, 398.
- Huber, M. L., et al., 2006. Correlation for the Vapor Pressure of Mercury. *Industrial & Engineering Chemistry Research*. 45, 7351-7361.
- Huck, P. M., et al., 1989. Modelling of radium-226 leaching from barium-radium sulfate sludges. *Waste Management*. 9, 157-163.
- Huerta-Diaz, M. A., Morse, J. W., 1992. Pyritization of trace metals in anoxic marine sediments. *Geochimica et Cosmochimica Acta*. 56, 2681-2702.
- Humphries, C., The "Global Chemical Experiment". Harvard Magazine, Harvard, 2018.
- Hurtado-Bermúdez, S., et al., 2019. Levels of radionuclide concentrations in benthic invertebrate species from the Balearic Islands, Western Mediterranean, during 2012–2018. *Marine Pollution Bulletin*. 149, 110519.
- Hussain, N., et al., 1995. Bio-volatilization of polonium: Results from laboratory analyses. *Aquatic geochemistry*. 1, 175-188.
- Hutchins, D. A., Bruland, K. W., 1998. Iron-limited diatom growth and Si:N uptake ratios in a coastal upwelling regime. *Nature*. 393, 561-564.
- IAEA, 2001. Generic models for use in assessing the impact of discharges of radioactive substances to the environment. International Atomic Energy Agency, Vienna.
- IAEA, Radiation protection and management of radioactive waste in the oil and gas industry. Safety Report Series. International Atomic Energy Agency, Vienna, 2003, pp. 139.

- IAEA, Application of the concepts of exclusion, exemption and clearance safety guide. Safety standards series. International Atomic Energy Agency, Vienna, 2004a.
- IAEA, 2004b. Sediment distribution coefficients and concentration factors for biota in the marine environment. Internat. Atomic Energy Agency, Vienna.
- IAEA, Analytical Methodology for the Determination of Radium Isotopes in Environmental Samples. IAEA Analytical Quality in Nuclear Applications. International Atomic Energy Agency, Vienna, 2010, pp. 74.
- IAEA, Management of NORM Residues. International Atomic Energy Agency, Vienna, 2013, pp. 66.
- IAEA, 2014a. The environmental behaviour of radium. International Atomic Energy Agency.
- IAEA, 2014b. Handbook of parameter values for the prediction of radionuclide transfer to wildlife.
- IAEA, 2014c. Modelling of Biota Dose Effects: report of working group 6 Biota Dose Effects Modelling of EMRAS II topical heading reference approaches for Biota Dose Assessment, Environmental Modelling for Radiation Safety (EMRAS II) Programme. International Atomic Energy Agency, Vienna.
- IAEA, 2014d. Radiation protection and safety of radiation sources: international basic safety standards. International Atomic Energy Agency, Vienna.
- IAEA, Regulations for the safe transport of radioactive material. Specific Safety Requirements, Vol. SSR-6 (Rev 1). International Atomic Energy Agency, Vienna, 2018, pp. 165.
- IAEA, Guidelines on Soil and Vegetation Sampling for Radiological Monitoring. Technical Reports Series No. 486. International Atomic Energy Agency, Vienna, 2019, pp. 266.
- ICRP, 2007. The 2007 recommendations of the International Commission on Radiological Protection. International Commission on Radiological Protection, Oxford.
- ICRP, 2008. Environmental Protection - the Concept and Use of Reference Animals and Plants. ICRP Publication 108. 38.
- ICRP, 2009. Environmental Protection: Transfer Parameters for Reference Animals and Plants. ICRP Publication 114. 39.
- ICRP, 2017. Dose coefficients for non-human biota environmentally exposed to radiation. ICRP Publication 136. 46.
- ICSC, Dimethyl mercury. 2016, pp.  
[https://www.ilo.org/dyn/icsc/showcard.display?p\\_version=2&p\\_card\\_id=1304](https://www.ilo.org/dyn/icsc/showcard.display?p_version=2&p_card_id=1304).
- IOGP, 2016a. Guidelines for the Management of Naturally Occurring Radioactive Material (NORM) in the Oil and Gas Industry. IOGP, London.
- IOGP, Managing Naturally Occurring Radioactive Material (NORM) in the oil and gas industry. International Association of Oil and Gas Producers London, 2016b, pp. 68.
- IPIECA, Mercury management in petroleum refining. An IPIECA Good Practice Guide, London, United Kingdom, 2014.
- ISO, Water quality — Determination of the activity concentration of radionuclides — Method by high resolution gamma-ray spectrometry. ICS: 17.240, Vol. 10703:2007, 2007, pp. 20.
- Jakimska, A., 2011. Bioaccumulation of metals in tissues of marine animals, part I: The role and impact of heavy metals on organisms. Polish Journal of Environmental Studies. 20, 1117-1125.
- Jas, E., et al., 2017. Out of sight out of mind – subsea pipeline decommissioning. The APPEA Journal. 57, 79-87.
- Jensen, L. K., et al., 2016. Individual and molecular level effects of produced water contaminants on nauplii and adult females of *Calanus finmarchicus*. Journal of Toxicology and Environmental Health, Part A. 79, 585-601.

- Jia, G., et al., 2020. The fate of the main naturally occurring radionuclides in mussels (*Mytilus edulis*) and their radiological impact on human beings. *Environmental Monitoring and Assessment*. 192, 217.
- Jiang, L., et al., 2021. Improving the regulatory health risk assessment of mercury-contaminated sites. *Journal of Hazardous Materials*. 402, 123493.
- Johansen, M. P., et al., 2012. Assessing doses to terrestrial wildlife at a radioactive waste disposal site: Inter-comparison of modelling approaches. *Science of The Total Environment*. 427-428, 238-246.
- Johnson, W. P., et al., 2015. Total- and methyl-mercury concentrations and methylation rates across the freshwater to hypersaline continuum of the Great Salt Lake, Utah, USA. *Science of The Total Environment*. 511, 489-500.
- Jones, F. V., Israni, K., Environmental risk assessment utilizing Bow-Tie methodology. Society of Petroleum Engineers - SPE/APPEA Int. Conference on Health, Safety and Environment in Oil and Gas Exploration and Production 2012: Protecting People and the Environment - Evolving Challenges, Vol. 2, 2012, pp. 1239-1245.
- Jones, P., et al., 2015. Lead-210 and Polonium-210 disequilibria in the northern Gulf of Mexico hypoxic zone. *Marine Chemistry*. 169, 1-15.
- Jonsson, S., et al., 2012. Mercury Methylation Rates for Geochemically Relevant HgII Species in Sediments. *Environmental Science & Technology*. 46, 11653-11659.
- Kalnicky, D. J., Singhvi, R., 2001. Field portable XRF analysis of environmental samples. *Journal of Hazardous Materials*. 83, 93-122.
- Kamal, M. S., et al., 2018. Oilfield scale formation and chemical removal: A review. *Journal of Petroleum Science and Engineering*. 171, 127-139.
- Kearns, J., Turner, A., 2016. An evaluation of the toxicity and bioaccumulation of bismuth in the coastal environment using three species of macroalgae. *Environmental Pollution*. 208, 435-441.
- Khormali, A., et al., 2016. Experimental analysis of calcium carbonate scale formation and inhibition in waterflooding of carbonate reservoirs. *Journal of Petroleum Science and Engineering*. 147, 843-850.
- Kibogy, J., Using Handheld XRF Technology to Determine Surface Mercury Concentration - The Yeh/Kibogy Method. In: C. U. S. A. Inc., (Ed.), API Industrial Hygiene TF Workshop, Hyatt Hotel, Denver CO, 2010.
- Kim, C. S., et al., 2004. Geological and anthropogenic factors influencing mercury speciation in mine wastes: an EXAFS spectroscopy study. *Applied geochemistry : journal of the International Association of Geochemistry and Cosmochemistry*. 19, 379-393.
- Koppel, D. J., et al., 2019. Assessing the Risk of Metals and Their Mixtures in the Antarctic Nearshore Marine Environment with Diffusive Gradients in Thin-Films. *Environmental Science & Technology*. acs.est.9b04497.
- Kozin, L. F., Hansen, S., *Mercury Handbook: Chemistry, Applications and Environmental Impact*. Royal Society of Chemistry, 2013.
- Krieger, K., 2005. NORM Contamination—Now You See It, Now You Don't. *Health physics : the radiation safety journal*. 89, S20-S21.
- Krivan, V., Haas, H. F., 1988. Prevention of loss of mercury(II) during storage of dilute solutions in various containers. *Fresenius' Zeitschrift für analytische Chemie*. 322, 1-6.
- Landsberger, S., et al., 2013. Determination of 226Ra, 228Ra and 210Pb in NORM products from oil and gas exploration: Problems in activity underestimation due to the presence of metals and self-absorption of photons. *Journal of Environmental Radioactivity*. 125, 23-26.

- Lang, D., et al., Mercury arising from oil and gas production in the United Kingdom and UK continental shelf. University of Oxford, 2012.
- Langmuir, D., Riese, A. C., 1985. The thermodynamic properties of radium. *Geochimica et Cosmochimica Acta*. 49, 1593-1601.
- Lansens, P., et al., 1990. Long-term stability of methylmercury standard solutions in distilled, deionized water. *Analytica Chimica Acta*. 229, 281-285.
- Lariviere, D., et al., 2006. Radionuclide determination in environmental samples by inductively coupled plasma mass spectrometry. *Spectrochimica Acta Part B: Atomic Spectroscopy*. 61, 877-904.
- Lavoie, R. A., et al., 2013. Biomagnification of Mercury in Aquatic Food Webs: A Worldwide Meta-Analysis. *Environmental Science & Technology*. 47, 13385-13394.
- Le Roy, E., et al., 2018. The  $^{226}\text{Ra}$ -Ba relationship in the North Atlantic during GEOTRACES-GA01. *Biogeosciences*. 15, 3027-3048.
- Leckie, S. H. F., et al., 2016. Sedimentation-induced burial of subsea pipelines: Observations from field data and laboratory experiments. *Coastal Engineering*. 114, 137-158.
- Leermakers, M., et al., 1991. Storage and stability of inorganic and methylmercury solutions. *Fresenius' Journal of Analytical Chemistry*. 336, 655-662.
- Lepland, A., et al., 2000. Accumulation of barium in recent Skagerrak sediments: sources and distribution controls. *Marine Geology*. 163, 13-26.
- Levin, L. A., et al., 2001. The Function of Marine Critical Transition Zones and the Importance of Sediment Biodiversity. *Ecosystems*. 4, 430-451.
- Li, J., et al., 2017. Scale formation and control in oil and gas fields: A review. *Journal of Dispersion Science and Technology*. 38, 661-670.
- Liem-Nguyen, V., et al., 2016. Effects of Nutrient Loading and Mercury Chemical Speciation on the Formation and Degradation of Methylmercury in Estuarine Sediment. *Environmental Science & Technology*. 50, 6983-6990.
- Linderoth, S., Morup, S., 1992. Stability and magnetic properties of an iron-mercury alloy. *Journal of Physics: Condensed Matter*. 4, 8627-8634.
- Ling, S. D., et al., 2018. Pollution signature for temperate reef biodiversity is short and simple. *Marine pollution bulletin*. 130, 159-169.
- Linstrom, P. J., Mallard, W. G., NIST Chemistry WebBook, NIST Standard Reference Database Number 69. 2005.
- Liu, Q., 2013. Mercury concentration in natural gas and its distribution in the Tarim Basin. *Science China Earth Science*. 56, 1371-1379.
- López-Antón, M. A., et al., 2012. Analytical methods for mercury analysis in coal and coal combustion by-products. *International Journal of Coal Geology*. 94, 44-53.
- Lothongkum, A. W., et al., 2011. Simultaneous removal of arsenic and mercury from natural-gas-co-produced water from the Gulf of Thailand using synergistic extractant via HFSLM. *Journal of Membrane Science*. 369, 350-358.
- Ma, M., et al., 2019. Mercury methylation by anaerobic microorganisms: A review. *Critical reviews in environmental science and technology*. 49, 1893-1936.
- Ma, Y., et al., 2014. Absorption characteristics of elemental mercury in mercury chloride solutions. *Journal of Environmental Sciences*. 26, 2257-2265.
- Mabuchi, H., 1963. On the volatility of some polonium compounds. *Journal of Inorganic and Nuclear Chemistry*. 25, 657-660.

- MacIntosh, A., Marine ecotoxicological effects of offshore petroleum infrastructure-associated contaminants: A systematic review. Department of Earth and Environmental Sciences, Vol. Master of Research. Macquarie University, Sydney, 2020, pp. 73.
- Macreadie, P. I., et al., 2018. Eyes in the sea: Unlocking the mysteries of the ocean using industrial, remotely operated vehicles (ROVs). *Science of The Total Environment*. 634, 1077-1091.
- Manouchehri, S., Subsea Pipelines and Flowlines Decommissioning: What We Should Know for a Rational Approach. ASME 2017 36th International Conference on Ocean, Offshore and Arctic Engineering, Vol. Volume 5B: Pipelines, Risers, and Subsea Systems, 2017.
- Manoukian, S., et al., 2010. Effects of two offshore gas platforms on soft-bottom benthic communities (northwestern Adriatic Sea, Italy). *Marine Environmental Research*. 70, 402-410.
- Mat Çatal, E., et al., 2012. 210Po and 210Pb variations in fish species from the Aegean Sea and the contribution of 210Po to the radiation dose. *Marine Pollution Bulletin*. 64, 801-806.
- Mathews, T., Fisher, N. S., 2008. Evaluating the trophic transfer of cadmium, polonium, and methylmercury in an estuarine food chain. *Environmental Toxicology and Chemistry*. 27, 1093-1101.
- Matsuyama, A., et al., 2016. Distribution and characteristics of methylmercury in surface sediment in Minamata Bay. *Marine Pollution Bulletin*. 109, 378-385.
- Matsuyama, A., et al., 2018. Chemical characteristics of dissolved mercury in the pore water of Minamata Bay sediments. *Marine Pollution Bulletin*. 129, 503-511.
- Matyskin, A. V., On the solubility of radium sulfate and carbonate. Chalmers University of Technology, Gothenburg, Sweden, 2016.
- McAuliffe, C. A., 1977. *The Chemistry of Mercury* The Macmillan Press Ltd., London
- McCready, R. G. L., et al., 1980. Preliminary studies on the chemical, physical, and biological stability of Ba/RaSO<sub>4</sub> precipitates. *Hydrometallurgy*. 5, 109-116.
- McDevitt, B., et al., 2019. Emerging investigator series: radium accumulation in carbonate river sediments at oil and gas produced water discharges: implications for beneficial use as disposal management. *Environmental Science: Processes & Impacts*. 21, 324-338.
- McDonald, P., et al., 1996. Technological enhancement of natural radionuclides in the marine environment. *Journal of Environmental Radioactivity*. 32, 67-90.
- McKay, S., et al., 2020. NORM inventory forecast for Australian offshore oil and gas decommissioned assets and radioactive waste disposal pathways. *The APPEA Journal*. 60, 19-33.
- McLean, D. L., et al., 2017. Using industry ROV videos to assess fish associations with subsea pipelines. *Continental Shelf Research*. 141, 76-97.
- McLean, D. L., et al., 2019. An assessment of fish and marine growth associated with an oil and gas platform jacket using an augmented remotely operated vehicle. *Continental Shelf Research*. 179, 66-84.
- McLean, D. L., et al., 2020. Fish-habitat associations on a subsea pipeline within an Australian Marine Park. *Marine Environmental Research*. 153, 104813.
- Miao, Z., et al., 2012. Sulfate reduction in groundwater: characterization and applications for remediation. *Environmental geochemistry and health*. 34, 539-550.
- Moggridge, B. J., et al., 2019. Integrating Aboriginal cultural values into water planning: a case study from New South Wales, Australia. *Australasian Journal of Environmental Management*. 26, 273-286.
- Momoshima, N., et al., 2001. Formation and Emission of Volatile Polonium Compound by Microbial Activity and Polonium Methylation with Methylcobalamin. *Environmental Science & Technology*. 35, 2956-2960.

- Monnin, C., 1999. A thermodynamic model for the solubility of barite and celestite in electrolyte solutions and seawater to 200°C and to 1 kbar. *Chemical Geology*. 153, 187-209.
- Morel, F. M. M., et al., 1998. The Chemical Cycle and Bioaccumulation of Mercury. *Annual Review Ecology and Systematics*. 29, 543-566.
- Mussig, S., Rothmann, B., Mercury in Natural Gas - Problems and Technical Solutions for its Removal. In: S. o. P. Engineers, (Ed.), *SPE Asia Pacific Oil and Gas Conference and Exhibition*. Society of Petroleum Engineers, Kuala Lumpur, Malaysia, 1997.
- Nancekievill, M., et al., 2018. Development of a Radiological Characterization Submersible ROV for Use at Fukushima Daiichi. *IEEE Transactions on Nuclear Science*. 65, 2565-2572.
- NCBI, PubChem Compound Summary for CID 409301, Methylmercury chloride. National Center for Biotechnology Information 2021.
- Ndungu, K., et al., 2016. Effects of organic matter addition on methylmercury formation in capped and uncapped marine sediments. *Water Research*. 103, 401-407.
- Nelson, A. W., et al., 2015a. Understanding the Radioactive Ingrowth and Decay of Naturally Occurring Radioactive Materials in the Environment: An Analysis of Produced Fluids from the Marcellus Shale. *Environmental Health Perspectives*. 123, 689-696.
- Nelson, A. W., et al., 2016. Partitioning of naturally-occurring radionuclides (NORM) in Marcellus Shale produced fluids influenced by chemical matrix. *Environmental Science: Processes & Impacts*. 18, 456-463.
- Nelson, A. W., et al., Naturally-Occurring Radioactive Materials (NORM) Associated with Unconventional Drilling for Shale Gas. *Hydraulic Fracturing: Environmental Issues*. American Chemical Society, 2015b, pp. 89-128.
- Nelson, P. F., et al., 2012. Atmospheric mercury emissions in Australia from anthropogenic, natural and recycled sources. *Atmospheric Environment*. 62, 291-302.
- Nengkoda, A., et al., Understanding of Mercury Corrosion Attack on Stainless Steel Material and Gas Well: Case Study. International Petroleum Technology Conference, Doha, Qatar, 2009.
- Nevado, J. J. B., et al., 2011. Comparison of gas chromatographic hyphenated techniques for mercury speciation analysis. *Journal of Chromatography A*. 1218, 4545-4551.
- NHMRC, NRMMC, Australian Drinking Water Guideline. National Water Quality Management Strategy, Vol. Version 3.5. National Health and Medical Research Council, National Resource Management Ministerial Council, Canberra, 2011, pp. 1172.
- NOPSEMA, Environment plan decision making. In: NOPSEMA, (Ed.), Guideline, 2019, pp. 30.
- NOPSEMA, ALARP & Acceptable for environmental impacts and environmental risks. Fact Sheet. National Offshore Petroleum Safety and Environmental Management Authority, Perth, 2020a, pp. 2.
- NOPSEMA, Environment plan assessment. Policy. National Offshore Petroleum Safety and Environmental Management Authority, Perth, 2020b, pp. 16.
- NOPSEMA, Environment plan content requirement. Guidance Note. National Offshore Petroleum Safety and Environmental Management Authority, Perth, 2020c, pp. 41.
- NOPSEMA, Section 572 Maintenance and removal of property. National Offshore Petroleum Safety and Environmental Management Authority, 2020d, pp. 14.
- NOPTA, National Electronic Approvals Tracking System. 2021.
- Norton, S., Schofield, K., 2017. Conceptual model diagrams as evidence scaffolds for environmental assessment and management. *Freshwater Science*. 36, 231 - 239.
- Nys, C., et al., 2018. A framework for ecological risk assessment of metal mixtures in aquatic systems. *Environmental Toxicology and Chemistry*. 37, 623-642.

- O'Connor, D., et al., 2019. Mercury speciation, transformation, and transportation in soils, atmospheric flux, and implications for risk management: A critical review. *Environment International*. 126, 747-761.
- O'Rear, D. J., et al., Process, method, and system for removing mercury from pipelines. 2016. Oceaneering, Subsea NORM inspection tool. In: I. Oceaneering International, (Ed.), 2017.
- OECD, Test No. 201: Freshwater Alga and Cyanobacteria, Growth Inhibition Test. OECD Guidelines for the Testing of Chemicals, Section 2. OECD Publishing, Paris, 2011, pp. 25.
- Oil & Gas UK, Decommissioning of pipelines in the North Sea region. The UK Oil and Gas Industry Association Limited, 2013, pp. 52.
- Oil & Gas UK, 2014. Guidance on Risk Related Decision Making. Oil & Gas, UK, London, UK.
- Olsvik, P. A., et al., 2012. Low impact of exposure to environmentally relevant doses of  $^{226}\text{Ra}$  in Atlantic cod (*Gadus morhua*) embryonic cells. *Journal of Environmental Radioactivity*. 109, 84-93.
- Paranjape, A. R., et al., 2017. Recent advances in the study of mercury methylation in aquatic systems. *FACETS*. 2, 85-119.
- Parker, J. L., Bloom, N. S., 2005. Preservation and storage techniques for low-level aqueous mercury speciation. *Science of the Total Environment*. 337, 253-263.
- Phillips, E. J. P., et al., 2001. Sulfate-Reducing Bacteria Release Barium and Radium from Naturally Occurring Radioactive Material in Oil-Field Barite. *Geomicrobiology Journal*. 18, 167-182.
- Pillay, A. E., et al., 2010. Radioactivity in Oily Sludge and Produced Waste Water from Oil: Environmental Concerns and Potential Remedial Measures. *Sustainability*. 2, 890-901.
- Pirrone, N., et al., 2010. Global mercury emissions to the atmosphere from anthropogenic and natural sources. *Atmospheric Chemistry and Physics*. 10, 5951-5964.
- Pojtanabuntoeng, T., et al., 2011. Effect of mercury on corrosion in production wells in Gulf of Thailand. *Corrosion Engineering, Science and Technology*. 46, 547-553.
- Pradella, N., et al., 2014. Fish assemblages associated with oil industry structures on the continental shelf of north-western Australia: FISH ASSEMBLAGES ON OIL INDUSTRY STRUCTURES. *Journal of Fish Biology*. 84, 247-255.
- Psaltaki, M., et al., 2013. TRS Cs CRwo-water values for the marine environment: analysis, applications and comparisons. *Journal of Environmental Radioactivity*. 126, 367-375.
- PTTEP AA, Jabiru and Challis Fields (Decommissioned State) Environment Plan Summary. PTTEP Australasia (Ashmore Cartier) Pty Ltd, 2014, pp. 18.
- Ram, R., et al., 2019. The aqueous chemistry of polonium (Po) in environmental and anthropogenic processes. *Journal of Hazardous Materials*. 380, 120725.
- Real, A., Garnier-Laplace, J., 2020. The importance of deriving adequate wildlife benchmark values to optimize radiological protection in various environmental exposure situations. *Journal of Environmental Radioactivity*. 211, 105902.
- Regoli, F., et al., 2019. Application of a Weight of Evidence Approach for Monitoring Complex Environmental Scenarios: the Case-Study of Off-Shore Platforms. *Frontiers in Marine Science*. 6.
- Remaili, T. M., et al., 2016. The impact of sediment bioturbation by secondary organisms on metal bioavailability, bioaccumulation and toxicity to target organisms in benthic bioassays: Implications for sediment quality assessment. *Environmental Pollution*. 208, 590-599.
- Rompalski, P., et al., 2019. Determination of mercury content in hard coal and fly ash using X-ray diffraction and scanning electron microscopy coupled with chemical analysis. *Arabian Journal of Chemistry*. 12, 3927-3942.

- Rosain, R. M., Wai, C. M., 1973. The rate of loss of mercury from aqueous solution when stored in various containers. *Analytica Chimica Acta.* 65, 279-284.
- Roseborough, D., et al., 2006. The surface behavior of mercury on iron systems. *Metallurgical and Materials Transactions B.* 37, 1049.
- Rouse, S., et al., 2018. Commercial fisheries interactions with oil and gas pipelines in the North Sea: Considerations for decommissioning. *ICES Journal of Marine Science.* 75, 279-286.
- Rouse, S., et al., 2019. Benthic Conservation Features and Species Associated With Subsea Pipelines: Considerations for Decommissioning. *Frontiers in Marine Science.* 6.
- Ruhland, D., et al., 2019. AF4-UV-MALS-ICP-MS/MS, spICP-MS, and STEM-EDX for the Characterization of Metal-Containing Nanoparticles in Gas Condensates from Petroleum Hydrocarbon Samples. *Analytical Chemistry.* 91, 1164-1170.
- Ryzhov, V. V., et al., 2003. Regular variations of the mercury concentration in natural gas. *Science of The Total Environment.* 304, 145-152.
- Sainal, M. R., et al., Mercury Removal System for Upstream Application: Experience in Treating Mercury From Raw Condensate. E&P Environmental and Safety Conference, Vol. SPE-106610-MS. Society of Petroleum Engineers, Galveston, Texas, U.S.A., 2007.
- Sainsbury, K., et al., 1997. Experimental management of an Australian multi-species fishery: examining the possibility of trawl-induced habitat modification.
- Sakamoto, M., et al., 2018. Health Impacts and Biomarkers of Prenatal Exposure to Methylmercury: Lessons from Minamata, Japan. *Toxics.* 6, 45.
- Salbu, B., Skipperud, L., 2009. Speciation of radionuclides in the environment. *Journal of Environmental Radioactivity.* 100, 281-282.
- Salvá, C., Gallup, D., Mercury Removal Process is Applied to Crude Oil of Southern Argentina. SPE Latin American and Caribbean Petroleum Engineering Conference, Lima, Peru, 2010.
- Santos Limited, WA-8-L Production Equipment Abandonment. 2018, pp. 68.
- Santos Limited, Bayu-Undan to Darwin Gas Export Pipeline Decommissioning & Preservation Environment Plan. Santos NA Darwin Pipeline Pty Ltd, 2020, pp. 443.
- Schaefer, J. K., et al., 2004. Role of the Bacterial Organomercury Lyase (MerB) in Controlling Methylmercury Accumulation in Mercury-Contaminated Natural Waters. *Environmental Science & Technology.* 38, 4304-4311.
- Schartup, A. T., et al., 2018. A Model for Methylmercury Uptake and Trophic Transfer by Marine Plankton. *Environmental Science & Technology.* 52, 654-662.
- Scheuhammer, A. M., 2012. Ecotoxicology of mercury in fish and wildlife recent advances. *Mercury in the Environment : Pattern and Process /.* 223-238.
- Schickling, C., Broekaert, J. A. C., 1995. Determination of mercury species in gas condensates by on-line coupled High-performance liquid Chromatography and Cold-vapor atomic absorption spectrometry. *Applied Organometallic Chemistry.* 9, 29-36.
- Schmidt, A. P., 2000. Naturally occurring radioactive materials in the gas and oil industry: origin, transport and deposition of stable lead and 210Pb from Dutch gas reservoirs. Univ, Utrecht.
- Schroeyers, W., 2017. Naturally Occurring Radioactive Materials in Construction : Integrating Radiation Protection in Reuse (COST Action Tu1301 NORM4BUILDING). Elsevier Science & Technology, Cambridge, UNITED KINGDOM.
- Shafawi, A. B., Mercury Species in Natural Gas Condensate. Department of Environmental Sciences, Vol. Doctor of Philosophy. University of Plymouth, Plymouth, 1999.
- Shaw, J. L., et al., Decommissioning offshore infrastructure: a review of stakeholder views and science priorities. WAMSI, Perth, Western Australia, 2018, pp. 74.

- Shell Australia Pty Ltd, Prelude FLNG Environment Plan. 2016, pp. 103.
- Silakorn, P., et al., The Technological Solution for Collecting Samples from Pipe Walls for Subsequent Analysis of Contaminants. SPE Symposium: Decommissioning and Abandonment, Vol. Day 2 Tue, December 04, 2018, 2018.
- Silakorn, P., et al., Alternative Method for Mercury Detection Using Square Wave Anodic Striping Voltammetry. SPE Symposium: Decommissioning and Abandonment, Vol. Day 1 Tue, December 03, 2019, 2019.
- Simpson, S., Batley, G., 2016. Sediment Quality Assessment: A Practical Guide. CSIRO Publishing.
- Sirelkhatim, D. A., et al., 2008. Distribution of 226Ra–210Pb–210Po in marine biota and surface sediments of the Red Sea, Sudan. Journal of Environmental Radioactivity. 99, 1825-1828.
- Skoog, D. A., et al., 2007. Principles of Instrumental Analysis. Thomson Brooks/Cole.
- Skyllberg, U., et al., 2021. Chemical speciation of mercury, sulfur and iron in a dystrophic boreal lake sediment, as controlled by the formation of mackinawite and frambooidal pyrite. Geochimica et Cosmochimica Acta. 294, 106-125.
- Slowey, A. J., Brown, G. E., 2007. Transformations of mercury, iron, and sulfur during the reductive dissolution of iron oxyhydroxide by sulfide. Geochimica et Cosmochimica Acta. 71, 877-894.
- Smith, A. L., NORM: The Lessons to be Learned, New Challenges and Innovative Thinking with Decommissioning and Radioactive Waste SPE International Conference on Health, Safety and Environment in Oil and Gas Exploration and Production. SPE International, Rio de Janeiro, Brazil, 2010, pp. 19.
- Smith, A. L., First Correlation of NORM with a Specific Geological Hypothesis. SPE European Health, Safety and Environmental Conference in Oil and Gas Exploration and Production, Vol. All Days, 2011.
- Snell, J. P., et al., 1996. Performance improvements in the determination of mercury species in natural gas condensate using an on-line amalgamation trap for solid-phase micro-extraction with capillary gas chromatography-microwave-induced plasma atomic emission spectrometry. Analyst. 121, 1055-1060.
- Spangenberg, J. V., Cherr, G. N., 1996. Developmental effects of barium exposure in a marine bivalve (*Mytilus californianus*). Environmental Toxicology and Chemistry. 15, 1769-1774.
- Stauber, J., et al., 2020. Application of Bioavailability Models to Derive Chronic Guideline Values for Nickel in Freshwaters of Australia and New Zealand. Environmental Toxicology and Chemistry. etc.4885.
- Stewart, G. M., Fisher, N. S., 2003. Experimental studies on the accumulation of polonium-210 by marine phytoplankton. Limnology and Oceanography. 48, 1193-1201.
- Stewart, G. M., et al., Chapter 8 The Bioaccumulation of U- and Th-Series Radionuclides in Marine Organisms. In: S. Krishnaswami, J. K. Cochran, Eds.), Radioactivity in the Environment. Elsevier, 2008, pp. 269-305.
- Suter II, G., et al., 2017. A weight of evidence framework for environmental assessments: Inferring qualities. Integrated Environmental Assessment and Management. 13, 1038-1044.
- Suter II, G. W., 2006. Ecological Risk Assessment. CRC Press, Boca Raton.
- Tao, H., et al., 1998. Mercury speciation in natural gas condensate by gas chromatography-inductively coupled plasma mass spectrometry. Journal of Analytical Atomic Spectrometry. 13, 1086-1093.
- Tchounwou, P. B., et al., 2003. Review: Environmental exposure to mercury and its toxicopathologic implications for public health. Environmental Toxicology. 18, 149-175.
- Tessier, A., Campbell, P. G. C., Partitioning of trace metals in sediments: Relationships with bioavailability. In: R. L. Thomas, et al., Eds.). Springer Netherlands, 1987, pp. 43-52.

- Thakur, P., Ward, A. L., 2020. 210Po in the environment: insight into the naturally occurring polonium isotope. *Journal of Radioanalytical and Nuclear Chemistry*. 323, 27-49.
- Todd, V. L. G., et al., 2018. Quantitative analysis of fish and invertebrate assemblage dynamics in association with a North Sea oil and gas installation complex. *Marine Environmental Research*. 142, 69-79.
- Todd, V. L. G., et al., 2020. Characterizing the first wave of fish and invertebrate colonization on a new offshore petroleum platform. *ICES Journal of Marine Science*. 77, 1127-1136.
- Torres, M. E., et al., 1996. Barite fronts in continental margin sediments: a new look at barium remobilization in the zone of sulfate reduction and formation of heavy barites in diagenetic fronts. *Chemical Geology*. 127, 125-139.
- Tung, A., A Comparison of Stakeholder Engagement Strategies for Offshore Decommissioning Projects in the United Kingdom and Australian Landscape. *Offshore Technology Conference*, Vol. Day 3 Wed, May 06, 2020, 2020.
- Uddin, S., et al., 2018a. 210Po concentration in selected calanoid copepods in the northern Arabian Gulf. *Marine Pollution Bulletin*. 133, 861-864.
- Uddin, S., et al., 2018b. 210Po concentration in selected diatoms and dinoflagellates in the northern Arabian Gulf. *Marine Pollution Bulletin*. 129, 343-346.
- Uğur, A., et al., 2011. Spatial and temporal variability of 210Po and 210Pb in mussels (*Mytilus galloprovincialis*) at the Turkish coast of the Aegean Sea. *Chemosphere*. 83, 1102-1107.
- Underwood, J. N., et al., 2020. Extreme seascape drives local recruitment and genetic divergence in brooding and spawning corals in remote north-west Australia. *Evolutionary Applications*. 13, 2404-2421.
- UNEP, Global Mercury Assessment - Sources, Emissions, Releases and Environmental Transport. United Nations Environment Programme, Switzerland, 2013.
- UNSCEAR, 1994. Sources and effects of ionizing radiation: United Nations Scientific Committee on the Effects of Atomic Radiation. United Nations, New York.
- UNSCEAR, 2008. Sources and effects of ionizing radiation: UNSCEAR 2008 report to the General Assembly. United Nations, New York.
- US DOE, RESRAD-BIOTA: a tool for implementing a graded approach to biota dose evaluation. Washington, D. C., USA, 2004.
- US DOE, A graded approach for evaluating radiation doses to aquatic and terrestrial biota. Vol. DOE-STD-1153-2019. Office of Public Radiation Protection, Washington, DC. , 2019.
- US EPA, Method 7473 (SW-846): Mercury in Solids and Solutions by Thermal Decomposition, Amalgamation, and Atomic Absorption Spectrophotometry. Washington, Dc., 1998a.
- US EPA, SW-846 Test Method 7473: Mercury in Solids and Solutions by Thermal Decomposition, Amalgamation, and Atomic Absorption Spectrophotometry. Washington, Dc., 1998b.
- US EPA, 2001. Lead and Lead Compounds; Lowering of Reporting Thresholds; Community Right-to-Know Toxic Chemical Release Reporting. Rules and Regulations., 66, 47.
- US EPA, Field Portable X-Ray Fluorescence Spectrometry for the Determination of Elemental Concentrations in Soil and Sediment. 2007a.
- US EPA, SW-846 Test Method 7471B: Mercury in Solid or Semisolid Waste (Manual Cold-Vapor Technique). 2007b.
- US EPA, SW-846 Test Method 7474: Mercury in Sediment and Tissue Samples by Atomic Fluorescence Spectrometry. 2007c.
- US EPA, SW-846 Method 6020B: Inductively Coupled Plasma - Mass Spectrometry. Washington DC., 2014a.

- US EPA, SW-846 Test Method 3200: Mercury Species Fractionation and Quantification by Microwave Assisted Extraction, Selective Solvent Extraction and/or Solid Phase Extraction. Vol. Revision 1, 2014b.
- US EPA, SW-846 Test Method 6010D: Inductively Coupled Plasma-Optical Emission Spectrometry (ICP-OES). 2018.
- Valeur, J. R., 2011. Environmental Impacts of NORM Disposal—With Emphasis on Discharges to Sea. SPE Projects, Facilities & Construction. 6, 124-131.
- van Dam, R. A., et al., 2017. Development and implementation of a site-specific water quality limit for uranium in a high conservation value ecosystem: A Site-Specific Water Quality Limit for Uranium. Integrated Environmental Assessment and Management. 13, 765-777.
- van Dam, R. A., et al., 2019. How Specific Is Site-Specific? A Review and Guidance for Selecting and Evaluating Approaches for Deriving Local Water Quality Benchmarks. Integrated environmental assessment and management. 15, 683-702.
- van Dam, R. A., et al., 2010. Aquatic toxicity of magnesium sulfate, and the influence of calcium, in very low ionic concentration water. Environmental Toxicology and Chemistry. 29, 410-421.
- Vetter, O. J., et al., 1982. Prediction of Scale Problems Due to Injection of Incompatible Waters. Journal of Petroleum Technology. 34, 273-284.
- Vives i Batlle, J., et al., 2011. The estimation of absorbed dose rates for non-human biota: an extended intercomparison. Radiation and Environmental Biophysics. 50, 231-251.
- Wang, Y., et al., 2015. Quantitative proteomic analysis reveals proteins involved in the neurotoxicity of marine medaka *Oryzias melastigma* chronically exposed to inorganic mercury. Chemosphere. 119, 1126-1133.
- Warne, M. S., et al., 2014. Revisions to the derivation of the Australian and New Zealand guidelines for toxicants in fresh and marine waters. Environmental Science and Pollution Research. 21, 51-60.
- Warne, M. S., et al., Revised Method for Deriving Australian and New Zealand Water Quality Guideline Values for Toxicants. Canberra, 2018, pp. 54.
- Waters, J. M., et al., 2010. Australia's marine biogeography revisited: Back to the future? Austral Ecology. 35, 988-992.
- Whicker, F. W., Schultz, V., 1982. Radioecology: nuclear energy and the environment. CRC Press.
- White, G. J., Rood, A. S., 2001. Radon emanation from NORM-contaminated pipe scale and soil at petroleum industry sites. Journal of Environmental Radioactivity. 54, 401-413.
- White, W. M., 2020. Geochemistry. John Wiley & Sons.
- Wiener, J. G., 2013. Mercury exposed: Advances in environmental analysis and ecotoxicology of a highly toxic metal. Environmental Toxicology and Chemistry. 32, 2175-2178.
- Wilhelm, S. M., 1999a. Avoiding Exposure to Mercury During Inspection and Maintenance Operations in Oil and Gas Processing. Process Safety Progress. 18, 178-188.
- Wilhelm, S. M., 1999b. Generation and Disposal of Petroleum Processing Waste That Contains Mercury. Environmental Progress 18, 130-143.
- Wilhelm, S. M., 2001a. Estimate of Mercury Emissions to the Atmosphere from Petroleum. Environmental Science & Technology. 35, 4704-4710.
- Wilhelm, S. M., Mercury in Petroleum and Natural Gas: Estimation of Emissions from Production, Processing, and Combustion. Tomball, Texas, 2001b, pp. 79.
- Wilhelm, S. M., Bloom, N., 2000. Mercury in Petroleum. Fuel Processing Technology. 63, 1-27.
- Wilhelm, S. M., et al., 2007. Mercury in Crude Oil Processed in the United States (2004). Environmental Science & Technology. 41, 4509-4514.

- Wilhelm, S. M., et al., 2006. Identification and Properties of Mercury Species in Crude Oil. *Energy & Fuels.* 20, 180-186.
- Wilhelm, S. M., Nelson, M., 2010. Interaction of Elemental Mercury with Steel Surfaces. *The Journal of Corrosion Science and Engineering.* 13.
- Williams, A., et al., 2010. Scales of habitat heterogeneity and megabenthos biodiversity on an extensive Australian continental margin (100–1100 m depths). *Marine Ecology.* 31, 222-236.
- Williamson, R. B., et al., 1999. Effect of burrowing by the crab *Helice crassa* on chemistry of intertidal muddy sediments. *Environmental Toxicology and Chemistry.* 18, 2078-2086.
- Wisnubroto, D. S., 2005. Implementasi Bss Pada Pengelolaan Norm-Tenorm. *Puslitbang Teknologi Maju-BATAN* Jogjakarta. 45-52.
- Woodside Energy Ltd, Browse FLNG Development Draft Environmental Impact Statement EPBC 2013/7079. 2014.
- Woodside Energy Ltd, OKHA Floating Production Storage and Offloading Facility Operations Environment Plan. 2019, pp. 867.
- Woodside Energy Ltd, Echo Yodel and Capella Plugging for Abandonment Environment Plan. In: Developments Division, (Ed.). Woodside Energy Ltd., 2020a, pp. 786.
- Woodside Energy Ltd, Nganhurra Operations Cessation Environment Plan. Woodside Energy Ltd, 2020b, pp. 1065.
- Wu, P., et al., 2019. The importance of bioconcentration into the pelagic food web base for methylmercury biomagnification: A meta-analysis. *Science of The Total Environment.* 646, 357-367.
- Yan, Q., et al., Geochemical Characteristics of Mercury in Oil and Gas. 2017 International Conference on Environmental and Energy Engineering (IC3E 2017), Vol. 63. Earth and Environmental Science, 2017, pp. 012024.
- Yang, Y., et al., 2020. Characterization, formation and development of scales on L80 steel tube resulting from seawater injection treatment. *Journal of Petroleum Science and Engineering.* 193, 107433.
- Ye, Y.-j., et al., 2019. Effects of temperature, salinity, and pH on 222Rn solubility in water. *Journal of Radioanalytical and Nuclear Chemistry.* 320, 369-375.
- Yu, L.-P., Yan, X.-P., 2003. Factors affecting the stability of inorganic and methylmercury during sample storage. *Trends in Analytical Chemistry.* 22, 245-253.
- Zettlitzer, M., Kleinitz, W., Mercury in steel equipment used for natural gas production-- amounts, speciation and penetration depth. *Oil gas European Magazine* Vol. 23, 1997, pp. 25-30.
- Zettlitzer, M., et al., Determination of elemental, inorganic and organic mercury in North German gas condesates and formation brines. SPE International Symposium on Oilfield Chemistry. Society of Petroleum Engineers, Inc., Houston, Texas, 1997.
- Zhang, C., et al., 2014. Effects of sediment geochemical properties on heavy metal bioavailability. *Environment International.* 73, 270-281.
- Zhang, Y., et al., 2016. Observed decrease in atmospheric mercury explained by global decline in anthropogenic emissions. *Proceedings of the National Academy of Sciences.* 113, 526-531.
- Zhong, Q., et al., 2019. The 210Po / 210Pb disequilibrium in a spring-blooming marginal sea, the Southern Yellow Sea. *Journal of Environmental Radioactivity.* 207, 15-26.
- Zougari, M. I., 2010. Shear driven crude oil wax deposition evaluation. *Journal of Petroleum Science and Engineering.* 70, 28-34.

## Attachment A - Problem Statement



### NDRI Problem Statement

'Risk-based Marine Impact Assessment of Naturally Occurring Radioactive Materials (NORMs) and Mercury from Decommissioning Oil and Gas Infrastructure'

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## 1. Problem Statements

### Proposed Problem Statement 1 – Understanding thresholds of NORM and Mercury in the environment

#### Statement

- o The Australian oil and gas industry-seeks to understand the threshold levels at which NORM in subsea production equipment and Mercury in subsea gas transmission systems in the offshore environment become a concern during and after decommissioning, relative to the receiving environment, specifically: levels/dosage/concentration, form/chemical species and likely behaviour (how are they likely to leach over time and what are the likely pathways) if NORM and mercury is present in the O&G infrastructure.

#### Project Objective

- **Objective 1:** Consolidate the state of knowledge on protective standards and impact of NORMs/Mercury exposure on the marine environment at various concentrations.
- **Objective 3:** Science-based understanding of NORM/Hg location, concentration and amount, speciation, mobility and quantified risk in marine receptors, considering degradation scenarios of left in-situ subsea infrastructure.

#### Background

- We are looking for a literature review on impacts and pathways, noting that previous efforts to identify this information has identified less than a dozen published papers. It is noted that some NDRI partners have work underway, such as research into ecological frameworks for closure of offshore petroleum structures, which could support this literature review.
- We are interested in understanding the levels of NORM and Hg that have been noted in the environment, specifically those of concern. In this context, we are keen to understand the form/chemical specie and how they are likely to leach, over what time and where. For example, only when pipelines break down and, if they do break down, how have these been absorbed/consumed within the environment.
- We are interested in understanding the potential application of guidelines, as proposed by the study. It would be useful to see if any research papers exist for prediction of Hg concentrations in sediment after pipeline degradation or a simulated degradation has taken place. Similarly, in relation to NORM, has research been performed on leaching of NORM scale.
- We are keen to recognise that this isn't a one-size fits all situation for the application of guidelines. It would be valuable to compare, for example, the ANZACC/ARMCANZ/ARPANSA guidelines to other protective standards or guidelines. In the context of this project, we are seeking feedback from the researchers on guidelines that are relevant or irrelevant for decommissioning offshore oil and gas structures. It is noted that industry participants have views on how well some of these guidelines suit and the challenges with application. It is noted that NORM and Hg are managed differently under regulation, with Hg having specific total limits/trigger values under some guidelines. Exceedence prompts site specific analysis and monitoring.



### Proposed Problem Statement 2 – Understanding the Fate of NORMs and Mercury

#### Statement

- The Australian oil and gas industry seeks to understand the processes and exposure pathways whereby individual contaminants might (or might not) reach receptors after decommissioning.

#### Project Objective

- **Objective 4:** Develop a COPC framework to quantify the combined impact and risk of NORM and mercury to marine receptors of left in-situ subsea infrastructure.

#### Background

- This Problem Statement is considered critical/of highest priority for the overall study.
- The project should seek to delineate exposure pathways under nominated decommissioning scenarios, describe known protective standards, and impact assessment. It should articulate the processes whereby contaminants might (or might not) reach receptors during decommissioning alternatives should be a focus for this study.
- It is suggested that this project should not be treating impacts and risk in a 'combined' way.
- Industry will provide support in relation to what represents standard practice for preparation for decommissioning, such as cleaning of equipment, in order to establish a real-world set of alternatives and processes.

### Proposed Problem Statement 3 – Understanding of Tools and Procedures

#### Statement

- The Australian oil and gas industry seeks to improve its tools and procedures for measuring Mercury and NORM associated with equipment/infrastructure in-situ to inform decommissioning decisions and long term monitoring objectives.

#### Objectives

- **Objective 2:** Determine the applicable standards and guidelines for the identification and measurement of NORM and mercury types and quantities in oil and gas subsea infrastructure.

#### Background

- The ability to identify and measure NORM and mercury in the O&G infrastructure is key towards enabling a characterisation of the environmental impacts and risks. The focus should be on existing methods and approaches, not on improving tools and procedures.
- The industry is seeking to understand standards, guidelines, procedures and technologies for in-situ/subsea measurement of NORM and Mercury. Some companies possess this capability in-house and some companies rely on external speciality providers.
- Some companies, such as BHP, have developed tooling to perform subsea measuring of NORM using spectral techniques. It is not known whether there is similar technology for Hg measurements. Surface measurement of Hg is still a difficult task to perform in a lab and still can't be performed externally to a pipeline subsea to our knowledge. There is a HG JIP being performed in the US for the purpose of identifying and possibly developing guidelines for best practice in surface measurement of Hg.



## 2. Additional Comments and Questions

- It is suggested that Problem Statement 2 should be addressed first.
- NDRI will need to confirm alignment of timelines between Projects 2.1 (metal degradation), 2.2 (non-metal degradation) and 2.3/2.4 (this project) and the extent to which those projects can offer insights into potential degradation rates. If timeline on those projects is out of sync, it is suggested this project should use a predicted release scenario to inform potential release rates into the marine environment of COPC.
- Proposed decommissioning scenarios:
  - For each decommissioning scenarios, consider 1) the removal activities; and, 2) decommissioning in-situ.
  - What are the exposure pathways?
  - What are the concentrations in the marine environment resultant from the exposures?
  - What are the impacts to the marine receptors from decommissioning?
- Decommissioning of anodes
  - Anodes (containing mercury) are used for cathodic protection of O&G infrastructure. Anodes compositional information can be provided. Anodes can be decommissioned in-situ or removed.
- Decommissioning of pipelines
  - For the purposes of decommissioning, pipelines are flushed and cleaned. Residual contaminants could remain in the pipelines.
  - Pipelines could be severed/cut for decommissioning. Are there exposure pathways due to the severing/cutting? For pipeline removal, what are the exposure pathways during removal activities?
  - Pipelines that are decommissioned in-situ can be 1) plugged/capped, or 2) open-ended. How do the degradation of metal and non-metal components of the pipelines affect the release rate of residual NORM and mercury to the environment? How does the change in burial status (e.g., pipelines on the seabed can be buried over time due to natural sedimentation) affects the release rate and exposure pathways of the NORM and mercury in the pipelines?

## Attachment B – Study Basis (Assessment Phase)

TR-13266-2



Curtin University

OIL AND GAS  
INNOVATION CENTRE

**Confidential**

# Risk Based Marine Impact Assessment of NORM and Hg from Decommissioning Oil & Gas Infrastructure

Curtin University Oil & Gas Innovation Centre

## Assessment Phase Study Basis

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# Assessment phase plan

## Aim and Objectives

The assessment phase aims to integrate and operationalise the understandings captured in the literature review into a best practice ecological risk assessment framework for mercury and NORM in offshore pipelines. The understanding developed in the literature review will guide the identification of practical metrics for a risk framework, and guide the assessment of how site-specific environmental variables influence risk. This will be progressed as far as possible within the data, desktop, and time constraints.

This will be achieved by four objectives:

1. Define management objectives specific to the ecological risk of mercury and NORM in subsea pipelines
2. Develop an ecological risk framework for mercury and NORM specific to the decommissioning of subsea pipelines
3. Assess how site-specific environmental variables influence the ecological risk of mercury and NORM
4. Identify technology options for the in-situ measurement of mercury and NORM.

Each step will also define the key phase 2 research gaps and data needs.

## Deliverable

At the end of the assessment phase, the research team will produce an assessment report that consolidates existing scientific knowledge of mercury and NORM hazard impacts and exposure in an ecological risk framework for offshore pipeline decommissioning that can be used to proactively predict risk in subsea pipelines.

## Assessment Phase Execution

The assessment phase of this NDRI project is a 9-month period between April and December 2021 and will be executed to align with the three stated objectives. Objective 1 will be addressed first followed by a parallel assessment of objectives 2 and 3.

In accordance with the current project schedule the assessment phase reporting was targeted for October 2021 and the findings, inclusive of the draft risk framework, presented to NDRI/IP/ISAB in November 2021. This sequence enabled an incorporation of feedback with a final revision and issue of the assessment documentation before the end of 2021.

We are proposing the same milestone targets but will combine the NORM and mercury assessment work into a single report to simplify the deliverables and review process.



### Objective 1. Define management objectives specific to the ecological risk of mercury and NORM in subsea pipelines

Ecological risk assessments support environmental decision making by giving risk managers an approach to assess environmental impacts. This occurs within a legislative framework defining the environmental outcome for activities that pose a risk to ecosystems. To develop an ecological risk framework, management objectives need to be defined to guide the collection of lines of evidence that can support an assessment of risk acceptability. To support this, uncertainties around existing guideline documents also need to be clarified.

The management objectives are envisioned to be statements that if met will demonstrate adherence to the legislative requirements for decommissioning (such as for acceptable environmental impacts, equal or better environmental outcomes than complete removal, and ensuring accord with ecologically sustainable development principles). These will be specific to the risk that mercury and NORM from subsea pipelines pose to ecosystem receptors that have been identified in the literature review.

Possible management objectives may include:

1. Ensure ecosystem habitat function and structure will be maintained or enhanced compared to baseline conditions following pipeline decommissioning.
2. Ensure water and sediment quality will be maintained following pipeline decommissioning.
3. Ensure the community's ongoing use of marine resources (i.e. commercial, recreational, Indigenous use) is not negatively impacted by decommissioning actions.

Uncertainties around existing environmental guidelines relevant to mercury and NORM need to be clarified. For example, it remains unclear whether IAEA/ARPANSA exemption and exclusion criteria should apply to NORM in pipelines, whether the National Assessment Guidelines for Dredging are applicable in this context, and whether the water quality management framework is a suitable approach to assessing contaminant risk for future exposure scenarios. These questions will be answered through consultation with relevant stakeholders including NOPSEMA, APRANSA, and Department of Agriculture, Water, and Environment advisory members.

#### Outcomes

- Defined management objectives against which risk will be considered including:
  - Specifying how legislative requirements for decommissioning apply to mercury and NORM risks to ecosystem receptors
  - The risk assessment framework will be developed to assess against these requirements
- Clarification around the use of existing guideline documents in the context of subsea pipeline decommissioning

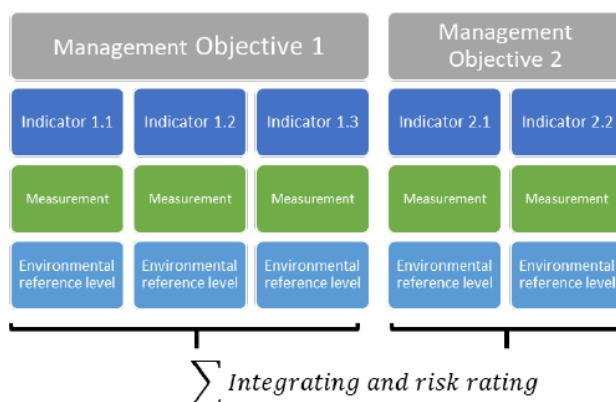


**Objective 2. Develop an ecological risk framework for mercury and NORM specific to the decommissioning of subsea pipelines**

Ecological risk assessment frameworks are simple, flexible procedures for conducting and evaluating ecological risk. Such frameworks provide increased transparency and confidence in assessment outcomes.

An ecological risk assessment framework will be developed by defining relevant lines of evidence and indicators. These are measurable aspects of NORM or mercury's exposure paths or hazard impacts that can be used to inform their risk against the management objectives defined in Objective 1. For the risk assessment framework, environmental reference levels will be identified for indicators that can be used to quantify potential impact. Alternatively, reference levels based on comparisons to background or control sites will be explored. Each reference level will be justified based on published evidence describing the relationship between indicators and impact to the management objective. Finally, possible mechanisms to integrate multiple lines of evidence to assess risk will be explored. This may be simple logic tables or more complicated regression or Bayesian-based approaches where data availability allows.

This process is summarised visually below:



**Outcomes**

- A risk assessment framework for mercury and NORM in offshore pipelines that includes:
  - Justified indicators that can be used to predict ecological impact of mercury and NORM to ecosystem receptors
  - Environmental reference levels that relates contaminant concentration to hazard impact
  - Possible methods for the integration of multiple lines of evidence into a single assessment of risk against the management objectives
- Measurement requirements for different indicators
- Gap analysis outlining priorities for future research



**Objective 3.** Assess how site-specific environmental variables influence the ecological risk of mercury and NORM

Environmental factors are known to affect contaminants' exposure pathways or hazards to ecosystem receptors. This may have significant impacts to the risk of mercury and NORM within individual assets (which exist in heterogenous environments) or between different environments.

To understand the potential influence of environmental variables on the risk of mercury and NORM, likely contamination scenarios and environmental parameters describing pipelines in Australian oil and gas-producing basins need to be established. Mercury and NORM concentration ranges in contaminated products were identified in the literature review, but ranges of environmental parameters need to be determined. These ranges may be consolidated from available sources, such as field development plans, environment plans, marine bioregional plans, and published studies. And may include physical and chemical variables such as pH, temperature, asset depth (e.g. water pressure) and sediment characteristics (e.g. particle size, presence of organic carbon, redox status).

The relationship between environmental variables and the risk of mercury and NORM will then be explored using previously published research. For example, mercury methylation is highly variable and controlled by the presence of anaerobic bacteria whose activity is related to the sediment organic content, redox condition, and temperature. These environmental variables may therefore control the rate at which mercury is methylated.

The defined environmental parameter-risk relationship will be used to understand how risk varies along environmental gradients (possibly including depth and temperature) and between oil and gas-producing basins. This will help to define which components contribute uncertainty in the risk framework and which may be easily addressed with additional research.

**Outcomes**

- A description of the relationship between environmental parameters and their influence to mercury and NORM risk in the marine environment
- An understanding of the indicators that most contribute to risk or have the greatest uncertainty
- An understanding of environmental conditions that increase or decrease ecological risk
- Recommendations for future research that can reduce risk assessment uncertainty



**Objective 4. Identify technology options for the in-situ measurement of mercury and NORM.**

The identification and quantification of mercury and NORM in oil and gas infrastructure may be an important step to understanding its overall risk. It is also important to the verification of mitigation options and assurance of the assets 'as is' condition for decommissioning. While good laboratory-based techniques exist for the measurement of mercury and NORMs, no *in-situ* subsea measurement techniques currently exist.

A brief overview of technology options, their benefits and limitations, and their connection to the risk assessment framework developed in Objective 2 will be developed. This may be achieved through target surveys of operators, suppliers, and researchers. Alternatively, existing work being undertaken by operators on this topic could be expanded by connecting measurement options and limitations with risk assessment needs.

**Outcomes**

- An overview of technology options for the *in-situ* measurement of mercury and NORM in subsea pipelines, their readiness level and likely effectiveness against requirements