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Research Paper

An approach to assess potential environmental mercury release, food web bioaccumulation, and human dietary methylmercury uptake from decommissioning offshore oil and gas infrastructure

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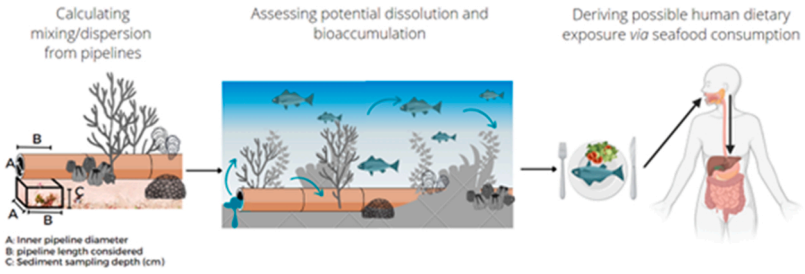
HIGHLIGHTS

- Calculated pipeline mercury threshold to avoid exceeding environmental guidelines.
- Novel application of fisheries management programme for contaminant tracking.
- Marine mercury contamination could impact future food webs.
- Offshore decommissioning plans could benefit from additional risk assessments.

GRAPHICAL ABSTRACT



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ABSTRACT

Subsea pipelines carrying well fluids from hydrocarbon fields accumulate mercury. If the pipelines (after cleaning and flushing) are abandoned in situ, their degradation may release residual mercury into the environment. To justify pipeline abandonment, decommissioning plans include environmental risk assessments to determine the potential risk of environmental mercury. These risks are informed by environmental quality guideline values (EQGVs) governing concentrations in sediment or water above which mercury toxicity may occur. However, these guidelines may not consider e.g., the bioaccumulation potential of methylated mercury. Therefore, EQGVs may not protect humans from exposure if applied as the sole basis for risk assessments. This

List of abbreviations: CA, Comparative Assessment; CR, Concentration Ratio; EFSA, European Food Safety Authority; ERA, Environmental Risk Assessment; EQGVs, Environmental Quality Guideline Values; EWI, Estimated Weekly Intake of methylmercury; EwE, Ecopath with Ecosim; FAO, Food and Agriculture Organisation of the United Nations; FS, Food Standards (for mercury); Hg^0, Elemental mercury; IAEA, International Atomic Energy Agency; ICES, International Council for the Exploration of the Sea; K_d, Partitioning coefficient; M/O ratio, Model versus Observed ratio; TWI, Tolerable Weekly Intake of methylmercury; SQGV, Sediment Quality Guideline Value; UNCLOS, United Nations Convention on the Law of the Seas; WHO, World Health Organisation; WQGV, Water Quality Guideline Value; XRF, X-Ray Fluorescence.

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paper outlines a process to assess the EQGVs' protectiveness from mercury bioaccumulation, providing preliminary insights to questions including how to (1) determine pipeline threshold concentrations, (2) model marine mercury bioaccumulation, and (3) determine exceedance of the methylmercury tolerable weekly intake (TWI) for humans. The approach is demonstrated with a generic example using simplifications to describe mercury behaviour and a model food web. In this example, release scenarios equivalent to the EQGVs resulted in increased marine organism mercury tissue concentrations by 0–33 %, with human dietary methylmercury intake increasing 0–21 %. This suggests that existing guidelines may not be protective of biomagnification in all circumstances. The outlined approach could inform environmental risk assessments for asset-specific release scenarios but must be parameterised to reflect local environmental conditions when tailored to local factors.

1. Introduction

1.1. Offshore oil and gas infrastructure decommissioning

Oil and gas infrastructure nearing the end of their life may be decommissioned in different ways, where options include the complete or partial removal, or in situ abandonment, of submerged structures [1]. The preferred approach depends on, among other factors, the environmental impacts and benefits associated with each option. In some jurisdictions, for example, the operator of the facility to be decommissioned must undertake a comparative assessment (CA) of each of the abandonment options for the infrastructure. These assessments need to consider all receptors potentially impacted by the decommissioning activities in the form of environmental risk assessments (ERA). The objective of CAs and ERAs is to determine decommissioning scenarios with minimal impact on decommissioning operations and ecosystems, and to mitigate potential human risk. For example, the EU directives 2008/56/EU (Marine strategy framework directive [2]) and 2011/92/EU [3] outline the need to identify, describe, and assess the direct and indirect impacts of each 'project' on biota, the environment, material assets, and cultural heritage. Special focus should be given to oil and gas pipelines, as well as contamination by oil or gas exploration. Similar legislation exists in Australia, necessitating environment plans to demonstrate acceptable environmental impacts/risk levels for any decommissioning activity.

In situ decommissioning, where subsea infrastructure is cleaned and abandoned in the marine environment, is considered a cost and time effective decommissioning option for pipelines. The structures can be abandoned entirely or in parts, and different cleaning protocols are available for the pipelines' inner surface before abandonment [4,5]. This is important because contaminants are known to accumulate in subsea infrastructure which may pose an unacceptable risk to the marine environment [6,7]. A plethora of international treaties and legislations have been developed over the years to ensure protection and sustainable use of the marine environment that apply to in situ decommissioning decisions. These include: (1) The 'United Nations Convention on the Law of the Sea' (UNCLOS), which outlines the rights and responsibilities of nations for the use of the ocean, and that any abandoned/disused installations must be removed with consideration for the environment [8]. It also states that the exploitation of continental shelf may not interfere with navigation, fishing, or conservation works, and that all abandoned/disused structures must be removed entirely, as previously stated under the 'Convention on the Continental Shelf 1958' [9]; (2) the 'Convention on the Prevention of Marine Pollution by dumping of wasted and other matters 1972' (London Convention) and the 'London Protocol', which aim to prevent marine pollution by regulating the dumping of wastes and other matters [10]; and (3) the 'Minamata Convention on Mercury' which is designed to protect human health and the environment from anthropogenic emissions and releases of mercury and mercury compounds [11]. The regulatory threshold to justify in situ abandonment of pipelines (e.g., in Australia) is currently governed by the need to demonstrate equal or better environmental and health and safety outcomes than complete removal, as well as meeting all further regulatory and legal requirements [12].

1.2. Mercury in the marine environment

Mercury is a contaminant of concern for offshore industries, because it is a naturally occurring heavy metal present in oil and gas reservoirs [13,14]. Other processes associated with the anthropogenic release of mercury include mining and smelting, coal burning, cement production, and artisanal gold mining [15]. In the marine environment mercury can form a range of chemical species [16], depending on local factors such as the sediment redox potential, pore-water sulphide concentration, sediment organic matter content, pH, and sediment texture [17–20]. Many mercury species are not very water soluble, but within abandoned pipelines some (such as elemental mercury; Hg^0) may leach into the dissolved phase over time. Upon release, they will disperse and speciate depending on the local environment leading to their partitioning between the water column and sediment phase [7]. In the sediment, site-specific parameters will influence its bioavailability by forming e.g., insoluble sulphide complexes in the presence of relevant pore water sulphide concentrations, or more bioavailable species [17].

The speciation of mercury, together with environmental parameters, affect its availability for methylation by certain bacterial communities. If methylated, mercury forms toxic and bioaccumulative organomercurials (e.g., mono- and di-methylmercury, herein methylmercury) [21]. Methylmercury has a higher propensity to bioaccumulate and biomagnify than other mercury species [22], along with greater toxicity to marine organisms [23]. Exposure to methylmercury can lead to developmental neurotoxicity in foetuses, as well as various adverse effects after birth [24], Due to its biomagnification, it may adversely affect high trophic level organisms such as marine mammals and large predators in marine food webs [25].

Mercury is a contaminant of global concern, as outlined by the Minamata Convention on Mercury [26]. The convention's goal is the reduction of mercury emissions from sources such as the industrial processess, artisanal small-scale gold mining, medical and municipal waste incineration, and sulphide ore roasting [27,28]. Oil and gas extraction is also identified as an activity that may release mercury. During hydrocarbon extraction, mercury remains in the produced petroleum derivatives (mostly as Hg^0 and sulphide-bound mercury) [29,30]. Mercury accumulates in the production infrastructure through deposition to pipeline surfaces via condensation, adsorption to steel and into corrosion products, as well as in e.g., sludges and produced water [4,31,32]. Historically, studies assessing the impact of oil and gas production on the marine ecosystem have focused on contamination occurring during the infrastructures' operational life, rather than the potential environmental contamination of in situ decommissioned pipelines. If pipelines are to be abandoned in situ, guidelines on the acceptable levels of residual contaminants, such as Hg, must be developed. A detailed review of subsea pipeline associated mercury and the ecological risk has recently been published, providing further insights into these aspects [7].

1.3. EQGVs and consumer protection

Environmental quality guideline values (EQGVs) are values below which there is expected to be a low probability that a pollutant will have a negative impact on the environment, based on a particular measure of

impact [33]. EQGVs for mercury in the marine environment exist for a range of jurisdictions to support contaminant ecological risk assessments. For the sediment compartment, the most common guidelines (referred to as sediment quality guideline values; SQGVs) range from 0.13 to 0.7 mg/kg dry sediment [34–36], although some nations are more stringent, with a SQGV of 0.07 mg/kg dry sediment [37,38]. The water quality guideline values (WQGVs) ranged from 0.016 to 1.4 μg/l [35,39], with more variation between the nations than was the case for the SQGVs. An overview of the minimum EQGVs of various nations can be found in Table S1. These values are based on toxicity data obtained from laboratory and field studies of species from a range of different taxonomic groups [40]. However, these values do not necessarily protect against the potential long-term impacts arising from mercury biomagnification, affecting marine organisms or seafood consumers.

Seafood consumers are protected by regulations on mercury in landed fish. These include the food standards (FS) for total mercury in tissue and the tolerable weekly intake (TWI) for methylmercury. The FS describe the acceptable levels of mercury in fish muscle tissue for commercially sold species and recommended fish consumption for humans per week. Due to the bio-accumulating and -magnifying nature of mercury, these FS often account for the trophic position of the species in question and provide separate values for species that are more or less likely to accumulate high amounts of mercury (Table S1). The TWI aims to limit human dietary exposure to methylmercury due to its highly toxic nature, and because most of the mercury measured in fish tissue is methylmercury [41]. Most countries implement the TWI of 1.3 μg methylmercury/kg bodyweight (herein μg/kg) as outlined by the European Food Safety Authority (EFSA) [42], which is more conservative than the previously determined TWI of 1.6 μg/kg provided by the FAO/WHO [43].

1.4. Aims and objectives

The sustainable development principle of intergenerational equity, that the present generation should ensure that the health, diversity, and productivity of the environment are maintained or enhanced for the benefit of future generations, requires that the long-term fate and impacts of mercury are considered in any decommissioning activity [44]. The current EQGVs were developed to protect the local ecosystem from potential toxic effects but do not necessarily protect against biomagnification impacts. They are also not intended to be used as the sole point of comparison for environmental impact and risk assessment studies that consider human health.

This paper aims to demonstrates an approach to assess the potential biomagnification impacts from mercury releases to marine food webs. We use existing EQGVs as input mercury concentrations to explore how protective EQGVs are of mercury biomagnification impacts in the marine food web. In this study, EQGVs were first used to back-calculate pipeline mercury threshold concentrations that describe the concentration of mercury in pipelines that when released to the environment will not exceed EQGVs for sediments and seawater. Then, a hypothetical marine food web was used to outline the applicability of the modelling programme Ecopath with Ecosim (EwE) for the determination of mercury bioaccumulation in a food web using generic parameterisation, upon which site-specific assessment can be built. Four trials describing hypothetical release scenarios were selected to determine the application to calculate future biota mercury concentrations. The derived biota mercury concentration data was then used to showcase the potential of this method to assess human risk by assessing the possible exceedance of the FS for mercury in foods by calculating dietary exposure to methylmercury, using published global weekly fish consumption data. The example implementation of this study's approach does not intend to draw conclusions on the direct health implications or to discern the actual intake of methylmercury of future generations of seafood consumers. Rather, the data needs for each step for future site-specific parametrisations are discussed, to inform the local implementations of such methods.

2. Methods

To determine pipeline mercury threshold concentrations that would not exceed the sediment and water EQGVs under conservative release scenarios (defined as estimates that err on the side of caution, also referred to as a 'worst case scenario' [45]) a simple calculation is proposed (part A). Modelling is then proposed to determine the bioaccumulation potential of mercury in a marine food web and the potential for future exceedance of FS and TWIs (part B). The following steps were taken:

A. The pipeline mercury threshold concentrations are calculated using different approaches, to provide output values in area, length, and mass-based units, allowing for the comparison to data derived from different measurement techniques such as acid digest, pipeline pigging, or X-ray fluorescence (XRF) assessment of pipeline coupons. This simplifies direct comparison between actual measurements and threshold values with the same unit of measurement, standardising the approaches.
B. A hypothetical food web designed in EwE, using the Ecotracer contaminant tracking tool, was used to assess the mercury accumulation of different contamination scenarios. The output was converted to muscle tissue concentrations and compared to current FS for mercury in fish. The muscle tissue concentrations then informed the estimated weekly ingestion of methylmercury (EWI) for humans from seafood to determine the potential future human risk of increased methylmercury via dietary uptake.

Where possible, realistic parameterisations and considerations are given for each step in the outlined approach. However, these are not specific to a particular location or food web, Additionally, various data gaps necessitated the use of parameterisations that may be unrealistically simplistic. For the purpose of demonstrating the approach, these were selected to ensure they represented a conservative exposure scenario (i.e., assumed that all mercury was bioavailable). The assumptions are highlighted throughout the methods section and further requirements are discussed in more detail in the discussion.

2.1. A: pipeline mercury threshold calculation

To simplify the approach, a 1 kg sediment compartment (hereafter called the sediment box) underlying a pipeline is postulated, to allow for the calculation of a pipeline mercury threshold (Fig. 1). This assumes that at the end of pipe decay all the mercury falls in this sediment box.

After determining the mercury concentration in the box, the resulting water contamination can be calculated using a partitioning coefficient (K_d value). A K_d value can be used to approximate the behaviour of mercury in the marine environment, with a sediment-water K_d value describing the relationship between the solid and dissolved phase [46], assuming an equilibrium between the contaminant fraction in the

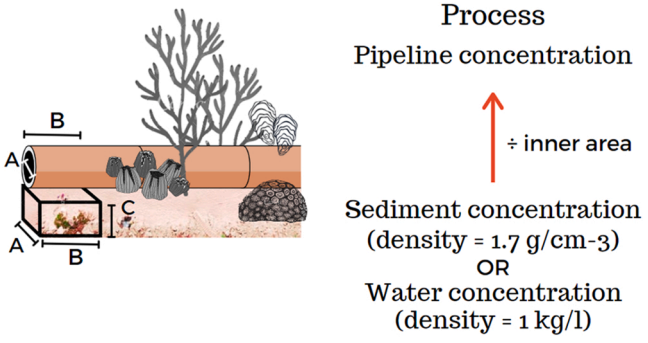

Fig. 1. Area-based approach to determining the pipeline mercury threshold according to the SQGV and WQGV.

sediment and the water column:

$$K_d(l/kg) = \frac{Hg\ concentration\ per\ unit\ sediment\ mass(kg/kg)}{Hg\ concentration\ per\ seawater\ volume(kg/l)} \quad (1)$$

The recommended K_d value for open ocean of 4×10^3 by the IAEA [47] was applied in the following work, a value below commonly reported empirical values (typically $> 10^4$) [48,49]. This value will most likely overestimate the amount of mercury in the dissolved phase, but follows the approach recently discussed in an environmental plan for the decommissioning of a gas export pipeline [50]. As recommended by the IAEA, K_d values higher and lower than the recommended value were used (here 4×10^2 and 4×10^4), to determine the percentage change between the K_d values. The determined difference for the IAEA recommended K_d values was between 1 % and 10 % from the 4×10^3 K_d value (data not shown) indicating that a tenfold change in K_d value induced only a 10 % change in resulting maximum permissible pipeline concentration and thus supporting the use of the chosen K_d value. No change in steel density due to corrosion is considered, and the calculations used the API 5L pipeline dimensions (Table S2). The calculations were done in Excel (Microsoft, Version 2201) and the Tidyverse package for R [51, 52] was used for data visualisation.

2.1.1. Area/Length-based calculation

The diameter of the pipeline (A) is fixed to represent the width of the underlying sediment box. Pipelines used for subsea oil and gas infrastructures range from 2 ″ diameter for gas pipelines to 14 ″ for export pipelines. Additionally, the wall thickness varies, typically being between 6 and 40 mm for frequently used pipelines (see Table S2 for the pipeline dimensions considered here). The depth of the sediment box (C) varied, with 5 cm used to derive the minimum and 10 cm the maximum permissible concentration. 5–10 cm is a common depth for taking sediment samples (shown as 'min' and 'max' in the results), and the density of marine sediment/water (δ) is known. Thus, the length of the sediment box (B):

$$\mathbf{B} = (\mathbf{A} * \mathbf{C})/\boldsymbol{\delta} \quad (2)$$

B also represents the length of the pipeline section. Using this value, and the inner diameter of the pipeline, the inner surface area (α) of the pipeline can be determined:

$$\boldsymbol{\alpha} = \mathbf{2}\boldsymbol{\pi} \boldsymbol{r} \boldsymbol{B} \quad (3)$$

The resulting information can then be applied to determine either the area- or the length-based pipeline threshold values for mercury contamination considering the guideline value for either the sediment or water compartment.

2.1.2. Mass-based calculation

Given the known mass of the pipeline material, Eqs. (1)–(3) derived above can also be converted to provide an output value as weight/length unit (kg/m) in the mass-based approach for comparison with data of mercury concentration obtained from e.g., pigging (Fig. 2). This approach considers the pipeline parameters to compute the weight of the pipeline segment.

The output derived from length-based approach was divided by the weight of the steel pipe:

$$Compartment_{max}(mg/kgsteel) = Compartment_{max}(mg/m)/Steelweight(kg/m) \quad (4)$$

2.2. B: food web modelling

EwE is a mass balance food web modelling programme, which allows for a static mass balance (Ecopath), as well as temporal (Ecosim) and spatiotemporal modelling (Ecopath). In addition, a tool for the tracing of contaminants has been developed (Ecotracer), using already established models and equations. A detailed description of the programmes main equations and underlying parameters can be found in the publication by Christensen and Walters [53].

The Ecotracer tool within EwE simulates the transport of any contaminant through the food web, solving the contaminant dynamic equation simultaneously with the outlined EwE equation [54]. It allows for a varied contaminant influx over time and considers different decomposition/outflow methods. The underlying assumption is that the contaminant is either within the environment (the water compartment), or the species (functional group). Each functional group is considered as a compartment and can thus have different contaminant concentrations.

2.2.1. 'Anchovy Bay'

A hypothetical food web called 'Anchovy Bay' was used for this study (Fig. 3) which was previously part of a course taught at the University of British Columbia [55]. This food web has been verified, is mass-balanced, and spans 4 trophic levels, which can be seen as representative of most marine food webs. The model was selected to demonstrate a potential method for the determination of mercury bioaccumulation in marine food webs that can be used to assess environmental risks related to activities, such as offshore oil and gas decommissioning. Due to a lack of mercury concentration and decommissioning data for locations globally, the hypothetical 'Anchovy Bay' model was thus chosen, to demonstrate the applicability of Ecotracer, as it has been used in past contaminant tracking examples (see Walters and Christensen [54]) and will now be applied to decommissioning-related contaminant accumulation in marine food webs. Input data, as well as the dietary matrix, output parameters and statistical assessments of the model can be found in Table S3.

Four different trials were run with the 'Anchovy Bay' model, to assess the impact of different release scenarios on the bioaccumulation of mercury in the food web over a total of 200 years via Ecosim. All concentrations of mercury influx are based on literature-derived values,

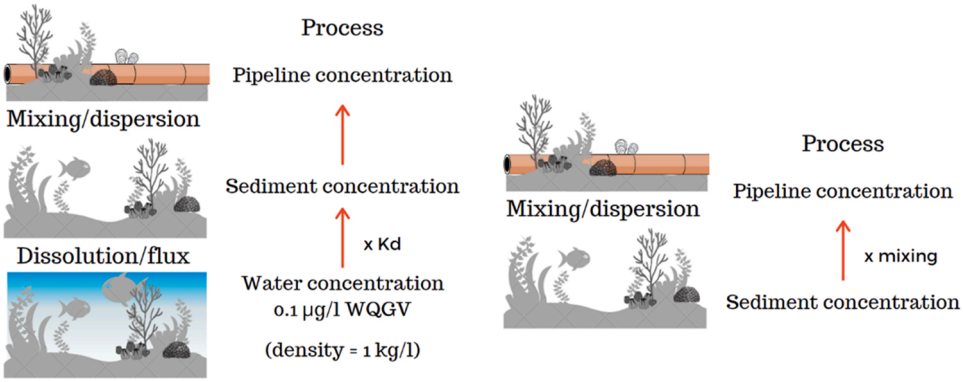

Fig. 2. Mass-based approach to determining the pipeline mercury threshold according to the WQGV (left) and SQGV (right).

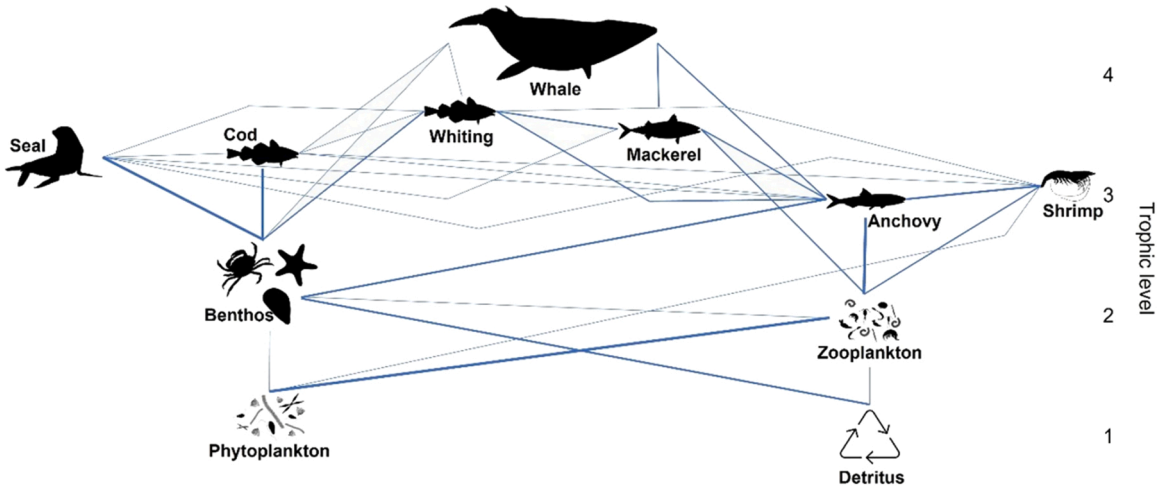

Fig. 3. 'Anchovy Bay' food web structure. The blue lines indicate food web interaction, the line thickness indicates the proportion of biomass transferred between the functional groups.

converted to the model environment. Here it was assumed that all mercury that was added directly entered the water compartment and was available for incorporation into the model food web (i.e., conservatively assuming its complete availability to be methylated and bioaccumulate). Although this is not environmentally representative, as only approximately 15 % of the total mercury in the water column is accounted for by methylmercury [56–59], and with most of the methylation occurs in marine sediments [60,61]. Direct uptake rates for lower trophic level organisms were calculated from real world examples (see Eqs. (5)–(7) for details), which was then validated against actual biota concentrations (see Eqs. (8) and (9)). One 1 km long representative pipeline with a diameter of 35.56 cm, and a wall thickness of 1.91 cm was used for all calculations of equivalent releases in the trials. The food web was defined as a closed system with an area of 1 km^2 and a depth of 100 m. All input values are listed in Table S4.

Trial 1 – Background exposure: This trial represents the current environmental conditions of existing oceanic mercury concentration, providing an insight into the potential future mercury bioaccumulation if the current rate of anthropogenic and natural mercury released into the environment is unchanged in the next 200 years. The initial environmental concentration was calculated from the global ocean mercury mass of 68 million metric tons, resulting in an average concentration of 0.19 t/km^2 provided by Neff et al., [62]. They further determined an annual global oceanic anthropogenic influx of up to 8800 t/year (0.000024 t/km^2/year). No release of pipeline-associated mercury is considered in this trial.

Trial 2 – Release of 11 mg mercury/m pipeline (mg/m) (0.005 t) over 1 year: In addition to the trial 1 assumptions, it is assumed that a 1 km pipeline segment in the modelled ecosystem is contaminated with 11 mg mercury/m pipeline (herein mg/m). When considering the K_d value of 4×10^3 kg/l [47],this results in a dissolved mercury concentration equivalent to the most stringent WQGV listed in Table S1 of 0.05 µg/l, herein referred to as 'low WQGV'. This amount of mercury is released into the modelled ecosystem in a single-release event after 50 years as an input of 0.005 t/km^2, before returning to the previous influx value.

Trial 3 – Release of 22 mg/m mercury (0.01 t/km^2) over 1 year: In addition to the trial 1 assumptions, it is assumed that a 1 km pipeline segment in the modelled ecosystem is contaminated with 22 mg/m mercury [63], which is equivalent to 0.01 t dissolved mercury in the food web model, or 0.1 µg/l considering a K_d value of 4×10^3 [47]. This is a WQGV for mercury for various nations (Table S1). This amount of mercury is released into the modelled ecosystem in a single-release event after 50 years as an input of 0.01 t/km^2/y, before returning to the previous influx value.

Trial 4 – Release of 86 mg/m mercury (0.04 t/km^2) over 1 year: In addition to the trial 1 assumptions, it is assumed that a pipeline segment in the modelled ecosystem is contaminated with 86 mg/m mercury, which is the equivalent of the least stringent WQGV listed in Table S1 of 0.4 µg/l, herein referred to as 'high WQGV'. This amount is released into the modelled ecosystem in a single-release event after 50 years as an input of 0.04 t/km^2/year, before returning to the previous influx value.

Trials 2–4 represent a concentration series as all underlying assumptions remain the same as outlined in trial 1 with the addition of varying mercury concentrations in the single release event. This was done to determine the rate at which increased mercury emissions led to increased biota mercury concentrations. Incremental release trials (where an equivalent amount of mercury to trials 2–4 was released over 100 years) were also run (data not shown) but the final accumulated mercury concentration did not vary. To this end, the percentage increase in muscle mercury concentration between trial 1 and trials 2–4 were examined, rather than the actual mercury accumulation. It must be noted that the food web used here is a simplified representation. Mercury biomagnification will be highly sensitive to local environments and ecosystems and so these results should not be interpreted as being predictive of impacts from real mercury releases. For the derivation of the equivalent pipeline mercury concentration of trials 2–4, the sediment water partitioning of mercury was taken into consideration, and a complete distribution of mercury in the respective overlaying water column was assumed. The additional mercury concentrations modelled to be released into the 'Anchovy Bay' environment are within the range of those measured in oil and gas pipelines of different diameters [7].

To derive the initial concentrations for the functional groups, the literature-derived value mercury concentration of 83 ng/g dry weight for phytoplankton by Bełdowska and Kobos [64] was used, converting it to 0.004 mg/kg wet weight by applying the dry-to-wet weight conversion factor of 20 [65], as well as the initial environmental concentration of 0.19 t/km^2 [62]. Then, the uptake rate was computed following the calculations discussed by Booth et al., [66], to match the modelled concentration to the literature derived organism mercury concentration. This approach allows one to tailor the model more accurately to environmental conditions and can be done with more robust data for regions/systems of interest, where present. Briefly, the primary producer direct uptake rate (u_i) is derived by:

$$\boldsymbol{u_i = CR_i \times (P_i/B_i + m_i + d_i)} \quad (5)$$

Based on the concentration ratio CR_i, the production over biomass

ratio $\frac{P}{B}$, the excretion rate m_i, and the physical decay rate d_i. The concentration ratio is computed with the contaminant concentration within the group A_i, the biomass of the group B, and the initial environmental concentration C_0:

$$\boldsymbol{CR_i = (A_i/B_i)/C_0} \tag{6}$$

To derive the initial concentration for all other functional groups in the 'Anchovy Bay' model, the model was allowed to equilibrate with only the above-mentioned parameters (see Table S4). The thus derived initial concentrations for the remaining groups were then used for all further trials and for the calculation of the direct uptake rate of the zooplankton and benthos functional groups. For consumer functional groups, the uptake rates are derived from the predation losses and consumption uptakes as follows:

$$\boldsymbol{u_i = \frac{\left[\left[C_{i,eq} \times \sum_{j=predator} Z_i + m_i + d_i\right] - \left[AE_i \times \sum_{j=prey} \frac{Q_{ij}A_j}{B_j}\right]\right]}{B_i \times C_0}} \tag{7}$$

Here the losses from predation include the concentration of the functional group of interest C_i and sum of the group's mortality, excretion, and physical decay rates. The gains due to consumption of other functional groups includes the assimilation efficiency of the functional group of interest AE_i and the sum of uptake as consumption rate of j prey Q_{ij} with a concentration A_j and biomass. In addition, the demethylation rate by whales and seal was set to 15 % and 25 %, respectively [67].

To determine the validity of the model-derived initial concentrations from the direct uptake rate of phytoplankton as outlined above, literature-based muscle concentrations were examined taking species not directly listed in the model but of the same trophic level into account. Additionally, data from publicly accessible databases was used, creating an average concentration for each species observed. The dietary matrix based trophic levels for all species were derived from FishBase and SeaLifeBase (Table S5) [68,69]. Only studies published in English were reviewed for this purpose, where the trophic level of the examined species could be determined, muscle samples were analysed, it was evident whether the concentration was given in dry or wet weight, and where clear distinctions were made between the data obtained in the study and data taken from literature for comparative purposes. Additionally, studies were excluded that examined mercury concentrations in food products, as no clear indication was given on the number of freeze-thawing cycles and overall sample preparation/handling. Where dry weight mercury concentrations were used, the values were divided by the wet-to-dry conversion factors published by Cinnirella et al. [70] and necessary unit conversion were conducted. In addition, the modelled (M) versus mean observed (O) mercury concentration ratio was calculated for each trophic level, following the example of Li et al. [71] where:

$$\boldsymbol{M/O\ ratio = M/O} \tag{8}$$

The ratio between the modelled (M) and observed (O) muscle tissue concentration for a given species allows one to assess how comparable the model derived initial mercury concentrations are to biota samples. The closer to one the M/O ratio value is, the more similar the modelled and the observed concentration are and thus the more accurately the model initial concentrations represent the concentration measured in biota samples. The M/O value was used to determine when the modelled value output is predictive of actual environments and where the model over- or under-estimated the biota concentration. Additionally, the normalised mean bias of the M/O ratio, as outlined by Li et al. [71] was determined:

$$\boldsymbol{Normalised\ mean\ bias = \sum_{1}^{n}(M-O) \Big/ \sum_{1}^{n} O} \tag{9}$$

As a hypothetical marine ecosystem has been used, and fish tissue concentrations were taken from multiple studies representing multiple ecosystems, the M/O ratio was used to validate that the EwE and Ecotracer model outputs were adequately parameterised to give representative outputs reflecting generic biota tissue concentrations (see Section 4.2 for further details).

2.2.2. Methylmercury EWI

To determine the actual weekly fish consumption for each nation in 2019, the FAO database was used [72]. All searches were based on the following selection criteria: Countries – "Select All"; Elements – "Food supply quantity (kg/capita/yr)"; and Years – "2019". The items were grouped into three (Items – "Demersal Fish", "Marine Fish, Other", and "Pelagic Fish" OR "Crustaceans", "Cephalopods", "Molluscs, Other" OR "Meat, Aquatic Mammals", "Aquatic Animals, Other") and will be referred to as "fish", "benthos" and "mammals" herein. It should be noted, however, that the database does not allow for the accurate separation between freshwater and marine species in all cases (e.g., for demersal and pelagic fish and crustaceans). This is not thought to affect the accuracy of the present findings as average consumption values are used for this study, and the intention was to demonstrate an assessment method that should be tailored to individual exposure scenarios. The respective consumption rate was then applied to the functional group it is associated with, assuming that no seafood from other sources was consumed. To account for the dietary diversity in terms of fish species consumed, anchovy and mackerel were defined as oily fish, with the remaining fish species defined as white fish in accordance with the food standards agency definition [73] when assessing the weekly consumption of methylmercury via dietary intake. To account for the higher reliance on marine resources of coastal indigenous communities, the fish consumption of such communities was derived from Cisneros-Montemayor et al. [74]. The publication, however, made no distinction between different types of seafood and thus the data was applied to all functional groups. However, the muscle tissue concentrations published in this work can be used to derive the weekly methylmercury intake for communities with a different dietary matrix.

Comparing the EwE output to e.g., the FS or the TWI, more accurately, literature derived data regarding the distribution of total mercury in the functional groups was used. Of the total body burden of mercury, the following percentages were noted to accumulate in muscle tissue for each functional group in the 'Anchovy Bay' food web: 2 % for whales [75], 5 % for seals [76], 50 % for all fish species [77], 15 % for shrimp, and 100 % for the benthic [78–80] and planktonic species [81,82], and detritus. This approach assumes that the mercury released into the marine environment disperses homogenously within the model, and that no migration occurs. The hypothetical model used here was selected for the simplicity of the food web, lending itself as a proof-of-concept model for the method presented. The results of trial 1 was then used as baseline, deriving the percentage increase in muscle tissue mercury concentration for trials 2–5.

The output obtained from the Ecotracer trials was further used to calculate the methylmercury EWI based on the weekly fish consumption data. The fraction of total mercury in muscle tissue that represented methylmercury was 95 % [41]. Additionally, the bioaccessibility, the fraction of ingested methylmercury that is released from the food matrix into a soluble form within the gastrointestinal tract, for the species (or related species) modelled in the 'Anchovy Bay' food web was obtained from Bradley et al. [83] (Table 1). The data here refers to raw samples, as gaps in the published data did not allow for the assessment of other preparation methods. Moreover, the data for cooked samples frequently presented with higher bioaccessibility rates of methylmercury than raw samples, thus using raw sample bioaccessibility data provided a degree of conservatism to the study.

Cysteine is the most abundant form of protein- or peptide-bound thiol in biological systems [84]. It is the major complexing agent in (fish) muscle tissue [85], including for methylmercury [86]. The EWI

Table 1
Bioaccessibility (%) of methylmercury from raw samples from the species of the 'Anchovy Bay' food web model to human stomach epithelial cells as published by Bradley et al. [83].

Species	Bioaccessibility
Whale	98[a]
Seal	98[a]
Cod	77
Whiting	100[b]
Mackerel	80
Anchovy	100[b]
Shrimp	100
Benthos	100[c]

a Values refer to tuna.
b data refers to meagre.
c data refers to scallops.

was calculated based on the average fish consumption of women of childbearing age and children, using the mercury accumulation in muscle tissue (AC; μg/kg), the fraction of total mercury that is methylmercury (F; %), the weekly intake (WI; kg), the absorption rate though stomach cells (AB; %, known to be 79 % for cysteine bound methylmercury [83]), and the persons weight (W; kg). In this case, the weight of the average person used (62 kg), was in accordance with Walpole et al. [87]:

$$\boldsymbol{Consumed_{MeHg} = (ACxFxWIxAB)/W} \quad (10)$$

To account for the difference in white and oily fish consumption, it was assumed that ≤ 280 g (two portions) per week of oily fish were consumed [88], with the rest of each nations fish consumption being made up of white fish. In the case of nations consuming too little fish a week to account for 2 portions of oily fish, only one portion (140 g) was assumed. Where this was not feasible, 50 % of the consumed weekly fish was assumed to be oily. For shellfish or marine mammal intake rates, the FAO database derived consumption rate was split equally between the respective functional groups from Ecotracer. As no distinction in consumption rates of different types of seafood were provided for the indigenous communities [74], an even split between all functional groups was assumed. In addition, the overall mean intake values were used to generalise the mean global methylmercury intake via seafood using the following consumption rates: Crustaceans 6 g, fish 23.5 g, marine mammals 1 g, and for the indigenous community a mean consumption rate of 1.42 kg was used. Here, too, the results derived from trial 1 were then used as baseline, determining the percentage increase in weekly ingested methylmercury for trials 2–5.

2.3. Limitations of the modelling approach

The outlined approach relies on simplifications of complex environmental processes, due to the lack of data and the use of a hypothetical food web. These simplifications were selected for the purpose of demonstrating the approach and providing perspective on the data requirements for local implementation. The assumptions, some of which may be unrealistic for local implementations, are highlighted below for transparency and to outline where future research focus is needed, in order to make the method presented suitable for site-specific use. Assumptions made in this work include:

1. Calculating seawater mercury concentrations from solid-phase mercury concentrations using an open-ocean K_d value.
2. Assuming there is no sediment mixing and dilution post-pipeline release, and the potential for dispersion of aqueous mercury.
3. Representing all aqueous mercury as bioavailable for uptake by marine biota.
4. Using a closed food web model that does include mercury removal mechanisms by ocean currents or allow for the migration of organisms in and out of the food web.
5. That all consumed seafood originated from the modelled food web.

These limitations are described to encourage proponents seeking to adapt this approach to their context to apply an appropriate complexity to ensure an appropriate level of confidence in the modelling outcome.

3. Results

Pipeline mercury threshold concentrations were computed for the highest and lowest published guideline values (Table S1) to give a range representing the different EQGVs adopted by different nations. The EQGVs applied to the calculations are 0.05 μg/l [37] and 0.4 μg/l [89] for the water compartment, and 0.13 mg/kg [89] and 0.15 mg/kg [36] for the sediment compartment, representing the lowest applied EQGVs for different nations. A sampling depth of 5–10 cm was used to parameterise the sediment box below the pipeline, and the previously discussed K_d value of 4000 l/kg used [47]. The WQGV of 0.4 μg/l was also used to inform the potential future mercury accumulation in marine food webs (as outlined in Section 2.2.1, trial 4).

3.1. A: Pipeline mercury threshold calculation

This calculation investigated the pipeline mercury threshold concentrations to remain within the EQGVs. Overall, the low SQGV (0.13 mg/kg) was the most sensitive threshold value when deriving residual pipeline mercury concentrations. The area-based pipeline mercury threshold concentrations derived here were not affected by the pipeline parameters and was purely determined by changes in the referenced EQGV. The pipeline threshold concentration before exceeding the low SQGV was 7 mg/m^2, compared to 108.2 mg/m^2 before exceeding the low WQGV (Table 2). These differences likely reflect the choice of K_d value.

For the derivation of the length-based threshold concentration, the thinner the wall thickness and the larger the pipe diameter, the higher the permissible mercury concentration (mg/m). The reverse relationship was observed for the mass-based pipeline mercury threshold concentration (mg/kg). This can be seen in Fig. 4. For the length- and mass-based calculations, the pipeline diameter and wall thickness influenced the pipeline mercury threshold concentration (Fig. 4). Here, the derived pipeline threshold concentration to remain below the low SQGVs were 2.00 mg/m (for a pipeline with 4 ″ diameter and 1.1 cm wall thickness) or 0.003 mg/kg (14–18 ″ diameter, 3.5–4.5 cm wall thickness), for the length- and mass-based calculation, respectively, considering a sediment sampling depth of 5 cm. The pipeline threshold concentrations derived for remaining below the low WQGV were 31.28 mg/m (4 ″ diameter, 1.1 cm wall thickness) and 0.39 mg/kg (18 ″ diameter, 4.5 cm) for the length- and mass-based output, respectively.

Table 2
Pipeline mercury threshold concentrations to remain below the EQGVs (Table S1; low WQGV: 0.05 μg/l, high WQGV: 0.4 μg/l, low SQGV: 0.13 mg/kg, high SQGV: 0.15 mg/kg) for different pipeline configurations (Table S2) and sampling depths.

		Low EQGV			High EQGV		
		Area-based [mg/m^2]	Length-based [mg/m]	Mass-based [mg/kg]	Area-based [mg/m^2]	Length-based [mg/m]	Mass-based [mg/kg]
WQGV	Min	108.20	31.28	0.39	865.8	250.00	3.13
	Max	108.20	151.13	2.45	865.8	1290.00	19.59
SQGV	Min	7.00	2.00	0.03	8.1	2.30	0.03
	Max	7.00	9.80	0.16	8.1	11.30	0.18

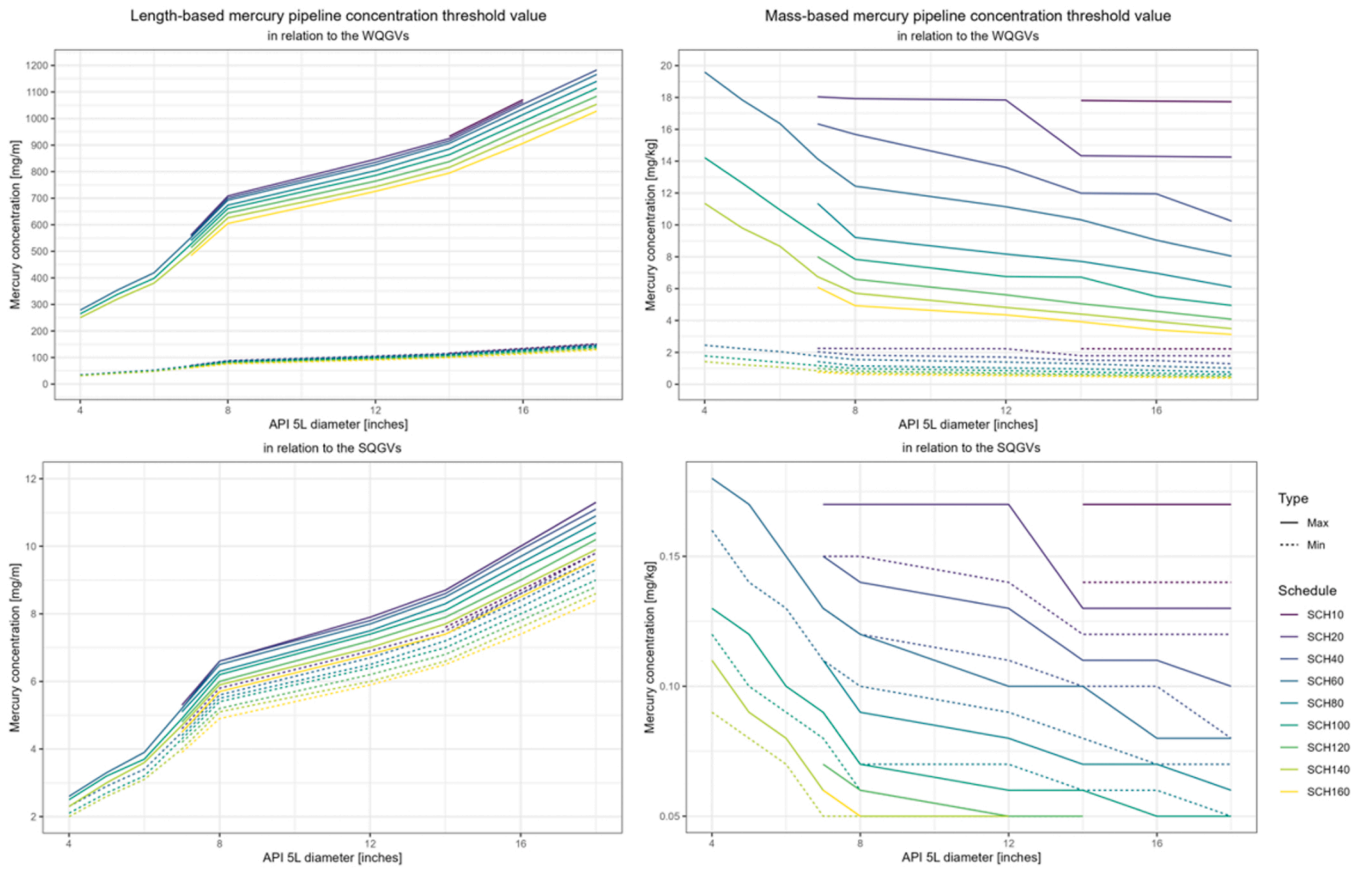

Fig. 4. Pipeline mercury threshold value calculation for WQGV (top) and SQGV (bottom). The length- (left) and mass-based (right) mercury pipeline threshold values for the low (continuous lines) and high (dashed lines) EQGVs for different pipeline schedules (colours).

3.2. B: food web modelling

3.2.1. Muscle tissue mercury accumulation

To determine the representativeness of initial mercury concentration used in the 'Anchovy Bay' model, publicly available mercury biota concentrations were obtained from literature, as well as databases for the North Sea [90,91] and US coasts [92]. 'Anchovy Bay' modelled mercury concentrations were within the distribution of measured mercury concentrations for species of comparable trophic levels (Fig. 5). The M/O ratio for whiting and shrimp were closest to one. All derived M/O values were between 0.13 and 3.0, with the model derived values for zooplankton, benthos, shrimp, mackerel, cod, and whale exceeding the literature derived ones (M/O ratio > 1) and the modelled concentrations for phytoplankton, anchovy, whiting, and seal remaining under those provided by literature (M/O ratio < 1). This was further supported by the determined normalised mean bias (Table S5).

The final mercury muscle tissue concentration in the different functional groups of the 'Anchovy Bay' ecosystem increased with increasing influx concentrations (i.e., from trial 1 to trials 2–4, Table 3). In trial 1, mackerel and both marine mammal species, exceed the 0.5 mg/kg value. Cod and whiting were found to exceed the 1 mg/kg FS in trial 1. Relative to background input (trial 1), the additional influx of mercury modelled in trials 2–4 is 0.5 %, 1%, and 4 %, respectively, eliciting increased biota concentrations of up to 33 %, compared to trial 1. Due to the different feeding behaviours of the functional groups, as well as the modelled detoxification process for whales and seals, the species with highest muscle tissue concentration in the scenarios are cod, whiting, and mackerel. Due to the short generation time and feeding habits of the lower trophic level organisms, the overall accumulation of mercury here does not increase significantly, even with increased annual mercury influx. For trials 2–4, the concentrations are shown as percentage increase from trial 1 (actual concentrations in Fig. S1).

3.2.2. Hypothetical weekly dietary intake of methylmercury

The average seafood consumption rates of non-indigenous communities were: 1 g/week aquatic mammals, 235 g/week fish, 59 g/week benthic and non-fish species. These values were used for all further calculations of methylmercury intake for non-indigenous communities. The average seafood consumption of coastal indigenous communities was 1.43 kg/week (Fig. 6).

The EWI in trial 1 was calculated for the consumption rates of seafood by non-indigenous communities, and only the consumption of white fish led to an exceedance of the 1.3 μg/kg TWI [42] (Table 4). The EWI from all seafood groups would also exceed the TWI. No increase in EWI was determined between trial 1 and 2. Between trials 1 and 3, an increase in EWI from white fish and total seafood consumption (of 6.7 % and 4.2 %, respectively) was determined. Between trials 1 and 4, an increase in ingested methylmercury of 16.7 % overall was calculated, and both oily and white fish led to an increased intake in this trial (14.3 % and 20 %, respectively). Considering the seafood consumption rates of coastal indigenous communities, only the overall seafood consumption data could be assessed, highlighting an exceedance of the TWI [42]. An increase in EWI for trials 2–4, compared to trial 1, was noted (of 2.5 %, 5.2 %, and 20.5 %, respectively).

4. Discussion

The work conducted here demonstrated an approach to assess the potential biomagnification impacts of mercury released into the marine environment and explores how protective existing EQGVs are of food web impacts. This work has focussed on remaining below the predefined

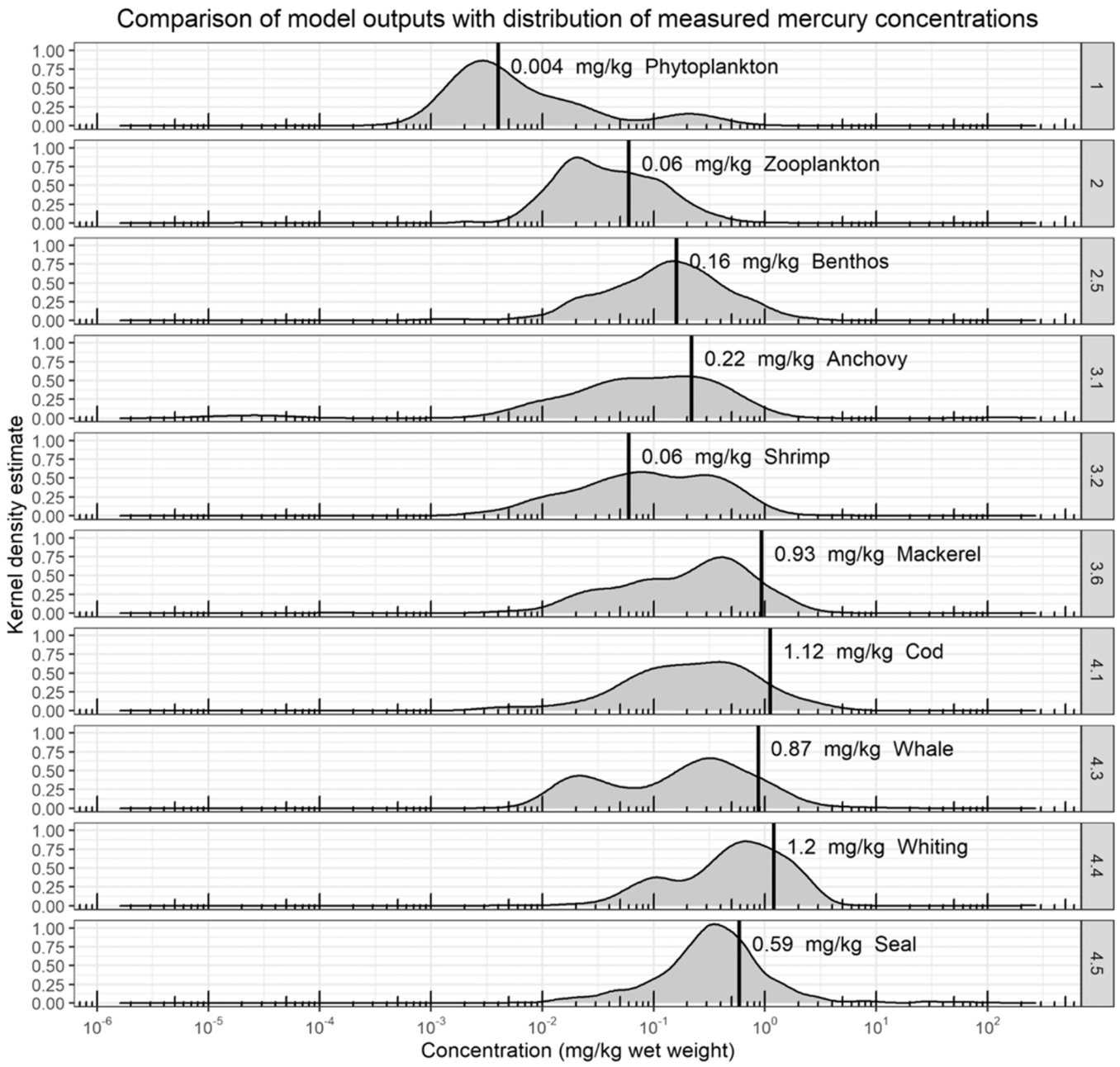

Fig. 5. Density distribution of mercury concentration in trophic levels of marine organisms sampled in the global oceans (data can be found in Table S6), compared to the model-derived initial concentration in 'Anchovy Bay' (reference lines and text). Trophic levels listed in the facet labels on the right.

EQGVs to outline the pipeline mercury threshold values that would require no additional testing or analysis in the context of environmental protection.

4.1. A: Pipeline mercury threshold calculation

This work has focussed on determining threshold values for pipeline mercury concentrations which, upon release from a corroding pipeline decommissioned in situ would not exceed existing guideline values. It should be noted that the exceedance of the EQGVs only indicates that further investigation is required to better quantify and understand the environmental risk [93]. Local parameterisations reflecting site-specific exposures should be adopted to calculate tailored threshold values.

The pipeline mercury threshold data provided output comparable to different measurement techniques currently used in the industry, such as X-Ray fluorescence, pigging dust analysis, or acid digests [4,94,95]. Previous publications measuring mercury in uncleaned pipelines have found concentrations of 10 g/kg in pigging dust, 10–100 g/kg concentration of scale on steel pipeline surfaces, or < 5 g/m pipeline [31,96,97]. Comparing these literature derived values to the threshold concentrations presented in Table 2, it may seem as though the published values far exceed the threshold values used for the calculations presented. However, this does not take into consideration that the literature-derived values were obtained from contamination product analysis (such as measuring the mercury concentration in pigging dust or cleaning solution), whilst computed values here are indicative of the maximum allowable pipeline-associated mercury concentration following cleaning.

A recent environmental plan submitted for the decommissioning of the Griffin gas export pipeline on the Northwest shelf of Australia calculated a pre-cleaning pipeline mercury concentration of 98 mg/kg. Their studies investigating the efficacy of cleaning options suggests that

Table 3
Total mercury concentration (mg/kg) in muscle tissue in all 'Anchovy Bay' function groups in trial 1, and the percentage increase in muscle tissue mercury concentration in trials 2–4 (cf Fig. S1).

	Trial 1 Background (mg/kg)	Trial 2 0.005 t/km^2/year (% increase)	Trial 3 0.01 t/km^2/year (% increase)	Trial 4 0.04 t/km^2/year (% increase)
Whale	0.89[a]	2.25	4.49	20.22
Seal	0.61[a]	1.64	4.92	19.67
Cod	1.15[b]	2.61	5.22	20.00
Whiting	1.23[b]	3.25	5.69	21.14
Mackerel	0.95[a]	3.16	5.26	21.05
Anchovy	0.22	4.55	9.09	22.73
Shrimp	0.06	0.00	16.67	16.67
Benthos	0.16	0.00	6.25	18.75
Zooplankton	0.06	0.00	16.67	33.33
Phytoplankton	0.00	0.00	0.00	25.00

a exceeds 0.5 mg/kg FS for low accumulation species.
b exceeds 1 mg/kg FS for high accumulation species.

post-cleaning the pipelines will have a concentration range of 0.26–0.010 mg/kg. Based on the approach outlined in Section 2.1, this would be unlikely to exceed the WQGVs or the higher SQGV [50]. These findings highlight the importance of mitigation actions such as pipeline cleaning prior to in situ abandonment, where such actions are planned.

4.1.1. Requirements for the determination of site-specific threshold values

The approach outlined here highlights the gaps and limitations that need to be addressed to improve the environmental relevance of the derived threshold values. Key gaps include the consideration of existing levels of contamination, the potential for dispersion and mixing of released mercury, the extent to which mercury in bioavailable or methylated, and the partitioning of mercury between the solid and aqueous phase.

This approach assumes an uncontaminated environment for the calculation of the pipeline threshold value. This may be more applicable to regions of low environmental contamination levels such as the Baltic Sea (0.6 ng/l water mercury concentration [98]) or the North Sea (0.5 ng/l water mercury concentration [99]). Regions with elevated background concentrations may require an adjusted approach to ensure relevant environmental concentrations are below levels at which it may cause harm.

Mixing and aqueous dispersion should be included in site-specific applications of this approach. Quantifying sediment transport and settling rates provide an estimate of solid-phase mixing, whilst the rate of mercury dissolution should be based on laboratory experiments and considered against ocean current-based dispersion.

Speciation is known to affect toxicity, and research on mercury speciaition in marine waters suggests that only up to ~ 22 % of total mercury is present as methylmercury [100], with the remainder being inorganic mercury adsorbed to organic or particulate matter [60]. The fractions could be estimated through laboratory experiments or in situ measurements in scenarios reflecting local conditions. It should be considered, however, that mercury speciation may change over longer time frames, so environmental transformations should be considered to better understand exposures.

Describing the partitioning of mercury between solid and aqueous phases is an important consideration to ensure the protection of marine ecosystems. This was achieved in the present study by applying a K_d value. Future research should consider the use of regional and mercury-species specific values where available. Other marine mercury K_d values published in the literature range from 10^4 to 10^6 l/kg [17,48,49,101], or higher for species such as metacinnabar. Alternative approaches, such as describing the flux and dispersion of mercury from sediment the overlaying waters offer a more sophisticated approach but may have higher data and experimental requirements [102].

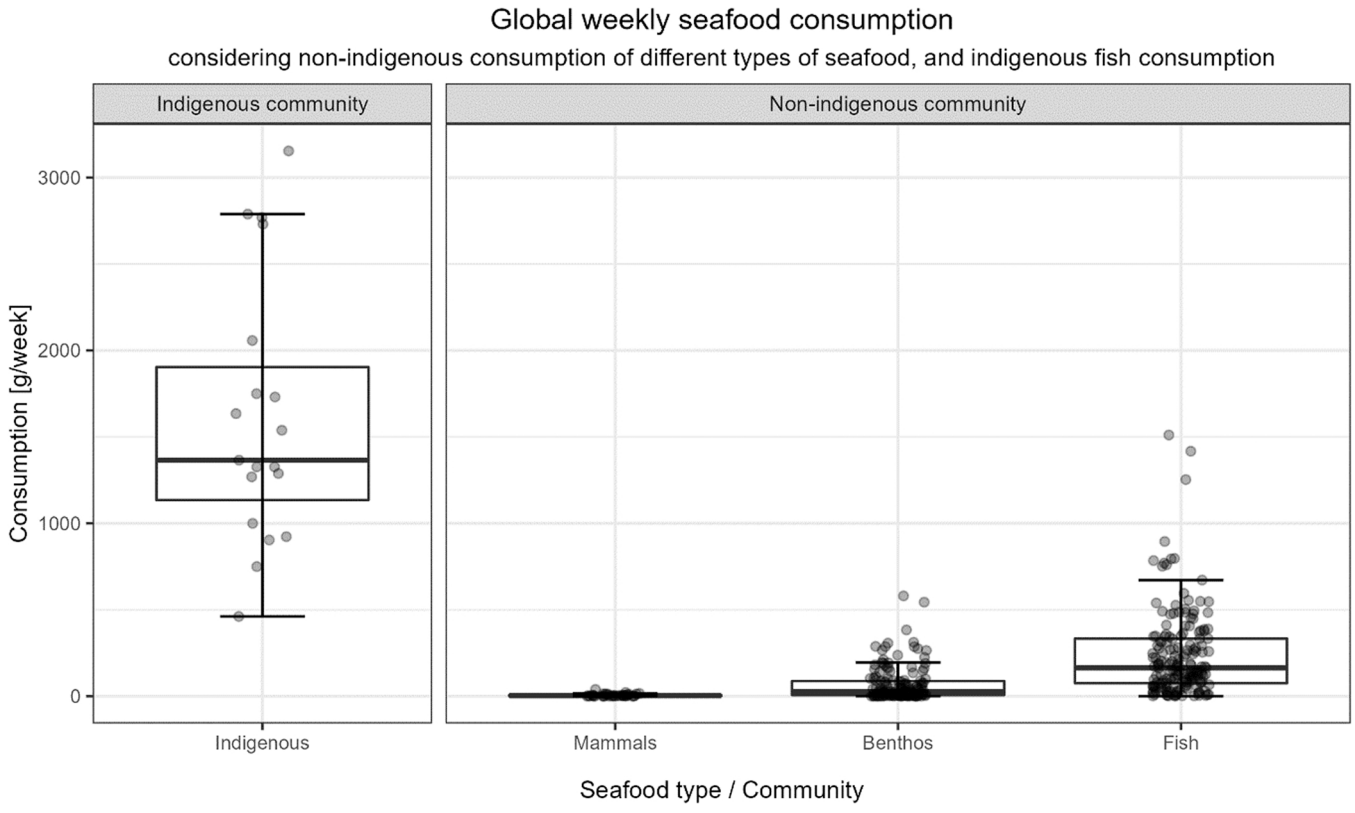

Fig. 6. Weekly consumption of seafood (g/week) for different nations [72], each dot representing the data for a nation or the overall fish consumption of coastal indigenous communities [74].

Table 4
Weekly methylmercury intake based on the trial 1model output (μg/kg), and the average seafood consumption for non-indigenous [72] and indigenous communities [74]. For trials 2–4 the percentage increase in weekly methylmercury intake from trial 1 is given.

	Trial 1 Background (μg/kg)	Trial 2 0.005 t/km^2/year (% increase)	Trial 3 0.01 t/km^2/year (% increase)	Trial 4 0.04 t/km^2/year (% increase)
Crustacean	0.01	0.00	0.00	0.00
Oily fish	0.07	0.00	0.00	14.29
White fish	0.15[a]	0.00	6.67	20.00
Marine mammals	0.01	0.00	0.00	0.00
Total non-indigenous consumption	0.24[a]	0.00	4.17	16.67
Total indigenous consumption	10.29[a]	2.53	5.15	20.51

a exceeds TWI of 1.3 μg/kg [42].

4.2. B: food web modelling

The background mercury concentration presented here for some of the species in the 'Anchovy Bay' were found to already exceed the FS. Although this may not be representative of all marine food webs currently, this is supported by e.g., the United Nations most recent report on global mercury assessment, where the mean mercury concentration in almost all marine species examined in the published case studies exceeded the 0.5 mg/kg FS. This held true for higher trophic level fish, such as swordfish, when examined in different oceanic basins. Additionally, all shark species examined in the report exceeded the 0.5 mg/kg FS, and 60% also exceeded the higher (1.0 mg/kg) FS [100]. This was supported by literature derived data from different environments (see Table S6). Moreover, the ICES data centre for North Sea species found that species such as cod (*Gadus morhua*), common dab (*Limanda limanda*), common mussel (*Mytilus edulis*) and manila clam (*Lajonkairia lajonkairii*) presented with mercury concentrations that exceeded at least one of the FS [103], with similar observations being made by others. [23,91,104]. In the modelled approach, comparison of modelled initial concentrations and literature derived data showed that whilst some degree over- or under-estimation occurred within the modelled species (Table S6), the concentrations still fell within the range observed in biota samples (Fig. 5). Thus, the focus was placed on relative changes in mercury biota concentrations with increasing contaminant influx in the model environment rather than the actual concentration itself.

The 'Anchovy Bay' model showed that with an increased mercury influx of 0.5 %, 1 %, and 4 % into the model environment between trials 2–4, the functional groups accumulated, an average of 2.6 %, 5.1 %, and 20.5 % additional mercury in muscle tissue. This example highlights that certain species with a population average below the FS, may exceed them in the future, with mercury releases equivalent to the EQGVs.

Considering the EWI calculated in this example, the percentage increase with increasing mercury influx into the system was more variable than that observed for the muscle concentration (Table 4). Between trials 2 and 4, the EWI increased by 0 %, 4.2 %, and 16.7 % for the non-indigenous community, but followed the muscle tissue trajectory more closely for coastal indigenous communities around the world (2.5 %, 5.2 %, and 20.5 %, respectively). The increased EWI by the non-indigenous community was influenced only by white fish (in trials 3 and 4) and oily fish (in trial 4), whilst the consumption of crustaceans and marine mammals did not affect the overall EWI. Studies assessing the EWI of different populations has found that parts of the Amazonas [105], Finnish [106], Italian [107], Spanish [108], and Taiwanese [109] populations may already be exceeding the TWI though the consumption of seafood. This is unsurprising given the wide distribution of organism mercury tissue concentrations (Fig. 5). There is a linear relationship between the mercury added to the Anchovy Bay food web and the percent increase in the fish functional groups but not the shrimp or benthos. This increased to a maximum of 20–30 % increased methylmercury intake compared to background levels.

It should be noted that in the present approach, the maximum potential uptake of (methyl-)mercury by marine organisms from the environment and humans through seafood consumption were calculated, thus presenting a conservative, i.e., 'worst case scenario'. This does not necessarily imply that a specific decommissioned offshore asset poses a risk currently, but that evidence suggests the potential for it to do so in the future. Such findings further underline the need to address the potential implications of environmental mercury release for both current and future communities. While the environmental guideline values may be protective of immediate adverse effects, already increasing background concentrations of mercury highlight the need to further determine the long-term accumulation and biomagnification potential of mercury in different marine food webs.

4.2.1. Requirements for site-specific food web modelling to assess mercury biomagnification and dietary exposure of humans

The approach described in this study uses a generic food web model that does not represent any specific local ecosystem. Applications of this method should thus incorporate site-specific data to improve the representativeness of the food web and mercury input components [54]. This includes updating the food web structure and improving the mercury fluxes to and from the ecosystem (e.g., considerations discussed in Section 4.1.1, along with mercury bioavailability, sediment sequestration, and biota excretion rates) [110,111]. Food web structure will have an impact on the biomagnification of contaminants, but it is not clear how different food webs will affect the transfer of mercury among organisms. Moreover, sub-acute chronic exposure to mercury is likely to lead to adverse health effect on the organisms before reaching lethal concentrations. In fish, such effects may be developmental alterations during embryogenesis, as well as impeding the larval predator avoidance and prey-capture abilities [112–116]. A summary of sub-acute effects observed in laboratory experiments with aquatic species can be found in Table S7. However, only few publications examine adverse effects in wild-caught species, thus highlighting the need for a better understanding of how laboratory-derived results translate into environmentally applicable insights. When assessing mercury in marine mammals, most publications focus on measuring concentrations and species in various organs and tissues [117]. Although marine mammals accumulate the majority of methylmercury in the liver [118], it will also accumulate in muscle tissue and the brain in the long run [119]. Interestingly, these organisms can detoxify methylmercury by binding it to selenium, forming the insoluble and non-toxic mercuric selenide crystals [120]. However, this detoxification pathway can induce selenium deficiency, which in turn has deleterious health effects if it occurs over a longer period [121]. Such interactions are not considered in many models, including Ecotracer, further affecting the representative food web biomass.

The toxicity of the accumulated mercury on the marine organisms should be considered [54,122], and spatial considerations should be incorporated, to reflect the effects of e.g., foraging ranges or dispersion and dilution from the contaminant source, to parameterise the model for a specific site. This may include more refined calculations describing mercury partitioning based on its transformation in local sediment, sediment binding or sequestration, and demethylation. Such processes may affect mercury biomagnification beyond the possible scope of the generic example applied in this study. Some of these processes may be addressed through integrating the spatial scale into Ecotracer by using Ecospace in EwE, but other parameters must be addressed outside of the model.

When calculating dietary methylmercury exposure, the potential

diversity of seafood sources and their preparation should also be considered. Seafood is a globally traded commodity and thus species caught from a contaminated food web may not affect the local community, or locally consumed fish may originate from a polluted food web, even if the local ecosystem is pristine. Moreover, the work of Bradley et al. [123] demonstrates that mercury bioaccessibility in seafood and its absorption by humans varies between species and method of preparation. Thus, factors such a local preparation methods and species preferences should be considered for local release scenarios. An attempt of this has been made for the present example, but gaps remain in the data for many commercially relevant seafood species.

Validating food web modelling is important, given the approach in this study aims to predict future release scenarios. These include deriving the concentration ratio between internal and environmental contaminant concentration or used the median concentration value where no concentration ratio could be derived. These values were then assessed for distribution normality, and Ecotracer was used to predict the observed versus predicted contaminant root mean square deviation to understand whether Ecotracer successfully reproduced real world concentration scenarios [66]. Others fit the model through restricted maximum likelihood, obtaining coefficient estimates and marginal (representing the variance explained by fixed factors) and conditional (representing the variance explained by fixed and random effects) R^2 values [124].

5. Summary

With many offshore petroleum production assets nearing the end of their operational life, decommissioning activities will only increase. Discussions about the way to decommission these assets need to consider the impacts and risks to the environment, human safety, and cost. Environmental guidelines are typically based on ecotoxicological data from laboratory and field-based studies. However, impacts such as biomagnification and dietary exposure to humans also need to be considered.

This desktop study aimed to demonstrate an approach to assess the potential biomagnification impacts from mercury releases equivalent to the EQGVs. In this demonstration, increased mercury input equivalent to pipelines contaminated with mercury below levels that would exceed EQGVs were found to increase marine organism tissue concentrations by 0–33 % (Table 3) and corresponding dietary methylmercury intake by 0–20 % (Table 4). This implies that further research is needed to be able to better characterise biomagnification risks from residual mercury in offshore oil and gas infrastructure decommissioned in situ. Environmental managers and regulators should consider the range of recommendations provided to tailor the approach outlined in this study to local environmental and release scenarios. Considering the bioaccumulation insights gained from this study, the applied software was found to be an applicable and easy-to-use tool in marine contaminant tracing and should be considered for site-specific assessments as a complementary line of evidence to existing EQGVs.

Environmental implications

Past research has shown that mercury can accumulate in offshore hydrocarbon pipelines, deeming such materials ‘hazardous’. The focus of environmental risk assessments, however, is predominantly on direct environmental impact and potential health implications for workers. This research paper has highlighted the importance of also focussing on potential future implications for food webs and the resulting impact this might have on seafood consumers.

CRediT authorship contribution statement

Rebecca von Hellfeld: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Software, Validation, Visualization, Writing – original draft, Writing – review & editing. **Christoph Gade:** Methodology, Writing – original draft, Writing – review & editing. **Darren J. Koppel:** Conceptualization, Investigation, Validation, Visualization, Writing – review & editing. **William J. Walters:** Formal analysis, Methodology, Software. **Fenny Kho:** Conceptualization, Writing – review & editing. **Astley Hastings:** Conceptualization, Funding acquisition, Investigation, Methodology, Project administration, Resources, Supervision, Writing – review & editing.

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Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Some authors have undertaken research funded by the oil and gas industry. These funding sources have not influenced the approach or conclusions of this study. The authors have no further potential conflicts of interest to disclose.

Data Availability

Data will be made available on request.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.jhazmat.2023.131298.

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