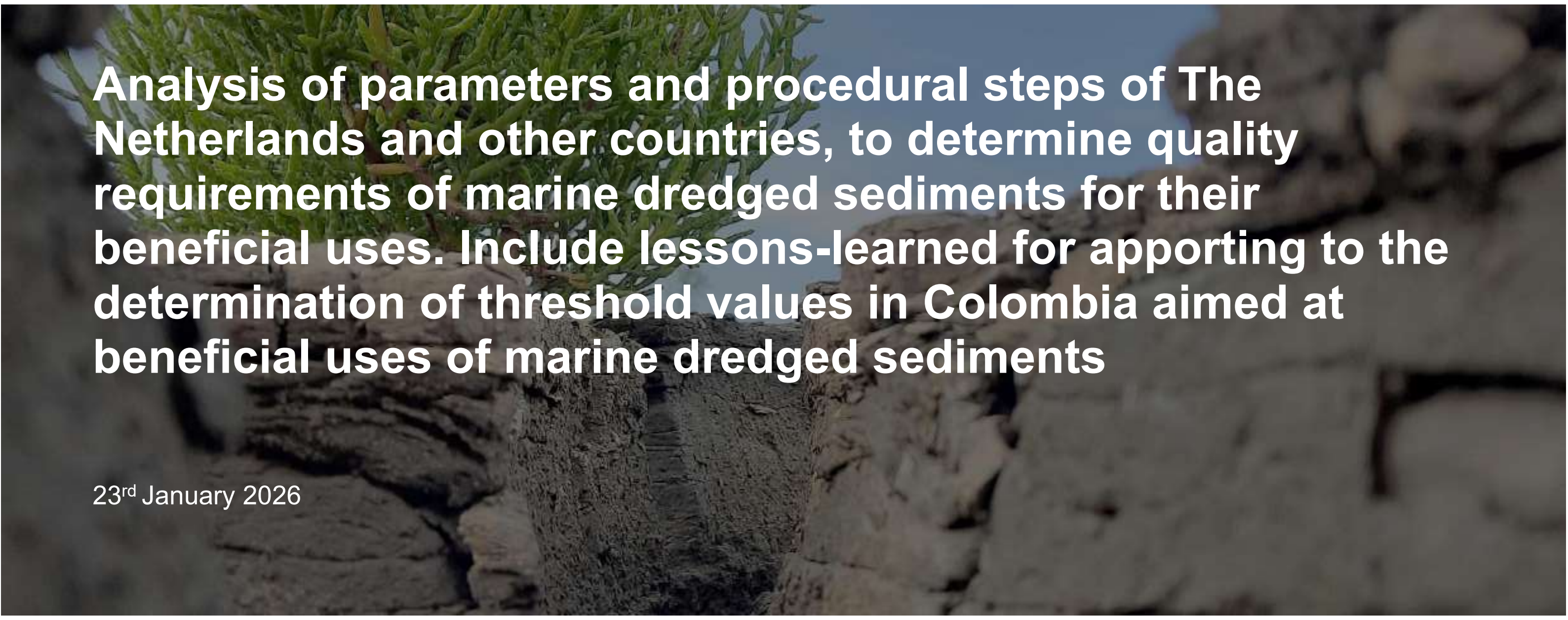
# Policy and technical advice on the beneficial uses of marine dredged sediments in Colombia, including nature-based solutions



## Analysis of parameters and procedural steps of The Netherlands and other countries, to determine quality requirements of marine dredged sediments for their beneficial uses. Include lessons-learned for apporting to the determination of threshold values in Colombia aimed at beneficial uses of marine dredged sediments

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The project “Policy and Technical Advice on the Beneficial Uses of Marine Dredged Sediments in Colombia, including Nature-Based Solutions” is part of the collaboration between the Government of the Netherlands, through the Partners for Water program, and the Colombian Ministry of Environment, the National Planning Department (DNP) and the Ministry of Transport. The project was carried out by a consortium consisting of Arcadis, Fundación Herencia Ambiental Caribe, JESyCA, and Netics, together with government entities from both Colombia and the Netherlands.

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# 1 Introduction

## 1.1 Background

Within the partnership between the Government of Colombia and the Government of the Netherlands in the field of water and climate change adaptation, beneficial use of (marine) dredged materials is one of the priorities, on the specific request of the Colombian Ministries of Transport (MinTransporte), Environment (MinAmbiente) and the National Planning Department (DNP). Beneficial uses of marine dredged sediments are not yet performed in Colombia, and dredged sediments have been dumped in offshore deposit areas previously approved by the environmental authority. Building on increased attention to beneficial use and to further improve the normative guidelines for dredged materials in Colombia, this assignment focuses on policy and technical advice for the beneficial uses of marine dredged sediments, including nature-based solutions (NbS).

The policy partnership with the Government of Colombia on the topic of dredging and beneficial use of dredged materials has a long history, supported through several projects in the last decade:

- The Plan Nacional de Dragados Marítimos (PNDM, 2017) consisted of a conceptual analysis and main recommendations to achieve, in the short, medium and long term, improvements to: (i) the institutional order, (ii) the technical and environmental regulations, (iii) the financing, (iv) the dredging contracting methodologies in the maritime access channels to the ports and (v) the maintenance dredging strategy by port area and the capital dredging strategy for the two coasts, Atlantic and Pacific, including beneficial use of dredged material. The PNDM also included an international comparison regarding the above aspects, in order to have a reference that would allow the Colombian Government entities to make qualified decisions, among which the use of materials from dredging occupies a prominent place. Specifically mentioned were the lack of uniformity in the basic criteria for the formulation of designs and works, a lack of clarity regarding the final disposal or beneficial use of dredged material, additional costs for unnecessary transport to dispose of materials offshore that could be reused, and imprecision regarding the final values of the projects.
- A webinar series on dredging and reuse of dredged materials in Colombia, meant to support the stakeholders on the topic of dredging, including key principles such as Building with Nature and reuse of dredged materials for other purposes (2020/2021)
- In the project Re-use of dredged materials in the Colombian context, opportunities for enlarging the range of beneficial uses of dredged materials have been analyzed, including examples of legal frameworks and normative requirements in other countries, a case study for opportunities for beneficial uses of dredged materials resulting from capital and maintenance dredging activities in the Buenaventura port zone and an analysis of barriers and enablers for the application of re-use of dredged materials in Colombia (2022).

Following these projects, the National Development Plan 2022-26 indicated in Article 240 the need to use the dredged material, complying with the environmental regulations issued for this purpose, prioritizing uses in the recovery of areas affected by coastal erosion, and in the recovery of mangrove areas or zones affected by flooding. Besides, in July 2023 the Colombian government issued the CONPES 4118 (National Port Policy), which states that the disposal of dredged materials offshore or onshore may have a negative impact on marine and coastal ecosystems. Given these statements in these normative instruments, it is the responsibility of MinAmbiente to establish a regulatory framework of environmental and technical guidelines for the use of dredged marine sediments in Colombia.

Given this need, the governments of the Netherlands and Colombia agreed to launch the current project “Policy and technical advice on beneficial uses of dredged marine sediments in Colombia, including nature-based solutions”.

## 1.2 Project objectives and deliverables

The project focuses on further improving the normative guidelines for marine dredged materials in Colombia, proposing a set of rules and parameters in the principal guidance document with respect to dredging (Guía de manejo ambiental de proyectos de infraestructura, modos marítimos y fluvial, INVIAS 2022) and including a case study for the port zone of Barranquilla showing what can be done with the dredged material.

The basis for improving the Colombian guidelines lies is providing technical advice in the form of practical rules and parameters for beneficial use of dredged material, drawn from the experience of the Netherlands and other countries. Accompanying this technical advice is the capacity building of major stakeholders in Colombia regarding the major technical components of the study.

The project consists of 5 main deliverables:

1. Rules and parameters applied in **the Netherlands** regarding dredging and reuse of dredged sediments.

2. Rules and parameters applied in **other countries: Australia/New Zealand, Japan, Brazil, Peru, Mexico, USA (Florida), Canada, Spain, Costa Rica and Panama.**
3. Characteristics of sediments in the port zone of Barranquilla.
4. Additions to the INVIAS guidelines.
5. Case study port zone Barranquilla.

## 1.3 Structure of this deliverable

This report is Deliverable 2, highlighting the physicochemical and biological parameters for the assessment of sediment quality in **other countries,** and to compare them to the parameters of The Netherlands. These parameters are based on international approaches that have been used as reference to elaborate analytical frameworks aimed at determining substance threshold values in sediments. Each country has chosen one or more of these frameworks to determine their Sediment Quality Guidelines (SQG), which allow them to identify whether dredged sediments comply with threshold values for dumping them in sea or land. Based on this, it is to discuss whether these threshold values are also acceptable for the beneficial uses of the sediments. Details on toxicity risks for organisms, including human health, are key to understanding the justification of the threshold values.

The structure of this report is as follows:

- Chapter 2: Description of approaches and frameworks to develop Sediment Quality Guidelines (SQGs). This includes a brief description of their methodology, the advantages and limitations and the current applications where the approach is used. This forms the basis for the SQGs that are described in chapter 3.
- Chapter 3: Description of the SQGs used in the studied countries. The main SQG types are described (Purpose-driven, step-by-step, international based regulations, custom approaches) and their main advantages and limitations addressed. For each focus country the governance structure, the framework and procedures and the approach used to define the SQGs in that country are explained.
- Chapter 4: The threshold values included in the SQGs are described for several countries that show similar environmental conditions to those of Colombia. The threshold values are compared and differences explained. Lastly, detection limits (LOD) are described.
- Chapter 5: Discussion of international experiences and key insights from the comparison of SQGs. The chapter addresses the diversity of regulatory approaches, the role of assessment systems, and the current challenges and opportunities for incorporating beneficial reuse of dredged sediments into existing regulations.

Appendices:

- Appendix A – Detection limits
- Appendix B – Overview table of institutional framework per country
- Appendix C – Glossary of technical terms
- Appendix D – Glossary of Acronyms

# 2 Research approaches and analytical frameworks to develop sediment quality guidelines

## Chapter 2 – Executive summary

A variety of approaches have been developed to formulate sediment quality guidelines (SQGs). These approaches have been extensively reviewed and summarized by CCE, 1995 and Long et al. 1992 among others.

The primary approaches used in developing guidelines include:
Sediment Background (SB)
1. Spiked-Sediment Bioassay (SSB)
2. Equilibrium Partitioning (EqP)
3. Tissue Residue (TR)
4. Screening Level Concentration (SLC)
5. Sediment Quality Triad (SQT)
6. Apparent Effects Threshold (AET)
7. Weight of Evidence (WOE)

Each approach is discussed in four key areas: a brief description of its methodology, its major advantages, limitations, and its current applications. All of these approaches support the development of numerical SQGs. However, additional procedures focus on site-specific sediment quality assessments, such as the International Joint Commission (IJC) sediment assessment strategy and benthic community structure assessments. These site-specific approaches are not considered here. The key takeaway of this chapter are the different approaches that exist to formulate sediment quality guidelines.

## 2.1 Approach types

### 2.1.1 Sediment Background (SB) Approach

This method evaluates sediment contamination by comparing chemical concentrations in the area of interest to those found in sediments from locations that reflect natural, undisturbed conditions. In some cases, historical data or sediment core profiles can help define baseline concentrations. A site is generally considered contaminated if one or more substances are present at levels significantly higher than background levels, often defined as exceeding the average by two or more standard deviations. Implementing this method requires careful selection of sampling sites, attention to sample processing and analysis techniques, and strict adherence to quality control procedures.

The key strength of this method is its straightforwardness. It involves chemical analyses that are commonly performed in standard laboratories and provides a direct comparison of results with background-based criteria. It is tailored to local environmental conditions, demands minimal data, and does not depend on toxicity testing.

However, the method has notable drawbacks. It does not incorporate direct biological or bioavailability data into the guideline development process. For human-made organic compounds, background concentrations are theoretically zero, but measurable amounts are frequently detected due to long-range atmospheric transport. Natural hydrocarbon and inorganic inputs from geologic sources can also complicate assessments. Additionally, background sites must be selected carefully to ensure similarity in clay mineralogy and content, as well as organic matter content, as these factors strongly influence contaminant concentrations in sediments. While this method may allow for setting guidelines that reflect current background levels, it remains uncertain whether those levels are adequately protective of aquatic life.

### 2.1.2 Spiked-Sediment Bioassay (SSB) Approach

This method involves adding known quantities of contaminants to clean sediments under controlled conditions to observe biological responses in test species. These responses may include mortality, impaired growth or reproduction, or physiological effects. The goal is to establish a clear link between chemical exposure and biological outcomes. This technique has been applied successfully with various sediments and individual contaminants or simple mixtures. Often, numerical guidelines are generated by applying a safety margin to the lowest concentration at which effects are observed in sensitive species, although other approaches may also be used.

This technique is valuable because it can be applied to a wide range of chemicals and sediment types, and it allows for the generation of accurate dose-response data. It also considers factors that influence chemical bioavailability, such as the amount of organic carbon or the presence of acid-volatile sulfides in the sediment. As a result, guidelines based on this method are generally well-supported scientifically.

The main limitation of this approach is that it has only been applied to a small number of species and contaminants, such as certain metals, pesticides, and polycyclic aromatic hydrocarbons (PAHs). Expanding the database to include a broader range of substances would require significant investment, which may not be feasible for many regulatory agencies. Additional challenges include uncertainties related to the spiking process, how long it takes for chemicals to bind with the sediment, and how these factors affect bioavailability, each of which can influence the interpretation of results. Lastly, the approach involves serious societal constraints, raising sediment levels in toxic experiments is not friendly to animals and cannot be performed on site.

### 2.1.3 Equilibrium Partitioning (EP) Approach

This method has been widely used, particularly in the U.S and Netherlands, to develop sediment guidelines for non-polar hydrophobic organic compounds. It assumes that the distribution of contaminants between sediment particles and the water in the sediment pores can be predicted based on chemical properties. The theory also assumes a continuous state of equilibrium between the sediment and pore water. Laboratory toxicity studies support this assumption, showing that biological impacts often correlate with the concentrations of these contaminants in pore water.

The method uses established water quality criteria developed for protecting marine life as the foundation for sediment guideline development. These criteria, originally designed for organisms living in the water column, are extended to species living in or near the sediment. Guidelines are calculated by combining the water quality criteria with partitioning values that describe how a chemical divides between water and sediment. For organic compounds without a charge, the calculation is:

**SQG = Kp × FCV**

Where:

- SQG is the sediment guideline (typically in micrograms per kilogram),
- Kp is the sediment/water partition coefficient (liters per kilogram), and
- FCV is the final chronic value for water quality (micrograms per liter).

Currently, this method is primarily used for substances like PAHs, polychlorinated benzenes, PCBs, dioxins, and furans. U.S. Environmental Protection Authority (EPA) has developed guidelines for several such compounds, including fluoranthene and dieldrin. Adjustments based on the sediment's organic carbon content and clay content can help predict toxicity for many of these chemicals. The influence of sulfides on metal bioavailability and the role of dissolved organic carbon in pore water are also being explored as factors in refining these guidelines. This method has the advantage of being broadly applicable, particularly because it considers site-specific factors that affect contaminant bioavailability. It is also practical, as it relies on existing data like water quality standards and partition coefficients. The approach is scientifically robust and has already been used in various regulatory contexts.

Still, there are several drawbacks. It does not account for how different chemicals might interact (e.g., combined toxicity effects). Development of metal-specific guidelines is still in progress. The method is currently valid only for sediments with a certain minimum organic carbon content, and even then, the toxicity relationship has only been confirmed for low-carbon sediments. Another issue is the variability and uncertainty in partition coefficients, which can lead to wide confidence intervals in calculated guidelines. Finally, the assumption that sediments are always in equilibrium may not hold true under real-world conditions, limiting the method's reliability in some cases.

### 2.1.4 Tissue Residue (TR) Approach

Tissue residue is the concentration of a chemical or chemical group in an organism's tissue or a portion of an organism's tissue. The Tissue Residue Approach, also known as the biota-water-sediment equilibrium method, estimates acceptable sediment concentrations for individual chemicals or chemical groups, based on their potential to produce tissue residues in aquatic organisms. The method involves establishing a link between the levels of contaminants found in sediments and those accumulated in aquatic biota. Additionally, it requires understanding how contaminant residues in aquatic organisms may impact predators, including humans and wildlife, that consume them. Several techniques are available for setting safe thresholds for contaminant concentrations in edible tissues of aquatic species.

A key benefit of this method is its straightforward application. Sediment quality targets can be calculated directly from tissue-based thresholds designed to protect human health or wildlife, assuming reliable sediment-to-organism bioaccumulation factors (BAFs) are available. Importantly, this method also explicitly considers the potential for long-term build-up of persistent pollutants.

However, aside from challenges shared with the Equilibrium Partitioning Approach, this method is limited by a lack of established tissue residue benchmarks for wildlife protection, as well as the absence of detailed dose-response data for many contaminants. As a result, the derived sediment guidelines often prioritize human health protection, potentially overlooking risks to sensitive wildlife species such as marine mammals that consume large amounts of aquatic prey. Nevertheless, advancements have introduced protocols to develop tissue-based guidelines for wildlife protection, including specific guidelines for compounds like dioxins and furans.

This approach has been used in the development of water quality criteria for substances like DDT, mercury, and PCBs, particularly to safeguard human health.

### 2.1.5 Screening Level (SL) Approach

The Screening Level Concentration Approach is a biologically based method for creating sediment guidelines aimed at protecting benthic fauna. It uses paired chemical and biological data collected from field studies to estimate the highest contaminant levels that a predefined proportion of benthic species can tolerate without experiencing adverse effects.

This method calculates the SL by first determining a species-specific level concentration (SSLC) for each organism with sufficient data. This is done by analyzing the distribution of contaminant levels at all sites where a species is found, typically requiring data from at least ten locations, and defining the 90th percentile of that distribution as the SSLC. These SSLCs are then used to generate a cumulative distribution, and the SL is set at the 5th percentile, representing a concentration that 95% of species can tolerate.

Advantages of this method include its broad applicability and reliance on commonly available data. It can be used to develop guidelines for any contaminant that can be measured analytically, and it is based on biological responses of organism's native to the environment in question, allowing for the development of regionally relevant guidelines.

Despite its strengths, the SLCA is built on several assumptions that can limit its usefulness. It assumes species distributions are mainly shaped by sediment contaminant levels, though other variables, such as habitat features, unmeasured pollutants, or biological interactions, may also influence species presence. The approach does not consider detailed dose-response data or sublethal effects, focusing instead on species presence or absence. Moreover, it requires a comprehensive database, ideally containing contaminant data from 20 or more sites and occurrence data for at least 20 species, which may be unavailable for many substances.

Another limitation is the lack of a direct causal link between any single contaminant and observed biological effects. In real-world settings, organisms are exposed to complex mixtures of chemicals, making it difficult to attribute biological responses to specific substances. As a result, SLCs are based on statistical associations rather than experimental evidence of toxicity. Also, potential sampling biases, such as dredging methods that favor certain species, can affect the results.

### 2.1.6 Sediment Quality Triad (SQT) Approach

The Sediment Quality Triad Approach was originally created to assess site-specific sediment quality conditions but has since been adapted for use in developing sediment guidelines. This approach integrates three types of data: sediment chemistry, toxicity tests (bioassays), and in-situ biological community information. Chemical and physical analyses of sediments provide insight into contaminant levels and potential confounding environmental factors. Toxicity tests help evaluate whether the contaminants present are causing harmful effects. Biological monitoring, such as examining benthic community composition or abnormalities in resident organisms, reveals how sediment conditions may be affecting ecosystem health.

By bringing these three lines of evidence together, the method allows for a comprehensive assessment of sediment quality. It is particularly useful for distinguishing natural variations in biological communities from those caused by contamination. For example, differences in benthic communities may result from factors like grain size or water depth rather than chemical exposure; the triad method helps separate these influences.

A major benefit of this approach is its holistic nature. It can be applied to any detectable contaminant, captures both short- and long-term effects, and does not depend on knowing the exact mechanisms through which toxicity occurs. It offers a weight-of-evidence strategy for interpreting environmental impacts.

However, there are several drawbacks. The method lacks standardized statistical tools for integrating the three data types, and no uniform metrics have been established for summarizing each line of evidence. It is also resource-intensive, requiring a large dataset and complex sampling and analysis efforts. Furthermore, the selection of reference sites, used for comparison, can be problematic if the sites themselves are degraded. The method focuses mainly on acute toxicity and does not systematically incorporate measures of contaminant bioavailability or sublethal/chronic effects. Although not originally intended for guideline development, the SQTA has proven useful for identifying contaminated areas in need of remediation, determining the extent of impacted zones, and confirming sediment conditions after cleanup efforts.

### 2.1.7 Apparent Effects Threshold (AET) Approach

The Apparent Effects Threshold Approach was originally developed for use in the Puget Sound region of Washington State. It links contaminant levels in sediments to biological responses observed in field studies, particularly among bottom-dwelling organisms. The objective is to define the contaminant concentration above which statistically significant adverse biological responses (typically at a confidence level of $p \leq 0.05$) are consistently observed. These responses can include toxicity in sediment-dwelling or water-column organisms, shifts in species abundance, or changes in community composition.

This method is conceptually similar to the Screening Level Concentration Approach, as both rely on paired sediment chemistry and biological data. However, the AET approach is better suited for guideline development because it encompasses a broader array of biological indicators, including more sensitive metrics. For metals, contaminant concentrations are typically normalized to dry weight, while for organics, either dry weight or total organic carbon (TOC) normalization is used.

One of the method's strengths is its flexibility in using both laboratory bioassays and field observations, allowing for the development of thresholds tailored to specific species, endpoints, or locations. It can be applied to most substances that can be detected analytically. The method has proven to be effective in the Puget Sound region (U.S.), where it has been used to predict ecological impacts from sediment contamination.

However, one key limitation is the reliance on site-specific data that establish connections between sediment chemistry and biological effects. At present, suitable datasets are only available for limited geographic areas such as Puget Sound, parts of California, the Great Lakes, and sections of the Atlantic coast. Applying this method elsewhere would require large-scale data collection efforts.

Like other co-occurrence methods, the AET approach does not prove direct cause and effect but instead identifies statistical associations, which introduces some uncertainty. Another challenge is that the AET values can only remain the same or increase as new data are added, this makes them less flexible than other methods and raises the possibility of underestimating risks if AETs are used directly as regulatory benchmarks.

Additionally, the method may sometimes be overly conservative. This can occur when a chemical is consistently found alongside another toxic compound that is responsible for the observed effects, which is especially problematic when evaluating regional datasets dominated by such co-occurrences (e.g., DDT in Puget Sound).

The AET approach has been widely adopted in Washington State for evaluating sediments for dredging and disposal, and it forms the basis of legally enforceable sediment quality standards set by the Washington Department of Ecology. These standards are used to regulate pollution discharges, identify contaminated sites, and guide remediation efforts.

While a scientific advisory panel acknowledged the method's usefulness for site-specific assessments, it advised against applying AETs as universal or nationally standardized guidelines.

### 2.1.8 Weight of Evidence (WOE) Approach

The Weight of Evidence approach was initially created to provide informal tools for interpreting coastal sediment data collected under the National Oceanic and Atmospheric Administration (NOAA) National Status and Trends Program. It compiles information from several established approaches to guideline development, including equilibrium partitioning, spiked-sediment toxicity tests, and various co-occurrence-based methods such as the AET, SLC, and SQT approaches.

In this method, all input data are weighted equally, regardless of their origin. Data are first screened for suitability, focusing on whether they include both chemical concentrations and matching biological effects, the methodology used, the relevance of the measured biological endpoints, and the level of agreement between chemical and biological results. Datasets that show no clear relationship between chemical levels and biological responses are kept in the database but are excluded from the statistical analysis.

For the included data, each entry records the concentration of a specific contaminant, the type and location of the observed biological response, and whether the response was consistent with the presence of the contaminant (classified as no effect, low gradient, no match, or a "hit", indicating a clear association). Data without signs of toxicity are considered to reflect background conditions. Entries showing biological effects at elevated contaminant concentrations are sorted in order of concentration to identify threshold values.

The primary advantage of this method is its integration of multiple lines of evidence into a single framework for evaluating sediment quality. It organizes diverse data sources to show how contaminant concentrations relate to biological outcomes. Additionally, it uses only existing data, eliminating the need for new fieldwork or experiments, and the database is scalable and adaptable to new regions or contaminants.

## 2.2 Approach comparison

The approaches explained in chapter 2 all have strengths and limitations. When developing SQGs, it is crucial to identify the appropriate approach. A comparison between the explained approaches, including their strengths and limitations, is shown in Table 2-1 (Macdonald et al. 1994).

*Table 2-1: Summary of the strengths and limitations of approaches for deriving numerical sediment quality assessment guidelines (MacDonald et al, 1994)*

| Approach | Strengths | Limitations |
|---|---|---|

| | | |
|---|---|---|
| **SBA** | • Sufficient data are generally available. | • Not based on biological effects. |
| **SSBA** | • Based on biological effects.• Suitable for all classes of chemicals and most types of sediments.• Supports cause and effect evaluations. | • Sufficient data are not generally available.• Implementation costs are high.• Spiking procedures are not yet standardized. |
| **EqPA** | • Based on biological effects.• Suitable for all classes of chemicals and most types of sediments.• Bioavailability is considered.• EPA will support research to validate this approach.• Supports cause and effect evaluations. | • Few sediment quality criteria are currently available.• Water quality criteria not available for some substances.• In situ sediments rarely at equilibrium. |
| **TRA** | • Simple to apply.• Bioaccumulation is considered.• Protocol for TRGs available. | • Tissue residue guidelines not available.• In situ sediments rarely at equilibrium. |
| **SLCA** | • Based on biological effects.• Sufficient data available.• Suitable for all classes of chemicals and most sediments. | • No cause-effect relationships.• Large database required.• End point insensitive.• Bioavailability not considered. |
| **SQTA** | • Based on biological effects.• Integrates chemistry, bioassay, and in situ effects.• Provides weight of evidence. | • Difficult to derive numerical SQAGs.• Labour intensive and expensive.• Statistical TRIAD criteria not established.• Extensive site-specific data needed.• No cause-effect relationships.• Bioavailability not considered. |
| **AETA** | • Based on biological effects.• All biological data considered.• Suitable for all chemicals and sediments. | • Extensive site-specific data required.• No cause-effect relationships.• Risk of under/over protection.• Not applicable broadly.• Bioavailability not considered. |
| **WEA** | • Based on biological effects.• All biological data considered.• Suitable for all chemicals and sediments.• Weight of evidence.• Data summaries for sediment quality.• May use existing data. | • Large database required.• No cause-effect relationships.• Bias risk from data amalgamation.• Bioavailability not considered. |

To evaluate the different approaches, they can be assessed based on several criteria (MacDonald et al. 1994). The criteria are organized into several categories: practicality, cost effectiveness, scientific defensibility and applicability. Eventually, MacDonald made an overall assessment for each approach based on the suitability to derive SQGs in Florida. This evaluation didn't focus specifically on the conditions in Florida, so it is generally applicable. The evaluation can be seen in Table 2-2.

*Table 2-2: Evaluation of approaches for deriving sediment quality assessment guidelines (MacDonald et al., 1994)*

| Evaluation Criteria | SBA | SSBA | EqPA | TRA | SLCA | SQTA | AETA | WEA |
|---|---|---|---|---|---|---|---|---|
| **Supports dev. of numerical SQA** | Y | Y | Y | Y | Y | Y | Y | Y |
| **Feasible near term** | Y | N | Y/N | Y | Y/N | N | N | Y |
| **Expensive to implement?** | N | Y | Y | N | Y | Y | Y | Y |
| **Require new data?** | N | Y | Y | N | Y | Y | Y | Y |
| **Considers bioavailability?** | N | N | Y | Y | N | N | N | N |
| **Provides cause/effect?** | N | Y | Y | Y | N | N | N | N |
| **Based on biological effects?** | N | Y | Y | N | Y | Y | Y | Y |
| **Considers data from South East?** | Y | Y | Y | Y | Y | Y | Y | Y |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| **Uses weight of evidence?** | N | N | N | N | N | Y | Y | Y |
| **Supports ranges vs absolutes?** | N | N | N | N | N | Y | Y | Y |
| **Considers mixture contaminants?** | N | N | N | N | N | Y | Y | Y |
| **Requires field validation?** | N | Y | Y | Y | Y | Y | Y | Y |
| **Site-specific conditions?** | N | Y | Y | Y | Y | Y | Y | Y |
| **Supports monitoring programs?** | Y/N | Y | Y | Y/N | Y | Y | Y | Y |
| **Supports problem identification?** | Y | Y | Y | Y | Y | Y | Y | Y |
| **Supports regulatory programs?** | N | Y | Y | Y | Y | Y | N | Y |
| **Overall Assessment** | * | ** | **** | *** | ** | **** | *** | **** |

## 2.3 Analytical frameworks to establish substance threshold values in sediments

The analytical approaches that are introduced and explained in Section 2.1 form the basis for establishing sediment quality guidelines (SQG's). The analytical frameworks that are used in the guidelines can be formulated based on one or more approaches. This chapter will elaborate on the main existing frameworks and SQG categories. The development of the frameworks is explained including the approach that is used. Additionally, the threshold values are explained in detail.

Table 2-3 provides an overview of the SQG categories, along with its analytical framework and the developers.

*Table 2-3: Overview of analytical methods along with the sources explaining the approach*

| SQG Category | Analytical framework | Developers |
|---|---|---|
| **Theoretical** | Equilibrium Partitioning (EP) | Di Toro, Mahony et al. (1991) |
| | | Di Toro, Zarba et al. (1991) |
| | | Ankley et al. (1996) |
| | | NYSDEC (1998) |
| | | Di Toro and McGrath (2000) |
| **Empirical** | Screening-Level Concentration (SCL) | Persaud et al. (1993) |
| | | Von Stackelberg and Menzie (2002) |
| **Empirical** | Effects Range-Low (ERL) and Effects Range-Median (ERM) | Long et al. (1995) |
| | | USEPA (1996) |
| **Empirical** | Threshold-Effects Level (TEL) and Probable-Effects Level (PEL) | MacDonald et al. (1996) |
| | | Smith et al. (1996) |
| | | USEPA (1996) |
| **Empirical** | Apparent-Effects Threshold (AET) | Barrick et al. (1988) |
| | | Ginn and Pastorok (1992) |
| | | Cubbage et al. (1997) |
| **Empirical** | Consensus-Based Evaluation (CBE) | Swartz (1999) |
| | | MacDonald, DiPinto et al. (2000) |

| | | |
|---|---|---|
| | | MacDonald, Ingersoll et al. (2000) |
| **Empirical** | Logistic Regression Modeling (LRM) | Field et al. (1999, 2002) |

The frameworks of Equilibrium partitioning, Screen-level concentration and Apparent-effects threshold are those that exclusively tied to the respective approaches described above. Therefore, both these approaches and frameworks are named the same. The other frameworks are described below.

### 2.3.1 Effects Range-Low (ERL) and Effects Range-Median (ERM)

The ERL/ERM analytical framework is a well-established framework for evaluating sediment quality based on empirical evidence of contaminant effects. It was developed using large datasets that correlate sediment contaminant concentrations with observed biological responses. This framework is based on the WOE approach. Please note that since large datasets are used, the WOE approach combines multiple data sources that are gathered using different individual approaches. The threshold criteria are defined as follows:

- Effects Range-Low (ERL): This is the concentration below which adverse effects are rarely observed. It represents the 10th percentile of concentrations from affected samples, serving as a lower threshold for potential concern.
- Effects Range-Median (ERM): This represents the 50th percentile of concentrations where biological effects were observed, indicating a level above which harmful effects are frequently or always encountered.

The ERL/ERM criteria are used as informal benchmarks to assess the potential for adverse effects in marine and estuarine environments. If a sediment sample has contaminant levels below the ERL, it is considered low risk; concentrations above the ERM suggest a high probability of harmful effects. Intermediate values require further consideration.

The reliability of ERL and ERM values can be evaluated by assessing consistency across data sources. The method also allows users to estimate the likelihood of biological impacts at varying contaminant concentrations, providing a probabilistic basis for environmental decision-making. The framework is widely used due to its simplicity, transparency, and compatibility with existing datasets. It facilitates screening-level assessments and supports decision-making in sediment management and site prioritization for remediation.

However, like other co-occurrence frameworks, ERL/ERM criteria are based on associations rather than causality. They do not account for site-specific factors such as bioavailability, sediment characteristics, or the presence of contaminant mixtures, which may influence the actual risk posed by a given concentration.

### 2.3.2 Threshold Effect Limit (TEL) and Probable Effect Limit (PEL)

This framework defines two key concentration thresholds to interpret sediment contamination levels, based on observed biological effects in benthic organisms. The Threshold Effect Limit (TEL) represents a contaminant concentration at which early signs of adverse biological responses begin to appear. In contrast, the Probable Effect Limit (PEL) marks the concentration above which significant toxic effects are consistently observed in a large proportion of benthic organisms. This framework is based on the WOE approach, which combines multiple data sources that are gathered using different individual approaches.

These thresholds help environmental managers and engineers make informed decisions regarding the ecological consequences of contamination and the potential need for site remediation or further investigation. Concentrations below the TEL are typically considered safe, while those above the PEL are regarded as harmful and likely to cause negative impacts. Values falling between these limits represent a zone of uncertainty where additional assessment and expert judgment are required.

The TEL and PEL criteria have been calculated for various contaminants based on laboratory studies where benthic organisms were exposed to sediments from coastal waters. For example, the state of Florida developed specific equations to derive these thresholds, grounded in toxicological responses observed under controlled conditions.

Numerous similar thresholds have been developed by different agencies, often using slightly different terminology. As of 2001, several organizations had published limits for metals and other substances with similar conceptual frameworks. These are summarized in reference tables comparing the various sediment quality benchmarks and their corresponding contaminant concentrations associated with toxic effects.

TEL/PEL and ERL/ERM are both paired, effects-based sediment benchmarks, but they're derived differently. TEL/PEL are typically calculated from combined chemistry effects datasets to define a lower "rare effects" threshold (TEL) and an upper "frequent effects" threshold (PEL) using a specific derivation method. ERL/ERM are derived by compiling concentrations associated with observed effects and setting ERL near the low end of the effects range and ERM around the median of the effects range. Because the underlying datasets and statistics differ, TEL/PEL and ERL/ERM can yield different numbers and shouldn't be used interchangeably without stating which set you applied.

### 2.3.3 Consensus-Based Empirical SQG

To reduce the uncertainty associated with interpreting sediment contamination in the range between TEL and PEL values, the Consensus-Based Sediment Quality Guideline (SQG) framework was developed, notably by the Wisconsin Department of Natural Resources (WI DNR). This method integrates findings from multiple SQG sources to provide more robust and widely accepted threshold values.

This framework typically defines three effect concentrations:

- Threshold Effect Concentration (TEC): The concentration below which harmful effects on benthic organisms are unlikely. It is derived similarly to TEL values.
- Midpoint Effect Concentration (MEC): An optional midpoint estimate to provide context between the lower and upper thresholds.
- Probable Effect Concentration (PEC): The concentration above which toxic effects are likely to occur, similar in concept to PEL.

The TEC and PEC values may be determined either from average values across multiple guidelines or derived from laboratory data specific to the site being assessed. When developing site-specific values, toxicity testing is conducted using local sediment and resident species to account for geographic and ecological variability. Risk tolerance and desired conservatism can also guide the selection of appropriate thresholds, either by choosing average values from published sources or identifying concentrations associated with specific percentiles of observed toxic effects.

### 2.3.4 Logistic Regression Modeling (LRM)

Logistic regression modelling is a statistical tool used to quantify the relationship between sediment contaminant concentrations and the likelihood of adverse biological responses. This method estimates the probability of an observed effect, such as reduced survival, reproduction, or changes in community structure, based on contaminant levels in sediments.

In the context of sediment quality assessment, logistic regression is particularly useful for defining thresholds with statistical confidence. By analyzing paired data on contaminant concentrations and biological outcomes from field studies or laboratory toxicity tests, the logistic model produces a probability curve that indicates the chance of observing biological effects at different contaminant levels.

The general form of the logistic model is:

$$P = 11 + e - (\beta_0 + \beta_1 C)P = \frac{1}{1 + e^{-(\beta_0 + \beta_1 C)}} P = 1 + e - (\beta_0 + \beta_1 C)1$$

Where:

- P is the predicted probability of a toxic effect occurring,
- C is the contaminant concentration,
- $\beta_0$, $\beta_1$ are model coefficients estimated from the data.

This model allows the derivation of contaminant concentrations corresponding to specific probabilities of effect (e.g., 10%, 50%, 90%). These probability-based thresholds can then inform sediment quality evaluations and help classify sites into different risk categories. Logistic regression also enables comparison between contaminants and locations, supports weight-of-evidence approaches, and can be used to validate or refine existing guidelines like TECs, PECs, TELs, or PELs.

Ultimately, logistic regression provides a rigorous, quantitative basis for determining how sediment contamination may affect ecological health and can improve confidence in risk-based decision-making for sediment management.

# 3 Analysis of parameters and guidelines in other countries

## Chapter 3 – Executive summary

Building on increased attention and a desire to further improve the normative guidelines for dredged materials, this chapter focuses on the procedural steps in other countries than Colombia regarding the regulation for maritime dredging and beneficial use of sediments. The focus countries of this study are the following:

- The Netherlands
- Canada
- USA (specifically the state of Florida)
- Australia
- New Zealand
- Spain
- Japan
- Brazil
- Chile
- Peru
- Mexico
- Panama
- Costa Rica

**The key takeaways of this chapter are:**

- Overall, the procedures have many similarities and can be divided into 3 categories:
    - Purpose-driven guidelines (e.g. The Netherlands) (Section 3.1 and Element 1 report[1])
    - Step by step approach with screening values, followed by detailed analysis (Canada, USA, Australia/New Zealand) (Section 3.2, 3.3 and 3.4)
    - Unique custom approaches (Spain, Japan) (Section 3.5 and 3.6)

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[1] Arcadis, 2025. Element 1: *Information, experiences, and lessons learned in the licensing, operation, monitoring and control of maritime dredging activities in the Netherlands, as well as in the determination of requirements to approve in this country the uses of marine dredged sediments*

    - Regulation based on water and soil quality or international regulations (Brazil, Panama, Mexico, Costa Rica, Peru, Chile) (Section 3.7 to 3.12)

Internationally, the following conventions are relevant, determining the basis requirements for dredging and waste dumping in international waters that prevail over national guidelines:

- The London convention[2] 1975 (Colombia included), followed by the London protocol 1996 (Colombia not included) (revised and modified several times, latest official accepted amendment in 2022[3]):
  *"Contracting Parties to the Protocol are obliged individually and collectively to protect and preserve the marine environment from all sources of pollution. This includes taking effective measures, according to their scientific, technical and economic capabilities, to prevent, reduce and where practicable eliminate pollution caused by dumping or incineration of waste or other matter at sea"*

Additionally, there are many region-specific legally binding agreements that include protocols on dumping, pollution, dredging and EIA requirements. As an example below several conventions are stated, but many more exist. This includes the OSPAR convention (North-East Atlantic, 1998), Barcelona convention (Mediterranean, 1976), Lima convention (Southeast pacific, 1981) and Cartagena convention (Gulf of Mexico & Caribbean Sea, 1986).

The sediment quality guidelines of each focus country are outlined in detail below. The section begins with an overview of the country's governance structure, highlighting the relevant institutions, their responsibilities, and the applicable legal framework. This is followed by a description of the sediment quality framework and the official procedures. Finally, the approach/methodology used to assess sediment quality is explained, including the basis on which the guidelines are established.

In Southern America, only Brazil has dedicated sediment quality threshold values. The other countries use separate approaches for certain chemicals but mainly rely on international guideline values. Table 3-1 shows the methods and approaches used to determine water and sediment quality criteria and which countries apply these.

*Table 3-1: Methods and approaches for south American countries (Developed by the authors of this report)*

| Method/Approach | Principles | Where Applied |
|---|---|---|
| **Toxicity tests** | Expose organisms to contaminated matrices to indicate toxicity | All South American countries |
| **Benthic structure** | Evaluate community composition and abundance for contamination | All South American countries |
| **Equilibrium Partitioning** | Predict interstitial water concentrations and compare to criteria | Brazil, Chile |
| **AET** | Identify concentration of chemicals above which effects occur | Brazil, Argentina |
| **SQT** | Integrates chemistry, toxicity, benthic fauna metrics | Brazil, Argentina, Chile |
| **SQG** | Identify chemical ranges and relate to effects using weight of evidence | Brazil |

## 3.1 The Netherlands

The procedural guidelines of The Netherlands regarding beneficial use of sediment have been described in detail in Element 1. In short, The Netherlands focuses on purpose-driven guidelines where the final purpose of the sediment decides which regulations are triggered. For regulation on dredging (site allocation, planning, environmental permits) it relies on international guidelines (OSPAR, London convention and the EU water framework directive).

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[2] https://www.imo.org/en/OurWork/Environment/Pages/London-Convention-Protocol.aspx

[3] Amendment of 2022 accepted by all parties except China, Canada and Finland: https://climate.law.columbia.edu/sites/climate.law.columbia.edu/files/content/LC%2046-17.pdf?utm_source=chatgpt.com

## 3.2 Canada

### 3.2.1 Governance structure

Canadian sediment quality management is a cooperative effort across federal and provincial levels. The Canadian Council of Ministers of the Environment (CCME) is a key institution, bringing together Canada's 14 federal, provincial, and territorial environmental agencies to develop nationally consistent guidelines and standards. CCME's Water Quality Task Group, which is supported by scientists at Environment and Climate Change Canada (ECCC), has led the development of sediment quality guidelines that are used as policy benchmarks across the country. ECCC provides technical expertise and manages certain federal activities. Provincial ministries implement and enforce sediment management, often by adopting or referencing the CCME guidelines in their own regulations and criteria.

Several laws and policies regulate sediment quality in Canada. At the federal level, the Canadian Environmental Protection Act (CEPA) empowers ECCC to control the dumping of dredged materials at sea through a permit system. The Fisheries Act also prohibits the release of polluting substances, to protect Canada's fisheries resources. The CCME sediment quality guidelines are used as reference values within legislations of the provinces. This means that the CCME's SQGs are not enforceable by law but used as guidelines to help the legal framework of the provinces.

In Canada not all provinces directly use the guidelines set up by the CCME. For example, Quebec developed new criteria for the assessment of sediment quality (EC, 2007), based on the approach of the framework and SQGs of CCME. This was needed because naturally high concentrations of certain metals in St. Lawrence were found, therefore the same standards could not be applied. This illustrates that site-specific sediment quality frameworks and regulations are sometimes required (and improve accuracy).

### 3.2.2 Framework and procedures

Canada has established a comprehensive framework for assessing sediment quality in both marine / coastal environments and for freshwater ecosystems via the CCME's national guidelines (CCME, 1995)[4]. Both marine and freshwater guidelines use two threshold values to initially assess the sediment quality: the Threshold Effect Level (TEL) and the Probable Effect Level (PEL). The TEL represents a concentration below which adverse effects on sediment-dwelling organisms are expected to be negligible or rare, whereas the PEL represents a higher concentration above which adverse biological effects are expected to occur more often. In other words, concentrations under the TEL are considered safe for the aquatic ecosystem (unlikely to cause harm), concentrations above the PEL are likely to pose ecological risks, and the intermediate range between TEL and PEL is an uncertainty zone where effects may occur. These guideline values are not hard standards but inform a weight-of-evidence evaluation. If a sediment's contaminant levels are above the TEL but below the PEL, other evidence (toxicity tests, bioassessments) will guide the final decision. An overview of the general framework can be seen in Figure 3-1.

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[4]

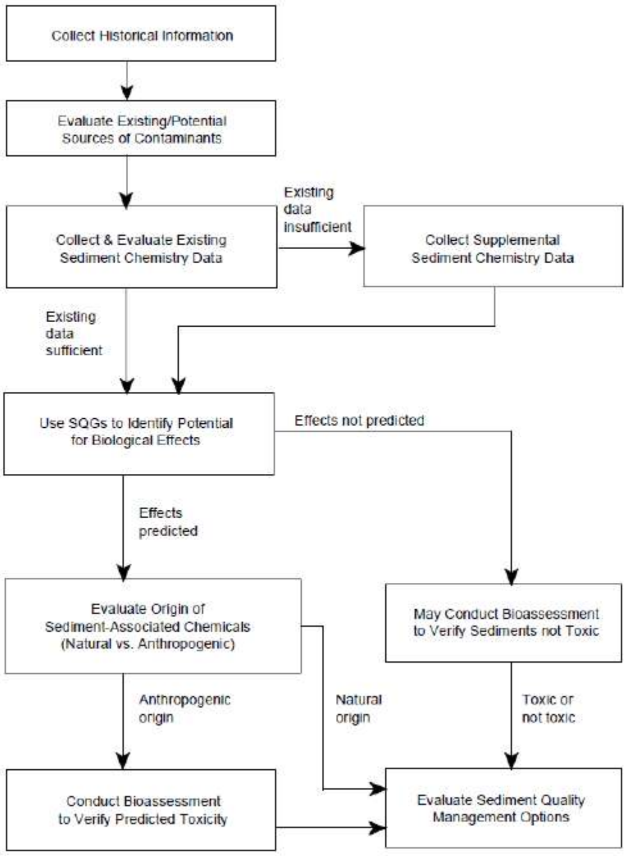

*Figure 3-1: Overview of the governmental approach to sediment quality in Canada (CCME, 1995)*

### 3.2.3 Approach

The sediment quality guidelines in Canada are developed by the Canadian Council of Ministers of the Environment (CCME). They established a protocol in 1995 which uses a weight-of-evidence (WOE) approach to develop the guideline (CCME EPC-98E) (CCME, 1995). This approach used a large database, which consisted of sediment chemistry and biological effects data. It evaluates toxicity test information to identify the concentration per substance.

Canada was one of the first to develop a guideline using this WOE approach. The process to develop the Canadian SQGs is schematized in Figure 3-2. In case that the Colombian government decides to determine SQGs based on the WOE approach, it is highly relevant to follow and study this schematization. It describes the approach from start to finish to determine country-specific SQGs.

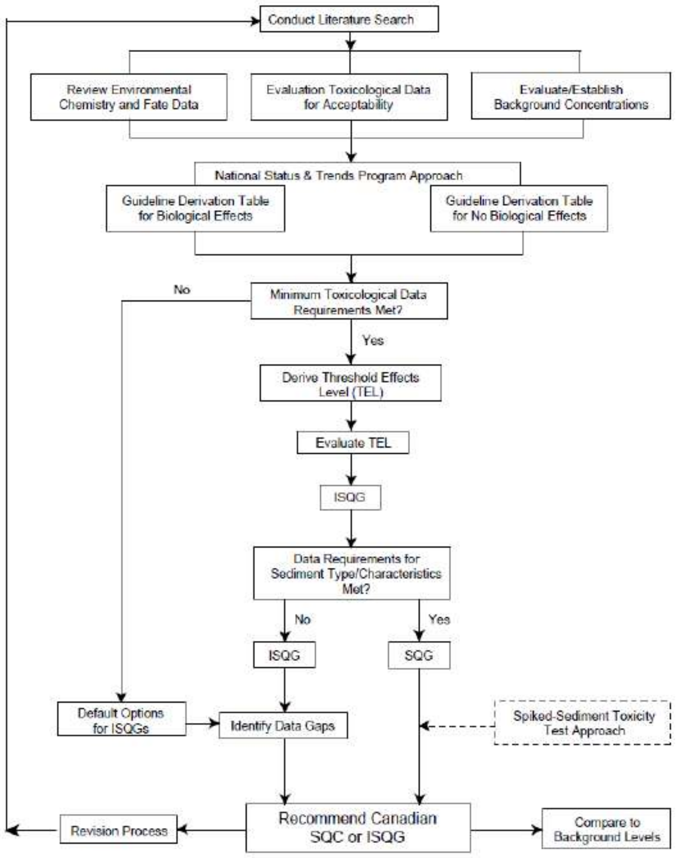

*Figure 3-2: Overview of the sediment quality assessment of Canada (CCREM, 1987)*

Canada's initial approach to establishing water quality guidelines involved a comprehensive review and comparison of existing standards developed by other countries (CCREM, 1987). In some cases, these external guidelines were adjusted using scientific evidence to better reflect environmental conditions specific to Canada.

Before a standardized method for sediment quality guideline development was in place, MacDonald et al. (1992) proposed a practical interim strategy. They recommended reviewing effect-based sediment quality benchmarks from other regions and determining their relevance for Canadian use. Where appropriate, these values could be modified using available scientific data to improve their applicability. If the adapted values were found to be suitable, they could be adopted as provisional sediment quality guidelines.

This recommendation was consistent with the early approach taken by the Canadian Council of Ministers of the Environment (CCME) for introducing preliminary water and soil quality guidelines. From 1999 to 2002, this approach led to several updates and revisions of individual contaminant guidelines, refining the framework for sediment quality protection in Canada.

*Table 3-2: Overview of the derivation of threshold values per component in the Canada sediment guidelines*

| **Canada: The framework and its underlying principles** | | |
|---|---|---|
| **Component** | **Derivation lower value** | **Derivation higher value** |
| **General framework** | Weight of evidence (WOE) approach | |
| **Toxaphene, Nonylphenol, PCB's, PCCD, PAH, Heptachlor epoxide, lead, lindane, mercury, DDT, dieldrin, Endrin, cadmium, chlordane,<br>Chromium, Zinc, Copper** | Validated with SSB + EP, but maintained TEL | Checked, but maintained PEL |
| **Other substances** | TEL, MacDonald et al. (1992) | PEL, MacDonald et al. (1992) |

## 3.3 USA: Florida

In the United States the U.S. Army Corps of Engineers and the U.S. Environmental Protection Agency (EPA) are responsible for issuing permits for dredging projects under the Clean Water Act, the Marine Protection, Research and Sanctuaries Act, and the National Environmental Policy Act. There are no national standards for evaluating sediment quality. In effect, the Clean Water Act grants individual U.S States the authority to establish surface water quality standards. Therefore, each State uses their own sediment quality guidelines.

Among the most widely referenced sediment quality benchmarks in the U.S. are those developed by the National Oceanic and Atmospheric Administration (NOAA). NOAA is a federal scientific agency focused on the conditions of the oceans, major waterways, and the atmosphere. In the context of sediment quality, NOAA established the Effects Range Low (ERL) and Effects Range Median (ERM) values through Long et al. (1995), based on a large database of paired sediment chemistry and biological effects data. These values are not regulatory standards but are widely used as screening tools to indicate concentrations below which adverse effects are rarely observed (ERL) and above which effects are more frequent (ERM). Most sediment quality regulations within states are based on the criteria of NOAA (Long et al. (1995)) or those of MacDonald (1994; 1996).

Since the sediment quality guidelines are applicable on state-level in the USA, this study will focus on only one state in more detail. The state Florida shows most environmental similarities to Colombia, hence this state is chosen to be analyzed and described in more detail.

### 3.3.1 Governance structure

Florida’s sediment quality management operates under a multi-tiered regulatory framework involving state and federal institutions. The Florida Department of Environmental Protection (FDEP) is the lead state agency, responsible for environmental management and oversight of water and sediment quality. At the federal level, the U.S. Environmental Protection Agency (EPA) provides guidance but has not set enforceable sediment quality criteria, leaving states like Florida to establish their own guidelines. The U.S. Army Corps of Engineers (USACE) plays a key role for dredging activities. Under the Clean Water Act (Section 404) and Rivers and Harbors Act (Section 10), the USACE permits dredge-and-fill operations in navigable waters, while FDEP must certify that such projects comply with state water quality requirements. Thus, Florida’s sediment governance is a collaborative system, dredging projects typically require both state-level approval (from FDEP) and a federal permit from USACE. Other entities like regional Water Management Districts and NOAA (in marine sanctuaries) may also be involved in review, but FDEP and USACE are the primary regulators.

Dredging in Florida’s waters are regulated through the Environmental Resource Permitting (ERP) program administered by FDEP. The ERP program covers activities in wetlands or surface waters, including dredging and filling. Through this program, FDEP evaluates proposed projects for potential water quality impacts (such as turbidity and contaminant release) and issues permit to protect aquatic environments. In coastal waterways, the Florida Coast Management Program (under the Coastal Zone Management Act) also ensures that dredging and disposal align with state coastal resource policies. Federal laws like the Marine Protection, Research, and Sanctuaries Act (for ocean dumping of dredged material) and the National Environmental Policy Act (NEPA) further layer into the oversight, typically requiring environmental impact assessments for larger dredging projects. Overall, FDEP, USACE, and EPA work together. This governance structure provides checks at multiple levels.

### 3.3.2 Framework and procedures

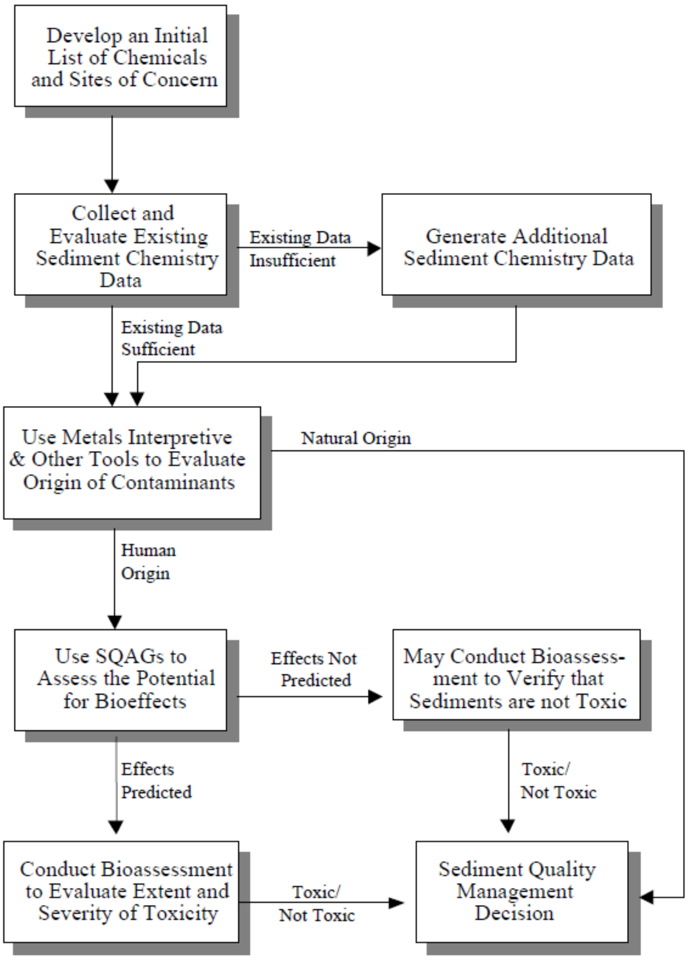

*Figure 3-3: Overview of the sediment quality assessment of Florida (MacDonald et al. 1994)*

Florida has developed a comprehensive framework for assessing sediment quality in both coastal/marine and freshwater environments (MacDonald et al. 1994). The framework that is developed applies Sediment Quality Assessment Guidelines (SQAGs) in combination with tools like the metals interpretive tool (published in 1988) and bioassessments. A schematization of this framework is shown in Figure 3-3 (MacDonald et al. 1994)

Additionally, the following steps should be conducted when a site-specific sediment quality assessment is done.

Collect information on historical land and water use
1. Collect and evaluate existing sediment chemistry data
2. Collect supplemental sediment chemistry data
3. Evaluate the origin of sediment-associated contaminants
4. Conduct preliminary assessment of the potential for biological effects of sediment-associated contaminants
5. Conduct biological assessment of sediment quality
6. Make management decisions based on the weight of evidence

This framework ensures that SQAGs are not used in isolation, they are a first-tier indicator, to be followed by site-specific biological evaluation if a potential problem is flagged.

In the states Florida and North Carolina regulations state that when dredged sediment is used beneficially, more fine-graded sediment (finer than sand) is allowed when compared to standard beach nourishment projects. Beneficially placed dredged sediment is permitted to include up to 10% fine-grained material (finer than sand) than what is found in the native beach sand. Standard beach nourishment projects are restricted to a maximum of 5% more fine-grained material than the native sand content (CSO,2022).

### 3.3.3 Approach

In the absence of federal standards, the Florida Department of Environmental Protection developed their own guidance documents. The main guideline of Florida: "A*pproach to the Assessment of Sediment Quality in Florida Coastal Waters*" is published in 1994 by D.D. MacDonald (FDEP, n.d.). MacDonald applied a modified version of the WOE Approach. This method integrates data from spiked-sediment bioassays, field-collected sediment chemistry and existing literature and databases. The guidelines are split into volume 1 and volume 2. Volume 1 describes the development and evaluation of the general SQAGs. Volume 2 describes the application of the SQAGs. In volume 2, the method is applied to Florida coastal sediment data, which resulted in the identification of several contaminated areas (Miami, Jacksonville, Tampa Bay, Pensacola). These areas are then considered the highest priority to investigate further.

Florida did not adopt NOAA's ERL/ERM criteria but created its own TEL/PEL framework based on a similar principle: using field data to statistically define thresholds of concern. Additionally, please note that although the threshold values are called TEL/PEL, these are different values than the Canadian threshold values. The Canadian TEL/PEL are developed by CCME and are based on data in North America. The guidelines in Florida adapted the Canadian methodology but customized it using Florida-specific data. MacDonald played a role in the development of both guideline systems.

The guidelines of Florida (MacDonald, 1994) are used on a global scale and form the basis of SQGs of many other countries. To this day, Florida still uses the same SQG threshold values as defined by MacDonald in 1994.

*Table 3--3: Overview of the derivation of threshold values per component in the Florida/USA sediment guidelines*

| **Florida: The framework and its underlying principles** | | |
|---|---|---|
| **Component** | **Derivation lower value** | **Derivation higher value** |
| **General framework** | Weight of evidence (WOE) approach | |
| **Organics,<br>Metals, metalloids, organometallic** | TEL (MacDonald, 1994) | PEL (MacDonald, 1994) |

## 3.4 Australia / New Zealand

Australia and New Zealand share a common set of sediment quality guidelines established under the joint framework (Australian and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand)((ANZECC and ARMCANZ, 2000a), (ANZECC and ARMCANZ, 2000b). Both countries will therefore be described in this section.

### 3.4.1 Governance structure

In Australia, the key legal instrument is the Environment Protection (Sea Dumping) Act 1981 (ACT 2023), which regulates the loading and disposal of dredged material at sea. All ocean disposal of dredged sediments in Australian waters requires a permit from the federal environment department, or from the Great Barrier Reef Marine Park Authority (GBRMPA) if within the Great Barrier Reef (GBR) Marine Park. Sensitive areas like the GBR have additional protection under the GBR Marine Park Act 1975. Before a dredging permit is granted, it is typically required to develop a sediment Sampling and Analysis Plan (SAP) for approval. The SAP ensures that sediment sampling and testing will meet the guidelines requirements, and that the data collected will be adequate for decision-making. Detailed requirements for sediment sampling, chemical analyses, and biological testing are set out in the National Assessment Guidelines for Dredging (2009). These guidelines list contaminants of concern (based on international lists) and their Screening Levels (trigger values), and they provide procedures for cases where a contaminant has no established guideline. If such a contaminant exceeds regional ambient background levels, additional bioavailability, toxicity, and bioaccumulation tests are required. Notably, if dredged material is found unsuitable for ocean disposal, beneficial land-based uses can be considered under the purview of state or territory environmental regulators, ensuring dredged sediments are managed by the appropriate authority depending on the use or disposal option. Often the contaminants of concern do not leach in land disposal sites and the spoil may even gain a solid waste classification, rather than industrial hazardous waste (for example, EPA NSW, 1997).

In New Zealand, the same sediment quality guidelines are applied. Ocean disposal and dredging activities are managed under New Zealand's environmental legislation in line with London Convention/Protocol obligations. For instance, disposal of dredged material in the coastal zone requires resource consent under the Resource Management Act 1991, and in the exclusive economic zone it is regulated by the Exclusive Economic Zone (EEZ) Act 2012 via the Environmental Protection Authority (EPA). New Zealand authorities use the Interim Sediment Quality Guideline Low/High (ISQG-Low and ISQG-High) values from ANZECC (2000) as key benchmarks in decision-making. Institutional responsibility in New Zealand is shared among the Ministry for the Environment (which co-develops the guidelines with Australia) and agencies that issue permits (regional councils for coastal waters and the EPA for the EEZ).

Both countries' governance frameworks thus ensure that dredged material is assessed against the same scientifically derived criteria before approval for ocean disposal.

## 3.4.2 Framework and procedures

Australia and New Zealand employ a tiered assessment framework for evaluating sediment quality. This stepwise approach means that an exceedance of a guideline trigger value leads to progressively more detailed investigations, rather than an automatic rejection. The framework is outlined in the ANZECC/ARMCANZ (2000) Water Quality Guidelines and updated in 2013 (Simpson et al., 2013).

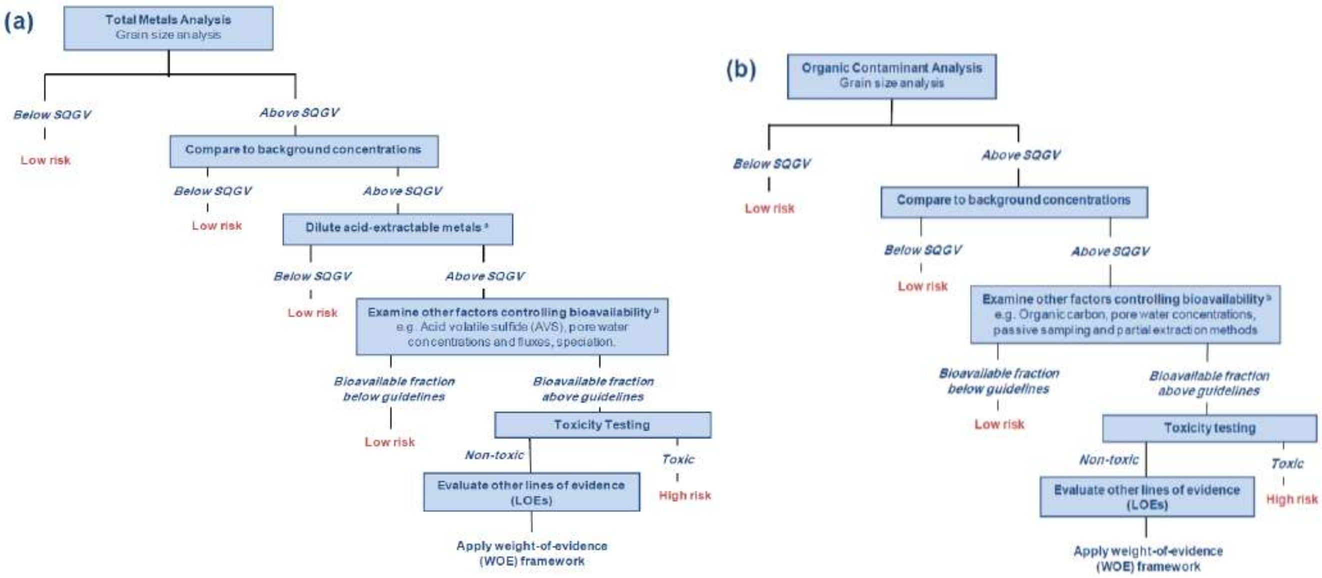

*Figure 3-4: Overview of the sediment assessment approach in Australia/ New Zealand (Simpson et al., 2013).*

At the first step, the measured values for the total contaminant concentration in the sediment is compared to the SQGV. These measurements are made on the sediment fraction <2mm to exclude materials with low capacity for binding contaminants (e.g. coarse sand). If total metal or organic contaminant levels exceed the SQGV, they should be compared to background concentrations in reference sediments of similar grain size from suitable locations. Exceeding the SQGV is acceptable if the concentrations remain below the natural background levels. If this is not the case, other factors that control bioavailability need to be examined. If the bioavailable fraction is above the guidelines, toxicity testing needs to take place. If the results show non-toxicity, other lines of evidence (LOE) need to be evaluated and applied to a weight of evidence (WOE) framework. This framework is designed to integrate the major LOEs, which are chemistry, toxicity, bioaccumulation and ecology. The scores for each LOE are placed in a decision matrix and an overall assessment is made. Table 3-4 provides an overview of the WOE approach.

*Table 3-4 Overview of the WOE approach (Source)*

| Line of Evidence | 3 (Significant Adverse) | 2 (Possible Adverse) | 1 (No Adverse) |
|---|---|---|---|
| **Chemistry - Sediment** | > SQG-high | > SQG-low < SQG-high | < SQGV-low |
| **Chemistry - Pore water** | > WQG[5] HC10 | < WQG HC10 | < WQG HCS |
| **Toxicity** | ≥ 50% effect | 20–50% effect | < 20% effect |
| **Bioaccumulation** | Sig. different (p<0.05) and >3× control | Sig. diff. (p<0.05) and <3× control | Not sig. diff. from control |

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[5] Water quality guideline

| Line of Evidence | 3 (Significant Adverse) | 2 (Possible Adverse) | 1 (No Adverse) |
|---|---|---|---|
| **Ecology** | Significant and high effects on abundance/diversity | Moderate effects | No significant effects |
| **Weight of Evidence** | **Significant adverse effects** | **Possible adverse effects** | **No adverse effects** |

Investigations of the sediment combine assessments of:

- Sediment chemistry (such as exceedances of sediment quality guidelines), including contaminant bioavailability tests (for example, pore-water measurements, acid volatile sulfide tests, passive sampling methods and approaches that mimic biotic responses to hydrophobic organic contaminants);
- Toxicity testing (for example, of multiple species, varying exposure pathways, and acute and chronic endpoints such as survival, growth, reproduction or avoidance, and biomarkers of effects);
- Bioaccumulation or biomagnification; and
- Benthic community structure and function.

Toxicity identification evaluation (TIE) and other assessments of causality may also be of value. In Australia, the requirements for sampling, testing, components, etc., are described in the National Ocean Disposal Guidelines for Dredged Materials. They therefore exclude parameters included in the marine and freshwater quality standards of Australia/New Zealand. The procedure here is that where contaminants are found for which there are no Screening Levels, and these contaminants are present at levels exceeding regional ambient baseline levels in sediments of comparable grainsize, the bioavailability, toxicity and, where appropriate, bioaccumulation testing, will be required for those substances.

### 3.4.3 Approach

The most widely recognized method for establishing sediment quality guidelines (SQGs) today involves the use of large-scale biological effects databases. Numerous international SQG values (SQGVs) have been proposed (e.g., Buchman, 2008), providing a basis for evaluating the potential risks posed by sediment-associated contaminants. This method forms the foundation of the interim sediment guidelines used by Australia and New Zealand. The SQGVs are adapted from the TEL/PEL used in Canada and the ERL/ERM used in the US. Therefore, the updated guidelines use empirically derived SQGVs, based on toxicity rankings and biological effects data from extensive North American datasets.

In Australia and New Zealand, the sediment guidelines within the broader water quality framework (ANZECC and ARMCANZ, 2000a) were revised by (Batley et al., 2008), and (Simpson et al., 2013). Although some SQGVs were updated, the main advancement was the introduction of a structured decision-making framework that incorporates multiple lines of evidence. This WOE approach is especially useful when sediment chemistry and toxicity results are inconclusive, aligning the region with current global best practices in sediment quality assessment.

Two reference values are provided: a lower threshold (SQGV-low) and an upper threshold (SQGV-high). These are conceptually aligned with the ERL/ERM framework, whose regulation is typically based on the lower threshold, which acts as a preliminary screening tool. Concentrations below this level are assumed to pose minimal risk, while those above the upper value are more likely to be harmful. If sediment contaminant levels exceed the lower SQGV, additional investigation is required. The adopted strategy is to apply the best-available international data and refine it using knowledge of local background conditions and new region-specific effects data as it becomes available.

It is important to note that, unlike water quality guidelines, these SQGVs are not grounded in direct cause-and-effect relationships. This can sometimes lead to misunderstandings about the ecological significance of sediment chemistry data. The empirical approach identifies the 10th percentile and median values of observed effects from compiled datasets. Because sediments often contain complex mixtures of contaminants (e.g., metals with organics), observed toxicity in a sample is attributed equally to all components during ranking. This can result in conservative values. For example, a sample with low zinc and high PAHs might still implicate zinc in the toxicity, even if it's not contributing to the effects, leading to an overly protective zinc SQGV.

In the revised ANZECC/ARMCANZ guidelines, copper sediment values were derived using a Species Sensitivity Distribution (SSD) approach, based on chronic toxicity data from multiple species. This method, similar to that used for water quality, estimates the concentration level for protecting 95% of species (HC5). To improve relevance, adjustments were made for factors like organic carbon, grain size, and metal bioavailability (e.g., AVS/SEM). While SSD provides a more robust and ecologically meaningful guideline than older empirical methods, it requires extensive data, limiting its broader application. For local conditions, the more tailored Site-Specific Biotic (SSB) approach can be used, though it demands significantly more resources.

*Table 3--5: Overview of the derivation of threshold values per component in Australia/New Zealand sediment guidelines*

| **Australia & New Zealand: The framework and its underlying principles** | | | |
|---|---|---|---|
| **Component** | **Derivation lower value** | **Derivation higher value** | **Specific comments** |
| **General framework** | Based on WOE approach | | |
| **Organics** | TEL (Mac Donald, 2000) | PEL (Mac Donald, 2000) | 10th percentile value |
| **Metals, metalloids, organometallics and other organic compounds** | ERL (Long et al. 1995) | ERM (Long et al. 1995) | |
| **Copper** | SSD (Simpson et al. 2013) | SSD (Simpson et al. 2013) | Custom approach |

# 3.5 Spain

Spain is included in this research due to several reasons. The country performed significant research in the field of sediment quality, with numerous site-specific studies conducted to develop SQGs. Some of these research efforts are carried out in collaboration with South American countries, enhancing their relevance to this study. Moreover, Additionally, as a Spanish-speaking country, Spain's official documentation is easily accessible to Colombian authorities. The following chapters will provide a more detailed overview of Spain's sediment quality guidelines and regulatory framework.

## 3.5.1 Governance structure

The guidelines concerning sediment quality are described in the report: “Directrices para la caracterización del material dragado y su reubicación en aguas del dominio público marítimo-terrestre", published by the Comisión Interministerial de Estrategias Marinas (CIEM, 2021). These replace the original guidelines of 1994. The guidelines are based on three conventions for the protection of the marine environment: the London protocol, the OSPAR Guidelines and the UNEP(DEPI)/MED guidelines.

Permits for dredging and placement of dredged material are given by different authorities, depending on the location. For an overview of the relevant authorities, see table 36-.

*Table 3--6: Overview of the governance structure in Spain regarding sediment reuse*

| Location | Permit from authority | Additional reports |
|---|---|---|
| **Inside port** | Port Authority | - Maritime Authority: if it affects navigation<br>- Environmental and Fisheries Authorities<br>- Coastal Authority: if the material is reused for beach nourishment.<br>- Waste Authority if the material might be hazardous |
| **Outside port** | Coastal Authority (permission to dredge) | Environmental and Fisheries Authorities |
| **Outside port** | Maritime Authority (permission offshore disposal) | Environmental and Fisheries Authorities |

## 3.5.2 Framework and procedures

The decision process stated in the regulations are summarized in Figure 3-5. The steps are as follows:

1. Check if characterization is required:

Article 8: "*Projects with a total volume of up to 10,000 m³ are exempt from characterization, provided there are no significant sources of contamination and local information reasonably confirms the sediment is not polluted.*"

2. If characterization is indeed required, check:
    a. Preliminary characterization (grain size, organic matter, solids, TPT test)
    b. Chemical analysis if needed (metals, organics) (Articles 15–18)
    c. Biological tests if contamination is near thresholds (Article 19)
3. Classification of sediment (apply to fine sediments (<63um) with more than 10% organic matter)
    a. Category A: Clean
    b. Category B: Moderate contamination
    c. Category C: More contamination, but not hazardous
    d. Category R: Potentially hazardous (Article 24)
4. Decide on:
    a. Beneficial use (beach nourishment, port fill, etc.)
    b. If not possible → evaluate placement or confinement (Article 26–29)
5. Permits: Apply for required permits based on location and use
6. Execution & Monitoring

The beneficial reuse of sediment is included in Figure 3-5. If the sediment quality is according to category A or B, it is allowed to be beneficially reused or to be directly disposed into the sea.

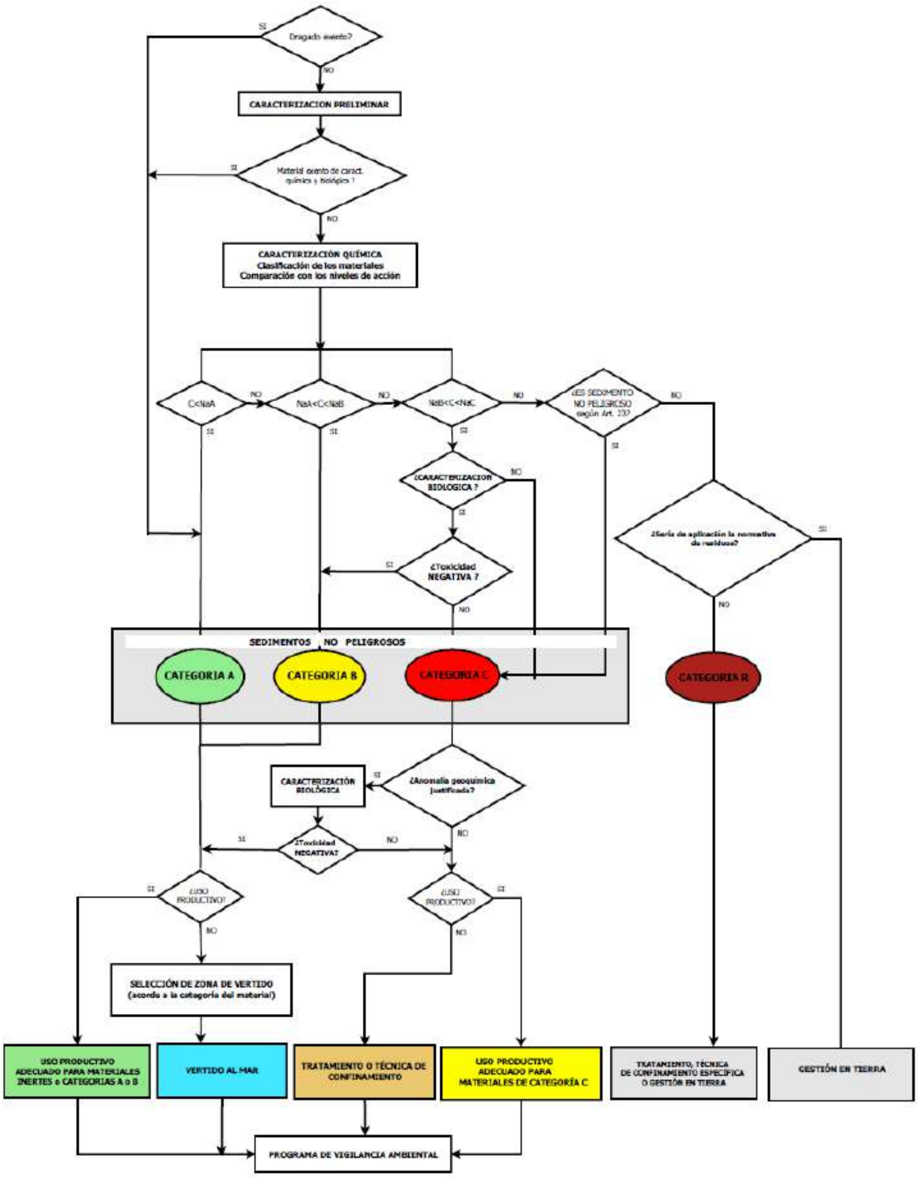

*Figure 3-5: Overview of the sediment assessment approach in Spain (CIEM, 2021).*

### 3.5.3 Approach

The current guidelines in Spain replace the original guidelines of CEDEX (1994). The guidelines are based on three conventions for the protection of the marine environment: the London protocol, the OSPAR Guidelines and the UNEP(DEPI)/MED guidelines.

A three-category action level-based approach is used, based on background values in Spanish coastal sediments, the anthropogenic load in dredged material, normalization techniques, validation of bioassay techniques and bioavailability of contaminants in dredged material from different locations.

*Table 3-7: Overview of the derivation of threshold values per component in the Spain sediment guidelines*

| **Spain: The framework and its underlying principles** | | |
|---|---|---|
| **Component** | **Derivation lower value** | **Derivation higher value** |
| **General framework** | Weight of evidence (WOE) approach | |
| **Organics, Metals, metalloids, organometallic and organic compounds** | CEDEX (1994) and local conditions | CEDEX (1994) and local conditions |

## 3.6 Japan

### 3.6.1 Governance structure

Japan has set extensive control of marine dredging activities. In Japan, the management of dredged sediment in marine environments is governed by the "*Law Relating to the Prevention of Marine Pollution and Maritime Disaster*" (1970, amended 2004[6]). This law implements the London Convention/Protocol on ocean dumping and requires that any sediment dumped at sea (e.g. from port dredging) meet strict criteria for hazardous substances and be placed only in designated disposal areas. Dredging projects (e.g. at commercial and fishing ports) must obtain a permit from the Ministry of the Environment (MOE) for ocean disposal, which is granted only after an EIA confirms the material will not harm the marine environment. In 2004, Japan amended this law to strengthen the permitting and assessment process for dredged material disposal at sea, in line with the London Protocol's requirements.

### 3.6.2 Framework and procedures

The law's Enforcement Ordinance (Cabinet Order) and associated MOE Ministerial Ordinances outline specific sediment quality standards ("judgement standards") that dredged material must satisfy for ocean dumping or use in marine land reclamation. For example, the system involves norms for background values for turbidity (2mg/l for fishing water, 5mg/L for open water).

When bottom sediment is dredged and brought up onto land, it becomes soil. There is a view that bottom sediment is part of soil, but there is no consensus on this. For dioxins, the bottom sediment environmental standard is 15% of the soil environmental standard. However, bottom sediment contamination poses a higher health risk than soil contamination (Hosomi, 2005) and the criteria for determining water bottom sediment are higher than the soil environmental standard. Japan has not yet defined a sediment quality standard specifically and instead adapts its existing surface water standards, to sediment quality requirements (Kusuma, 2022).

The framework (see Figure 3-6) follows a step-by-step approach where it matches the conservative guidelines for characterizing dredged materials.

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[6] https://faolex.fao.org/docs/pdf/jap73561.pdf

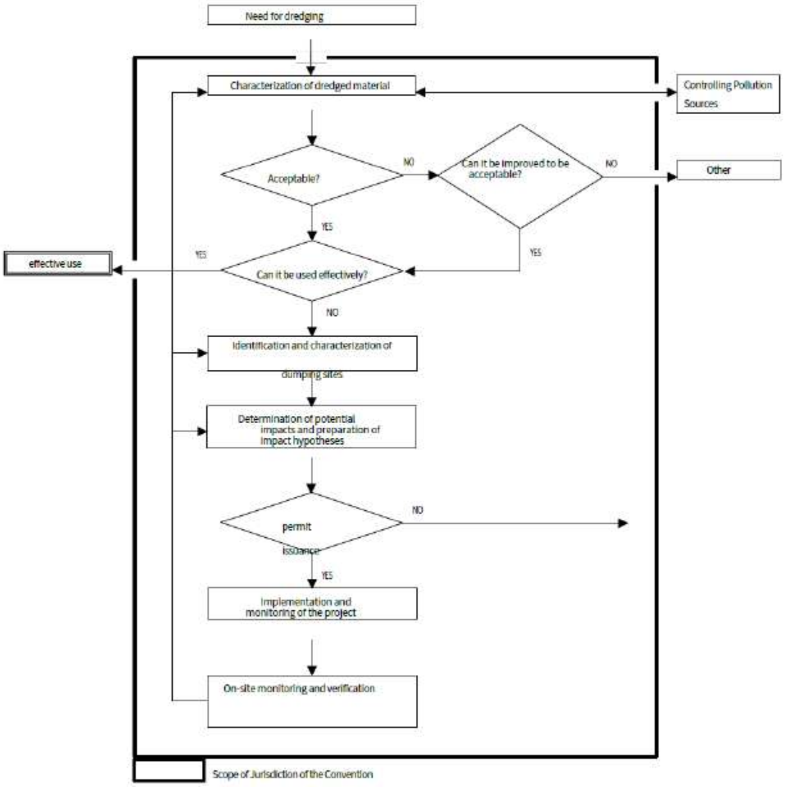

*Figure 3-6: Sediment quality assessment approach in Japan source: Technical guideline for marine disposal and effective utilization of dredged sediment, june 2006, port and harbour ministry of land infrastructure, transport tourism (translated)*

Environmental quality standards (EQS) for the living environment have established values for biochemical oxygen demand (BOD), chemical-oxygen demand (COD), dissolved oxygen (DO), and other variables. To prevent eutrophication, EQS for nitrogen and phosphorus were established for lakes and reservoirs. Provisional guideline values have also been set for sediments contaminated by mercury and polychlorinated biphenyl compounds (PCBs). Japan also introduced an EQS for dioxins in sediments at 150 pg-TEQ/g, under the Dioxin Law. If dioxin levels exceed this, authorities implement control measures to reduce health risks. Unlike PCBs/Hg (which demand removal), dioxin-contaminated sediment can be managed through a risk-based approach. Options include capping it in place or solidifying it (perform leaching tests).

### 3.6.3 Approach

Japan uses its aquatic life water quality criteria for assessing sediment. The Aquatic Life Water Quality Criteria (ALWQC) are calculated via:

- Screening chemicals based on ecological risk.
- Collecting toxicity data for aquatic species (e.g., fish, crustaceans).
- Selecting the lowest observed effect concentration (LOEC) for the most sensitive species.
- Applying safety factors to account for uncertainty.
- Validating criteria through site-specific monitoring and adaptive management.

Japan uses a safety factor of 10 on the water quality standard to derive safe concentrations in sediment. The water quality standards are documented using long-term dose-effect relations on fish and benthic species in Japan. Japan's vast experience in fisheries has led to a detailed ALWQC water quality framework. This complies with the equilibrium partitioning approach that calculates the dose effects using theoretical relations. For certain chemicals Japan uses organic carbon and pore water concentration to calculate the effective sediment standard (dioxins).

*Table 3-8: Overview of the derivation of threshold values per component in Japan sediment guidelines*

| **Japan: The framework and its underlying principles** | | | |
|---|---|---|---|
| **Component** | **Derivation lower value** | **Derivation higher value** | **Specific comments** |
| **General framework** | Equilibrium Partitioning | | |
| **Dioxins, mercury** | Equilibrium Partitioning | Equilibrium Partitioning | Additional |
| **Other Metals, metalloids, organometallic and organic compounds** | Equilibrium Partitioning | Equilibrium Partitioning | Based on the water quality standards |

# 3.7 Brazil

## 3.7.1 Governance structure

Brazil's dredged sediment management operates under a national environmental framework led by CONAMA (National Environmental Council) and enforced by IBAMA (Brazilian Institute of Environment and Renewable Natural Resources). CONAMA issues regulations including Resolution 454/2012, which replaced an earlier 2004 directive, to provide general guidelines and requirements for dredged material assessment. IBAMA (or delegated state environmental agencies) is responsible for the environmental licensing of dredging projects, ensuring compliance with these regulations. An Environmental Impact assessment (EIA) is conducted to evaluate the potential environmental impacts of the proposed dredging activity.

Every dredging project in Brazil requires an environmental license prior to execution. The project proponent must submit a Dredging Management Plan with detailed sediment characterization, proposed disposal or reuse sites, and an evaluation of potential environmental impacts with an EIA. This plan undergoes review by IBAMA, which may request further investigations before granting a permit. This assessment helps in identifying mitigation measures and is a prerequisite for obtaining environmental licenses. Under CONAMA Resolution 454/2012, dredging in “ports, bays, rivers, ponds, canals and sea areas” is regulated by a stepwise evaluation framework. CONAMA sets sediment quality guidelines (SQGs) that define two threshold levels (Level 1 and Level 2) for contaminants in sediments to be dredged. Level 1 represents the concentration below which there is a low likelihood of adverse effects on biota, whereas exceeding Level 2 indicates a high likelihood of adverse effects. These thresholds anchor the decision-making process during licensing.

If the dredged material is disposed on land, CONAMA Resolution 420/2009 is applicable. In such cases, dredged material may need to meet soil contamination criteria or be managed as waste, ensuring terrestrial disposal does not cause soil or groundwater pollution.

The Brazilian regulations consider beneficial reuse options and viability of disposal options. It directly copies these sentences from the guidelines of New Zealand/Australia. Dredging projects must comply with Brazilian environmental regulations. The CONAMA Resolution No. 454/2012[7] describes the procedure for dredging and reuse.

Overall, Brazil's governance structure emphasizes a permit-based, multi-institutional approach: CONAMA provides the legal framework, and IBAMA (with state agencies) implements it through licensing, oversight of sediment testing, and stipulating conditions for disposal or beneficial reuse.

## 3.7.2 Framework and procedures

Brazil follows a risk-based, stepwise assessment of dredged sediments. Prior to dredging, proponents must perform comprehensive sampling and analysis of the sediments. The required number of samples is proportional to the dredging project's size (e.g., 3 samples for volumes ≤25,000 m³, up to 30 samples for volumes ~2,000,000 m³) (Kanashiro, 2016). Sediment samples are analyzed for a suite of contaminants (heavy metals, organics like hydrocarbons and TBT, nutrients, etc.) and compared against the SQGs defined by CONAMA 454/2012. These are based on both American (ERL/ERM) and Canadian (TEL/PEL) criteria. If values exceed threshold 1, ecotoxicity or

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[7] https://www.icmbio.gov.br/cepsul/images/stories/legislacao/Resolucao/2012/res_conama_454_2012_materialserdragadoemaguasjurisdicionaisbrasileiras.pdf

bioaccumulation testing is required. If threshold level 2 is exceeded, the material should be sent to confined disposal units or licensed landfills. In some cases, disposal at marine landfills governed and legislated by the national government is permitted without prior investigation. The SQG of Brazil also describes testing methodology for lab and field testing of bioaccumulation. Lab testing is according to ASTM E1688. The field monitoring is done by collecting organisms from the dredged sediment and testing for maximum tolerance limits for human consumption, according to the regulations.

These empirically derived international guidelines were adopted as default criteria in Brazil, without considering the differences of the Brazilian coast and its biodiversity. Acknowledging this, Brazilian researchers have undertaken studies to develop site-specific SQVs better suited to tropical ecosystems and local species sensitivities. Brazil has developed site-specific guidelines for the following areas individually:

A. South-Eastern Brazil port zones (Estuarine systems of Santos and Paranaguá) (Choueri, 2009)
B. Ceará state
C. São Paulo state

The study conducted in zone A uses multivariate analysis on derived datasets from the origin locations. Basically, chemical concentrations associated to biological effects were established by identifying correlations among sediment chemical concentrations, toxicity endpoints, and benthic community descriptors through the application of Factor Analysis followed by PCA. Chemicals significantly correlated to biological effects were considered chemicals of concern and SQGs were derived for these contaminants. SQG-high of a given chemical of concern was determined by identifying its minimal measured concentration in the station where the factor (or factors) that associates this particular chemical as well as biological effects is relevant; above SQG-high all concentrations were related to negative biological effects in this study. The SQG-low of a chemical of concern was established through the identification of its highest measured concentration in the stations where the significant factors are not related to biological effects; below SQG-low, biological effects were not observed in this study. The “uncertainty area” (chemical levels in between SQGs-high and SQGs-low) included a range of concentrations in which biological effects were observed.

Study B focuses on the development of site-specific sediment quality values (SQVs) for Ceará state's semi-arid coastal regions, particularly Mucuripe Bay and Pecém Harbor. The researchers conducted comprehensive assessments before and after dredging activities, analyzing sediment samples for contaminants such as heavy metals (e.g., Cd, Cr, Cu, Zn), tributyltin (TBT), and hydrocarbons. They employed multivariate statistical methods to correlate chemical concentrations with observed toxic effects, enabling the derivation of SQVs tailored to local environmental conditions. The findings revealed that the international guidelines included in CONAMA 454/2012 were insufficiently protective for these tropical ecosystems, underscoring the necessity for localized benchmarks to effectively manage dredged materials and safeguard marine life.

Study C uses regional variations for calculating thresholds. Threshold values of metals were converted using geoaccumulation index (Igeo), which is used to classify the magnitude of the contamination related to an individual element, considering regional geological variations (Jiang et al., 2013, Kim et al., 2018). Igeo values were calculated as follows: $Igeo = \log_2 (C / 1.5 * B)$, where C is the threshold developed for a given metal, B is its background level, and the 1.5 factor is related to background variations due to geological processes. Median values of concentrations detected in insular zones of Ubatuba and São Sebastião cities (Cabras and Palmas islets, and Alcatrazes Archipelago) were used as a baseline for As, Cr, and Pb (Hoff et al., 2015), while median values of concentrations found within Laje de Santos Marine State Park were used as the baseline for Cu, Ni, and Zn (Moreira et al., 2017b). For Cd, they selected a background level determined in sediment cores collected at the industrial sites of Santos Estuarine System (Luiz-Silva et al., 2006). Igeo results are used to classify samples according to a qualitative scale of pollution ranging from 0 (unpolluted) to >5 (very strongly polluted), in which samples with Igeo above 1 show early signs of pollution (Müller, 1986). For PAHs, values were compared with reference values reported by Baumard et al. (1998) applied for samples from Mediterranean Coastal. The results of this study were compared with CONAMA Brazil standards and normalized as such.

### 3.7.3 Approach

Brazil’s sediment quality guidelines were initially derived from international benchmarks and scientific efforts have refined these values for local conditions. CONAMA 454/2012 Level 1 and 2 criteria were largely based on Sediment Quality Values (SQVs) from other jurisdictions, specifically the TEL/PEL and ERL/ERM approach that originated in Canada and the USA.

*Table 3-9: Overview of the derivation of threshold values per component in Brazil sediment guidelines*

| **Brazil: The framework and its underlying principles** | | | | |
|---|---|---|---|---|
| **Component** | **CONAMA (level 1 and 2) freshwater** | **CONAMA (level 1 and 2) saltwater** | **Case Study A** | **Case Study B** |
| **General framework** | TEL/PEL<br>ERL/ERM | TEL/PEL<br>ERL/ERM | Apparent effect threshold | Consensus-based evaluation |
| **As, Cr, Pb** | TEL/PEL (MacDonald et al., 1992) | ERL/ERM (Long et al.,1995) | AET, site specific | Site specific measurements (Hoff et al., 2015) |
| **Cu, Ni, Zn** | TEL/PEL (MacDonald et al., 1992) | | AET, site specific | WOE (Moreira et al., 2017) |
| **Cd** | TEL/ PEL (MacDonald et al., 1992) | | AET, site specific | Luiz Silva et al., 2006 |
| **PAH** | TEL/ PEL (MacDonald et al., 1992) | | AET, site specific | Baumard et al., 1998 |
| **Organics** | TEL/ PEL (MacDonald et al., 1992) | | AET, site specific | TEL/PEL (MacDonald, 1994) |
| **TBT** | TEL/ PEL (MacDonald et al., 1992) | SEDNET | | |
| **PCB** | | ERL/ERM (Long et al.,1995) | | |
| **Other Metals, metalloids, organometallic and organic compounds** | TEL/ PEL (MacDonald et al., 1992) | | AET, site specific | TEL/PEL (MacDonald, 1994) |

# 3.8 Chile

## 3.8.1 Governance structure

In Chile, the Ministry of Environment (Ministerio del Medio Ambiente, MMA) is the lead agency setting environmental policy and regulations. The Environmental Assessment Service (Servicio de Evaluación Ambiental, SEA) administers the Environmental Impact Assessment (EIA) system, which evaluates projects for environmental compliance. Oversight and enforcement are carried out by the Superintendence of Environment (Superintendencia del Medio Ambiente, SMA), which inspects projects and can sanction non-compliance. For marine activities, the marine authority of Chile (DIRECTEMAR) oversees permits for ocean disposal of dredged material in line with international conventions (e.g. the London Convention). Other agencies, such as the Ministry of Public Works or port authorities, may be involved in dredging operations, but environmental quality control remains under the MMA/SEA/SMA framework.

Chile does not have a specific national sediment quality standard. Environmental protection of water bodies is governed in a general sense by water quality standards and the EIA requirements, but no statute defines “clean” or “contaminated” sediment by numeric thresholds. Instead, any dredging-related pollution is managed under general prohibitions of environmental damage and the project-specific EIA stipulations. Relevant legislation includes the General Environmental Law 19300 and sectoral regulations (e.g., the Health Code for contaminants in seafood, which indirectly addresses human health concerns from sediment contaminants in edibles). Overall, governance relies on institutional oversight through EIA and enforcement, rather than a dedicated sediment quality law.

### 3.8.2 Framework and procedures

As mentioned earlier, Chile has no official sediment quality guidelines in law. A proposal to establish national sediment quality criteria was studied in the early 2000s, but it was never adopted as regulation. Consequently, Chile uses international benchmarks. Environmental authorities and consultants refer to guidelines from other jurisdictions, mostly the Canadian Sediment Quality Guidelines and the U.S. NOAA guidelines. In practice, the Canadian TEL/PEL threshold criteria and the U.S. NOAA ERL/ERM levels serve as the standard for Chile.

In the EIA for a dredging project, Chilean authorities expect a section on sediment quality. The proponent must discuss the contamination status and the disposal/application of dredged materials. In case that the TEL/PEL or ERL/ERM levels are exceeded the EIA might require that the dredged material should be disposed of on a waste facility on land instead of at sea. Conversely, if sediments meet the guideline levels for “no expected effects”, the material might be authorized for beneficial use or open-water disposal. Using this method Chile aims to protect the marine environment despite the absence of Chile-specific sediment quality criteria. However, since thresholds values based on data in North America are applied, it could be inaccurate for several substances in the environment of Chile.

### 3.8.3 Approach

Chile does apply the Equilibrium Partitioning approach to compare water quality criteria, but no sediment quality criteria are developed. Since mostly the Canadian Sediment Quality Guidelines and the U.S. NOAA guidelines are applied, the approach used to determine these guidelines can be found in Sections 3.2.3 and 3.3.3.

## 3.9 Peru

### 3.9.1 Governance structure

The governing party on dredging regulations in Peru is the General Secretariat, the Directorate of Water Resources Quality Management. Historically Peru lacked formal sediment quality guidelines. However, a recent policy plan (2021–2023) tasked the development of national Environmental Quality Standards for sediment. The first draft document has been started, where the Canadian sampling procedure for sediment sampling has been adopted. However, the evaluation of the sediments based on guidelines is not yet implemented. Currently Peru is in the process of establishing its own sediment quality criteria.

Article 34 provides that the management of waste from dredging works shall be carried out with the authorization of the National Institute of Natural Resources (INRENA), subject to a favorable technical opinion from the General Directorate of Environmental Health (DIGESA). This authorization is in addition to the authorization of dredging operations to be granted by the General Directorate of Captaincies and Coast Guards (DICAPI), in accordance with the provisions of Supreme Decree No. 028-DE-MGP, Regulation of the Law on Control and Surveillance of Maritime Activities, and the Single Ordered Text of Administrative Procedures (TUPA).

### 3.9.2 Framework and procedures

Peru is in the process of establishing its own guidelines. It intends to use a system similar to the Netherlands, by using a purpose-based system for water quality and soil quality. Based on the purpose and the receiving location, the norms that are in place will be determined. This includes:

- Soil environment[8] (general)
- Aquatic environment[9] (drinking water, recreation, industrial, natural etc.)

Despite this, most projects use the Canadian environmental quality standards as reference[10][11] by relying on screening values. Therefore, the framework and procedures can be found in Section 3.2.3.

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[8] https://sinia.minam.gob.pe/normas/aprueban-estandares-calidad-ambiental-eca-suelo-0

[9] https://sinia.minam.gob.pe/normas/aprueban-estandares-calidad-ambiental-eca-agua-establecen-disposiciones

[10] https://perulng.com/wp-content/uploads/2024/02/14.-Plan-de-Manejo-para-Contratistas-CMP-Obras-de-Dragado-para-la-Planta.pdf#:~:text=Las%20Normas%20de%20Calidad%20Ambiental,COMPA%C3%91%C3%8DA%20ser%C3%A1%20responsable%20de%20los

[11] https://cdn.www.gob.pe/uploads/document/file/7675508/6501104-1-protocolo-nacional-para-el-monitoreo-de-la-calidad-ambiental-de-

### 3.9.3 Approach

In the absence of local guidelines, Peruvian authorities and projects have relied on global standards. For example a study in 2022 applied the *Canadian Environmental Quality Guidelines* for sediments to assess heavy metal contamination in marine sediments along Peru's central coast (Loaiza, 2022). In practice, Peruvian environmental studies often reference benchmarks from the Canadian Council of Ministers of the Environment (CCME) or the U.S. EPA to interpret sediment data.

## 3.10 México

### 3.10.1 Governance structure

Mexico does not have a singular, published “sediment quality guideline” chart for dredged material, but it regulates dredging through its existing environmental laws and permits. Dredged sediments are managed under the “*General Law of Ecological Balance and Environmental Protection*” and the “*General Law for Prevention and Integral Management of Wastes*”, which require evaluating whether the material is hazardous. Prior to any dredging, an Environmental Impact Assessment is mandatory, and it must include analysis of the sediment to be removed. Authorities focus on whether dredged material contains contaminants at levels that could harm human health or ecosystems.

Request for dredging work needs to be carried out by the client. Based on a topographical survey the Deputy General Directorate of Oceanography, Hydrography and Meteorology will grant a permit for dredging. Next, an environmental permit is needed from the Secretaría de Medio Ambiente y Recursos Naturales (SEMARNAT), which is the Secretariat of Environment and Natural Resources (government department). For disposal at sea a permit from the secretary of health is needed. For disposal on land a permit from the “Presidencia Municipal”, meaning the municipal government office, is needed.[12]

### 3.10.2 Framework and procedures

If testing shows the sediment is not toxic and contaminant levels are low, it can be approved for beneficial reuse (e.g. land reclamation) or disposal at sea under permit. If the sediment is polluted above acceptable levels, it must be treated or disposed of in a confined facility. Notably, for ocean disposal, the Mexican Navy (SEMAR) and Health Ministry review results to ensure dumped material won’t cause marine pollution. This is in line with Mexico’s commitment to the London Convention, requiring that dredged material *“not cause direct or indirect harm to human health or the marine environment”.*

### 3.10.3 Approach

In practice, Mexico’s evaluations often reference global guidelines like its Latin American peers. While Mexico doesn’t formally promulgate NOAA or CCME numbers, environmental studies in the country may compare sediment data to NOAA’s ERL/ERM or CCME’s ISQG/PEL values to contextualize contamination levels. This helps determine safety for reuse. For instance, sediments with metals below NOAA’s ERL are generally considered safe, whereas those exceeding the ERM or other international criteria would be flagged for potential adverse effects. By using such benchmarks alongside its own norms, Mexico ensures its dredged sediment management aligns with global best practices even without an official domestic guideline chart.

## 3.11 Panama

Panama is included in this research due to its close geographical location and environmental similarities with Colombia, particularly its tropical coastal environments and major port activities. The governance structure is explained below, including the differences between the republic of Panama and the canal regulations.

In Panama there are two regimes of activities related to ports and maritime industries.  The Maritime Authority of Panama (Autoridad Marítima de Panamá, AMP) oversees dredging activities in the Republic’s coastal and marine

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sedimentos.pdf?v=1740405409#:~:text=partida%20el%20Decreto%20Supremo%20N%C2%B0,aprobaci%C3%B3n%20del%20ECA%20para%20sedimentos

[12] REQUERIMIENTOS DE ESTUDIOS Y PERMISOS PREVIOS A LA REALIZACIÓN DE UNA OBRA DE DRAGADO. 2013 https://www.semar.gob.mx/normateca/requerimientos_estudios_obra_dragado.pdf

waters (outside the Canal), including port areas. The Panama Canal Authority (Autoridad del Canal de Panamá, ACP) operates under a distinct constitutional and legal regime for the Panama Canal and adjacent waters.

In the absence of national guidelines, Panama likely adheres to international protocols for sediment assessment. The International Maritime Organization (IMO) provides guidelines for sampling and analyzing dredged material intended for disposal at sea, which Panama, as a member state, would follow. These guidelines assist in selecting substances of concern, field sampling, and conducting physical, chemical, and biological analyses.

Since both the AMP and ACP use international protocols for sediment quality assessments, the protocols and scientific approach that is used to define the SQGs are the same as the NOAA's Effects Range guidelines or Canadian's TEL/PEL. Since these are addressed in this report already (chapters 3.1 to 3.4), these will not be explained in detail again for Panama.

### 3.11.1 Panama republic

Panama is a party to global conventions on marine pollution (e.g. the London Convention), so it follows the Dredged Material Assessment Framework from those agreements when dumping at sea. This means Panamanian authorities apply internationally recommended testing and "action levels" to decide if dredged material is acceptable for ocean disposal or beneficial use.

The dredging project requirements for a contractor are established in the regulations under the administration of the Maritime Authority of Panama, the assignment procedure is stated in Resolution No. JD 019 - 2019. Also, Panama owns a special dredging commission whose function is to assess dredging proposals.

Panama does not have its own formal sediment quality guidelines for dredged material. The Environmental Impact assessment (EIA) for dredging must consider measures for both the dredging and disposal phases, as well as the details of the activity, including the use/purpose of the material. This (beneficial) use must then be approved by the Ministry of the Environment of the Republic. Dredging projects (e.g. Panama Canal or port dredging) include chemical analyses of sediments to measure contaminants like heavy metals and organics. The results are compared against global sediment quality criteria (such as NOAA's Effects Range guidelines or Canadian's TEL/PEL (Batista-Andrade et al., 2018).

In practice, if contaminant levels are below these recognized thresholds, the dredged sediment is considered relatively clean and may be safely reused. Common uses include land reclamation, construction fill, or habitat restoration. If contaminants exceed safe levels, the sediment is treated as contaminated. It may require confinement (e.g. in a contained disposal facility) or special disposal to prevent environmental harm.

It is important that since the Panama Maritime Authority is the custodian of the dredging material, it should grant the appropriate concession in accordance with Resolution JD-019-2019, which is its Concession Regulations.

### 3.11.2 Panama Canal

Dredging in the Panama Canal involves similar steps to other Panama tributaries via the republic. An environmental impact assessment (EIA) is needed to get granted dredging rights. The necessity of an EIA includes both the mining and construction industries as well.

Panamanian administrative regulations do not contemplate the beneficial use of dredged material through a special permit. Dredging activities are required to support the existing port activities in the country, and dredging itself is seen as a potentially polluting activity. Currently, beneficial use of dredged material besides land reclamation, landfills and spreading is not included within the regulations of Panama explicitly.

## 3.12 Costa Rica

Costa Rica is included in this research due to its close geographical location and environmental similarities with Colombia, such as tropical coastal ecosystems and biodiversity-rich marine environments.

Since Costa Rica adopts international SQGs, the underlaying scientific approach of the applied SQGs in Costa Rica is not described. Information about the approach used in international SQG can be found in previous chapters of this report.

### 3.12.1 Governance structure

Dredging projects must comply with Costa Rica's environmental regulations, which are overseen by the Ministry of Environment and Energy (MINAE). All dredging projects fall under MINAE, particularly through its National Technical Environmental Secretariat (SETENA) which reviews EIAs. The National System of Conservation Areas (SINAC) is involved in dredging projects that occur in or near protected marine areas. Lastly, the maritime and port authorities are the leading parties in port areas. These authorities ensure that environmental standards are met, and that biodiversity is protected. The environmental standards are based on international laws.

### 3.12.2 Framework and procedures

Costa Rica does not have specific laws about sediment quality. Instead, dredging is governed by general environmental and maritime laws. The Organic Law of the Environment (Law No. 7554) requires that any activity with significant environmental impact (including dredging) undergo an EIA and obtain an environmental license. In practice, this means any dredging project, whether in rivers, estuaries, or coastal waters, must be evaluated and authorized by SETENA before proceeding. Marine dredging and disposal activities are further constrained by Costa Rica’s commitments under international law.

Focusing specifically on sediment quality guidelines, Costa Rica adopts international guidelines such as those from the International Maritime Organization (IMO) or the London Convention, to assess and manage dredged materials. These standards provide comprehensive procedures for sampling, analysis, and evaluation to ensure environmental safety. In practice Costa Rica adopts mainly the Australia / New Zealand SQG criteria, but only screens on metal values.

# 4 Threshold values in The Netherlands, Florida (USA) and Brazil

## Chapter 4 – Executive summary

The threshold values in Sediment Quality Guidelines (SQGs) play a crucial role in conducting preliminary assessments of sediment quality. This chapter compares threshold values from several countries with geological and environmental conditions similar to those of Colombia. First, the countries that have relatively the same geological and environmental conditions are identified (Section 4.1). Then, the threshold values are compared to each other including a description of differences between countries (Section 4.2). For broader perspective and contrast, threshold values from The Netherlands are also included in the comparison. Lastly, substance toxicity levels and detection limits are explained (Sections 4.3 and 4.4), which provide more context and background information.

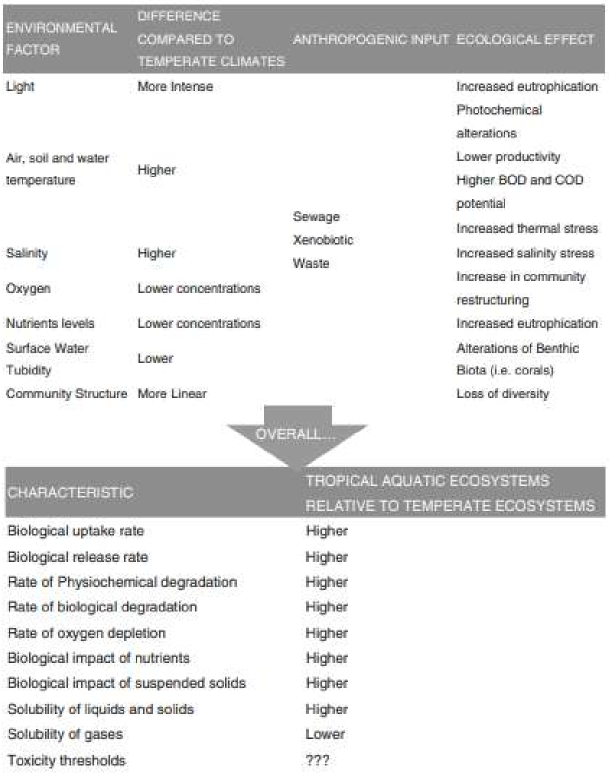
**The key takeaways of this chapter are:**

- Threshold values from other countries should and are compared to similar geographical and environmental conditions, as this impacts bioavailability of contaminants in sediments.
- Most guidelines use two levels, from which the lower level represents a concentration below which negative effects on sediment-dwelling organisms are expected to be negligible or rare. The higher level represents a higher concentration above which negative (biological) effects are expected to occur or have a high chance. The intermediate range between these levels is an uncertainty zone in which effects may occur. Thus, concentrations below level 1 are mostly considered safe, a concentration between levels 1 and 2 is alarming and should be investigated further. Above level 2 is considered a high chance of negative effects.
- Limits of Detection (LODs, also called Limits of Determination) represent the lowest concentration of a substance that can be reliably distinguished from zero by an analytical method. In sediment quality assessments LODs are essential because they define whether a contaminant is present at a level that can be confidently detected. The LOD can be a limiting factor in developing guidelines in Colombia. Since toxicity effects, especially in tropical areas, happen at very low concentrations, required LOD might not be possible in Colombia.
- Currently the sediment quality assessment guidelines and regulations are not focused on the beneficial use of dredged material, but rather on the prevention of negative impact on the environment. Some frameworks

mention the beneficial reuse of sediment, most regulations use the same criteria as when disposed directly in sea. In fact, most regulations do not distinguish further into the reuse of sediment on land compared to marine deposition.
- Element 2 selects threshold values from similar countries to Colombia. Compared to the Dutch and International guidelines we observe that threshold values for sediment reuse are required to be more strict in tropical countries compared to temperate climates, which is also highlighted in all the papers on this topic. However, the threshold values currently available in Southern America lack enough detail to be directly adopted by Colombia.
- If sediments are contaminated other practices exist to handle the contaminated material, such as:
    - Confined Aquatic Disposal (CAD)
    - Engineered Containment Facilities (ECF)
    - Confined Disposal Facilities (CDF)

## 4.1 Environmental conditions

To compare similar geological and environmental conditions to Colombia, one first needs to determine the most important conditions: bioavailability of contaminants in sediment. According to literature the following aspects are relevant:



| ENVIRONMENTAL FACTOR | DIFFERENCE COMPARED TO TEMPERATE CLIMATES | ANTHROPOGENIC INPUT | ECOLOGICAL EFFECT |
|---|---|---|---|
| Light | More Intense | Sewage<br>Xenobiotic<br>Waste | Increased eutrophication<br>Photochemical alterations |
| Air, soil and water temperature | Higher | | Lower productivity<br>Higher BOD and COD potential<br>Increased thermal stress |
| Salinity | Higher | | Increased salinity stress |
| Oxygen | Lower concentrations | | Increase in community restructuring |
| Nutrients levels | Lower concentrations | | Increased eutrophication |
| Surface Water Tubidity | Lower | | Alterations of Benthic Biota (i.e. corals) |
| Community Structure | More Linear | | Loss of diversity |

OVERALL...

| CHARACTERISTIC | TROPICAL AQUATIC ECOSYSTEMS RELATIVE TO TEMPERATE ECOSYSTEMS |
|---|---|
| Biological uptake rate | Higher |
| Biological release rate | Higher |
| Rate of Physiochemical degradation | Higher |
| Rate of biological degradation | Higher |
| Rate of oxygen depletion | Higher |
| Biological impact of nutrients | Higher |
| Biological impact of suspended solids | Higher |
| Solubility of liquids and solids | Higher |
| Solubility of gases | Lower |
| Toxicity thresholds | ??? |

*Figure 4-1: Challenges for comparing sediment guidelines (Kwok et al., 2014)*

The dose–response characteristics of sediment-bound chemicals in tropical aquatic environments have not been well studied, yet regulatory agencies in many countries have adopted SQGs, particularly those from North America, without adequate consideration of environmental differences that might make such guidelines more or less relevant (Kwok et al., 2014)

Based on the evaluated factors, the regions Ceará (Brazil), Florida (USA) and Sao Paulo (Brazil) exhibit the most environmental similarities to Colombia while having their own sediment quality framework available. All three regions are characterized by:

- **High Light Availability:** Due to their tropical locations.
- **Warm Air, Soil, and Water Temperatures**: Consistent with tropical climates such as colombia

- **Similar Salinity Levels**: Approximately 34–35 PSU.
- **High Oxygen Availability**: Typical of well-mixed tropical coastal waters such as colombia's.
- **Elevated Nutrient Levels**: Due to riverine inputs and coastal upwelling.
- **High Surface Water Turbidity**: Resulting from river discharge and coastal runoff.
- **Diverse Ecological Communities**: Including mangroves, coral reefs, and estuaries.

This makes sense, since it aligns with the knowledge gap of Brazilian scientists regarding tropical waters. The USA-guidelines are specifically designed for Florida originally and adopted by Canada, Peru and Chile to a certain extent. Although the guidelines of Florida are applied internationally by many different countries, the guidelines are based on North American historical data. This data was then validated on site-specific data in Florida to test the suitability of the guideline. For comparison's sake these values are also incorporated in the next chapter. This makes it possible to derive Colombia threshold values based on North Caribean datasets and East Atlantic Brazilian site-specific data.

*Figure 4-2: Topographical proximity of the comparison locations to Colombia*

Reflecting on the choose of geological origin., one can argue that climatic conditions along the equator are not always constant and Colombia's Pacific Ocean locations are not represented in the comparison. While this is true, none of the countries on our list has a better fit in terms of climate and geological origin.

On other hand, to determine the SQGs for the regions of Ceará and Sao Paulo (Brazil) the criteria included results from bioessays with the burrowing amphipod *Tiburonella viscana* (Moreira et al. (2021; 2022), which is also present in coastal ecosystems of Colombia. Meanwhile, the SQGs for Florida were based on information of 120 publications, out of which the 35% were from studies conducted in the southeastern and Gulf of Mexico regions of the United States (MacDonald, 1994).

## 4.2 Threshold values comparison

As explained earlier, this chapter compares threshold values from several countries with geological and environmental conditions like those of Colombia. The threshold values from The Netherlands are also included in the comparison.

Most SQGs make use of sediment quality threshold values to make an initial assessment. Most guidelines use two levels, from which the lower level represents a concentration below which negative effects on sediment-dwelling organisms are expected to be negligible or rare. The higher level represents a higher concentration above which negative (biological) effects are expected to occur or have a high chance. The intermediate range between these levels is an uncertainty zone in which effects may occur. Thus, concentrations below level 1 are mostly considered safe, a concentration between levels 1 and 2 is alarming and should be investigated further. Above level 2 is considered a high chance of negative effects. The exact definition of these levels differs between SQGs, but they are comparable. For example, the TEL/PEL could also be expressed as level 1 and 2. The same goes for the U.S. NOAA ERL/ERM levels and the SQGV/SQGV-High values in the Australian/New Zealand guidelines. In the table below these values are described as Level 1 and Level 2 for the sake of comparison. The units differ between substances and are specified in the table as either mg/kg or µg/kg. Table 4-1 -gives an overview of the threshold values (Level 1 & 2) for the countries of interest.

*Table 4--1: Comparison of threshold values for categorizing the beneficial application of marine dredged sediment for similar geological origins to Colombia. Additionally, the Netherlands is added for comparison.*

| **SQAGs** | | | **Florida (USA)** | **Ceará, Brazil** | **Sao Paulo, Brazil** | **The Netherlands: Surface water** | **Florida (USA)** | **Ceará, Brazil** | **Sao Paulo, Brazil** | **The Netherlands: Surface water** |
|---|---|---|---|---|---|---|---|---|---|---|
| **Category** | **Substance** | | **Level 1** | **Level 1** | **Level 1** | **Level 1** | **Level 2** | **Level 2** | **Level 2** | **Level 2** |
| **Trace metals and metalloids (SQAGs in mg/kg)** | Arsenic | As | 7.24 | | 8.10 | 29.00 | 41.60 | | | 85.00 |
| | Cadmium | Cd | 0.68 | | 0.40 | 4.00 | 4.21 | 1.10 | 0.80 | 14.00 |
| | Chromium | Cr | 52.30 | 44.50 | 24.90 | 120.00 | 160.00 | 59.00 | 31.50 | 380.00 |
| | Copper | Cu | 18.70 | 15.60 | 9.40 | 60.00 | 108.00 | 19.40 | 17.60 | 190.00 |
| | Lead | Pb | 30.20 | 24.00 | 6.70 | 110.00 | 112.00 | 47.30 | 16.60 | 580.00 |
| | Mercury | Hg | 0.13 | 0.04[a] | 0.20 | 1.20 | 0.70 | 0.07[a] | | 10.00 |
| | Nickel | Ni | 15.90 | 20.20 | 10.80 | 45.00 | 42.80 | 24.10 | 14.60 | 210.00 |
| | Silver | Ag | 0.73 | | | | 1.77 | | | |
| | Zinc | Zn | 124.00 | 543.00 | 52.60 | 365.00 | 271.00 | 603.00 | 190.20 | 2000.00 |
| | Antimony | Sb | | | | | | | | 15.00 |
| | Cobalt | Co | | | | | | | | 240.00 |
| | Molybdenum | Mo | | | | | | | | 200.00 |
| **Other Inorganic substances (mg/kg)** | Cyanide (free) | | | | | | | | | 20.00 |
| | Cyanide (Complex) | | | | | | | | | 50.00 |
| | Thiocyanates | | | | | | | | | 20.00 |
| **Organometallics (µg/kg dry weight, 1% OC)** | Tributyltin (as Tin) | TBT | | | | 115.00 | | 189.60 | | |
| **Organic compounds** | | | | | | | | | | |
| ***Aromatics (mg/kg)*** | Benzene | | | | | | | | | 1.00 |
| | Ethylbenzene | | | | | | | | | 50.00 |
| | Toluene | | | | | | | | | 130.00 |
| | Xylenes (total) | | | | | | | | | 25.00 |
| | Styrene (vinylbenzene) | | | | | | | | | 100.00 |
| | Fenol | | | | | | | | | 40.00 |
| | Cresols (total) | | | | | | | | | 5.00 |
| ***Polyclinic Aromatic Hydrocarbons (PAHs, in µg/kg)*** | Acenaphthene | | 6.71 | | | | 88.90 | | | |
| | Acenaphthylene | | 5.87 | | | | 128.00 | | | |
| | Anthracene | | 46.90 | | | | 245.00 | | | |
| | Fluorene | | 21.20 | | | | 144.00 | | | |
| | Naphthalene | | 34.60 | | | | 391.00 | | | |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 2-Methylnapthalene | | 20.20 | | | | 201.00 | | | |
| | Phenanthrene | | 86.70 | | | | 544.00 | | | |
| | Total Low Molecular Weight PAHs | | 312.00 | | | | 1442.00 | | | |
| | Benz(a)anthracene | | 74.80 | | | | 693.00 | | | |
| | Benzo(a)pyrene | | 88.80 | | | | 763.00 | | | |
| | Chrysene | | 108.00 | | | | 846.00 | | | |
| | Dibenz(a,h)anthracene | | 6.22 | | | | 135.00 | | | |
| | Fluoranthene | | 113.00 | | | | 1494.00 | | | |
| | Pyrene | | 153.00 | | | | 1398.00 | | | |
| | Total High Molecular Weight PAHs* | | 655.00 | | | | 6676.00 | | | |
| | Total PAHs* | PAH | 1684.00 | 925.70 | 182.60 | 8000.00 | 16770.00 | 1957.10 | 1138.60 | 40000.00 |
| ***Volatyle Chlorinated hydrocarbons (mg/kg)*** | Monochloroethene (vinyl chloride) | | | | | | | | | 0.10 |
| | dichloromethane | | | | | | | | | 10.00 |
| | 1,1-dichloroethane | | | | | | | | | 15.00 |
| | 1,2-dichloroethane | | | | | | | | | 4.00 |
| | 1,1-dichloroethylene | | | | | | | | | 0.30 |
| | 1,2-dichloroethylene (sum) | | | | | | | | | 1.00 |
| | dichloropropanes (sum) | | | | | | | | | 2.00 |
| | trichloromethane (chloroform) | | | | | | | | | 10.00 |
| | 1,1,1-trichloroethane | | | | | | | | | 15.00 |
| | 1,1,2-trichloroethane | | | | | | | | | 10.00 |
| | trichloroethylene (Tri) | | | | | | | | | 60.00 |
| | tetrachloromethane (Tetra) | | | | | | | | | 1.00 |
| | tetrachloroethylene (Per) | | | | | | | | | 4.00 |
| ***Chlorobenzenes (mg/kg)*** | Hexachlorobenzene | | | | | 0.044 | | | | |
| | Chlorobenzenes (sum) | | | | | | | | | 30.00 |
| ***Chlorophenols (mg/kg)*** | Pentachlorophenol | | | | | | | | | 5.00 |
| | Total chlorophenols | | | | | | | | | 10.00 |
| ***PCB's*** | Total Polychlorinated Biphenyls (PCBs; SQAGs in µg/kg) | PCBs | 21.60 | | | 139.00 | 189.00 | | | 1000.00 |
| ***Other chlorinated hydrocarbons*** | monochloroanilines (sum) *(mg/kg)* | | | | | | | | | 50.00 |
| | Chloronaphthalene (sum) *(mg/kg)* | | | | | | | | | 10.00 |
| | PCDD/PCDF (ng tox eq/kg)* | | | | | | | | | |
| ***Pesticides ( µg/kg)*** | Chlordane | | 2.26 | | | | 4.79 | | | 4000 |
| | Dieldrin | | 0.72 | | | | 4.30 | | | |
| | HCH (Alfa, Beta, Gamma) | | | | | | | | | 2000 |
| | Lindane | | 0.32 | | | | 0.99 | | | |

| | | | | | |
|---|---|---|---|---|---|
| | p,p'-DDD | 1.22 | | 7.81 | |
| | p,p'-DDE | 2.07 | | 374.00 | |
| | p,p'-DDT | 1.19 | | 4.77 | |
| | Total DDT | 3.89 | 20.00 | 51.70 | 4000 |
| | drins (sum) | | | | 4000 |
| | α-endosulfan | | | | 4000 |
| | Heptachlor | | | | 4000 |
| | Heptachlor epoxide | | | | 4000 |
| | MCPA | | | | 4000 |
| | Atrazine | | | | 6000 |
| | Carbaryl | | | | 5000 |
| | Carbofuran | | | | 2000 |
| ***Other substances (mg/kg)*** | Asbestos | | 100.00 | | 100.00 |
| | Cyclohexanone | | | | 45.00 |
| | **Phthalates** | | | | 60.00 |
| | Bis(2-ethylhexyl) phthalate (DEHP), type of Phthalates | 182.00 | | 2647.00 | |
| | Mineral oil | | 1250.00 | | 5000.00 |
| | Pyridine | | | | 0.50 |
| | Tetrahydrofuran | | | | 2.00 |
| | tetrahydrothiophene | | | | 90.00 |
| | tribromomethane (bromoform) | | | | 75 |

a- Normalised for <63um

## 4.3 Considerations on differences among the thresholds of heavy metals at each level

-As table 4-1 shows considerable differences among the values of substances at each level, the question arises as to why the thresholds for some substances are much lower than others. In this regard, below are described some considerations for heavy metals, that allow us to understand why some of them are more toxic than others.

Several heavy metals including cobalt (Co), copper (Cu), zinc (Zn), and molybdenum (Mo), are crucial for metabolic activity of plants and animals at low concentrations and hence are considered micronutrients. Meanwhile, other heavy metals including chromium (Cr), lead (Pb), arsenic (As), cadmium (Cd), mercury (Hg), are described as the most problematic heavy metals (Rahman & Singh, 2019). The toxicity of these substances depends on their type of chemical species present in a determined environment (speciation), which is dependent on the pH, redox conditions and organic matter content (chemical species can be inorganic or organic). The electric charge of each chemical species determines how mobile it is in relation to the charges of soil/sediment binding particles (clay, organic matter, and iron/aluminum hydr(oxides)) (Blume et al., 2016). Additionally, the molecular structure of each chemical species determines its toxicity level (Barratt, 2000; University of California, 1986).

Inorganic compounds of As, Cr and Cd have been classified as group-1 of human carcinogens by the International Agency for Research on Cancer, whereas Hg and Pb are known as potent neurotoxicants, and As is also reported to possess some neurotoxic effects (Rahman & Singh, 2019). According to heavy metal thresholds in different SQGs, like those of the table above, concentration-wise Cd and Hg are of higher concern, followed by As.

Some concerns have been raised about the possibility of toxic concentrations of certain elements being transported from plants to higher strata of the food chain (Peralta-Videa et al., 2009). Chromium usually exists as Cr(III) ($Cr^{3+}$) in anaerobic environments, and as Cr(VI) (oxyanions) in aerobic environments of neutral-alkaline pH (Ball & Izbicki, 2004; Rahman & Singh, 2019). Thereby, whereas much Cr(III) is immobilized by soil/sediment binding particles, Cr (VI) is highly mobile and highly reactive with other elements, and hence it is 100 times more toxic and 1000 times more mutagenic (Nriagu & Nieboer, 1988; Ball & Izbicki, 2004; Rahman & Singh, 2019). In mammal metabolism, much Cr(VI) is reduced to Cr(III) in saliva and gastric juice, hence the cases of Cr poisoning in humans are limited, and

Cr(VI) toxicity is much more common by prolonged exposure (Rahman & Singh, 2019). Furthermore, whereas Cr(III) is not required in plant metabolism, it is a micronutrient for animal metabolism, acting as an integral component of the glucose tolerance factor, and as a cofactor for the hormone insulin (Mertz, 1975; Spears, 2025).

Concerning Pb, which is mainly present as the species Pb(II), it is less mobile than other heavy metals, and upon its plant-uptake a major proportion remains in the root, but at high concentration it can enter the vascular system and be transported to other tissues. Pb can also be present in organic compounds such as triethyl lead and tetraethyl lead, and these organic forms are reported to be more toxic (Peralta-Videa et al., 2009).

Regarding Arsenic, it dominantly exists as As(V) (arsenate) in aerobic environments, and as As(III) (arsenite) in anaerobic ones, the latter being more mobile and 25–60 times more toxic (Peralta-Videa et al., 2009; Rahman & Singh, 2019). At low pH As is found complexed with iron whereas at high pH it is mostly bound to calcium, and the presence of iron and manganese oxides also increases As mobility and availability in soil (Peralta-Videa et al., 2009). On the other hand, in plant tissue As is a strong phytotoxic (Rahman & Singh, 2019); As(V) is reduced to As(III) and/or bio-transformed to less toxic organic compounds such as DMA, MMA, or as As(III) complexed with thiol groups (Peralta-Videa et al., 2009).

Cadmium is mainly present as Cd(II) ($Cd^{+2}$), and unlike many heavy metals in soil that are mainly bound to binding particles, Cd predominantly exists as $CdCO_3$ at high pH, and as CdS at low pH), and these complexes can readily transform to accessible form for plant uptake. In freshwater Cd is present as soluble forms $Cd^{+2}$, $Cd(OH)_2$, and $CdCO_3$; however, with increase of salinity in water, various chlorine species dominate (Rahman & Singh, 2019). This heavy metal is generally not phytotoxic, but its high concentration in plants represents delirious causes for their consumers. However, external factors such as Fe concentration can reduce the uptake of Cd (Peralta-Videa et al., 2009). In animals, Cd and Pb can interfere with Ca metabolism, and compete with Zn replacing it in proteins and enzymes. For instance, Cd(II) is likely a potent and competitive inhibitor of the enzyme phosphatase 1 (Rahman & Singh, 2019; Balali-Mood et al., 2021).

Mercury (Hg) mainly exists as the highly volatile elemental form (Hg(0)), as inorganic forms (Hg(I) and Hg(II), and as organic forms like methylmercury, ethylmercury and phenylmercury. Human exposure to Hg occurs mainly through inhalation of Hg(0) vapor and ingestion of organic forms. Inhaled Hg(0) enters the respiratory system and oxidize to Hg(II) (using catalase) on entry to the bloodstream. Organic Hg forms are 10 times more toxic than the inorganic ones and these organic forms are produced by some sulfate- and iron-reducing microorganisms, under anaerobic conditions (Rahman & Singh, 2019; Balali-Mood et al., 2021). Bearing in mind the accumulation of these organic forms throughout the food chain in aquatic ecosystems, consumption of fish is the chief source of human exposure (Rahman & Singh, 2019).

From the information above we can deduce that besides considering the heavy-metal contents in dredged sediments, their beneficial uses on land imply a risk of Cr toxicity, which is more likely in aerobic environments. By contrast, As and Hg toxicities are more likely in applications of dredged sediments in water bodies; and as Hg and Pb organic forms are more toxic, the higher the organic matter content in sediments, the higher the risk with these metals. Concerning Cd, regardless of the redox conditions it is highly mobile and toxic.

## 4.4 Detection limits

Limits of Detection (LODs, also called Limits of Determination) represent the lowest concentration of a substance that can be reliably distinguished from zero by an analytical method. In sediment quality assessments LODs are essential because they define whether a contaminant is present at a level that can be confidently detected. This has direct implications for environmental risk assessments, compliance with regulatory standards, and comparisons to sediment quality guidelines. Without clear knowledge of LODs, interpreting low-concentration data becomes uncertain and potentially misleading.

However, LODs have several important limitations. A result reported as "<LOD" does not mean the substance is absent, it means it was present below the detection capabilities of the method used. LODs are not universal values; they vary depending on the instrument, sample type, and laboratory protocol. This can lead to inconsistencies between datasets or studies unless the methods are standardized and clearly reported.

The determination of LODs also depends on both the analytical method and the type of substance:

*Table 4-2: Overview of typical analytical methods associated per contaminant*

| Contaminant | Typical Analytical Method(s) (Full Name) |
|---|---|
| **Ag, Al, As, Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb, Sb, Se, V, Zn** | ICP-MS, ICP-OES, AAS (Inductively Coupled Plasma Mass Spectrometry, Inductively Coupled Plasma Optical Emission Spectrometry, Atomic Absorption Spectroscopy) |
| **Mercury, Methylmercury** | ICP-MS, AAS (with cold vapor), sometimes GC-MS for methylmercury |
| **Tributyltin (TBT)** | GC-MS, GC-MS/MS (Gas Chromatography-Mass Spectrometry, Gas Chromatography-Tandem Mass Spectrometry) |
| **PAHs, TPHs, BTEX** | GC-MS, GC-MS/MS, GC-FID (Gas Chromatography-Flame Ionization Detector) |
| **Phenols, Phthalates, Carbamates, Herbicides/Fungicides** | LC-MS/MS, HPLC (Liquid Chromatography-Tandem Mass Spectrometry, High Performance Liquid Chromatography) |
| **Organochlorine/Organophosphate Pesticides, PCBs, Dioxins** | GC-MS, GC-MS/MS, GC-ECD (Gas Chromatography-Electron Capture Detector) |
| **Synthetic Pyrethroids** | LC-MS/MS, GC-MS |
| **Total Organic Carbon (TOC)** | TOC Analyzer, Combustion |
| **Ammonia, $NO_x$, Orthophosphate, Cyanide** | Colorimetry, Ion Chromatography |

These categories of methods differ in sensitivity and the range of substances they can target. Understanding these broad distinctions is important when interpreting results or comparing data across studies. Appendix A shows typical LODs that are easily achievable using typical test methods (Simpson et al., 2013).

## 4.5 Framework

A central insight from this report is the significant diversity in how countries regulate the beneficial reuse of dredged sediments. The comparison between frameworks revealed that no one-size-fits-all model exists. Each country approaches sediment management through its own combination of risk perception, policy maturity, and ecosystem sensitivity. For instance, a country like The Netherlands uses a purpose-driven framework, while the USA and Canada use a step-by-step approach using screening values, and Mexico relies on water quality criteria as proxies for sediment standards. Additionally, as seen in Brazil, international SQGs are not fully suitable for locations in Brazil, site-specific SQGs are definitely more suitable. These differences in applied frameworks and regulations show that no one-size-fits-all model exists.

Experiences during this analysis reinforced that threshold values alone do not determine safety or sustainability. The methodology behind the threshold, such as equilibrium partitioning or bioassay-based validation, is just as important. This underlines the importance for Colombia not only to choose appropriate threshold values, but also to establish a clear framework including a transparent justification process.

Another key takeaway was the practical value of assessment systems. Countries like Canada, the USA, and Australia / New Zeeland use frameworks that begin with conservative screening and escalate to more detailed analyses only when needed (step-by-step approach). Although this framework could be effective if the screening values (SQGs) are not exceeded, they require additional resources when the SQGs are exceeded. Especially for beneficial reuse, this framework can involve complex project-specific studies to assess if the sediment can be applied for that specific purpose. Therefore, a purpose-driven framework and regulations might be more suitable for reuse in nature-based solutions. To do this, it is advisable to evaluate more parameters than is currently done for international screening values. The parameters should be studied using country-specific data and incorporated into the official framework and regulations. Implementing screening values for each specific application will optimize resources for projects in the future and support beneficial reuse of sediment.

All research approaches discussed in this report can be applied in any country, depending on budget and scientific/technical criteria. Here, the most common approach is weight of evidence, which combines the available data regionally and internationally with specific testing in the target country. Such an approach can be setup from the start. However, key considerations here are that not all background data and testing is applicable to the geological origin of Colombia, which is why we mainly focused on a specific set of countries in this report. Moreover, the weight of evidence approach should still follow strict distinctions between different types of reuse applications, since these also have different type of implications to the environment. Specifying threshold values per reuse category allows for more reuse options and reduces the administrative load for conducting environmental declaration after a single screening value has been surpassed.

Another important consideration in adapting the framework is the distinction between salt and freshwater bodies. General screening values for saltwater bodies may not be applicable to freshwater ecosystem in terms of magnitude and substance types. In the Netherlands this distinction not based on chloride content but on predefined geometrical boundaries.

Finally the limit of detection values in this document are of great concern to Colombian parties responsible for accreditation of laboratories that should conduct value screening tests. Setting minimal accreditation values can mainly be based on appendix A, since these a typical equipment types used globally. If this turns out to be unrealistic, because of for example high investments costs, the next step is to zoom into the new Colombian regulations. The lowest value present in the regulations can be adopted to accredit instead of the international LOD value.

## 4.6 Beneficial use

Currently the sediment quality assessment guidelines and regulations are not focused on the beneficial use of dredged material, but rather on the prevention of negative impact on the environment. Some frameworks mention the beneficial reuse of sediment, most regulations use the same criteria as when disposed directly in sea, and land-based uses can be considered under the purview of regulators in Australia and New Zealand. The sediment should be non-toxic and contaminant levels should be low. The regulations of the states Florida and North Carolina include that when dredged sediment is used beneficially, more fine-graded sediment (finer than sand) is allowed when compared to standard beach nourishment projects.

Although beneficial reuse is mentioned in several frameworks, more study should be conducted, with specific attention on the inclusion of beneficial reuse into the existing frameworks and regulations. Beneficial reuse is not encouraged by having specific regulations (where applicable and safe to do so) to stimulate this activity. Especially purpose-driven frameworks and regulations offer more possibilities in terms of beneficial reuse of sediments. Applying this framework allows for different screening parameters, designed for specific (beneficial) end-use.

reuse of sediment.

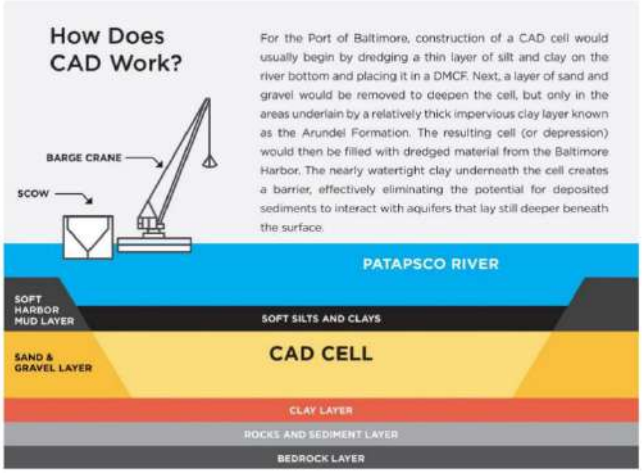
## 4.7 Contaminated sediment

Once the classification system for sediment is in place and sediment is classified as contaminated, most regulations do not offer any large scale options for reuse. This chapter discusses shortly the most common practices handling with contaminated sediments globally:

1. Confined Aquatic Disposal (CAD)

Confined Aquatic Disposal involves placing contaminated dredged sediment into a natural or man-made depression on the seafloor or riverbed. Once filled, the disposal area is covered (or “capped”) with clean material (typically sand or gravel) to isolate the contamination from the surrounding water. This method is widely used in countries like Canada and the United States, especially when suitable underwater depressions are available near dredging sites. CAD is considered cost-effective and minimally disruptive if well-designed, and it is often used in conjunction with long-term monitoring to ensure the cap remains stable and protective over time.

*Figure 4-4: left: CAD-cell in Port of New Bedford, Massachusetts*



For the Port of Baltimore, construction of a CAD cell would usually begin by dredging a thin layer of silt and clay on the river bottom and placing it in a DMCF. Next, a layer of sand and gravel would be removed to deepen the cell, but only in the areas underlain by a relatively thick impervious clay layer known as the Arundel Formation. The resulting cell (or depression) would then be filled with dredged material from the Baltimore Harbor. The nearly watertight clay underneath the cell creates a barrier, effectively eliminating the potential for deposited sediments to interact with aquifers that lay still deeper beneath the surface.

*Figure 4-3: CAD-cell explained.*

2. Engineered Containment Facilities (ECF)

An Engineered Containment Facility is a purpose-built structure, often made with steel sheet piling or concrete walls, designed to physically isolate contaminated sediments in the aquatic environment. Unlike CAD, which uses natural seabed features, an ECF is entirely human-made and may be constructed as an aboveground cell, a nearshore basin, or even a permanent in-water enclosure. One notable example is the Randle Reef project in Canada, where a large steel-walled containment area was built in a polluted harbor and filled with dredged material, which was then capped and integrated into future port development. ECFs are chosen when contamination is highly concentrated or mobile, and when more control over the disposal environment is needed.

*Figure 4-5: ECF facility at Rendal Reef, Canada: https://randlereef.ca/*

### 3. Confined Disposal Facilities (CDF)

Confined Disposal Facilities are large diked areas—on land, nearshore, or as artificial islands, used to contain dredged sediments, particularly those that are moderately to heavily contaminated. CDFs are widely used in the Netherlands, USA, and other countries with active dredging programs. Sediment is transported by pipeline or barge into the facility, where it is allowed to settle and dewater. Clean water is drained off and often treated, while the solids remain securely contained. Over time, these facilities may be capped and turned into usable land for ports, parks, or industrial zones. Well-known examples include De Slufter and IJsseloog in the Netherlands, which are massive structures designed to store millions of cubic meters of contaminated sediment while preventing environmental release.

*Figure 4-6: Baggerdepot the Slufter near Rotterdam, The Netherlands: https://www.portofrotterdam.com/nl/bouwen-aan-de-haven/lopende-projecten/de-slufter*

The preference for any of the three containment facilities is dependent on the owners space on land, cost-benefit analysis, water quality standards and local water depths among other factors.

Alternatively to containment of the contaminated dredging materials, one may consider cleaning the material for beneficial use. The Netherlands is one of the only countries in the world that is involved in such activities due to the high price of construction materials. Methods such as a) fine separation by gravity b) separation by hydrocyclonage c) thermal cleaning or d) immobilization are considered over the years. For now only thermal cleaning is considered economically feasible for large scale processing, but requires a lot of energy to deploy.

# 5 Discussion and experiences

## 5.1 Framework

A central insight from this report is the significant diversity in how countries regulate the beneficial reuse of dredged sediments. The comparison between frameworks revealed that no one-size-fits-all model exists. Each country approaches sediment management through its own combination of risk perception, policy maturity, and ecosystem sensitivity. For instance, a country like The Netherlands uses a purpose-driven framework, while the USA and Canada use a step-by-step approach using screening values, and Mexico relies on water quality criteria as proxies for sediment standards. Additionally, as seen in Brazil, international SQGs are not fully suitable for locations in Brazil, site-specific SQGs are definitely more suitable. These differences in applied frameworks and regulations show that no one-size-fits-all model exists.

Experiences during this analysis reinforced that threshold values alone do not determine safety or sustainability. The methodology behind the threshold, such as equilibrium partitioning or bioassay-based validation, is just as important. This underlines the importance for Colombia not only to choose appropriate threshold values, but also to establish a clear framework including a transparent justification process.

Another key takeaway was the practical value of assessment systems. Countries like Canada, the USA, and Australia / New Zeeland use frameworks that begin with conservative screening and escalate to more detailed analyses only when needed (step-by-step approach). Although this framework could be effective if the screening values (SQGs) are not exceeded, they require additional resources when the SQGs are exceeded. Especially for beneficial reuse, this framework can involve complex project-specific studies to assess if the sediment can be applied for that specific purpose. Therefore, a purpose-driven framework and regulations might be more suitable for reuse in nature-based solutions. To do this, it is advisable to evaluate more parameters than is currently done for international screening values. The parameters should be studied using country-specific data and incorporated into the official framework and regulations. Implementing screening values for each specific application will optimize resources for projects in the future and support beneficial reuse of sediment.

## 5.2 Beneficial use

Currently the sediment quality assessment guidelines and regulations are not focused on the beneficial use of dredged material, but rather on the prevention of negative impact on the environment. Some frameworks mention the beneficial reuse of sediment, most regulations use the same criteria as when disposed directly in sea, and land-based uses can be considered under the purview of regulators in Australia and New Zealand. The sediment should be non-toxic and contaminant levels should be low. The regulations of the states Florida and North Carolina include that when dredged sediment is used beneficially, more fine-graded sediment (finer than sand) is allowed when compared to standard beach nourishment projects.

Although beneficial reuse is mentioned in several frameworks, more study should be conducted, with specific attention on the inclusion of beneficial reuse into the existing frameworks and regulations. Beneficial reuse is not encouraged by having specific regulations (where applicable and safe to do so) to stimulate this activity. Especially purpose-driven frameworks and regulations offer more possibilities in terms of beneficial reuse of sediments. Applying this framework allows for different screening parameters, designed for specific (beneficial) end-use.

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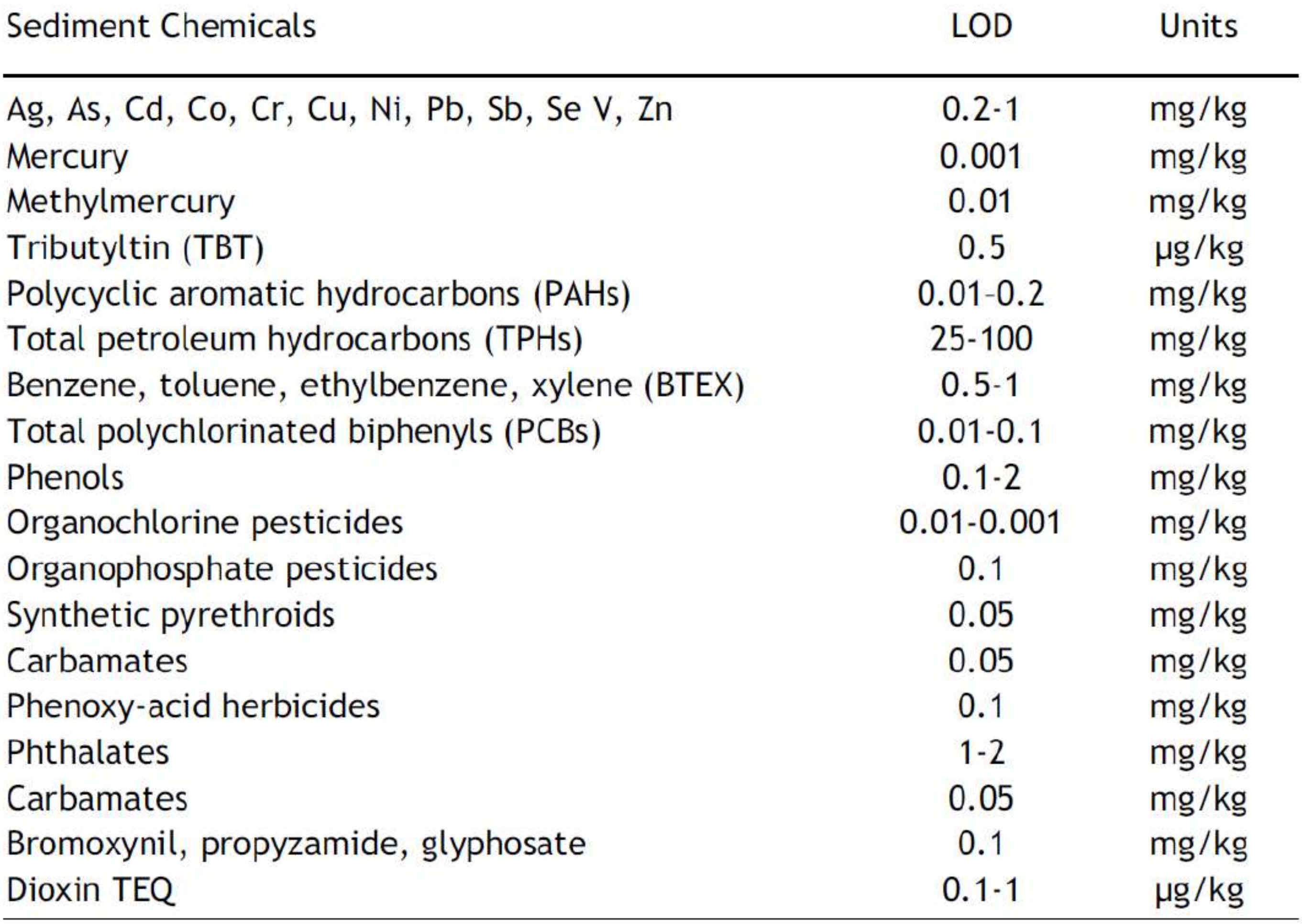
# Appendices

## Appendix A – Detection limits

*Table A-1: Typical limits of determination (LOD) for common contaminants in sediments (Simpson et al., 2005)*



| Sediment Chemicals | LOD | Units |
|---|---|---|
| Ag, As, Cd, Co, Cr, Cu, Ni, Pb, Sb, Se V, Zn | 0.2-1 | mg/kg |
| Mercury | 0.001 | mg/kg |
| Methylmercury | 0.01 | mg/kg |
| Tributyltin (TBT) | 0.5 | µg/kg |
| Polycyclic aromatic hydrocarbons (PAHs) | 0.01-0.2 | mg/kg |
| Total petroleum hydrocarbons (TPHs) | 25-100 | mg/kg |
| Benzene, toluene, ethylbenzene, xylene (BTEX) | 0.5-1 | mg/kg |
| Total polychlorinated biphenyls (PCBs) | 0.01-0.1 | mg/kg |
| Phenols | 0.1-2 | mg/kg |
| Organochlorine pesticides | 0.01-0.001 | mg/kg |
| Organophosphate pesticides | 0.1 | mg/kg |
| Synthetic pyrethroids | 0.05 | mg/kg |
| Carbamates | 0.05 | mg/kg |
| Phenoxy-acid herbicides | 0.1 | mg/kg |
| Phthalates | 1-2 | mg/kg |
| Carbamates | 0.05 | mg/kg |
| Bromoxynil, propyzamide, glyphosate | 0.1 | mg/kg |
| Dioxin TEQ | 0.1-1 | µg/kg |

# Appendix B – Overview table of institutional framework per country

| Country | Policy maker | Oversight | Framework Type | SQG Methodology |
|---|---|---|---|---|
| **The Netherlands** | Ministry of Infrastructure & Water and Ministry of Environment | Rijkswaterstaat & regional water authorities | Purpose-driven, mult-tier | Equilibrium Partitioning (EqP) approach & risk-based |
| **Canada** | Canadian Council of Ministers of Environment | Provincial institutions | Two-Tiered screening with guidelines | Weight of evidence (WOE) |
| **USA (Florida)** | U.S. Environmental Protection Agency (EPA)& National Oceanic and Atmospheric Administration (NOAA) | Federal & state permits, USACE (army corps) | Structured action-level framework, Two-tier classification | Weight of evidence (WOE) |
| **Australia & New Zealand** | Combined national ministries | Australia: Federal Department of Environment . GBRMPA ,States/territories<br>New Zealand: Regional Councils (coastal waters) and the national EPA (EEZ waters) | Two-Tiered screening with guidelines | Weight of evidence (WOE) |
| **Spain** | Ministry of Ecological Transition, documented by CEDEX. | Port Authority (inside ports), plus Maritime Authority if navigation is affected; Coastal Authority if beach nourishment; Environment & Fisheries authorities if ecological aspects; and Waste Authority if material is hazardous. | Structured action-level framework, Three-tier classification (Categories A, B, C) with clear decision steps. | Hybrid approach |
| **Japan** | Ministry of the Environment (MOE) | Ministry of the Environment (MOE) | Water-quality driven approach: pass/fail screening of values | Equilibrium Partitioning (EqP) approach: Sediment criteria are derived from water quality criteria for aquatic life. |
| **Brazil** | Federal state agency | BAMA (Brazilian Institute of Environment) and state environmental agencies | Structured action-level framework, Two-tier classification | Hybrid approach |
| **Chile** | Ministry of Environment (MMA) | Environmental Assessment Service (SEA). DIRECTEMAR (Maritime Authority under Navy) (Ministry of Public Works, port authorities) | No specific national framework | International benchmarks & risk assessment: No official guidelines. |
| **Peru** | General Directorate of Environmental Quality (Ministry of Environment), | Navy (DICAPI) authorizes dredging operations, and INRENA (Natural Resources Institute) with DIGESA (Environmental Health Directorate) had to authorize disposal of dredged material as waste. Thus, dredging fell under both maritime authority and environmental health oversight. A recent policy initiative (2021– | No specific national framework : project based screening using CCME approach | International benchmarks & risk assessment: No official guidelines. |

| Country | Policy maker | Oversight | Framework Type | SQG Methodology |
|---|---|---|---|---|
| | | 2023) is modernizing this structure, aiming for clear sediment guidelines. | | |
| **Mexico** | SEMARNAT | SEMARNAT (Environment Ministry) and state/local authorities: Navy's oceanography/hydrography directorate (which grants dredging permits). Health Ministry approves ocean disposal | No specific national framework : project based screening using NOAA or CCME approach (binary outcome) | International benchmarks & risk assessment: No official guidelines. |
| **Panama** | Maritime Authority of Panama (AMP) & Panama Canal Authority (ACP) | Maritime Authority of Panama (AMP) regulates dredging in national waters (ports, coasts), while the Panama Canal Authority (ACP) independently manages dredging within the Canal Zone under its own legal framework. Special Dredging Commission to review projects. | No specific national framework : project based screening using NOAA or CCME approach (binary outcome) | International benchmarks & risk assessment: No official guidelines. |
| **Costa Rica** | Ministry of Environment & Energy (MINAE) | Ministry of Environment & Energy (MINAE) oversees dredging impacts. The National Environmental Technical Secretariat (SETENA) evaluates and permits dredging projects through EIA. If dredging is in protected areas, the National System of Conservation Areas (SINAC) is involved. | No specific national framework : project based screening using NOAA or CCME approach (binary outcome) | Borrowed (ANZECC) approach: In practice, Costa Rica uses the Australian/New Zealand criteria (2000), particularly for metals. So methodology is effect-based (ERL/ERM-derived ISQG values). |
| **Colombia** | Ministry of Environment and Sustainable Development (MADS) via INVEMAR | Regional Autonomous Corporations (CAR) | No specific national framework : project based screening using NOAA or CCME approach (binary outcome) or using descrees | International benchmarks & risk assessment: No official guidelines. |

# Appendix C – Glossary of technical terms

| Term | Definition |
|---|---|
| **Apparent Effects Threshold (AET)** | Threshold above which adverse biological effects are consistently observed in field studies. |
| **Bioavailability** | The extent to which contaminants in sediments are available to aquatic organisms for uptake. |
| **Contaminant of Concern (CoC)** | Any chemical substance in sediment that may pose risk to organisms or human health. |
| **Dose-Response Relationship** | The relationship between the magnitude of exposure to a contaminant and the severity or type of adverse effect. |
| **Equilibrium Partitioning (EqP)** | Method that estimates sediment toxicity based on partitioning of contaminants between sediment solids and pore water. |
| **Geochemical Background** | Natural concentrations of substances in sediments, used as a reference to detect contamination. |
| **Hotspot** | A location with contaminant concentrations significantly higher than surrounding areas. |
| **Limits of Detection (LOD)** | The lowest quantity of a substance that can be reliably measured by an analytical method. |
| **Probable Effect Level (PEL)** | The concentration above which adverse effects on sediment-dwelling organisms are frequently expected. |
| **Screening Level Concentration (SLC)** | Concentration below which adverse biological effects are unlikely in most species. |
| **Sediment Quality Guidelines (SQGs)** | Benchmark concentrations used to assess contamination in sediments and determine safe disposal or reuse. |
| **Sediment Quality Triad (SQT)** | Assessment framework combining chemistry, toxicity tests, and benthic community analysis. |
| **Spiked-Sediment Bioassay (SSB)** | Laboratory test method where known quantities of contaminants are added to sediments to measure biological effects. |
| **Threshold Effect Concentration (TEC)** | Contaminant level below which harmful effects are unlikely. |
| **Threshold Effect Limit (TEL)** | Concentration below which adverse biological effects are rare. |
| **Tissue Residue Approach (TR)** | Method that assesses sediment quality by linking contaminant levels in sediments to residues in organisms. |
| **Total Organic Carbon (TOC)** | Organic carbon content of sediments, important for understanding contaminant binding and availability. |
| **Weight of Evidence (WOE)** | Approach combining multiple lines of evidence (e.g., chemistry, toxicity, ecology) to evaluate sediment quality. |

# Appendix D – Glossary of Acronyms

| Acronym | Full name | Description |
|---|---|---|
| **AET** | Apparent Effects Threshold | Empirical method defining concentration above which effects are consistently seen. |
| **ANZECC** | Australian and New Zealand Environment and Conservation Council | Joint authority that developed water and sediment guidelines in Australia and New Zealand. |
| **ARMCANZ** | Agriculture and Resource Management Council of Australia and New Zealand | Co-developer of ANZECC sediment quality guidelines. |
| **CCME** | Canadian Council of Ministers of the Environment | Developed national sediment and water quality guidelines in Canada. |
| **CONAMA** | Conselho Nacional do Meio Ambiente (Brazil) | National agency issuing environmental guidelines including dredging rules. |
| **CONPES** | Consejo Nacional de Política Económica y Social (Colombia) | Publishes Colombia's official policy documents like CONPES 4118. |
| **DNP** | Departamento Nacional de Planeación (Colombia) | Coordinates national development planning. |
| **EPA** | Environmental Protection Agency (USA) | Provides guidance on sediment and water quality in the U.S. |
| **EP** | Equilibrium Partitioning | Theoretical approach to derive sediment guidelines from water quality criteria. |
| **ERM** | Effects Range Median | Level above which effects are likely. |
| **ERL** | Effects Range Low | Level below which effects are rarely observed. |
| **FDEP** | Florida Department of Environmental Protection | State-level authority managing dredging permits and sediment quality. |
| **INVIAS** | Instituto Nacional de Vías (Colombia) | Infrastructure agency handling dredging and environmental guidance. |
| **ISQG** | Interim Sediment Quality Guideline | Australia's provisional values for safe sediment contaminant levels. |
| **LOD** | Limit of Detection | Analytical threshold for detecting a contaminant. |
| **London Convention / Protocol** | International treaty regulating dumping of waste at sea | Forms the basis for global dredged material regulation. |
| **MinAmbiente** | Ministerio de Ambiente (Colombia) | Environmental authority responsible for sediment regulation. |
| **MinTransporte** | Ministerio de Transporte (Colombia) | Oversees transport infrastructure including port dredging. |
| **NOAA** | National Oceanic and Atmospheric Administration (USA) | Developed ERL/ERM sediment quality screening values. |
| **OSPAR** | Oslo-Paris Convention | Regional convention for protecting the NE Atlantic marine environment. |
| **PAH** | Polycyclic Aromatic Hydrocarbons | Contaminants commonly monitored in sediment quality assessments. |
| **PCB** | Polychlorinated Biphenyl | Persistent organic pollutant included in SQGs. |
| **PEL** | Probable Effect Level | Upper threshold in sediment quality benchmarks. |
| **PNDM** | Plan Nacional de Dragados Marítimos (Colombia) | National dredging strategy emphasizing beneficial use. |
| **SLC** | Screening Level Concentration | Biologically based sediment guideline using species distribution data. |
| **SQT** | Sediment Quality Triad | Integrated framework using chemistry, toxicity, and biology. |

| Acronym | Full name | Description |
|---|---|---|
| **SSB** | Spiked-Sediment Bioassay | Lab method for deriving toxicity-based sediment limits. |
| **SQG** | Sediment Quality Guideline | General term for contaminant thresholds in sediment. |
| **SQGV** | Sediment Quality Guideline Value | Specific numerical values for a contaminant in sediment. |
| **TEC** | Threshold Effect Concentration | Lower threshold indicating minimal risk to organisms. |
| **TEL** | Threshold Effect Level | Lower threshold below which effects are rare. |
| **TOC** | Total Organic Carbon | Key sediment property affecting contaminant bioavailability. |
| **TR** | Tissue Residue | Contaminant level in organisms used to back-calculate sediment standards. |
| **WOE** | Weight of Evidence | Integrated approach for sediment quality evaluation. |

# Colophon

POLICY- AND TECHNICAL ADVICE ON THE BENEFICIAL USES OF MARINE DREDGED SEDIMENTS IN COLOMBIA, INCLUDING NATURE-BASED SOLUTIONS
ANALYSIS OF PARAMETERS AND PROCEDURAL STEPS OF THE NETHERLANDS AND OTHER COUNTRIES, TO DETERMINE QUALITY REQUIREMENTS OF MARINE DREDGED SEDIMENTS FOR THEIR BENEFICIAL USES. INCLUDE LESSONS-LEARNED FOR APPORTING TO THE DETERMINATION OF THRESHOLD VALUES IN COLOMBIA AIMED AT BENEFICIAL USES OF MARINE DREDGED SEDIMENTS

The project “Policy and Technical Advice on the Beneficial Uses of Marine Dredged Sediments in Colombia, including Nature-Based Solutions” is part of the collaboration between the Government of the Netherlands, through the Partners for Water program, and the Colombian Ministry of Environment, the National Planning Department (DNP) and the Ministry of Transport. The project was carried out by a consortium consisting of Arcadis, Fundación Herencia Ambiental Caribe, JESyCA, and Netics, together with government entities from both Colombia and the Netherlands.

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