UN environment programme

AGREEMENT ON THE CONSERVATION OF SMALL CETACEANS OF THE BALTIC, NORTH EAST ATLANTIC, IRISH AND NORTH SEAS

ASCOBANS/AC30/Doc.4.1

31 July 2026

30th MEETING OF THE ADVISORY COMMITTEE
Bonn, Germany, 1–3 September 2026
Agenda Item 4.1

INTERIM REPORT OF THE WORKING GROUP ON SHALLOW WATER MINING AND SMALL CETACEANS

(Prepared by the Working Group)

1. The 10th Meeting of the Parties to ASCOBANS (MOP10), through Resolution 10.7 *Shallow-water Mining and Small Cetaceans*, established a small working group (WG) to “review the available information on shallow water mining and its potential impacts on small cetaceans”. The Working Group on Shallow-water Mining and Small Cetaceans has five members[1] and presented its work plan to AC29 (ASCOBANS/AC29/Doc.4.8), which the meeting endorsed.

2. On 14 May 2026 the Secretariat, on behalf of the WG, shared the draft chapters of their report with ASCOBANS National Coordinators for feedback, encouraging them to share the draft chapters with relevant national colleagues and experts, as appropriate. By the deadline of 5 June 2026, feedback had been received from Poland, Germany, and the Netherlands. The WG then reviewed the comments and suggestions, and worked to include them in the chapters, as appropriate.

3. Some of the comments, however, would have required significant time commitment, which at this point of the process was not possible. Therefore, the report presented in the Annex is an ‘interim’ report for the review of the Advisory Committee. Parties, whose comments have not yet been addressed in this version, are invited to raise them again at AC30 so that a discussion can be had about what is realistic for the WG to undertake. Feedback is also invited from other AC30 participants.

4. The next step until AC31 is for the Working Group to finish the report and develop recommendations for the consideration of the Advisory Committee.

5. The WG Chair is stepping down after AC30. A new Chair needs to be identified, and more members would be welcome to join the WG to finalize the report in time for AC31 in 2027.

Action requested:

6. The Advisory Committee is requested to review the interim report in Annex 1 and provide feedback.

[1] Finland (Chair), Germany, OceanCare, WDC, and Chair of the ASCOBANS Jastarnia Group.

INTERIM REPORT OF THE WORKING GROUP ON SHALLOW WATER MINING AND SMALL CETACEANS

The ASCOBANS Working Group on Shallow-water Mining and Small Cetaceans was tasked with compiling an initial overview of relevant activities within the ASCOBANS Agreement area and assessing potential impacts on species covered by the Agreement.

Within this context, the present report provides an overview of three activities: the extraction of ferromanganese nodules, offshore carbon capture and storage (CCS) and sand and gravel (marine aggregate) extraction. While these activities differ in purpose and operational characteristics, all three involve industrial use of the seabed and associated marine environments and may therefore introduce pressures relevant to small cetaceans and their habitats.

Impacts may arise both directly and indirectly. Direct effects may include underwater noise, vessel activity, and disturbances associated with the exploration, construction, dredging, extraction, discharging, operational and monitoring phases of extraction projects. Indirect effects may occur through changes to seabed habitats and water column turbidity, as well as impacts on benthic communities, prey availability, and broader ecosystem processes. As small cetaceans rely on productive shallow-water environments for feeding, movement, and other key life functions, such pressures may be relevant to the conservation objectives of ASCOBANS.

The report therefore aims to summarize the main characteristics of these activities, outline available information on their ecological and biological impacts, and highlight aspects that may be of relevance to small cetaceans within the ASCOBANS area. It is intended to support the Working Group's ongoing review and to inform further discussion within ASCOBANS on these topics.

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Ferromanganese nodules in the shallow-water marine environment, their extraction and possible effects to biodiversity and small cetaceans

Introduction

Ferromanganese (FeMn) nodules form on the seafloor in both deep and shallow waters, and rising demand for the metals they contain - cobalt, nickel, manganese and copper among them - has put their extraction on the European policy agenda. Most attention to date has gone to deep-sea nodules in the Pacific, where exploration is governed internationally by the International Seabed Authority (ISA), but several European states are now exploring, and preparing to permit, extraction from shallow coastal waters as well.

The following chapter focuses on the Baltic Sea, which holds the majority of recorded European nodule occurrences, the most extensive shallow-water seabed mapping of any European sea area, and the most advanced shallow-water extraction proposal currently in preparation in Europe, in the Swedish part of the Bothnian Bay. It is also home to the Critically Endangered Baltic Proper harbour porpoise. The following sections cover what nodules are, their ecological role, where they occur, how they are extracted, the governance gap for shallow-water mining, environmental impacts, and what is known about effects on small cetaceans, and closes with recommendations.

Description of nodules

Shallow marine areas, such as the continental shelf, are rich in valuable minerals, including: (1) mineral-rich sands, (2) polymetallic nodules and phosphorites, (3) placer deposits containing metallic minerals or gemstones (e.g., gold and diamonds).

Polymetallic nodules, often referred to as manganese or **ferromanganese (FeMn) nodules**, are mineral concretions found on the sea floor both in the deep and shallow waters. They consist of concentric layers of iron and manganese hydroxides around a core, and form through precipitation in the boundary layer between bottom sediments and seawater. Their main components are iron oxyhydroxides and manganese oxides, along with associated metals such as nickel, cobalt, copper, titanium, and rare earth elements. They occur as rounded or irregular concretions, with sizes ranging from a few millimeters to several square meters (Figure 1.). Shape and size depend on the structure and composition of the seabed (Zhamoida et al., 2004). It is this metal content that has made nodules a target for mining interest: several of the metals they contain are used in batteries, electric motors and other low-carbon technologies, and rising demand linked to the energy transition has driven growing commercial and governmental interest in nodules as an alternative source of supply to terrestrial mining (Hein et al., 2020).

Figure 1. Ferromanganese nodules in the eastern Gulf of Finland. Photo: SYKE 2012

Growth rates differ markedly between settings. In the Baltic Sea where especially FeMn nodules are widespread (particularly in the Gulf of Finland), their formation and growth occur at comparably higher rates (0.014-0.028 mm/**yr**). In the deep sea, by contrast, FeMn nodules grow extremely slowly, ranging from 1 to 10 millimeters per million years (mm/Myr). (Grigoriev et al., 2013)

Occurrence of polymetallic nodules

Polymetallic nodules occur both in deep ocean environments - such as the vast abyssal plains of the Pacific's Clarion-Clipperton Zone (CCZ) (Fig. 2) - as well as in shallower waters, including the Baltic Sea (Fig. 3) and even some lakes. However, they are best known for their presence in the deep sea, where they form on sediment-covered floors at depths of several thousand meters.

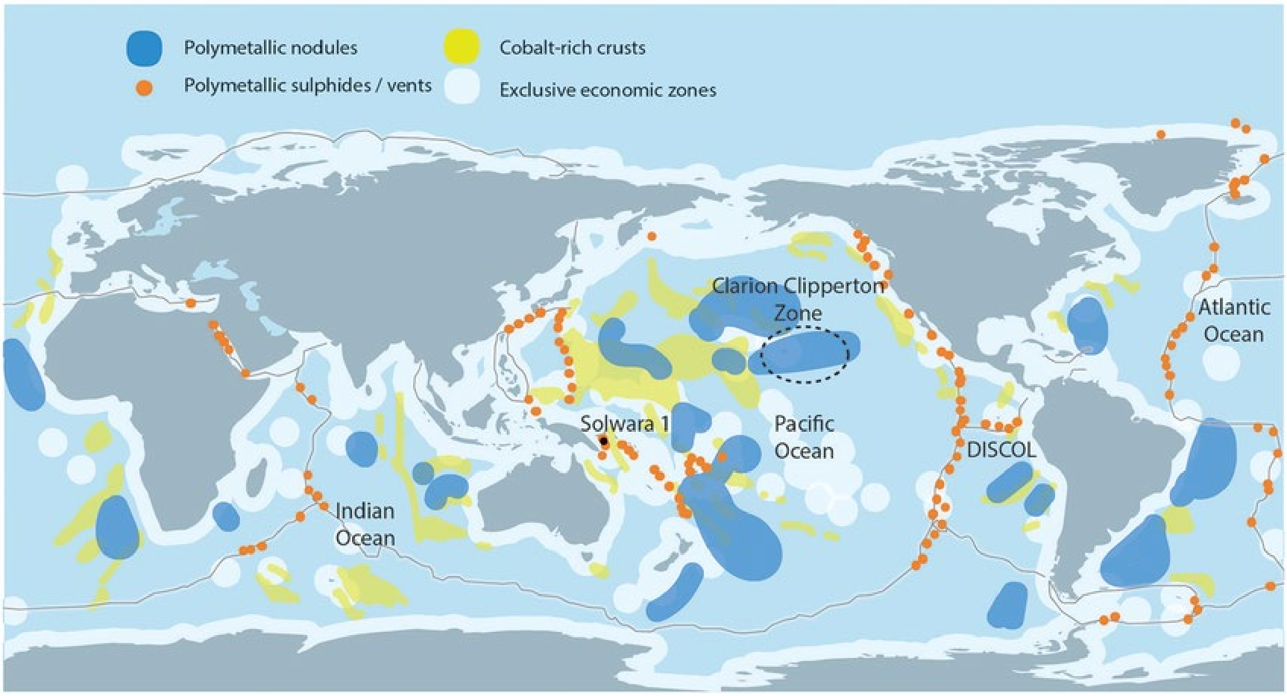

Figure 2. A world map showing the location of the three main marine mineral deposits in deep waters: polymetallic nodules (blue); polymetallic or seafloor massive sulfides (orange); and cobalt-rich ferromanganese crusts (yellow). Redrawn from a number of sources including Hein et al. (2013).

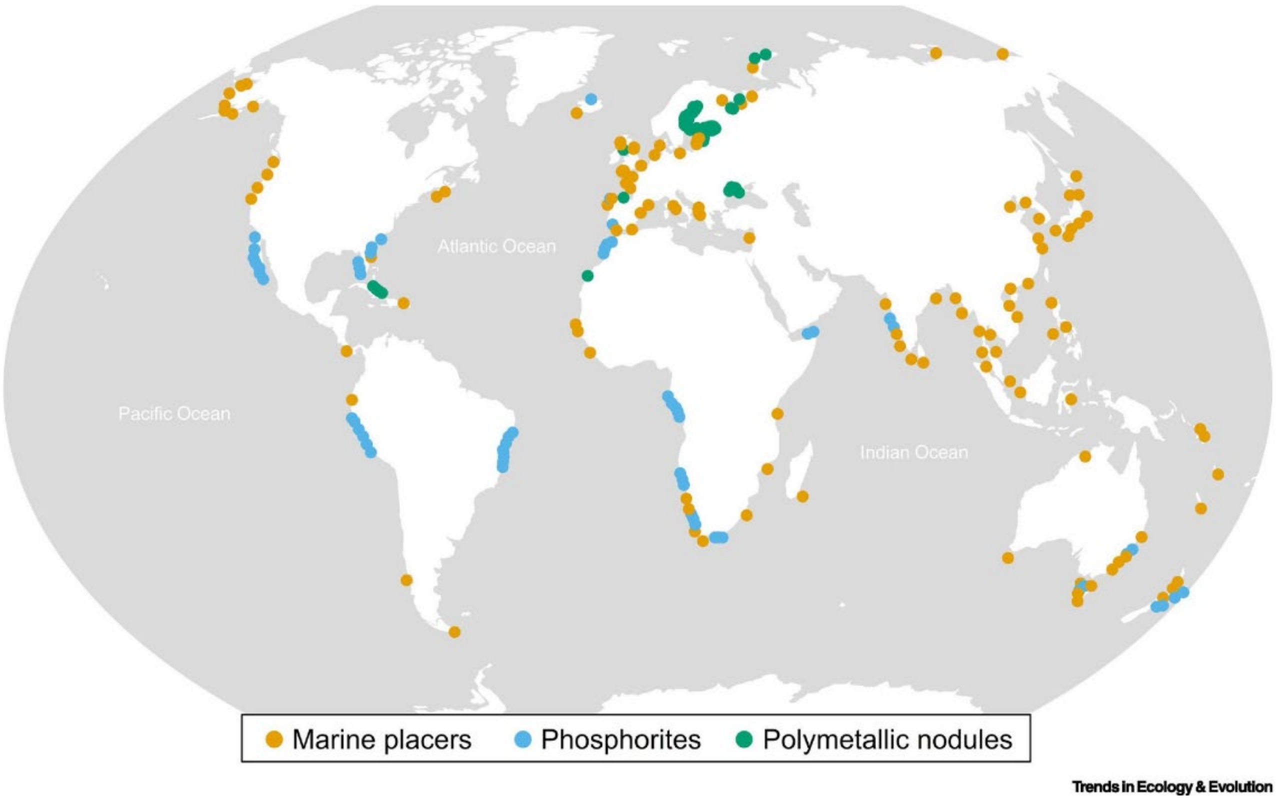

Figure 3. Mineral resources found in shallow waters: marine placers (such as metallic minerals or gemstones), phosphorites, and **polymetallic nodules (green sphere)**. These resources have extraction potential based on how close they are to the coast and the depth at which they are found. Map by Kaikkonen et al. 2022.

European occurrence

The MINDeSea project is a collaboration project between eight GeoERA (*Establishing the European Geological Surveys Research Area to deliver a Geological Service for Europe*) Partners and four Non-funded Organizations. It focused on mapping and assessing seafloor mineral resources, with the aim to identity and characterize deposits of Critical Raw Materials (CRMs) and strategic metals—such as Cobalt, Copper, and Rare Earth Elements—in the Atlantic, Mediterranean, Baltic, and Black Seas.

The project database lists a total of 296 polymetallic nodule occurrences in European waters, most of which are located in the Baltic Sea (184 occurrences, representing 62% of the total). The North-East Atlantic Ocean (including some data points from the Arctic Ocean) accounts for 23% of the occurrences (68 occurrences). The remaining 15% are distributed across five European marine regions, as shown in Fig. 4. (Ferreira et al., 2021)

Baltic sea

As stated previously, about 62% of all polymetallic nodules occurrences listed in the MINDeSEA database are located in the Baltic Sea. Of these, about half (99 occurrences in total) are located in the Gulf of Bothnia, 57 in the Gulfs of Finland and Riga, and the remaining 27 in the Baltic Sea proper, including Kiel Bay (Ferreira et al., 2021).

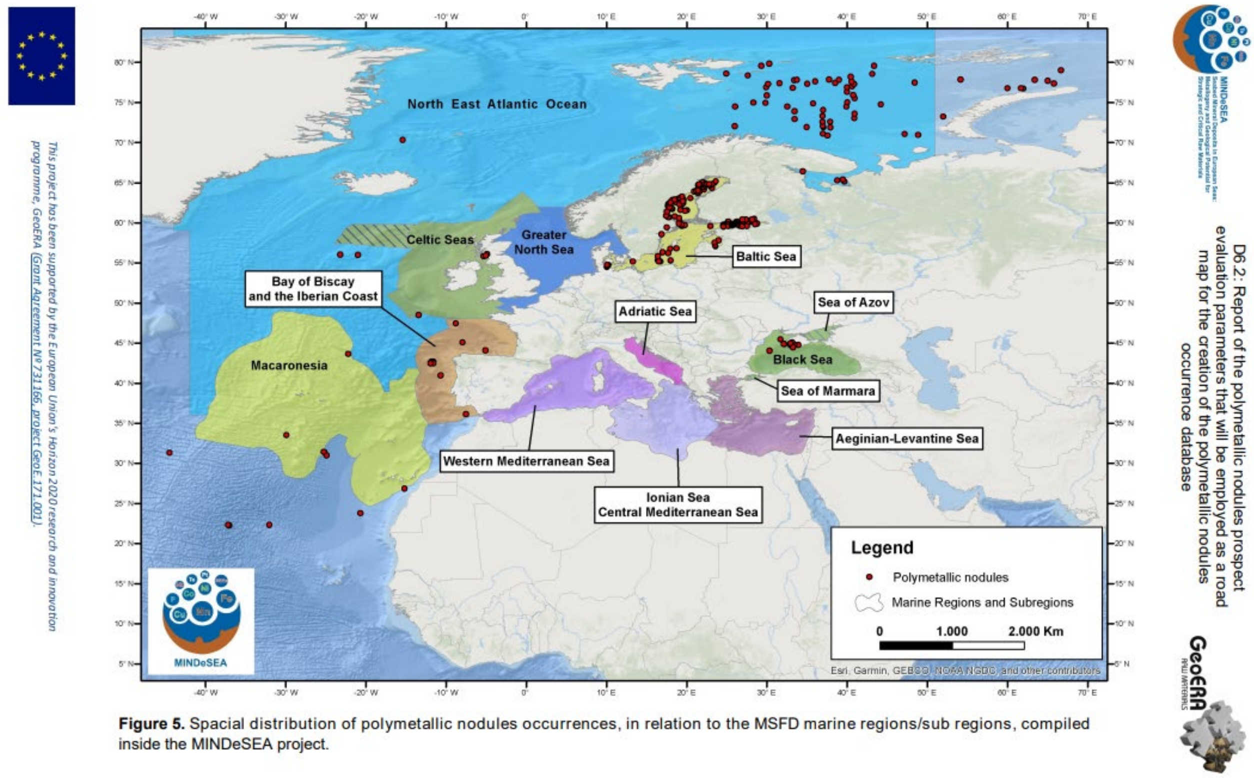

Figure 4. Polymetallic nodules occurrence in European seas (Ferreira et al. 2021)

The Gulf of Bothnia and the Bothnian Sea contain the largest share of registered data—54% of the total for the Baltic Sea. When looking at the distribution across the Exclusive Economic Zones (EEZs) of Baltic countries, the highest number of recorded nodule occurrences is found in Sweden's EEZ, which also accounts for 54% of all occurrences within Baltic EEZs (Ferreira et al., 2021 Fig. 5). This is, however, probably mostly due to the registration of samples in the MINDeSEA database and not to real occurrence. Observations from, for example, scientific diving and drop cameras, deployed in the Finnish Velmu programme, show that polymetallic nodules are observed in 6831 locations in all basins of the Finnish part of the Baltic Sea, with the exception of the southern Bothnian Sea (Kaikkonen et al., 2019) Fig. 6 and Fig. 7.

Poland

Between 1976 and 1990, the Polish Geological Institute conducted geological research in Polish maritime areas. In 1980, the threshold separating the Bornholm Basin from the Słupsk Furrow was mapped, revealing numerous Fe-Mn nodules on the seabed. Similar nodules have been identified in the Gulf of Finland and Gulf of Riga, both showing high Fe concentrations. However, samples from the Gulf of Finland are comparatively richer in sodium and calcium oxides (1.75% and 2.15%, respectively) and total sulfur (0.17%) (Baturin, 2019).

In spring 2025, the University of Gdańsk conducted a research cruise to study iron-manganese concretions in the southern Baltic Sea within Poland's Exclusive Economic Zone and to make a preliminary assessment of potential mineral resources. Covering over 300 nautical miles, the expedition collected water and sediment samples from 47 research stations and documented seabed conditions at depths exceeding 50 meters using underwater drones. (University of Gdańsk, 2025).

Currently, the project is in the laboratory and data analysis phase. As the scientific work is still ongoing, there is no confirmed timeline for publication yet, and no specific data can be shared at this moment.

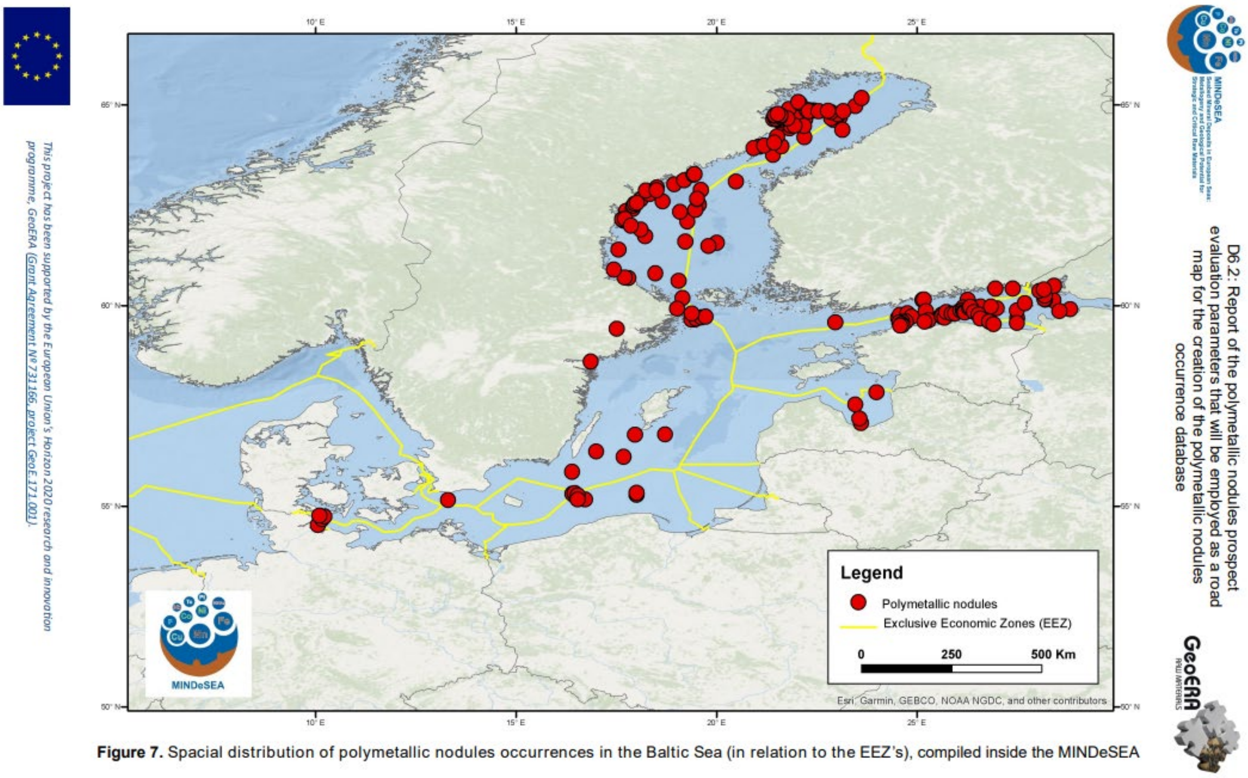

Figure 5. Spatial distribution of polymetallic nodules occurrences in the Baltic Sea (in relation to the EEZ's) compiled inside the MINDeSea project (Ferreira et al. 2021), https://geoera.eu/projects/mindesea2/

Russia

Russia has long explored ferromanganese deposits in the eastern Gulf of Finland as part of its mineral resource development efforts (Zhamoida et al., 2017).

Sweden

In Sweden, Scandinavian Ocean Minerals has been granted exploration permits to survey the seabed in four areas covering a total of 800 km^2 in the Bothnian Bay. Surveys include mapping with sonar, estimates of nodule density, sediment and benthic fauna samples and video mapping. In April 2026, the company was preparing to apply for a permit to mine iron-manganese nodules in the Bothnian Bay. Meanwhile, between 2023-2027, a major Formas-funded research project led by Francisco Nascimento at Stockholm University will investigate how Baltic Sea coastal ecosystems respond to pressures coming from seabed mining (Umeå University, 2025).

Finland

The distribution of FeMn concretions has been assessed in Finnish marine areas using data from 140,000 sites surveyed by the Finnish VELMU programme (2004-2018) (Fig. 6). Seabed-mapping records were analysed and spatial modelling applied to estimate the potential coverage, showing that concretions are more extensive than previously assumed: FeMn nodules were found at ~7,000 sites and predicted to cover over 11% of Finnish sea areas, as shown in Fig. 7 (Kaikkonen et al. 2019).

These findings reveal greater seafloor complexity in the northern Baltic Sea and support future work on the ecological and resource value of these mineral deposits.

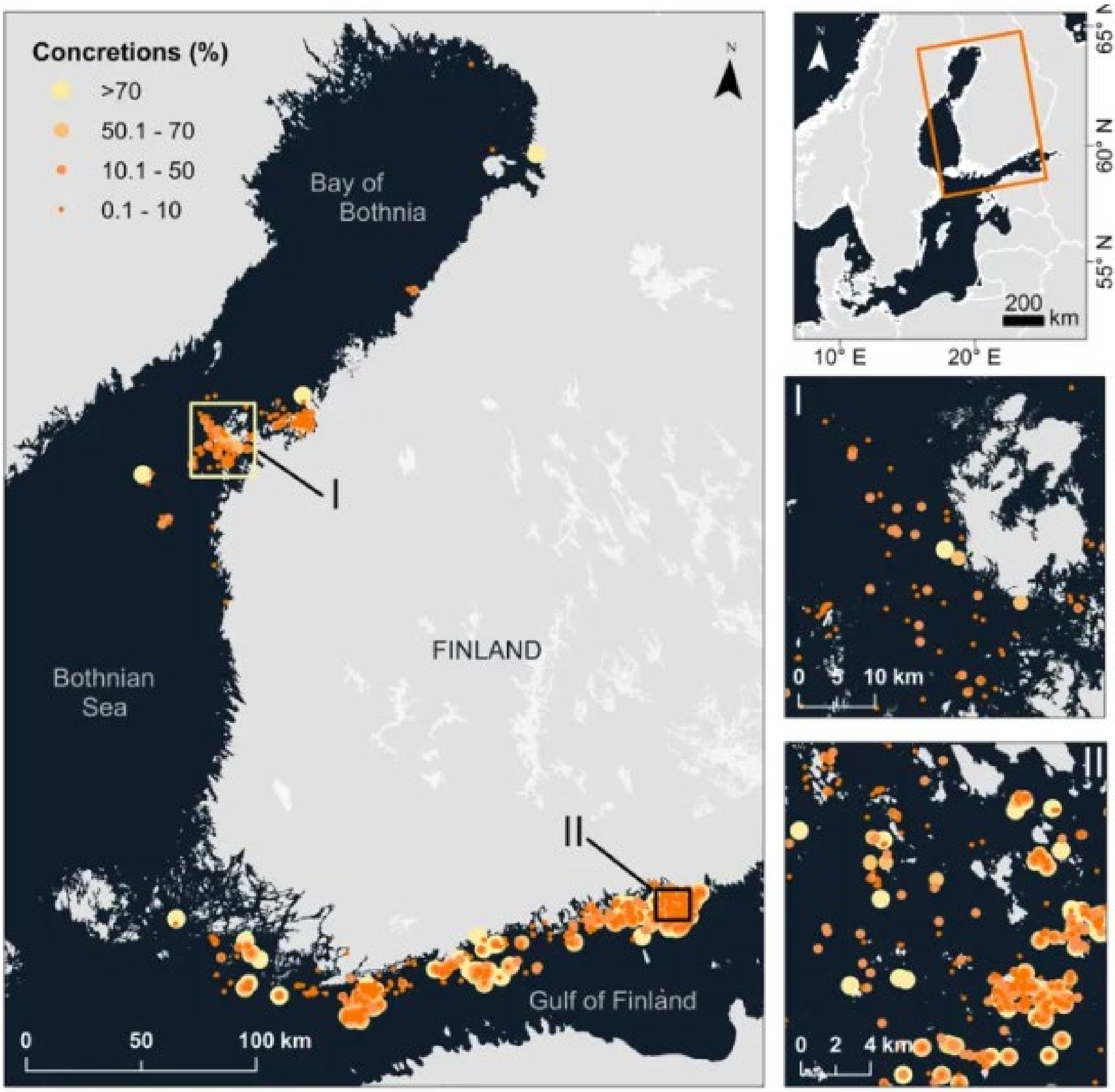

Figure 6. Concretions found along the Finnish marine areas. Larger, light yellow points indicate areas of concretion fields (>70% coverages) and small, orange points of smaller concretion occurrences (0.1–10%) (Kaikkonen et al., 2019).

Ferromanganese (FeMn) nodules as habitats in the Baltic Sea

FeMn concretions have been classified as habitat types by both HELCOM and Finland in assessments of threatened habitats and biotopes. These concretions create hard, complex substrates on otherwise soft seafloors, enhancing both geological and biological diversity. Increased habitat complexity correlates with higher benthic biodiversity, providing shelter and living spaces for various organisms.

Such habitats often support a relatively rich benthic fauna dominated by mussels, including the lagoon cockle (*Cerastoderma glaucum*), the Baltic clam (*Macoma balthica*), and the blue mussel (*Mytilus trossulus*). Other associated species include snails such as the spire snail (*Ecrobia ventrosa*), oligochaete worms, crustaceans (e.g., Ostracoda), and midge larvae. These concretions also offer protection against soft-bottom erosion and serve as adhesion substrates in areas affected by bottom currents (Kotilainen et al., 2018). In addition to binding environmental toxins, iron-manganese deposits also retain phosphorus, an essential nutrient for living organisms. Because the amount of phosphorus bound in sediments is higher than in the surrounding water, FeMn concretions may play a significant role in controlling the internal nutrient load of the Baltic Sea (Kotilainen et al., 2018).

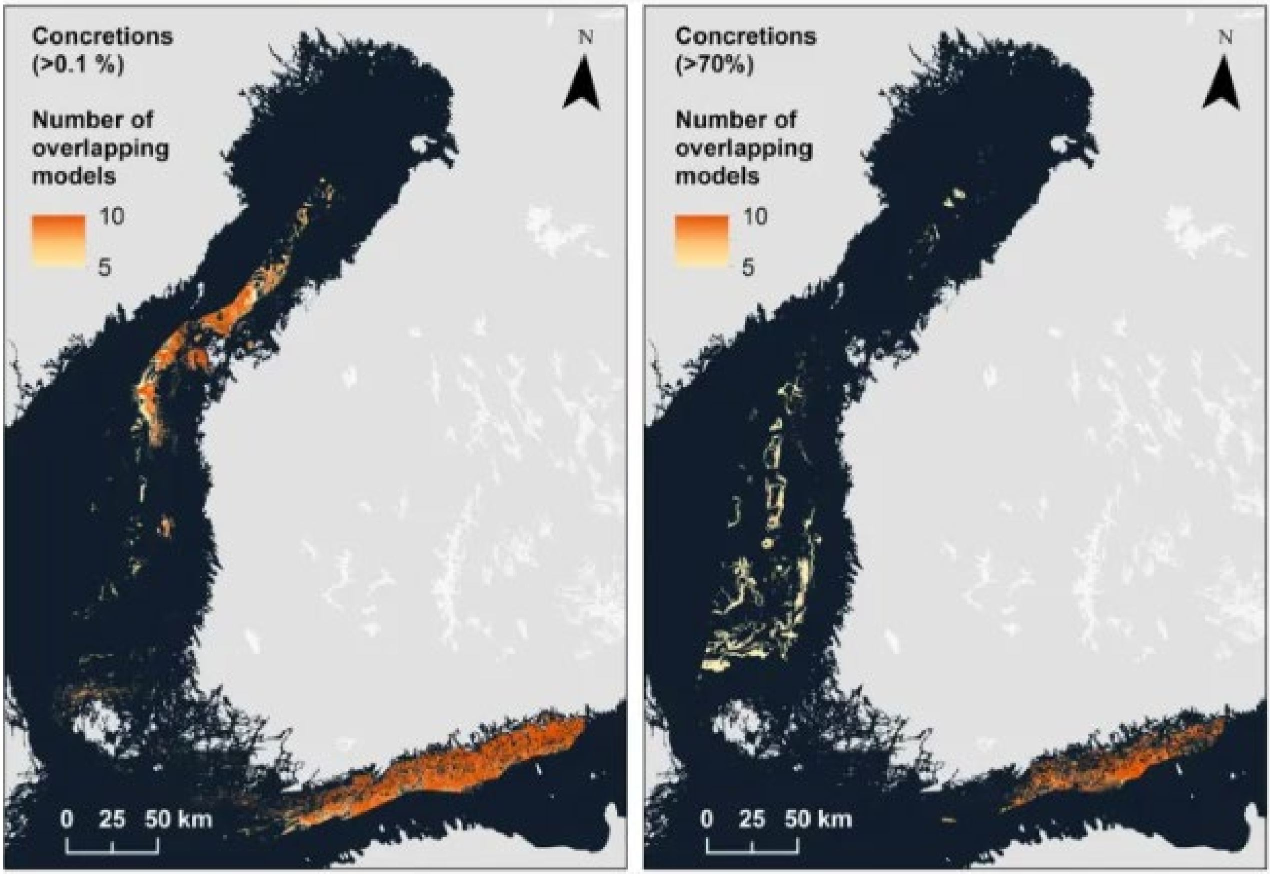

Figure 7. Concretion prediction maps for concretion occurrences (>0.1%) and concretion fields (>70%) (Kaikkonen et al., 2019).

Extraction

As noted in the introduction, FeMn concretions are a potential source of valuable elements such as cobalt, along with iron and manganese, and rising demand for green energy technologies , has increased interest in mining from unconventional sources such as the seafloor. The first trial of a prototype nodule-mining system occurred in 1970 on the Blake Plateau, off Florida in the Atlantic Ocean, at a depth of 1000 meters. The main depths where mining occur is 4000 to 5000 meters. (https://www.isa.org.jm/wp-content/uploads/2022/06/eng7.pdf). The International Seabed Authority (ISA) has entered into 15-year contracts with 21 contractors for the exploration for polymetallic nodules, polymetallic sulphides and cobalt-rich ferromanganese crusts in the international seabed area. The explored areas are in the Clarion-Clipperton Zone, the Indian Ocean, the Mid Atlantic Ridge and the Pacific Ocean (Fig.8).

Utilizing polymetallic nodules from shallow coastal areas is in many cases more cost-effective than exploiting deep-sea minerals. Coastal areas are easier and cheaper to access because specialized deep-sea equipment is not needed, and short distances to shore reduce logistics costs.

Nodules found in coastal waters contain the same metal potential as those in deep-sea environments. Additionally, coastal areas have generally been mapped and studied much more thoroughly than the deep sea. This reduces financial risk and makes environmental impact assessments easier. For these reasons, basic information about nodule occurrences is also easier to obtain from coastal waters than from deep-sea deposits.

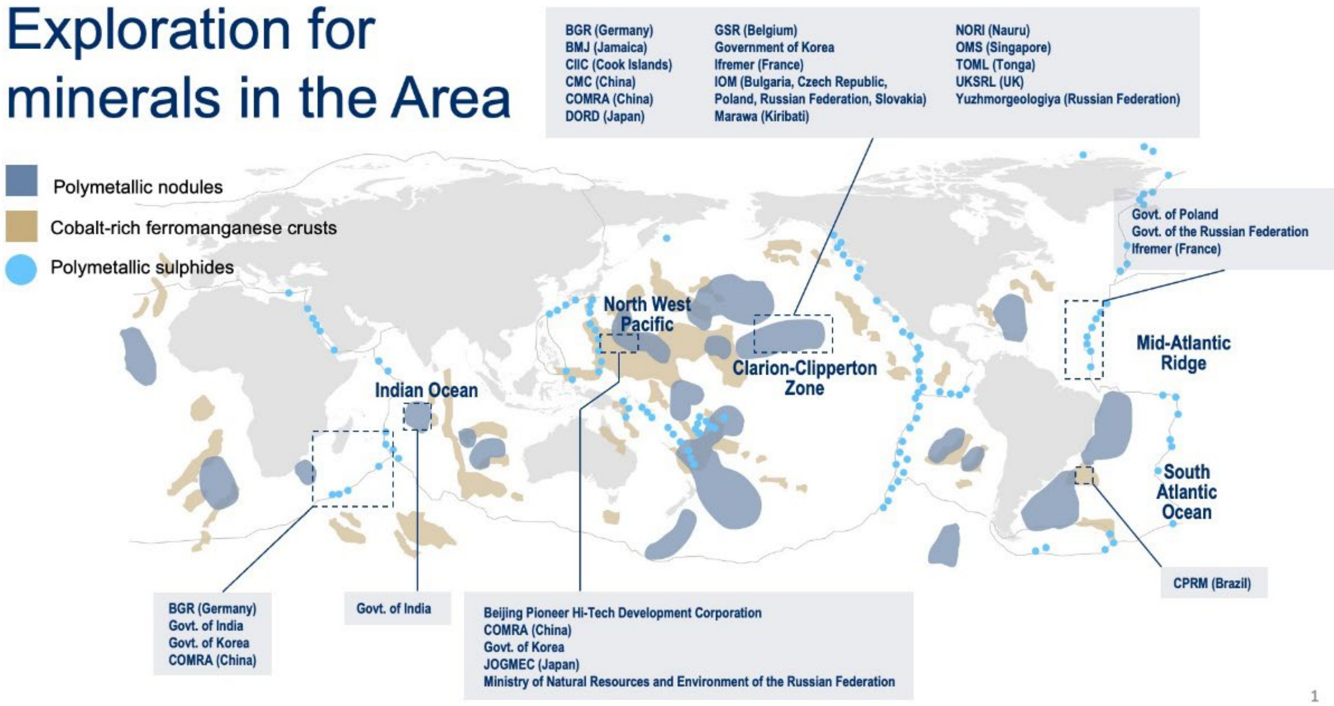

Figure 8. Location of deep-sea minerals and countries with exploration license (https://www.iucn.nl/en/story/the-impact-of-deep-sea-mining-on-biodiversity-climate-and-human-cultures/) – Original Source: International Seabed Authority https://isa.org.jm/exploration-contracts/exploration-areas/(NB. an updated map to follow).

The high density and relatively shallow depths mean that e.g. the Gulf of Bothnia is economically and logistically advantageous for nodule extraction. A comparable abundance of ferromanganese concretions was found also in the eastern Gulf of Finland, where the abundance of spheroidal concretions locally reaches 50–60 kg/m2 (wet weight).

Of the Baltic states, discussed before, Sweden and Russia are currently the only two countries where extraction itself, as distinct from exploration and seabed mapping, has been carried out or is concretely planned.

Sweden

Extraction by Scandinavian Ocean Minerals, SOM, is planned to be carried out using a soft suction technique where the aim is to be able to extract nodules with minimized environmental impact (Fig. 9). An application for permit to start extraction is planned for 2026.

Russia

Between 2006 and 2008, a Russian mining company dredged 56,000 tonnes of FeMn deposits from the eastern Gulf of Finland for metals enrichment studies. The operation has since been tested in the Vyborg Bay as well (Kostamo, 2021).

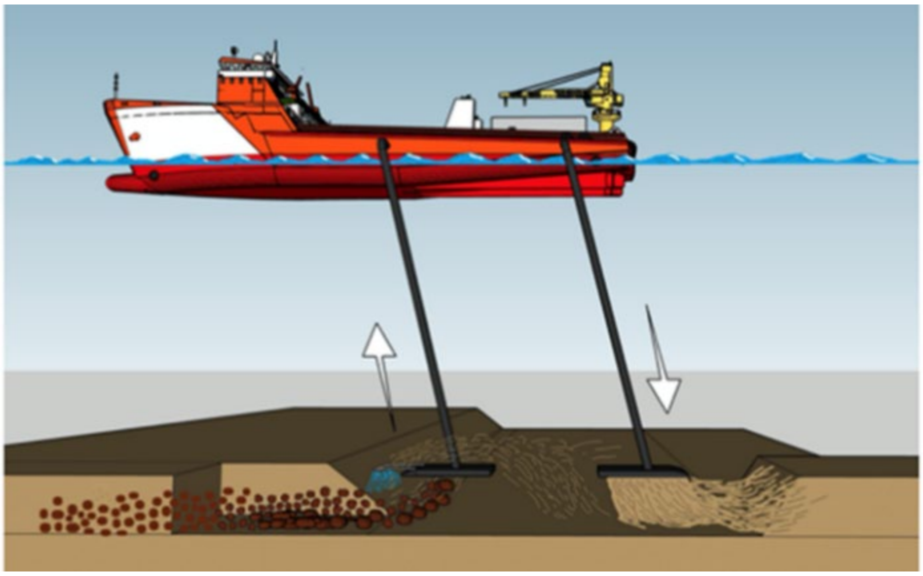

Figure 9. The plan is to use an airlift technique by vacuum cleaning the upper 5-10 centimetres of the sea floor. The sediment is brought up to a platform at the surface of the sea where it's being sieved. The particles larger than one millimetre are kept and the remains are released back to the sea floor. Image by [Scandinavian Ocean Minerals | Om oss]

Governance and regulatory framework

The ISA's Mining Code remains incomplete. Exploration regulations have applied for years, but the exploitation regulations that would govern commercial extraction have been under negotiation since 2014, missed a 2025 adoption target (ISA, 2026a), and remained unresolved as of the Council's March 2026 session, with a further session due in July 2026. Negotiations have been complicated by a separate, unresolved dispute over whether mining can proceed outside the ISA framework altogether, after a US-based company indicated it would seek a permit under US domestic law rather than through the ISA, a move the ISA Secretary-General said would violate international law.

Shallow-water mining has no equivalent regime, no national legislation, and no international process underway to create one (Kaikkonen & Virtanen, 2022). Where shallow-water projects are assessed, this happens through ordinary national and EU EIA procedures, not anything designed with seabed mining or small cetaceans in mind, the governance gap most relevant to ASCOBANS (ISA, 2026b; Earth Negotiations Bulletin, 2026).

Possible impacts of mining

A study by Kaikkonen et al. (2018) synthesizes empirical evidence from experimental seabed mining and analogous industries to assess the ecological effects of polymetallic nodule and ferromanganese concretion extraction. Using a problem structuring framework, the authors evaluated causal links between extraction pressures and ecosystem responses and highlighted the need for pressure specific expert elicitation, as well as improved integration of ecosystem services in impact and risk assessments (Table 1). (Kaikkonen et al.,2018)

Table 1. Pressures associated with seabed mineral extraction and their typical ecological consequences (taken from Kaikkonen et al., 2018).

Table 1
Pressures and associated state changes resulting from seabed mineral extraction with examples of ecosystem indicators, and affected underwater ecosystem services from deep and shallow seabed.

Pressure	State change	Indicators	Affected ecosystem services	References
Extraction of seafloor substrate	Loss of benthic fauna by direct removal	Macrofaunal abundance and diversity	Biodiversity, secondary production, trophic support, existence value	(Miljutin et al., 2011), (Borowski, 2001), (Newell et al., 2004), (Kenny and Rees, 1994, 1996)
	Habitat loss and degradation	Suitable habitat abundance	Habitat provision, existence value	(Vanreusel et al., 2016), (Khripounoff et al., 2006)
	Loss of fauna attached to nodules	Megabenthic fauna	Biodiversity, existence value, trophic support	(Bluhm, 2001), (Vanreusel et al., 2016)
	Changes sediment composition	Fish diversity, Incomplete recovery of fauna	Biodiversity, existence value, trophic support	(Hwang et al., 2013), (Desprez, 2000), (Van Dalfsen et al., 2000), (Waye-Barker et al., 2015), (Robinson et al., 2005)
Siltation and resuspension of sediment	Smothering of fauna	Macrofaunal recovery	Biodiversity, trophic support	(Hussin et al., 2012), (Desprez et al., 2009), (Cooper et al., 2007b), (Desprez, 2000), (Boyd and Rees, 2003)
	Smothering of benthic flora	Recovery of benthic flora	Biodiversity, trophic support, biomaterials for industrial and pharmaceutical uses	(Lyngby and Mortensen, 1996, McMahon et al., 2011, Onuf, 1994, Riul et al., 2008)
	Behavioral changes in animals		Biodiversity, fish catch	(Hecht, 1992)
	Changes in sediment composition	Sediment grain size and biochemical composition	Habitat provision, nutrient cycling	(Cooper et al., 2011, Le Bot et al., 2010)
	Incomplete recovery of fauna	Macrofauna, sediment composition	Nutrient cycling	(Desprez, 2000), (Van Dalfsen et al., 2000), (Waye-Barker et al., 2015), (Robinson et al., 2005)
	Changes in seabed geomorphology	Sediment composition	Nutrient cycling	(Le Bot et al., 2010, Uścinowicz et al., 2014), (Khripounoff et al., 2006)
	Loss of fish nursery grounds	Sediment composition, fish abundance	Biodiversity, fish catch	(Scharf et al., 2006)
Release of substances from the sediment	Sublethal effects on fauna	Feeding and reproduction of fauna, increased concentration of contaminants and their bioavailability after dredging	Biodiversity, fish catch	(Roberts, 2012, Simpson and Spadaro, 2016) (Bradshaw et al., 2012)
	Nutrient release from the sediment	Chl *a*, *Synechococcus* cell number, inorganic nutrient concentration	Chemosynthesis, Photosynthesis, aesthetic value	(Hyun et al., 1998, Lohrer and Wetz, 2003)
	Increased turbidity	Water clarity	Photosynthesis, aesthetic value	(Sharma et al., 2001)
Underwater noise	Disturbance to animals	Feeding and resting of animals	Fish catch, existence value	(Roberts et al., 2015, Robinson et al., 2011)
Underwater light	Disturbance to animals	Feeding and resting of animals	Fish catch, existence value	(McFarland, 1986)

Compared with terrestrial mining and costly deep-sea mining operations, shallow-water mining may save operational costs, as mining takes place closer to shore and uses existing technology. Dredging shallow seafloor minerals is efficient, with dredge rates for raw sediment material from the seabed up to 2500 m3 per hour. While these aspects render shallow resources attractive from an economic standpoint, mining shallow-water resources does not come without risks (Kaikkonen & Virtanen, 2022).

Shallow-water mining exerts additional pressures on vulnerable coastal ecosystems which are already burdened with cumulative impacts from human activities and the effects of climate change, making them less resilient to new human activities. Despite faster recovery times of shallow-water ecosystems compared with vulnerable and slow-growing deep-sea communities, the overall environmental footprint of mining will be significant also in shallow areas (Kaikkonen & Virtanen, 2022). Shallow-water seabed mining likely results in cascading impacts on the ecosystem both on the seafloor and water column, with ecosystem recovery varying from years to decades (Kaikkonen & Virtanen, 2022).

Mining operations can further have negative impacts on local people, either by direct disturbance from the operations and the required infrastructure on land, or via the degradation of the environment and subsequent impacts on ecosystem services (Kaikkonen & Virtanen, 2022).

Kaikkonen et al. (2021) present the first systematic evaluation of the ecological risks associated with seabed mining. Based on interviews with a multidisciplinary group of experts, the study provides a foundation for an ecological risk assessment model. It further demonstrates how qualitative information can be used to progress toward quantitative assessment by applying a causal probabilistic approach to estimate the impacts of seabed disturbance and direct sediment extraction on benthic fauna in the Baltic Sea. The results indicate that current knowledge of seabed mining impacts remains limited, highlighting the need for further research to support scientifically sound permitting decisions.

Additionally, a study by Dargahi (2023) highlights that the environmental and ecological risks of shallow water mining in the Baltic Sea are not sufficiently studied and recommends that the Baltic Sea seabed be kept free of mining until the evidence base improves. In his study, he employed models applied over a ten-year period (2000–2009) under prevailing environmental conditions to simulate seabed mining operations. The study focused on sediment concentrations near the seabed and their dispersion patterns. The use of integrated hydrodynamic, particle-tracking, and sediment-transport models in the Baltic Sea revealed important characteristics of near-bed suspended sediment concentrations.

The results show that shallow water mining would significantly disrupt the sediment transport equilibrium, with the Gotland Basin identified as the most vulnerable region.

At present, due to the limited exploitation of minerals in shallow marine areas, there is no national legislation governing such activities—unlike the Mining Code applied to deep-sea mining. This regulatory gap is one reason why the mining industry is increasingly turning its attention to the shallow marine areas (Kaikkonen & Virtanen, 2022).

Pathways of impact

Drawing together the evidence above and adapting the pressure-response structure used by Kaikkonen et al. (2018), four main pathways of impact can be distinguished.

Extraction of seafloor substrate directly removes and degrades benthic habitats, alters sediment composition, and causes the loss of associated fauna (Kaikkonen et al., 2018). These disturbances are reflected in reduced macrofaunal abundance, diminished habitat availability, and incomplete recovery of benthic and fish communities. Habitat loss can lead to local extinctions, particularly among low-mobility species or organisms that depend on concretions for shelter.

Additionally, specific to FeMn concretions, their removal disrupts the biogeochemical processes as these structures function as natural sinks for heavy metals and phosphorus and help regulate redox conditions at the seafloor. Microbial communities associated with the concretions contribute to pollutant degradation and play a role in concretion growth and metal cycling As a result, key ecosystem services - including biodiversity maintenance, trophic support, habitat provision, and existence value - may be substantially impaired (Kaikkonen et al., 2018)

Siltation and sediment resuspension lead to the smothering of benthic flora and fauna, behavioural changes in animals, altered sediment characteristics, and incomplete ecosystem recovery, which comes generally with seabed mineral extraction Indicators include macrofaunal changes, shifts in sediment grain size, and modified seabed morphology (Kaikkonen et al., 2018). These processes can reduce biodiversity, disrupt nutrient cycling, and diminish habitat quality. Dargahi's (2023) Baltic Sea modelling illustrates the scale this can reach: average sediment concentration was approximately 2 g/m3, with peaks up to 90 g/m3 on a single autumn day, whereas simulated mining activity raised concentrations to 300-500 g/m3, which are three to five times above background levels, with the Gotland Basin identified as the most vulnerable area to this disturbance.

Likewise general to seabed mineral extraction, release of substances from sediments, including nutrients and contaminants, causes increased turbidity, sublethal stress on fauna, and potential loss of nursery habitats. Relevant indicators include changes in nutrient concentrations, sediment and fish abundance, and behavioural shifts in marine animals. These impacts may affect fish production, primary productivity, and overall ecosystem health (Kaikkonen et al., 2018).

Underwater noise and light, a pressure common to underwater industrial operations generally, primarily disturb marine animals by altering their feeding and resting behaviour. Such disturbances may reduce fish catch potential and negatively influence ecosystem services related to biodiversity and aesthetic value (Kaikkonen et al., 2018).

Overall, the pressures associated with seabed mining can cause long-lasting alterations to seafloor ecosystems, highlighting the importance of comprehensive environmental assessments prior to mineral extraction activities.

Possible effects on small cetaceans

At present, there are no research projects specifically examining the effects of seabed mining in shallow waters on small cetaceans. However, it is already well established that seabed mining affects both benthic habitats and the species associated with them, suggesting that mining operations may also have impacts on small cetaceans.

The closest available evidence base comes from deep-sea mining, principally in the Clarion-Clipperton Zone, rather than from shallow-water contexts like the Baltic, so it should be read as analogous rather than directly applicable. The International Whaling Commission (IWC) convened an intersessional correspondence group to assess deep-sea mining's potential impact on cetaceans, which concluded that direct impacts are most likely to come from noise, for which an extensive evidence base already exists from other maritime activities, alongside potential indirect impacts from sediment plumes, ecosystem disruption and disturbance of prey. On this basis, the IWC Scientific Committee recommended that the IWC join the call for a moratorium on deep-sea mining pending further study, and that it collaborate with other bodies, including CMS, on the issue.

A 2025 CMS review of deep-sea mining's impacts on migratory species adds further detail relevant to small cetaceans specifically. Of 86 CMS-listed marine mammal species, at least 41 have ranges overlapping an ISA contract area or a proposed national mining area, including a number of species within ASCOBANS' remit, among them the harbour porpoise, white-beaked dolphin, Atlantic white-sided dolphin, bottlenose dolphin, common dolphin, Risso's dolphin, killer whale, both pilot whale species, and several beaked whales.

The review identifies noise as the impact pathway best supported by existing evidence: mining machinery and dynamic-positioning vessels are expected to operate continuously over months or years, introducing chronic noise into what are otherwise quiet acoustic environments, with the potential to mask communication, disrupt feeding and navigation, and induce physiological stress (Thaler, 2025).

The scale of impacts in a shallow-water context such as the Baltic are likely to depend on the scale, location, duration, and methods of extraction, and may differ in important ways from the deep-sea evidence above. For instance, Baltic harbour porpoise does not undertake the extreme dives, associated with decompression sickness risk in beaked whales, but operate in a shallower, already noisier and human-impacted soundscape, where increased underwater noise increases stress levels and impact important behaviours such as feeding and breeding. Habitat loss or disturbance and the release of nutrients and contaminants may affect prey species and, indirectly, small cetaceans through reduced food availability or increased contaminant exposure.

In addition, both research activities and mining operations generate noise and light pollution, which may have direct effects on small cetaceans, depending on how these activities are conducted. This combination of indirect, prey-mediated pathways and direct sensory disturbance represents a clear and currently unaddressed knowledge gap, and points to a clear need for dedicated research projects focused on the effects of shallow-water seabed mining on small cetaceans specifically.

Conclusion and recommendations

As demand for electricity and battery materials increases, so does the need for various critical metals. While such resources are found in deep-sea environments, mining activities there are constrained by significant environmental concerns. In addition, deep-sea mining is costly due to the distance from land and the high expense of extraction methods. As a result, interest in exploiting mineral resources located in shallow, near-shore marine areas have grown.

High-grade mineral deposits have been identified in these shallow coastal zones, where extraction is considerably more cost-effective and efficient than in deep waters or on land. However, the environmental risks in shallow marine areas are no less significant than in other ecosystems; a key challenge is the lack of adequate guidelines and regulatory frameworks. It is well established that mining in shallow marine environments affects seabed structures and benthic habitats. Consequently, such activities are also likely to have direct impacts on small cetaceans, for example through noise, as well as indirect impacts related to changes in prey availability and quality.

Below are some recommendations to improve impact assessment and to reduce or avoid potential adverse effects:

a) Increase research on the impacts of shallow-water marine mining on habitats and marine organisms and, through these effects, on small cetaceans
b) Study environmental impacts of different mining techniques, including noise
c) Determine rates, degrees and likelihood of habitat recovery, if recovery occurs at all
d) Develop guidelines based on acquired knowledge, including guidance for Environmental Impact Assessments (EIAs) that explicitly consider small cetaceans
e) Enhance post-impact studies to build a more robust understanding of mining impacts and increase the available evidence base. The additional insights gained should be used to improve guidelines, strengthen rules of procedure, and refine regulatory frameworks as knowledge evolves.
f) Enhance dialogue with mining operators to jointly identify areas of high biological importance, including those critical for small cetaceans, avoid activities in these areas, and develop methods to prevent or mitigate environmental impacts where activities take place, including environmental windows, impact thresholds, and life-observations of cetacean presence where needed.

Carbon capture and storage (CCS)

Carbon capture and storage (CCS) refers to technologies and processes designed to capture carbon dioxide (CO_2) from industrial processes or directly from the atmosphere and store it in geological formations to reduce the climatic impact of large-scale CO_2 emissions. The process typically involves three main stages: capture, transport, and storage (Bui et al., 2018; Krevor et al., 2023; Cames et al., 2024).

Captured CO_2 is typically compressed into a supercritical state, giving the gas the density of a liquid but the flow properties of a gas. This compressed CO_2 is then transported by pipeline or ship to specialised storage sites and injected deep underground through boreholes (Krevor et al., 2023). Offshore storage is increasingly preferred due to the large spatial requirements and regulatory constraints associated with onshore storage infrastructure. As a result, many CCS proposals focus on geological formations beneath the seabed (Bui, et al., 2018; Cames et al., 2024; Flöer et al., 2025).

Suitable storage sites generally include depleted or near-depleted oil and gas reservoirs, uneconomic coal deposits, or deep saline aquifers located beneath the seafloor. These formations must be overlain by at least one mechanically stable “caprock” layer with very low porosity and permeability to prevent upward migration of injected CO_2 (Krevor et al., 2023; Cames et al., 2024). In offshore settings, e.g. North Sea, injection typically occurs at depths of approximately 800 m or more below the seafloor within porous rock formations such as sandstone. (CDRmare 2025)

After injection, CO_2 drifts upward through pore spaces within the reservoir rock until it becomes trapped beneath the caprock. Several trapping mechanisms contribute to long-term storage: structural trapping, residual trapping, dissolution trapping, and mineral trapping/mineralization. These mechanisms operate over varying timescales — from within decades for structural and residual trapping, to centuries for dissolution trapping, and centuries to millennia for mineralisation — gradually stabilising the stored carbon (Bui, et al., 2018; Krevor et al., 2023; Pogge von Strandmann et al., 2019). (CDRmare 2025)

Although CCS is frequently proposed as a tool for climate mitigation, environmental risks must be considered. Particularly for offshore implementation, activities harmful to marine environments include site exploration (such as seismic surveys), infrastructure installation, potential leakage of stored CO_2 and formation fluids, and long-term monitoring operations. There is also a risk of formation water leaks, which is often highly saline and contaminated with pollutants, such as heavy metals. These activities may affect marine ecosystems and are therefore relevant to vulnerable species, including small cetaceans (Fendt et al., 2024; Cames et al., 2024; ECO2 Project, 2014).

Biological and ecological impacts

The ecological risks associated with offshore CCS arise from several phases of project development, including site exploration, infrastructure installation, injection operations, and long-term monitoring (Figure 10).

Impacts from seismic surveys

Before storage sites can be developed, geophysical surveys are conducted to identify suitable geological formations and assess their long-term stability. Offshore CCS will therefore involve seismic survey activity during exploration and monitoring phases (Cames et al., 2024; Fendt et al., 2024; IEAGHG, 2015).

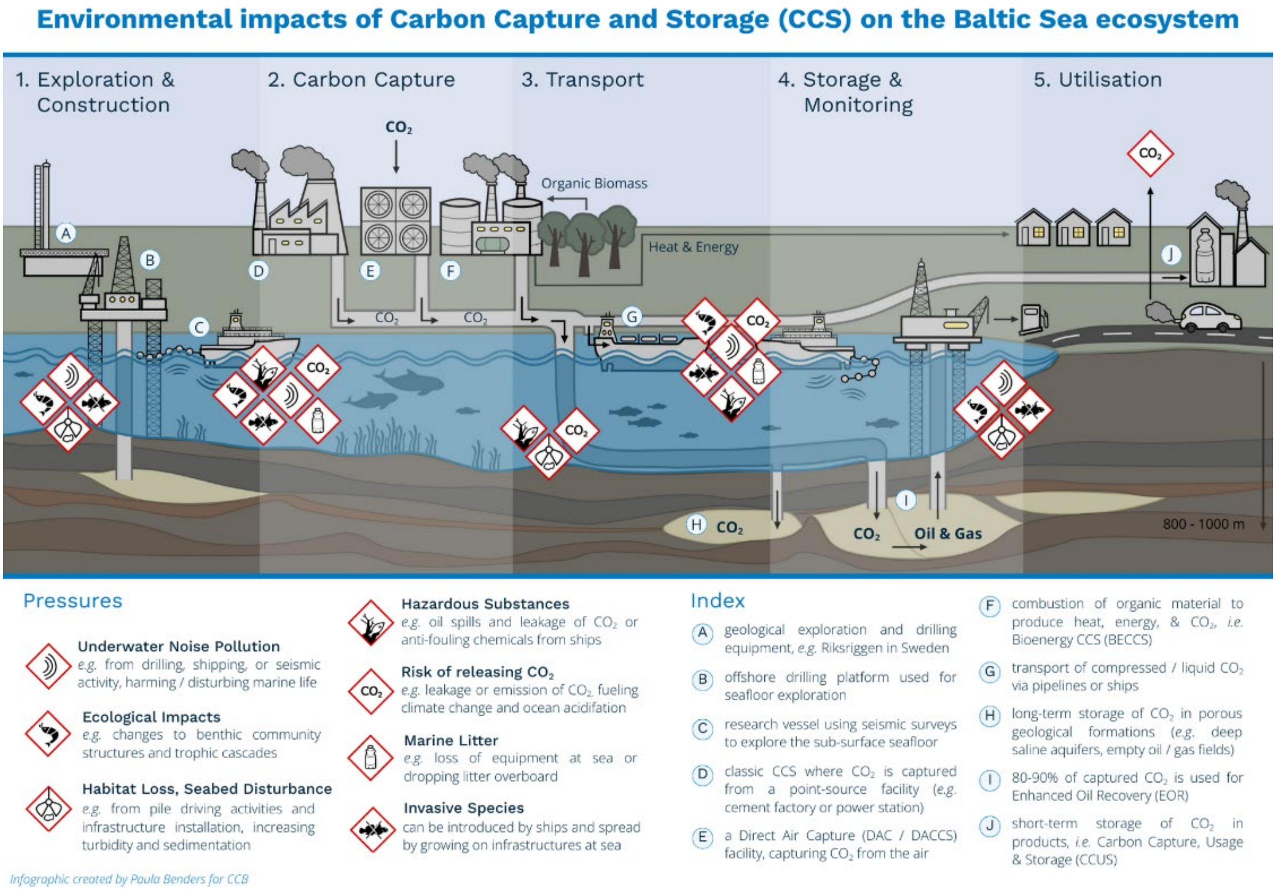

Figure 10. Environmental impacts of CCS on the Baltic Sea ecosystem. Infographic by Paula Benders, Coalition Clean Baltic, from https://www.ccb.se/keep-the-baltic-free-from-carbon-dumping .

Such surveys are known to produce intense underwater noise that can propagate over large distances and overlap with the hearing ranges of many marine species, including cetaceans. Noise exposure from seismic surveys has been associated with behavioural disturbance, displacement from habitat, disruption of communication and foraging behaviour, and, in extreme cases, temporary or permanent hearing impairment. For species protected under the ASCOBANS Agreement, such as the harbour porpoise (*Phocoena phocoena),* repeated exposure to seismic noise may reduce habitat suitability, particularly in key feeding or breeding areas (Fendt et al., 2024; Cames et al., 2024).

Impacts from CO_2 leakage

While CCS systems are designed to prevent leakage, geological uncertainties remain. Potential leakage pathways include faults, fractures in caprock layers, and poorly sealed abandoned wells (Krevor et al., 2023; Weber et al., 2023; Cames et al., 2024).

If CO_2 escapes into marine sediments or seawater, it can locally increase dissolved CO_2 concentrations and lower pH, resulting in localised acidification (Lichtschlag et al., 2021; ECO2 Project, 2014). Experimental and field studies have shown that elevated CO_2 concentrations can significantly alter benthic ecosystems, causing reductions in biodiversity, biomass, and microbial activity (Molari et al., 2018; Molari et al., 2019; Lichtschlag et al., 2021). Field studies near natural CO_2 seep sites have reported substantial declines in faunal biomass and changes in sediment biogeochemical processes under elevated CO_2 conditions (Molari et al., 2018; Rastelli et al., 2015).

Localised acidification may impair calcifying organisms, disrupt benthic food webs, and reduce the availability of prey species for higher trophic levels, including fish and marine mammals (Schade et al., 2016; Lichtschlag et al., 2021).

Behaviour of CO_2 in marine environments

Carbon dioxide behaves differently from many industrial pollutants when released into marine systems. CO_2-enriched seawater becomes denser than surrounding water masses and may accumulate near the seabed, particularly in stratified or semi-enclosed marine environments (Lichtschlag et al., 2021; ECO2 Project, 2014). This can prolong exposure for benthic organisms and alter local chemical conditions. Many marine organisms already operate near physiological thresholds due to ongoing ocean acidification; additional localised CO_2 inputs may therefore impair calcification, reproduction, larval survival, and ecosystem interactions (Schade et al., 2016; Lichtschlag et al., 2021; ECO2 Project, 2014).

Seabed disturbance and habitat impacts

Construction of offshore CCS infrastructure may involve drilling operations, installation of injection wells and platforms, subsea pipelines, anchoring systems, and monitoring equipment. These activities can disturb seabed habitats, resuspend sediments, and alter benthic communities Fendt et al., 2024; Cames et al., 2024; ECO2 Project, 2014). Although the physical footprint of individual installations may appear limited, cumulative impacts could become significant if multiple CCS storage sites are developed across a region such as the North Sea (Fendt et al., 2024; Cames et al., 2024).

Indirect and system-level ecological effects

Beyond localised environmental impacts, CCS deployment may also have broader ecological implications. The availability of CCS technologies may prolong reliance on fossil fuels by enabling the continued operation of carbon-intensive industries and infrastructure, thereby delaying emissions reductions. Such delays contribute to ongoing climate change, which poses major risks to marine ecosystems through warming, deoxygenation, acidification, sea-level rise, and increasing marine heatwaves and other (compound) extreme events (Fendt et al., 2024; Jacobson, 2019; Jacobson, 2025; Kazlou, Cherp & Jewell, 2024; IPCC, 2021). In addition, the substantial financial and energy investments required for CCS infrastructure may be diverted from mitigation strategies with more immediate and predictable benefits for marine ecosystems, such as renewable energy deployment or ecosystem restoration (Fendt et al., 2024; Jacobson, 2019; Jacobson, 2025). Finally, offshore CCS transfers part of the climate mitigation burden to long-term management of subsurface reservoirs beneath marine ecosystems. Monitoring, verification, and remediation of offshore storage sites may remain necessary for decades after injection ceases, creating long-term environmental governance challenges (Fendt et al., 2024; Cames et al., 2024).

Impacts on cetaceans

The principal impacts of offshore CCS activities on cetaceans are likely to arise from underwater noise generated during exploration, construction, and monitoring operations, as well as from indirect ecosystem changes affecting habitat quality and prey availability (Fendt et al., 2024**;** Cames et al., 2024**;** ECO2 Project, 2014)**.**

Direct impacts

Cetaceans may be directly exposed to noise from seismic surveys during site exploration and monitoring, from vessel traffic associated with construction and maintenance activities, and from drilling and installation operations (Fendt et al., 2024; Cames et al., 2024). Such noise can interfere with echolocation, communication, and foraging behaviour. In species such as harbour porpoises, which rely heavily on high-frequency echolocation for prey detection, noise disturbance may reduce feeding efficiency or cause displacement from important habitats (Fendt et al., 2024; Cames et al., 2024; Sarnocińska et al., 2020).

Indirect impacts

CCS infrastructure may also affect cetaceans indirectly through ecosystem changes. Potential indirect effects include changes in prey availability due to benthic ecosystem disturbance (Molari et al., 2018; Molari et al., 2019; Schade et al., 2016; Lichtschlag et al., 2021), habitat fragmentation caused by offshore infrastructure and increased vessel traffic (Fendt et al., 2024; Cames et al., 2024), and altered ecosystem dynamics resulting from localised acidification or sediment disturbance (Molari et al., 2018; Molari et al., 2019; Schade et al., 2016; Lichtschlag et al., 2021). Because many cetaceans occupy high trophic levels and depend on healthy prey populations, ecosystem-level disturbances may have cascading effects on their populations (Schade et al., 2016; Lichtschlag et al., 2021; ECO2 Project, 2014).

Overview of activities taking place within the ASCOBANS Agreement area

Belgium

Belgium is among the countries that have accepted the 2009 amendment to the London Protocol. As of 20 August 2024, Belgium was also among the European Economic Area (EEA) governments that had submitted a declaration of provisional application of the 2009 amendment, permitting the movement of CO_2 across national boundaries for the purpose of sub-seabed geological sequestration (Levina, 2024).

Denmark

Denmark already has CCS projects in various stages of construction or operation. Project Greensand, Denmark's first offshore CO_2 storage initiative, began injection of CO_2 in the North Sea in 2023. It aims to store up to 1.5 million tonnes of CO_2 annually by 2026 and up to 8 million tonnes by 2030 (Flöer et al., 2025). The general use of CCS is regulated in the Danish Subsoil Act (Undergrundsloven). Additionally, Denmark filed an executive order in 2024, which further regulates CCS operations in the country (Bekendtgørelse om geologisk lagring af CO_2 (BEK nr. 823 af 26/06/2024)).

Germany

Following the amendment to the Carbon Dioxide Storage Act (KSpG) in 2025, the now so-called Carbon Dioxide Storage and Transport Act (KSpTG), enables CCS on an industrial scale and storage in the continental shelf of the German EEZ in the North Sea and Baltic Sea, whereas previously only research storage had been permitted. The coastal seas are excluded.

The focus of German CCS activities has so far been primarily on the EEZ of the North Sea, with comparatively little attention paid to the Baltic Sea; research projects have likewise focused exclusively on the North Sea. There has been little research into possible geological storage structures in the Baltic Sea to date (most recent data Anthonsen et al. 2014).

Under the KSpTG, injection facilities are not permitted to be in Marine Protected Areas (MPAs), and storage facilities must be located outside of the MPAs' 8 km buffer zone. In addition, to protect the harbour porpoise, pile driving and noise-intensive seismic surveys are prohibited during the sensitive period (May-August) in areas of main harbour porpoise concentration or within a distance of less than 8 km of such areas. In addition to the KSpTG, the HSEG (High Seas Importation Act), was amended, and enforcement procedure is ongoing. regulates the export of CO_2 for storage in other countries.

In May 2024, the German federal government published the “cornerstones” of its national Carbon Management Strategy, committing to enable CCU/S for all emission sources except coal heating and power plants, with a focus on hard-to-abate industrial emissions. The strategy is party implemented but is part of an ongoing transformation process that includes further analyses, stakeholder dialogues, and implementation efforts. In August 2024, a new funding programme, “Federal Funding for Industry and Climate Protection (BIK)”, open to CCU/S applications, was introduced (Flöer et al., 2025).

The Netherlands

The Netherlands currently has two major CCS projects under implementation: Porthos and ARAMIS. The Porthos project involves an offshore pipeline from the Port of Rotterdam to depleted gas fields around 20 km off the North Sea coast, where CO_2 will be stored at a depth of approximately 3 km; the project is under construction and is expected to begin storing approximately 2.5 million tonnes of CO_2 per year for 15 years, adding to a total of ~37 million tonnes. The ARAMIS project is focused on building CO_2 transport and storage infrastructure, with the final investment decision expected in 2025/2026, storage is expected to begin in 2028/2029, and injection capacity planned to expand to 22 million tonnes per year by 2030. The Netherlands has also established a financial framework for scaling up CCS through its SDE++ subsidy programme (Flöer et al., 2025). For any CCS project, permits are required pertaining the Environment and Planning Act, Mining Act for pipeline, transport and storage, project decision and Environmental Impact Assessment.

Norway

Norway is one of the most advanced countries implementing CCS in Europe, with projects such as Sleipner and Snøhvit. The Longship project is intended to demonstrate the full CCS chain and includes the Northern Lights transport and storage component, beginning operation in 2025 with 1.5 million tonnes of CO_2 stored each year in its first phase. Northern Lights is expected to receive CO_2 from Yara’s ammonia plant in the Netherlands, Brevik cement plant in Norway, and from Ørsted’s two biomass power plants in Denmark from 2026 onwards. In its second phase, Northern Lights aims to store 5 million tonnes per year. The Brevik CCS project, also operating since 2025, is the first cement plant in Europe with a large-scale CO_2 capture unit installed (Flöer et al., 2025).

Poland

Poland is a Party to the London Convention only and is not listed among the countries that have accepted the 2009 amendment to the London Protocol (Levina, 2024).

Sweden

Sweden is among the countries that have accepted the 2009 amendment to the London Protocol. As of 20 August 2024, Sweden was also among the EEA governments that had submitted a declaration of provisional application of the 2009 amendment to allow movement of CO_2 across national boundaries for the purpose of sub-seabed geological sequestration (Levina, 2024). The Swedish government has initiated surveys to find suitable areas for CO_2 storage in the Swedish EEZ of the Baltic Sea, including overlapping with important areas for harbour porpoises. Permitting procedures for the surveys are currently ongoing.

United Kingdom

The United Kingdom already has CCS projects in various stages of development. The UK government has set a goal of capturing 20 to 30 million tonnes of CO_2 per year by 2030 and plans four major CCU/S clusters supported by up to GBP 22 billion in government funding: HyNet, the East Coast Cluster, Viking CCS, and Acorn. HyNet is planned to store 4.5 million tonnes of CO_2 per year; the East Coast Cluster 23 million tonnes by 2035; Viking CCS 10 million tonnes per year; and Acorn 5 to 10 million tonnes per year. Overall, 27 appraisal and storage licences are currently on the UK Continental Shelf (Flöer et al., 2025).

The Baltic Sea region

In the Baltic Sea region, the Helsinki Convention currently prevents offshore CO_2 storage. Article 11 of the Convention prohibits dumping in the Baltic Sea Area except for dredged material, and the definition of dumping encompasses disposal into the seabed. Because CO_2 is not listed as an exception, offshore CO_2 storage in the Baltic Sea Area is currently prohibited.

In June 2023, some Contracting Parties held an informal meeting to discuss a possible amendment of Article 11 in the context of carbon storage, acknowledging that the Article in its current form prohibits this activity. Any amendment would require acceptance by all Contracting Parties (Levina, 2024). In addition, at the time of the publication of this interim report (July 2026), there is an ongoing process within HELCOM to both look at the legal possibilities for CCS under the convention, and the ecological impacts of CCS. In the Baltic Sea region, environmental governance is coordinated through the Helsinki Commission (HELCOM) under the Convention on the Protection of the Marine Environment of the Baltic Sea Area (the Helsinki Convention).

EU initiatives

At EU level, the European Industrial Carbon Management Strategy, published in February 2024, expects that by 2030 at least 50 million tonnes of CO_2 will be captured each year. The strategy identifies CO_2 infrastructure as a key factor and notes that an EU-level regulatory package for cross-border CO_2 transport is under development. The EU Net-Zero Industry Act (NZIA) has also set a target of 50 million tonnes per year of CO_2 injection capacity by 2030. In addition, an investment atlas for potential CO_2 storage sites is expected by 2026 (European Commission, 2024a; Flöer et al., 2025).

Research and available guidelines relevant to CCS activities

Research initiatives on CO_2 storage in the Baltic and North Sea regions

Several research initiatives have investigated the feasibility and potential environmental implications of geological carbon storage beneath European seas, including the Baltic Sea and North Sea (O'Neill et al., 2014; GEOSTOR, n.d.; Flöer et al., 2025).

The BASTOR project assessed potential carbon dioxide storage sites in the southern Baltic Sea and estimated a regional theoretical storage capacity of approximately 16 Gt of CO_2 within Middle Cambrian sandstone formations beneath thick caprock layers. Dynamic modelling indicated, however, that the relatively limited permeability and porosity of some formations, particularly in the Swedish sector of the Dalders Monocline, could constrain injection rates and reduce practical storage potential. The study also evaluated possible failure mechanisms of geological seals, identifying three potential pathways for CO_2 migration: failure of the top seal, migration along bounding faults, and leakage across fault planes. Although these risks were considered low based on available geological data, the study emphasized that further site-specific geological and geophysical investigations would be required prior to commercial deployment (O'Neill et al., 2014).

In the Baltic States, geological research has also identified potential storage formations within the Baltic Sedimentary Basin, particularly in porous Cambrian sandstone formations capable of storing CO_2 in a supercritical state. Some structures in Latvia have been identified as having significant storage capacity, although additional geophysical surveys and reservoir characterization are necessary to confirm their long-term suitability (O'Neill et al., 2014). In the North Sea region, the GEOSTOR research consortium has investigated the feasibility of storing CO_2 in geological formations beneath the German North Sea. The project identified sandstone formations located approximately 1–4 km beneath the seabed, overlain by thick impermeable layers such as claystone or halite that could act as caprock barriers and prevent upward migration of stored CO_2. These formations were identified as potential storage reservoirs, although further site characterization and testing would be necessary before operational deployment (GEOSTOR, 2025). The GEOSTOR research program also examined environmental risks associated with CCS deployment, including potential leakage pathways along faults or abandoned wells, possible induced seismicity, and the need for monitoring technologies capable of detecting subsurface changes and micro-seismic activity associated with CO_2 injection (GEOSTOR, 2025).

These findings are relevant to the ASCOBANS Agreement area, as they suggests that offshore storage infrastructure and associated transport networks in the North Sea are likely to expand further in the near-term.

Regulatory frameworks governing offshore CO_2 storage

European Union regulatory framework

Within the European Union, geological storage of carbon dioxide is governed primarily by Directive 2009/31/EC on the geological storage of carbon dioxide (the EU CCS Directive). The directive establishes requirements for the entire CCS chain, including site selection, environmental risk assessment, monitoring, reporting, and long-term liability. Member states remain free to choose whether to allow geological storage of CO_2 in their territories, exclusive economic zones, and continental shelf, but if they do allow it, they must comply with the Directive (Directive 2009/31/EC; European Commission, 2023; Levina, 2024).

No exploration may take place without a permit, and the suitability of a geological formation must be determined through thorough characterisation and assessment of the potential storage complex and surrounding area, including modelling of CO_2 injection, risk assessment, identification of potential leakage hazards, and consideration of surrounding habitats and species as well as potential environmental and health impacts. Once exploration activities are concluded, operators must obtain a storage permit from the designated national authority, and draft permits must be forwarded to the European Commission for review. Storage permits are reviewed five years after issuance and thereafter every 10 years. Operators must develop corrective measures and contingency plans to address any potential leakage or environmental impacts, and long-term monitoring obligations remain in place until authorities determine that the stored CO_2 will remain permanently contained (Directive 2009/31/EC; Levina, 2024).

CCS projects are also required to undergo environmental impact assessment under the EU Environmental Impact Assessment Directive. The assessment framework takes account of project characteristics, location, and the type and characteristics of the potential impact. Developers must consider surrounding habitats and species and analyse potential environmental and health impacts when assessing whether a geological formation is suitable for storage. The permitting guide further notes that several EU, international, and national instruments may need to be considered together when applying for a CO_2 storage permit, including the Environmental Liability Directive, the TEN-E Regulation, the revised EU ETS Directive, the NZIA, the London Protocol, OSPAR, and relevant regional conventions. (Levina, 2024)

Directive 2011/92/EU of the European Parliament and of the Council of 13 December 2011 on the assessment of the effects of certain public and private projects on the environment, as well as national regulations implementing that directive, also apply. These may determine the refusal to approve a specific CCS activity if the authority concludes that the risks associated with such an activity are excessive and the risk mitigation measures proposed by the applicant are insufficient.

HELCOM and Baltic Sea governance

The Baltic Sea is recognised as a particularly sensitive marine ecosystem subject to multiple environmental pressures, including eutrophication, pollution, and habitat degradation. As a result, HELCOM has historically taken a precautionary approach toward activities that could introduce additional environmental risks.

Recent discussions within HELCOM working groups, including the Working Group on Sea-Based Pressures (WGSEA), have considered the need for regional environmental risk assessments related to carbon capture and storage activities. In 2025, the working group supported initiating a structured environmental analysis of the potential environmental and cumulative impacts of CCS in the Baltic Sea context, including the establishment of an expert correspondence group to advance this work. These discussions reflect growing recognition that any future CCS activities in the Baltic Sea would require comprehensive environmental assessment and regional coordination among Baltic Sea states.

The permitting guide notes that the Helsinki Convention is stricter than the London Protocol and other regional sea conventions in this respect. Article 11 prohibits dumping in the Baltic Sea Area except for dredged material requiring a prior special permit, and because CO_2 is not listed as an exception, offshore CO_2 storage in the Baltic Sea Area is currently prohibited.

OSPAR

OSPAR has adopted a Decision (2007/2) to ensure safe storage of carbon dioxide streams in geological formations together with guidelines for risk assessment and management of storage of CO_2 streams in geological formations (OSPAR Agreement 2007-12). OSPAR has also adopted a Decision (2007/1) to prohibit the storage of carbon dioxide streams in the water column or on the seabed, because of the potential negative effects. OSPAR has also published a report on impact assessment of the offshore oil and gas industry on the marine environment. There, it is noted that impacts from CCS are expected to be similar to pressures from offshore oil and gas activities, but that further research is needed. CCS monitoring will be reviewed in 2026 (Marappan et al., 2022).

The 2025 OSPAR Ministerial Meeting agreed a process to future-proof the OSPAR Convention by examining how the Convention can evolve to respond to new and emerging pressures, including carbon dioxide capture and storage (CCS) and hydrogen production and storage (HPS) linked to offshore oil and gas. Amending the Convention will allow OSPAR to take measures, where they are needed, to protect the OSPAR Maritime Area from these new, emerging and increasing pressures.

OSPAR 2026 agreed that the proposed amendments in relation to the storage of carbon dioxide will not be limited to offshore oil and gas industry and that another meeting cycle is required to further develop the work on potential amendments of the Convention.

In relation to CCS and HPS, a first draft has been prepared and will need further consideration to address pending issues and take into account the amendments on renewables

London Convention / London Protocol

The Contracting Parties to the London Convention and London Protocol have established international regulations for marine geoengineering and atmospheric carbon dioxide mitigation through sub-seabed geological storage. In 2006, they legally authorized and regulated permanent carbon capture and storage in sub-seabed formations for large emission sources, excluding enhanced oil recovery. To facilitate international cooperation, a 2009 amendment allowed countries to export carbon dioxide to share transboundary storage sites, though this requires formal acceptance by two-thirds of the parties to officially enter into force. To bridge this gap, a 2019 resolution approved the provisional application of the export rules, enabling countries to immediately export carbon dioxide for sub-seabed storage provided they submit formal declarations and bilateral agreements to the International Maritime Organization as an interim measure.[1]

[1] https://www.imo.org/en/ourwork/environment/pages/ccs-default.aspx

Available guidelines and risk management measures for CCS activities

Although CCS activities remain relatively limited in European marine environments, several guidelines and risk management principles have been identified through existing regulatory frameworks and research programs.

According to the permitting guide[2] (Levina, 2024), before a storage permit application can be launched, many countries require an exploration permit to establish whether storage is possible in terms of suitability and capacity. Detailed technical information must be collected on storage capacity, geology and containment, injectivity, monitoring technologies and approaches, and risk assessment (Levina, 2024).

Research programs such as GEOSTOR and BASTOR emphasise the importance of detailed geological characterization prior to CO_2 injection. In the North Sea and Baltic Sea regions, potential storage formations are typically porous sandstone reservoirs overlain by thick impermeable caprock layers that can act as geological barriers. Proper site characterization must include geophysical surveys, reservoir modelling, and evaluation of potential leakage pathways along faults or abandoned wells. Also, the proximity of Marine Protected Areas (MPAs) should be considered (and avoided).

Long-term monitoring is considered essential for ensuring the safety of geological CO_2 storage, and the GEOSTOR programme specifically investigated potential leakage pathways and geotechnical risks associated with CO_2 storage beneath the German North Sea, highlighting the need for careful evaluation of subsurface structures and historical drilling activity before selecting storage sites.

Monitoring programmes include seismic monitoring systems, seabed observations, and reservoir pressure measurements to detect changes within the storage formation and identify possible migration of injected CO_2. Such systems are designed to detect natural or injection-related seismic events in near real time, enabling operators to respond rapidly to unexpected changes in reservoir behaviour. The permitting guide sets out that the storage site development process includes operation, closure, monitoring during operation and post-closure, and long-term stewardship. Monitoring is therefore framed not only as an operational requirement but as part of the long-term management of geological CO_2 storage (Levina, 2024).

Offshore CCS projects must also consider potential conflicts with other uses of the marine space. Research conducted within the GEOSTOR program identified possible interactions between CO_2 storage projects and existing activities such as offshore wind energy development, shipping routes, fisheries, and marine conservation areas. Effective integration of CCS projects into marine spatial planning frameworks is therefore necessary to avoid conflicts with other marine activities and to ensure that environmentally sensitive areas are appropriately protected. Natura 2000 areas and wider environmental legislation are identified as relevant considerations in the permitting process (Levina, 2024)

Conclusions

[coming later]

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[2] https://www.globalccsinstitute.com/wp-content/uploads/2025/08/Permitting-in-Europe-Report-9.12-FINAL.pdf

Sand and gravel (marine aggregate) extraction

This chapter first outlines the most commonly used dredging vessel types for marine aggregate extraction and the associated noise emissions, then details the amount of marine aggregates currently extracted in ASCOBANS Party States and subsequently discusses the biological and ecological impacts of the activity and how it may affect small cetaceans.

We define marine aggregates, in accordance with Tillin et al. (2011), as "[a] mixture of sand, gravel, crushed rock or other bulk minerals used in construction and civil engineering."

Dredging vessel types and associated noise emissions

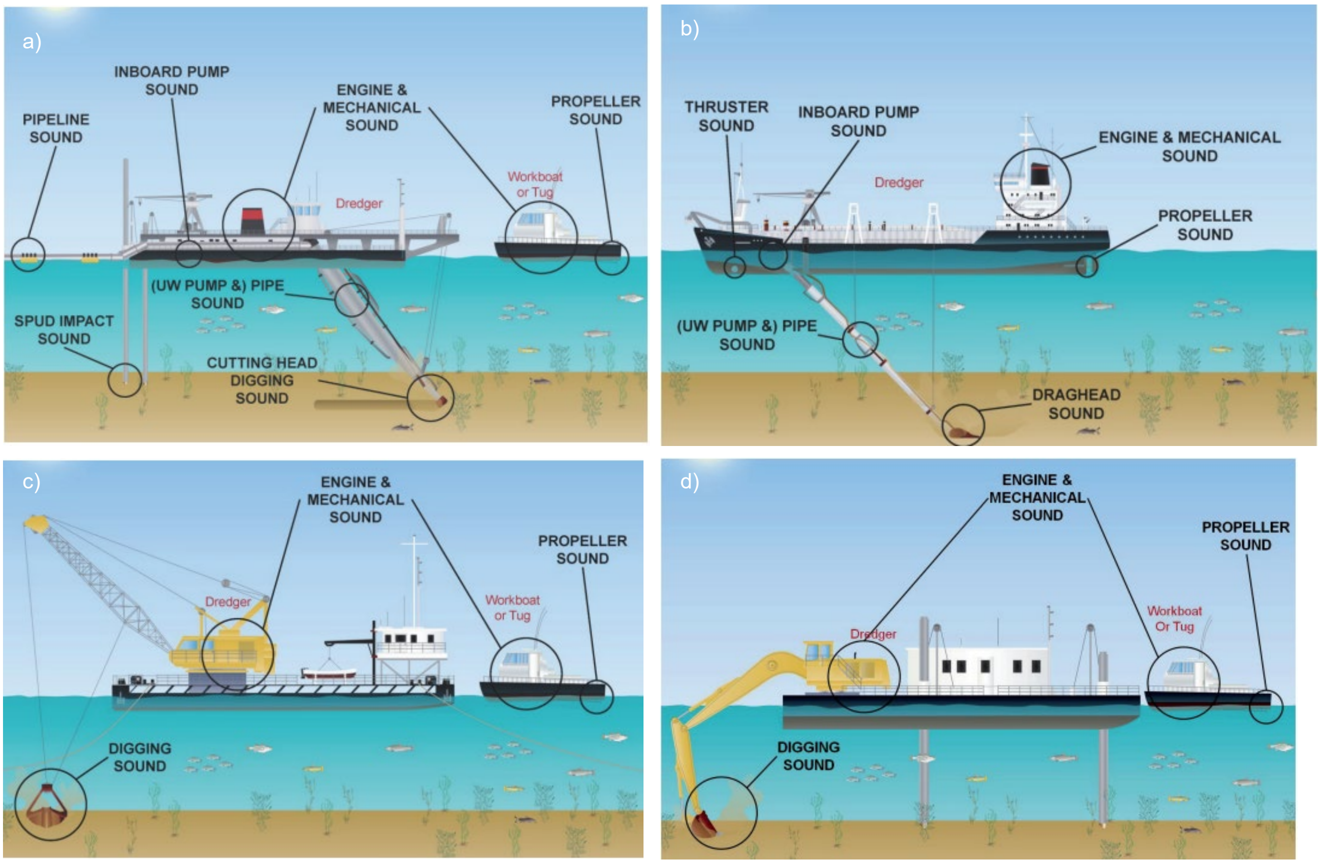

Figure 11. Sound sources emitting from a a) cutter suction dredger (CSD), b) trailing suction hopper dredger, c) grab dredger, and d) backhoe dredger. (Images sourced from CEDA, 2011, available at: https://dredging.org/media/ceda/org/documents/resources/cedaonline/2011-11_ceda_positionpaper_underwatersound_v2.pdf)

Marine aggregate extraction within ASCOBANS Party and Range States

The extraction and use of marine aggregates of the signatory Parties to the ASCOBANS Agreement between 2000 and 2024 is summarised in Figure 12.

Figure 12. Marine aggregate extraction of ASCOBANS Party States between 2000 and 2024, with their intended use (data source: the North Atlantic Marine Aggregates Application (NAMAAP) from ICES WGEXT).

Biological and ecological impacts of marine aggregate extraction in general

The main disturbances to marine wildlife from sand and gravel extraction likely result from the direct removal of superficial seabed sediment, damage to the seabed integrity, short-term changes to the water column (e.g., increased turbidity, release of nutrients and chemicals, including toxins and pollutants), and re-deposition of suspended sediments, which can smother flora and fauna through deposition of fine sediment from both surface and bottom sediment plumes (Newell et al., 1998; Desprez et al., 2010; Le Bot et al., 2010; Duclos, 2012; ICES, 2016).

The main group of organisms impacted are benthic and epibentic animals (as light levels at extraction sites may often be insufficient to allow plant or seaweed growth). Sandy habitats are generally more mobile than those on coarse sediments and characterised by smaller, shorter-lived species typical of environments exposed to disturbances, such as bristle worms, bivalve molluscs, crustaceans, and burrowing sea urchins (Tillin et al., 2011). These finer sediments tend to support less species richness and productivity than coarse sediments (Desprez et al., 2009). Gravel and coarse sediment habitats characterised by low sediment mobility support colonial species that attach to harder surfaces such as hydroids and bryozoans as well as mobile epifauna such as crustaceans (crabs, amphipods, isopods, shrimps), whelks, and scallops (Tillin et al., 2011).

Since few benthic invertebrates are able to escape entrainment (the unintentional removal of organisms within a hydraulic or mechanical dredger's suction field; Reine and Clarke, 1998; Todd et al., 2015), it is assumed that aggregate dredging has a direct impact on benthic communities within the dredge footprint. Studies found that, within the dredged area, species diversity can be reduced by 30-70% and biomass and animal abundance by 40-95% following activities (Newell et al., 1998). Such a reduction in benthic invertebrate biomass represents a loss of food to animal predators, including other benthic species, fish, and some marine mammals (Tillin et al., 2011).

Extraction from sand and gravel deposits can also lead to entrainment of fish, fish eggs, and larvae that either live in these habitats or use the seabed for spawning (e.g., herring, sand eels, plaice, sole, or black bream; Defra, 2002). In general, entrainment rates depend on several factors including depth, dredger type, dredging speed, and strength of the suction field, the latter of which is more pronounced - and therefore more of a risk for marine life - in hydraulic than mechanical dredgers (Reine and Clarke, 1998; Nightingale and Simenstad, 2001; Todd et al., 2015). Susceptibility is also species-specific. Benthic fauna and fish associated strongly with the seabed and dredged substrates are considered more at risk from entrainment than highly mobile species (Todd et al., 2015). Similarly, young fish, eggs and larvae of marine organisms are considered at higher entrainment risk because of their immobility or reduced swimming ability which doesn't allow them to actively avoid the suction field (Reine and Clarke, 1998; Nightingale and Simenstad, 2001; Drabble, 2012b). Dredging can therefore affect survival rates of organisms to adulthood, and hence population structure and growth (Todd et al., 2015).

A study around dredging activities in the eastern English Channel between 2005 and 2008, for example, documented both a reduction in abundance and changes in population structure of common sole (*Solea solea*), lemon sole (*Microstomus kitt*), thickback sole (*Microchirus variegatus*), and European plaice (*Pleuronectes platessa*) (Drabble, 2012a). After the first year of dredging, few of the before-abundant age 1 sole had survived to reach age 2, and after two years of dredging, fish born before the dredging dominated the samples (Drabble, 2012a). For plaice, results looked similar. Entrainment was considered an explanatory factor, if not necessarily the only one, while natural conditions alone could not account for the alterations (Drabble, 2012a; Todd et al., 2015). As effects are considered to be worst for eggs and larval stages, impacts of dredging can be reduced by implementing and enforcing temporal restrictions on dredging activities (known as “environmental windows”), to restrict activities in spawning and nursery grounds at critical times (Todd et al., 2015). While entrainment rates are mostly recognised as influenced by species traits such as mobility, avoidance reactions and habitat preferences, they can also vary with seasonality (Tillin et al., 2011). Over-wintering crabs, for example, often exhibit low activity and might therefore be unable to avoid a draghead during that period (Royal Haskoning, 2005).

Despite an increased recognition of their important role in the ecological functioning of marine ecosystems, microbial components of the seabed and meiofauna (benthic organisms smaller than 1mm) have not received significant scientific attention in recent assessments of aggregate extraction impacts (Tillin et al., 2011).

Regarding physical disturbances to benthic habitats, a dredging-inflicted change in sediment structure (most commonly a fining of the sediment) has been reported frequently in the literature (e.g., Kenny and Rees, 1996; Desprez, 2000; van Dalfsen et al., 2000; Weller et al., 2002; Hitchcock and Bell, 2004; Boyd et al., 2005; Cooper et al., 2005; Robinson et al., 2005; Desprez et al., 2010; Barrio Froján et al., 2011; Todd et al., 2015). Van Lancker et al. (2014), studying the Hinder Banks of Belgium, also reported an observable increase in mud, increased sandification (which, however, can be buffered by certain types of porous seabed, such as permeable coarse sands and gravel), and a decrease in long-lived organisms of the epibenthos, which need longer disturbance-free periods to establish themselves and grow. The last trend might also have been influenced by more intense fishing (van Lancker et al., 2014).

In addition to direct impacts caused by the draghead, sediment plumes can have wide-ranging impacts. They are formed by the draghead on the seabed, during the loading of sediment onto the vessel, and/or by discharges following onboard aggregate sorting (Tillin et al., 2011).

The deposition from sediment plumes can smother or bury marine organisms associated with the seabed. These impacts are highly species-specific and depend for example on a species' ability to tolerate or escape burial, which can vary with e.g., sediment type and temperature (e.g., Last et al. 2011; Todd et al., 2015). A smothering of organisms' eggs can cause mass mortality or delayed hatching (e.g., Berry et al., 2011), and accumulating sediment can reduce the availability of areas suitable for settlement for larvae, which might increase competition (e.g., Wilber et al., 2005; Todd et al., 2015).

Turbidity created by the plumes has the potential to impact the feeding ability of fish through e.g., visibility, with piscivorous fish likely to be more affected than planktivorous fish that visually detect their prey over shorter distances (Hecht and van der Lingen, 1992; Utne-Palm, 2002; de Robertis et al., 2003). Turbidity could also affect habitat choices (e.g., Wenger and McCormick, 2013), predator-prey relationships (e.g., Wenger et al., 2013) and increase anti-predator responses (Leahy et al., 2011). High concentrations of suspended sediments can negatively affect gill functioning in fish (Herbert and Merkens, 1961; Lake and Hinch, 1999; Au et al., 2004; Wong et al., 2013) and respiration rates in invertebrates due to clogging, lead to behavioural changes, and reduce the efficiency of filter-feeders faced with a lower food-to-sediment-ratio (e.g., Last et al., 2011; Todd et al., 2015).

There are indications that high concentrations of suspended sediments coming into contact with marine organisms' eggs within two hours of their disposal can increase the attachment between eggs, disrupt larval development, and compromise the eggs' ability to attach to surfaces (Griffin et al., 2009). Outside of the initial two hours, the effects of suspended sediments on egg hatching success were (under laboratory conditions) limited (Auld and Schubel, 1978; Kiørboe et al., 1981). In the wild, current strengths, water temperatures, contaminant levels, differences between species' eggs, and other factors could influence these effects (Todd et al., 2015).

Todd et al. (2015) observe that the loss of sensitive, crucial habitats such as seagrass beds as a consequence of dredging can be substantial. For instance, a review of 45 case studies worldwide found that 21,023 ha of seagrass beds were lost because of 26 dredging projects over a 50-year period (Erftemeijer and Lewis, 2006; Todd et al., 2015). The number is likely an underestimation, since there were other projects that did not assess the extent of loss (Erftemeijer and Lewis, 2006). Seagrass beds represent an indispensable part of ecosystems, primarily due to their high rate of primary productivity, nutrient cycling, their role in sediment stabilisation (Duarte, 2002; Orth et al., 2006), and through providing habitat, nursery grounds, and shelter for a variety of marine organisms and communities (Heck et al., 2003; Todd et al., 2015). Changes will therefore affect at least some elements within the food web, for varying lengths of time, and the capacity for recovery will be contingent on the species concerned and the extent of loss or damage sustained (Erftemeijer & Lewis, 2006). The factors influencing survival include smothering, burial depth, sediment type, nutrient load, and the presence of pollutants (Zieman and Zieman, 1989; Wilber et al., 2005; Erftemeijer and Lewis, 2006).

In recent years, the dredging-related impacts on seagrass beds have generally declined in comparison with the past (Erftemeijer and Lewis, 2006; Todd et al., 2015). This phenomenon may be attributed to an increase in knowledge, the capacity to predict the trajectories of sediment plumes, and the implementation of well-designed and effective mitigation measures (Erftemeijer and Lewis, 2006; Todd et al., 2015).

Oyster beds and other species that form large beds are similarly important components of their ecosystems. They provide habitat, shelter, and colonisation opportunities to other marine life, and can also be at risk from dredging-related smothering (Wilber et al., 2005). It is crucial that each case is assessed individually, taking into account the present sensitive environments, to ensure that dredging activities are designed to generate the least possible impacts (Todd et al., 2015).

In general, Tillin et al. (2011) summarise that the magnitude of the loss of seabed habitats and its concomitant consequences for benthic organisms and demersal fish populations is contingent upon the intensity and spatial scale of dredging activities. The ICES (2016) report concludes that, given the strong association between the distribution of marine organisms and hydrodynamic, morphological, and sediment parameters (McLusky and Elliott, 2004; Baptist et al., 2006; Degraer et al., 2008; Pesch et al., 2008), any physical changes in a seabed's features will lead to a response in the composition of its benthic assemblage. This will have consequences for the quality of habitat in a wider area, as well as for the dispersal and survival of fish larvae and eggs, and for the abundance of food available for e.g., marine mammals (ICES, 2016).

The recovery of dredging sites strongly depends on the nature and intensity of the extraction and hence the degree of initial disturbance to the seabed habitat and benthic assemblages (Tillin et al., 2011). Recovery time may depend on substrate type (and its mobility rate pre-extraction), the type of ecosystem (and its resilience or adaptability to disturbances), and the location of the site.

Further impacts on marine ecosystems and organisms from aggregate dredging can arise from extraction-related shipping activities such as vessel movements, boat noise, and vibrations created by the extraction process (e.g., Cefas, 2003; Tillin et al., 2011; ICES, 2016). Their impacts on higher trophic levels, such as marine mammals, are generally considered minor and temporary (see below). However, cumulative and synergistic effects can arise and affect various trophic levels through impacts on those species' prey and habitat, e.g., when a number of dredging sites are closely located to each other or to a species' key ecological habitat, when other anthropogenic impacts exist at the (or close to the) dredging site, or because removal of marine aggregates tends to be focused on a relatively small number of habitat types (Tillin et al., 2011). It is also important to consider wider activities occurring in the marine environment which can add additional environmental pressure. For example, fisheries using bottom trawling or construction activities related to e.g., pipelines, offshore energy production, or cable laying cause further disturbances to the seabed and cumulatively lead to a much bigger area of seabed disturbance than dredging might cause alone (e.g., Tillin et al., 2011).

Furthermore, the extraction processes depend on the availability of suitable wharves and jetties at which cargo can be landed. Those structures and their operational use can give rise to further environmental impacts such as habitat loss or deterioration, and disturb wildlife (Tillin et al., 2011).

Impacts on cetaceans

Despite existing concern, not much is known about the exact impacts of marine aggregate extraction on marine life, including cetaceans (Todd et al., 2015). Responses of marine mammals to disturbance caused by sand and gravel extraction may vary both temporarily and spatially among groups and include avoidance, tolerance and other behavioural changes, and may depend also on the quality of the occupied site, the distance to the disturbance, availability and quality of alternative habitat, predation risks, the density and presence of competitors, or investments made by the individual or group in a site before the disturbance happened (Gill et al., 2001; ICES, 2016).

Direct impacts

The main direct impacts on cetaceans have been related to a) collisions with vessels, which are considered possible but unlikely given the slow speed to the dredging ships, and b) noise emitted from the operation of vessels and the mechanical activity.

Collisions

Vessel movements are associated with every stage of marine aggregate dredging, from accessing the extraction site and frequently the extractive activities themselves to movements between extraction sites and dumping grounds and/or landing infrastructure. As a consequence, collisions with dredgers are possible, but only one incident elsewhere in the world, leading to the death of a southern right whale (*Eubalaena australis*) calf, has been documented in the literature so far (Best et al., 2001; Todd et al., 2015). Studies on collisions between ships and cetaceans document that both collision risks and severity of injuries generally increase when marine vessels exceed speeds of 10-14 kn (Laist et al., 2001; Vanderlaan and Taggart, 2007; Gende et al., 2011; Neilson et al., 2012; Lammers et al., 2013); the vessel size and type, on the other hand, prove less significant (Todd et al., 2015). Resting or feeding whales, as well as calves and juveniles, seem to be more at risk (e.g., Laist et al., 2001; Van Waerebeek et al., 2007; Neilson et al., 2012; Lammers et al., 2013). Panigada et al. (2006), for example, reported increased collision rates of fin whales during months when intensive feeding occurs, possibly due to the animals being distracted, and less focussed on vessel movements (Todd et al., 2015).

Since most marine aggregate dredgers operate either stationary or at slow speeds (at around 1-3 kn), the associated collision risk is considered minimal if dredging is well managed, avoids critical habitats, times when animals may be distracted, and areas where calves are abundant (Todd et al., 2015). Collision risks may be greater for dredgers in transit as they can reach speeds around 12-16 kn (Brunn et al., 2005; Todd et al., 2015). Nevertheless, collision risk is thought unlikely to be substantially increased by the additional presence of dredging vessels, as they typically represent only a minor proportion of overall shipping activity. Yet, Regional Environmental Assessments (REAs; a framework with which the potential cumulative and in-combination effects of regional marine aggregate extraction proposals are investigated in the UK) and similar processes and resources should be referred to for region-specific evaluations of this proportion and possible associated increases in collision risk (Tillin et al., 2011).

Noise

Sound emissions from marine aggregate extraction are generally broadband, with most energy below 1 kHz, but sound pressure levels can vary greatly depending, for example, on dredging methods, operational stage, or environmental conditions (Todd et al., 2015). Data is still limited, but it is currently assumed that noise emissions are unlikely to cause damage to marine mammals' auditory systems permanently or temporarily (e.g., Todd et al., 2015). According to the ICES (2016) report, underwater sound caused by aggregate extraction in the UK has been largely discounted as a significant impact. Similarly, in a 2019 study from the Netherlands, noise pollution from dredgers didn't rank among the top seven major underwater sound sources (Ainslie et al., 2009). However, masking of cetacean vocalisation and communication as well as behavioural disturbances are possible (Todd et al., 2015).

Acoustic masking can cause cetaceans to alter the duration, frequency, and sound level of their acoustic signals (e.g., Todd et al., 2015). Behavioural changes due to underwater noise can occur at large distances from the source and be costly for individuals, as they can affect energy expenditure and limit time spent feeding or resting (NRC, 2005; Todd et al., 2015). It is also possible that noise induces stress (Wright et al., 2007; Rolland et al., 2012), which in turn could reduce the animals' foraging efficiency and/or increase their susceptibility to diseases and the effects of toxins (Geraci and Lounsbury, 2001; Reynolds et al., 2005; Perrin et al., 2009).

Behavioural avoidance of areas affected by dredging noise (sometimes concurrently affected by other sound-emitting industrial activities, such as the installation of a pipeline or fishing) have been documented in minke whales off the coast of Ireland (Anderwald et al., 2013), harbour porpoises in the North Sea (at ranges of 600m from the dredger; Diederichs et al., 2010), and bottlenose dolphins in Aberdeen harbour (whose presence in foraging areas declined as dredging intensity increased despite the animals being accustomed to high levels of vessel disturbances; Pirotta et al., 2013)).

Indirect impacts

Noise

Indirect effects of noise from dredging activities are also possible, arising from sensitivities of certain fish and cephalopod species (e.g., CEFAS, 2009; Tillin et al., 2011). Even though no studies have looked at dredging noise in particular, avoidance of low-frequency vessel noise have been reported in several fish species (e.g., Mitson, 1995; Handegard et al., 2003; Mitson and Knudsen, 2003; de Robertis and Wilson, 2011; de Robertis and Handegard, 2013). Low-frequency noise can also alter breathing rhythms and movement of cephalopods (specifically, *Sepia officinalis*, *Loligo vulgaris*, *Loligo pealeii*, *Octopus vulgaris*, *Illex coindetii*, which can form an important part of the diet of marine mammals), and in some cases induce acoustic trauma in these prey species (e.g., Packard et al., 1990; Mooney et al., 2010; Andre et al., 2011; Solé et al., 2013).

Avoidance of the dredging area and noise impact zone by these mobile animals could influence the short-term availability and/or distribution of cetacean prey species (e.g., Cefas, 2003; Mooney et al., 2010; ICES, 2016 & 2022). Cetaceans can likely compensate for such small-scale changes in prey abundance by e.g., switching to different prey species, moving to alternative feeding grounds, or increasing foraging effort; for populations with restricted ability to move elsewhere or species with higher energy demands, this may be more problematic (Todd et al., 2015).

Impacts from sediment plumes

Sediment plumes in the water column created by outwash and screening might reduce water clarity and can cause short-term avoidance of the working area by some fish and marine mammals (Cook & Burton, 2010). However, the effects from turbidity on cetaceans are assumed to be minimal, since sediment plumes are generally localised and short-lived (e.g., Hitchcock et al., 1999; Newell et al., 2004; Tillin et al., 2011), and also because various marine mammals are known to reside often in turbid waters (Todd et al., 2015).

A case study from the Black Sea, however, concluded that the dumping of sediment from dredging employed in the maintenance of navigational canals and reconstruction of ports should be considered a source of disturbance for Black Sea harbour porpoises and bottlenose dolphins, not only because of the associated noise pollution, but also because of the decline in water transparency and the destruction and siltation of the benthic ecosystem and associated species, which reduces cetacean foraging capabilities (Birkun, 2002).

Dredging can lead to a short-term increase in biological productivity possibly associated with re-suspended organic nutrients from the sediment plume (Ingle, 1952; Biggs, 1968; Sherk, 1972; Walker & O'Donnell, 1981). However, this effect is often short-lived only – and followed by reductions in diversity, biomass, and abundance of marine organisms, as described above – and often only positive up to a certain concentration of suspended particles (Todd et al., 2015).

Chemical pollution

As sediments accumulate toxins and pollutants such as Persistent Organic Pollutants (POPs) and heavy metals over time (Cundy et al., 2003; Taylor et al., 2004), dredging disturbances can release contaminants into the water column, where they reduce water quality (at both the production and dumping sites) and can become bioavailable (e.g., Todd et al., 2015; Eggleton & Thomas, 2004; Roberts, 2012). If they do, there is the potential that they accumulate up the food chain (Todd et al., 2015) and have detrimental effects on e.g., the endocrine, immune, and reproductive systems of animals, including marine mammals (CMS, 2025). These effects are complex and site-specific, and pollutant loads are generally lower in coarse gravel and sand beds than in finer sediment types. Nevertheless, contaminated sediments must be managed strictly and disposed of correctly to minimise their impacts on the environment and wildlife in both time and space (Roberts, 2012; Todd et al., 2015).

Wider impacts to the ecosystems, prey species, and habitat degradation

More indirect effects on cetaceans might arise through additional harmful effects on their prey, including on their fitness, distribution and abundance, through for example entrainment of adult individuals, larvae and eggs, as well as through habitat degradation, as can result from contamination, impacts from sediment suspension on benthic, epibenthic, and infaunal communities, and alteration of seabed structure, porosity, or integrity (e.g., Todd et al., 2015; ICES, 2022; van Lancker et al., 2014).

Several fish species depend on the seabed for their feeding, among them plaice, sole, dab, gurnard (Triglidae), red mullet (Mullus spp.), haddock, whiting, and cod, who primarily feed on benthic organisms like bivalves, worms, crustaceans, and sea urchins. Top predators, including marine mammals, can be highly sensitive to changes in their abundance and diversity (ICES, 2022).

Dredging in fish spawning areas can be especially detrimental, but risks can be decreased through the use of environmental windows (Todd et al., 2015) and attuning dredging activities to known biological rhythms and threatened species' traits. A general avoidance of spawning or nursery grounds brings great benefits, as it minimised large-scale losses of species (Todd et al., 2015). Similar benefits can be gained by minimising dredging-associated sedimentation in the proximity to oyster and seagrass beds and other sensitive habitats (Todd et al., 2015).

Generally, dredged seabed areas are colonised quickly by opportunistic species, which likely attract higher trophic levels. If this re-colonisation includes species consumed by cetaceans, impacts on prey availability are likely only short-term and population-level effects are unlikely, but short-term changes to feeding behaviour and distributions of cetaceans are possible (Todd et al., 2015).

In summary, the ICES (2016) report concludes that assessing the impacts of marine aggregate extraction on higher trophic levels is complex, and that species-specific factors such as adaptability in diet may influence the significance of the impacts on a local scale. It is important to note that the ability of marine mammals to adapt can also be influenced by cumulative and/or synergistic effects of dredging and other anthropogenic activities and stressors impacting prey species and/or habitat (ICES, 2016). The latest ICES report (2022) adds that dredging has the potential to impact marine mammals, and its effects will be species- and location-specific, varying also with dredging equipment type. Assuming management procedures are implemented, expected effects on cetaceans are most likely to be acoustic masking and short-term behavioural alterations due to shifts in prey availability (Todd et al., 2015; ICES, 2022). The extent of negative impacts from the exclusion of prey from foraging areas within or close to dredging sites is species- and context-specific, and depends on the significance of the area, the cetaceans' ability to switch to alternative prey, and the availability of alternative foraging areas (ICES, 2022).

Available guidelines to reduce impacts during marine aggregate extraction activities

The Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT) recommends the use of the "ICES Guidelines for the Management of Marine Sediment Extraction" (ICES et al., 2003), which include mitigation measures to protect sensitive species and habitats, and limit the impacts upon the biota (Sutton & Boyd, 2009; ICES, 2022).

Conclusions

[coming later]

Conclusions

The information compiled in this report indicates that, in addition to shallow-water mining, both offshore carbon capture and storage and sand and gravel extraction are activities of relevance to the ASCOBANS Working Group on Shallow-water Mining and Small Cetaceans. Although the mechanisms and scale of potential impacts differ between the three activities, all may introduce pressures that could affect small cetaceans and the ecosystems on which they depend.

For CCS, relevant pressures may arise from exploration activities such as seismic surveys, infrastructure installation, vessel traffic, and monitoring operations, as well as from the potential for leakage of stored carbon dioxide and associated environmental changes. For sand and gravel extraction, key pressures include physical disturbance of the seabed, sediment suspension and deposition, entrainment of benthic organisms and early life stages of fish, underwater noise from dredging operations, and increased vessel movements. For FeMn nodule mining, impacts are similar to those of sand and gravel extraction, with the addition that the nodules provide a unique marine habitat that take centuries to form, and hence would need similar time spans to recover from mining activities.

In addition to direct disturbance, all three activities may lead to indirect effects through changes in benthic habitats and prey availability and distribution, which may, in turn, influence the distribution and behaviour of small cetaceans. The significance of such impacts is likely to depend on the location, scale, timing, and cumulative context of activities, including their overlap with ecological windows, habitats important to cetaceans or their prey species and additional anthropogenic pressures in the marine environment.

The next step until AC31 is for the Working Group to finish the report and develop recommendations for the consideration of the Advisory Committee.

As part of the Working Group's ongoing work, further efforts may focus on improving the overview of relevant activities where possible, identifying knowledge gaps, and considering potential measures to minimize risks to small cetaceans and their habitats. The current report represents an initial contribution to this process and provides a basis for further consideration within the Working Group and the Advisory Committee.

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