Marine Aggregate Extraction and the Marine Strategy Framework Directive: A Review of Existing Research

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Contents

I Summary

The purpose of this bibliographical review is to consider the environmental impacts of marine aggregate extraction in the context of the Marine Strategy Framework Directive (MSFD) and the overarching objective of achieving Good Environmental Status (GES) across a number of relevant descriptors. The review identifies gaps in current knowledge, and highlights the need for expert judgement where understanding is limited. In particular, this report calls attention to the need to account for seabed recovery and recolonization when seeking to understand the footprint of effects from aggregate dredging. Information from this study should be used to optimize the management of marine aggregate extraction and its sustainable development, thus addressing policy and management needs.

II Foreword

This bibliographic review was initiated by the ICES Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem (WGEXT). We gratefully acknowledge all members of WGEXT since 1991 for their contributions to this work.

1 Introduction

Global biodiversity is threatened by human activities, which are increasingly impacting marine ecosystems (Halpern *et al.*, 2008; Coll *et al.*, 2012; Korpinen *et al.*, 2012). These impacts are usually cumulative, and can lead to the degradation of habitats and ecosystem functionality (Crain *et al.*, 2008; Ban *et al.*, 2010; Korpinen and Andersen, 2016; Stelzenmüller *et al.*, 2018; Lonsdale *et al.*, 2020). Understanding the relationships between human pressures and ecosystems is the second major challenge identified by Borja (2014) for future research within the field of marine ecosystem ecology.

As a result of the intense use of marine aggregates, the environmental impacts of marine aggregate extraction have been investigated for decades (e.g. de Groot, 1979; Kenny *et al.*, 1998; Newell *et al.*, 1998; Desprez, 2000; Van Dalfsen *et al.*, 2004; Cooper *et al.*, 2007, 2008; Tillin *et al.*, 2011; Waye-Barker *et al.*, 2015).

Habitat modification (e.g. geomorphology and sediment type) is the most direct result of extracting material from the seabed, along with the removal of the associated benthic infauna. The relationship between activity and impact (Elliott *et al.*, 2017, 2020a) varies according to the pressure level (e.g. spatial extent, duration and/or frequency, and intensity of aggregate extraction), habitat type and component species, and their recovery potential (Foden *et al.*, 2010; Lambert *et al.*, 2014; Duarte *et al.*, 2015). The effects of sustained activity can ultimately change the abundance, biomass and function at a community or ecosystem level (Barrio-Froján *et al.*, 2011; Thrush *et al.*, 2016).

Indirect impacts also need to be considered, such as the footprint of sediments plumes (Boyd and Rees, 2003; Desprez *et al.*, 2010; Spearman, 2015), the impact of reduced food resources on foodweb structures (de Jong *et al.*, 2014), or the output noise of dredging vessels (Robinson *et al.*, 2011; Heinis, 2013). Finally, the effects of dredging can affect human welfare through the reduction of ecosystem services and societal benefits (Elliott *et al.*, 2014, 2020b; Smith *et al.*, 2016). In this review, the impacts on human welfare were not considered, with the focus being placed instead on changes in status in the marine environment.

The European Marine Strategy Framework Directive (MSFD) aims at Good Environmental Status (GES) in marine waters, following an ecosystem-based approach focused on 11 descriptors related to ecosystem features, human drivers, and pressures (EC, 2010).

Several documents concerning the MSFD note the need to incorporate extraction as a human impact factor. For instance, in Annex III of the MSFD, extraction of minerals (rock, metal ores, gravel, sand, shell) is mentioned as a human activity with pressures and effects on the marine environment (EC, 2016a; Patricio *et al.*, 2016; Elliott *et al.*, 2017).

This report reviews existing research on the various effects of marine aggregate extraction on the seabed and the water column, and the connection between research and criteria for GES, which is relevant for the following MSFD descriptors:

- Descriptor 1 (D1): biological diversity;
- Descriptor 3 (D3): commercial fish and shellfish resources;
- Descriptor 4 (D4): marine foodwebs;
- Descriptor 6 (D6): seabed integrity;
- Descriptor 7 (D7): hydrographical conditions;
- Descriptor 8 (D8): contaminants; and
- Descriptor 11 (D11): underwater noise.

Table 1.1 summarizes the impacts on the marine ecosystem, developed in different sections, and the links between these impacts and the descriptors. This review also highlights gaps where further knowledge is needed to fulfil MSFD requirements.

Table 1.1. Main impacts of marine aggregate extraction and links with the MSFD descriptors. Descriptors in parenthesis are considered to be less influenced by marine aggregate extraction.

Effects of aggregate extraction	Impact on	Potentially influenced MSFD descriptors:
Seabed removal	Topography/bathymetry	(D1), D6, D7
	Sediment composition	D1, (D3), D6
	Habitat and biological communities	D1, (D3), D4, D6
Sediment plumes	Turbidity	D3, D4, (D8)
	Deposition	D1, D3, D4, D6, (D8)
Ship activities	Underwater noise	D11

2 Impacts of marine aggregate extraction by MSFD descriptor

Since 2003, ICES guidelines for the management of marine sediment extraction have encouraged an ecosystem approach to the management of extraction activities and the identification of areas suitable for extraction (ICES, 2003). Moreover, these guidelines, which have been adopted by OSPAR, provide recommendations for the implementation of mitigation and monitoring programmes, which ensure that extraction methods minimize adverse effects, and preserve the overall quality of the environment once extraction has ceased. In each of the following sections, the potential impacts of marine aggregate extraction on the MSFD descriptors of the marine ecosystem are considered.

2.1 Descriptor 1: Biodiversity

"The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic, and climate conditions. Assessment is required at several ecological levels: ecosystems, habitats, and species." (EC, 2010)

The urgent need for large-scale spatial data on benthic species and communities has resulted in an increased application of modelling of biogeographical distribution (Reiss *et al*., 2014; Cooper *et al*., 2019). These new insights on benthic biodiversity provide vital context, and can help the management of marine aggregate extraction. Sand and gravel bottoms are targeted by the extraction industry, and they represent only a fraction of the overall diversity in marine habitats and species (i.e. variety of bottom types, habitats of common interest, presence of rare and endangered species). In general, the biodiversity of the seabed tends to increase with the size and heterogeneity of the sediment (microhabitats), and with the stability of the substrate. For instance, sandy bottoms, with low diversity in microhabitats, and, in particular, the mobile banks of coarse sand targeted by extraction, are typically poor in species number and biomass. In contrast, gravelly bottoms are the most biologically diverse among marine habitats, because the larger gravel size allows for the settlement of, and provides shelter for, many sessile and mobile organisms (Seiderer and Newell, 1999; Desprez, 2000; Cooper *et al*., 2007). Deep-water gravel habitats tend to be more diverse than those closer to the coast, with a varied and abundant epifaunal component, including sponges, tunicates, bryozoans, hydroids, and polychaetes. Biogenic reefs, which are under threat and are of high heritage value, can be associated with these gravel habitats.

According to a decreasing gradient of impact, Browning (2002) identified three main classes of anthropogenic pressures on biodiversity in the English Channel–North Sea area:

- maximal impact class, including fishing activity (threatened species, destruction of protected biotopes);
- higher to medium impact class, including many types of pollution; and
- lower to medium impact class, including marine aggregate extraction and the deposition of harbour maintenance sediments.

2.1.1 Effects

The potential impacts of marine aggregate extraction on key habitats and species of the European Directive Natura 2000 protected areas network are summarized in Table 2.1 from Posford Duvivier Environment and Hill (2001). A loss of 60% of the number of benthic species is generally observed within dredging sites (Newell *et al*., 1998, 2004a; Desprez, 2000; Boyd *et al*., 2002; Boyd and Rees, 2003; ICES, 2009, 2016a; Krause *et al*., 2010; Desprez *et al*., 2014).

Approaches to support the conservation of marine biodiversity include: measures of rarity and diversity, identification of the number and abundance of species and habitats in different locations, and the identification of biological indicators (Hiscock and Tyler-Walters, 2006).

The identification of key habitats and species requires ambitious mapping programmes for the biological characteristics of marine habitats, several of which have recently been developed at international, national, and regional scales (Coggan and Diesing, 2011; Coggan *et al*., 2012; Michez *et al*., 2015; Vanstaen *et al*., 2015; Vasquez *et al*., 2015; Delage and Lepape, 2016; Galparsoro *et al*., 2016; Baffreau *et al*., 2017; La Rivière *et al*., 2017). The area covered by these mapping programmes far exceeds that of extraction areas.

Habitat maps can provide information on the location and distribution of habitat types, threatened species, hot spots for habitat diversity, zones of high productivity, and spawning aggregation sites, and can ultimately aid in the selection of protected areas. Habitat maps also give important information for environmental impact studies on aggregate extraction, in order to evaluate and mitigate potential impacts. The ICES Working Group for Marine Habitat Mapping (WGMHM; ICES, 2016a) reports on national mapping initiatives, including the development of mapping techniques and modelling, data analysis, and habitat classification schemes. WGMHM also reviews practices regarding the use of habitat maps [e.g. for the MSFD (Cogan *et al*., 2009), marine spatial planning, and management of marine protected areas (MPAs)], and is a major contributor to the development of common and candidate OSPAR biodiversity indicators for benthic habitats.

The loss of structural biodiversity at extraction sites is local, and its duration varies according to the extraction strategy used. Effects can be pronounced, with a risk of cumulative effects, in coarse sediment areas, where intensive extraction takes place (Cooper *et al*., 2007; Lonsdale *et al*., 2020). Such effects may be counterbalanced by more extensive extractions (< 50% of the total licensed area), where an increase in the diversity of benthic communities can be linked to the diversification of habitats (Thrush *et al*., 2006; Hewitt *et al*., 2008; de Backer *et al*., 2014; Desprez *et al*., 2014). Cusson *et al.* (2014) observed that changes within the structure of community assemblages are generally independent of biodiversity.

Table 2.1. Potential impacts of marine aggregate extraction on key habitats and species of the European Directive Natura 2000 (S: short term, M: mean term, L: long term).

Potential impact	Habitats (Ann. I)	Species (Ann. II)	
	Sand and gravel banks	Fish	Mammals
Benthos and substrate loss	M	M	M
Turbidity	S	S	S
Sediment	ML	ML	

2.1.2 Recovery

The lower impact of extensive extraction favours benthic recovery, notably through the use of spatial and temporal zoning, which allows the recolonization by drift of adults/juveniles from surrounding deposits and by planktonic settlement from more distant sources (Newell *et al*., 1998; Birchenough *et al*., 2010).

In the case of intense deposition of fine sediments due to screening, the damage by dredging to functional diversity, and to the capacity of the macrofaunal assemblage to recover, is immediate and not so dependent on dredging intensity (Barrio-Froján *et al*., 2008). A return to the initial biodiversity can be artificially accelerated by creating a heterogeneous substrate with the seeding of shells or gravel (Collins and Mallinson, 2007; Cooper *et al*., 2010a), but the cost of this work may be considerable (Cooper *et al*., 2010b). Ecoengineering can facilitate restoration and have beneficial effects on biodiversity with the creation of new habitats and associated communities (Bouma *et al*., 2009; de Jong *et al*., 2015; Elliott *et al*., 2016).

2.1.3 Biodiversity and ecosystem functionality

Understanding the role of biodiversity in maintaining ecosystem functionality is a major challenge in marine ecosystem ecology (Borja, 2014). The study of the ecological function of biodiversity is very recent (Loreau *et al*., 2001; Bremner *et al*., 2003, 2006a, 2006b, 2008; Duffy *et al*., 2007; Cooper *et al*., 2008; Mouillot *et al*., 2013). However, it has been recognized as having fundamental implications for predicting the consequences of biodiversity loss on ecosystem function, i.e. translating structural biodiversity measures into functional diversity to generate better biodiversity–to–ecosystem function relationships (Strong *et al*., 2015). Furthermore, as a driver for functional composition and diversity, habitat heterogeneity should be explicitly included within studies trying to predict the effects of species loss on ecosystem function. The need to include functional biodiversity in impacts assessments has been highlighted by the Working Group on Good Environmental Status (EC, 2010).

Theoretically, a larger number of functional group types will provide a higher functional biodiversity organization to the system, and contribute to more stable and resilient ecosystems (Borja *et al*., 2009; Tomimatsu *et al*., 2013; Cusson *et al*., 2014; Strong *et al*., 2015). However, Törnroos *et al.* (2014) observed that while a decrease in species richness led to a global decrease in functionality, the functional richness remained comparatively high at the lowest level of specific richness, thus showing that a potential existed for species substitution to maintain the ecological functioning of marine benthic systems (Frid, 2011). Clare *et al.* (2015) confirmed that ecological functioning was statistically comparable between periods with significantly different species composition.

Differences in functional traits between habitats are more strongly influenced by differences in the density of organisms than by the presence/absence of individual traits, illustrating the importance of variations in organism density for functionality (Hewitt *et al.*, 2008).

The Marine Biodiversity and Ecosystem Functioning (MARBEF[1]) project demonstrated that alterations in the abundance of key species affect ecosystem functioning more than changes in species diversity (Heip *et al.*, 2016).

Overall, it is now fully recognized that understanding the entire ecosystem requires the study of all biodiversity components (Borja, 2014), from species to habitats, including foodwebs (D4) and complex biophysical interrelationships within the system.

2.1.4 Biodiversity indicators

For marine assessments, like the MSFD, biodiversity is defined at the level of species, communities, habitats, and ecosystems, as well as at the genetic level (Cochrane *et al*., 2010). Biodiversity can be seen as an overarching descriptor, and is too broad a topic to list all possible indicators. In any case, not all indicators can be applied everywhere. Therefore, there is a need for more guidance on which habitats and species to consider (EC, 2010).

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[1] https://www.marbef.org/

Sample Simpson, Shannon and Richness indices are useful indicators of changes in biodiversity, although their population equivalents do not always reflect biodiversity changes (Barry *et al.*, 2013).

For demersal fish communities, consisting mainly of mobile species, neither the habitat-level indicators, nor the single-species distribution indicator explicitly directed at sessile/benthic species, are pertinent. Appropriate fish biodiversity metrics cannot be derived to support this D1 indicator (Greenstreet *et al.*, 2012).

A catalogue of existing indicators of marine biodiversity has been established within the DEVOTES project (Development of innovative tools for understanding marine biodiversity and assessing good environmental status; Teixeira *et al.*, 2014). In this catalogue, 218 indicators are assigned to at least one of the eight criteria for determining GES (Commission decision, 2017) related to D1. Most of the indicators are related to abundance, biomass, distribution, diversity, and richness of species.

Ecosystem overviews for each European regional sea have been written to highlight the specific features of those areas, especially from the standpoint of biodiversity. However, it is still necessary to provide quantitative indicators with reference levels. In the European regional seas, the DEVOTES catalogue revealed that there are gaps mainly in indicators to address ecosystem structure, processes, and functions. Furthermore, the analysis of the indicator set revealed that there is considerable overlap between the MSFD indicators for biodiversity, foodwebs, and seabed integrity (D1, D4, and D6, respectively).

Impact indicators for major drivers of marine biodiversity loss are currently lacking (Woods *et al.*, 2016). Moreover, the value of an ecological indicator is no better than the uncertainty associated with its estimate. Indicator uncertainty is seldom estimated, although legislative frameworks such as the European Water Framework Directive stress that the confidence of an assessment should be quantified (Carstensen and Lindegarth, 2016). With increased knowledge and understanding about the strengths and weaknesses of competing index approaches, there is a need to unify approaches, with the aim of providing managers the simple answers they need to use ecological condition information effectively and efficiently (Borja *et al.*, 2009, 2016).

Elliott *et al*. (2018a) proposed an integrated approach to assess benthic habitats with OSPAR indicators relating to biodiversity (D1) and seabed integrity (D6) descriptors linked together. This method can be expanded to include other related indicators under the different descriptors [e.g. commercial fish and shellfish (D3), or foodwebs (D4)] where relevant. The concept is a first step towards integration of benthic indicators.

2.1.5 Conclusion

With respect to D1, WGEXT recognizes that extraction of marine aggregates can potentially be a serious threat to biodiversity (Table 2.2). This can notably occur through loss of habitat when extraction projects affect gravelly areas that are either small or underrepresented in the geographical area.

The ICES Guidelines for the management of marine sediment extraction (ICES, 2003), by OSPAR, provide guidance for the selection of appropriate extraction site locations and dredging protocols, with the aim of preventing harmful effects on habitats of prime importance for sensitive species (Table 2.2).

Table 2.2. Contribution to MSFD D1 according to the various impacts detailed in the ICES Guidelines for the management of marine sediment extraction (2003).

Effects of extraction	Impact on	Potentially influenced MSFD descriptors	Level of contribution of WGEXT guidelines to MSFD descriptors			
			Introduction	Baseline survey	Impact assessment	Mitigation
Seabed removal	Bathymetry/ topography	D1: Biological diversity is maintained - Habitat Criterion 1.6: Physical condition	"Ensuring that methods of extraction minimize the adverse effects on the environment and preserve its overall quality once extraction has ceased" and "protecting sensitive areas and important habitats"	<u>Dredging activity</u>: "spatial design and configuration of aggregate dredging (maximum depth of deposit removal, shape and area of resulting depression)"		
	Sediment composition	D1: Biological diversity is maintained: quality and occurrence of habitats, and distribution and abundance of species			<u>Physical impact assessment</u>: changes to sediment type, exposure of different substrates, transport and settlement of fine sediment from overflow. <u>Biological impact assessment</u>: changes to the benthic community structure.	
	Habitat and communities	D1: Biological diversity is maintained: quality and occurrence of habitats, and distribution and abundance of species		<u>Biological setting</u>: "fauna and flora within the area likely to be affected by aggregate" and "presence of any areas of special scientific or biological interest designated under local, national, or international regulations"	<u>Biological impact assessment</u>: "changes to the benthic community structure and to any ecologically sensitive species or habitats" and "effects on sites designated under local, national or international regulations"	"Agreeing exclusion areas to provide refuges for important habitats or species, or other sensitive areas" and "Selection of dredging equipment and timing of operations to limit impact upon the biota (birds, benthos, sensitive species and habitats"

2.2 Descriptor 3: Commercial fish and shellfish

"Populations of all commercially exploited fish and shellfish are within safe biological limits, exhibiting a population age and size distribution that is indicative of a healthy stock." (EC, 2010)

The proposed indicators of mortality and biomass are the basis for D3, whereas a third indicator, age and size distribution, should also be linked to the foodwebs descriptor (D4). Changes in, or loss of, a preferred sediment grain size can disturb mobile species. For instance, taxa such as herring (*Clupea harengus*), black bream (*Spondyliosoma cantharus*), and sandeel (Ammodytidae) require certain substrate conditions for spawning or breeding activity. Studies such as de Groot (1979) have highlighted the importance of historical spawning grounds for herring and the particular requirement for coarse gravel (ICES, 2011). This increases the vulnerability of herring to disturbance if marine aggregate extraction occurs within spawning areas. In addition, ovigerous female brown crabs (*Cancer pagurus*) prefer to overwinter on coarse gravelly material, and are, therefore, susceptible to direct dredging impacts. In contrast, certain species such as lobster (*Homarus gammarus*) may benefit from the creation of habitat mosaics as a result of extensive dredging activity.

Mobile species are more likely to be influenced by other impacts or anthropogenic activities outside the extraction licence area than by extraction activities. Therefore, direct predictions on the impact of marine aggregate extraction on mobile species are difficult. A study by Boyd *et al*. (2001) compared the commercial fish landings for fish caught in an aggregate zone to those obtained from ports distant to dredging. A localized decline in catches in Dover sole (*Solea solea*) was observed, which the study considered could be a result of the reduced abundance of prey items within the extraction area, as Dover sole derive much of their food from benthic species (Pearce, 2008; Desprez *et al*., 2014).

A study by Kenny *et al*. (2010) looked at the long-term trends in the ecological status of an aggregate producing region on the east coast of the UK, including fish stocks in its considerations. This study noted that long-term trends in fish stocks appear to be dominated by wider factors that govern trends at the North Sea scale, with observed changes being evident in both the North Sea and the aggregate producing region.

Stelzenmüller *et al*. (2010) investigated the vulnerability of 11 species of fish and shellfish to aggregate extraction. The authors calculated a sensitivity index (SI) for each species, and modelled their distribution around the UK. The selected species were likely to be affected by aggregate extraction, and had either commercial or conservational importance. They included sole, thornback ray (*Raja clavata*), plaice (*Pleuronectes platessa*), cod (*Gadus morhua*), whiting (*Merlangius merlangus*), and the bivalve mollusc queen scallop (*Aequipecten opercularis*). The highest sensitivity to aggregate dredging was observed in coastal regions, where nursery and spawning areas of four important commercial species occurred (cod, plaice, sole, and whiting).

In 2003, the Franco–British project CHARM (eastern channel habitat atlas for marine resource management) was initiated to support decision-making on the conservation, protection, and/or management (anthropogenic disturbances) of essential fish habitats such as spawning grounds, nurseries, or areas carrying biodiverse fish communities (Vaz *et al*., 2007). The enhanced understanding of species distribution provided by CHARM provides context to inform decisions regarding marine aggregate extraction. Similarly, an inventory of coastal areas of conservation importance was defined in France to protect commercially important fish resources and functional areas of prime importance for their life cycle, in order to maintain their renewal and the associated fishing activity (Delage and Le Pape, 2016).

Turbid plumes produced by aggregate extraction can affect the bentho-pelagic life cycle (eggs and larvae) of several commercially important species, such as flatfish, during their transit from spawning to nursery grounds (Barbut *et al*., 2019). While eggs can tolerate sediment concentrations > 100 mg l^{-1}, mortality of herring and cod larvae occurs at slightly lower levels (20 mg l^{-1}; Westerberg *et al*., 1996). For adults, an avoidance behaviour was observed in visual predatory fish, such as mackerel (*Scomber scrombrus*) and turbot (*Scophthalmus maximus*), with such behaviour being triggered at very low silt concentrations (3 mg l^{-1}) for herring and cod (Westerberg *et al*., 1996).

There have been few direct studies on changes in fish populations due to marine aggregate extraction (ICES, 2016b). Experimental fish monitoring in the eastern Channel between 2007 and 2011 showed a strong impact of an intensive aggregate extraction on fish presence, both for the number of species (–50%) and for abundance and biomass (–92%). In contrast, the impact of an extensive dredging programme (with spatial and temporal zoning) was limited, without any decrease in species number and biomass, and a reduction in abundance of 35% (Desprez *et al*., 2014). Dab (*Limanda limanda*) and whiting are the two fish species most adversely affected by dredging. In contrast, sole and rays appear to flourish in areas where the sediment had been modified by the deposition of sandy material, allowing permanent fishing activity. These observations are still qualitative, and further work is required to determine the impacts footprints.

The impact of aggregate extraction activities on the displacement of fishing activities was based primarily on anecdotal evidence until preliminary impact studies were carried out by Vanstaen *et al*. (2010) on aggregate sites in the UK over various periods. They concluded there was no evidence that marine aggregate extraction had significantly altered the spatial fishing distribution of fleets operating various mobile gears.

The effects of dredging intensity, and the distance between extraction sites and the distribution of fishing effort, were more recently investigated for a broad selection of French and English demersal fleets operating in the eastern English Channel. The most prominent result was that, for most of the fishing fleets and aggregate extraction sites, neither extraction intensity nor the proximity to the extraction site had a substantial deterring effect on fishing activities (Marchal *et al*., 2014). The distribution of fishing effort of French netters remained consistent over the study period, and increased substantially in the impacted area of an extraction site in Dieppe. The fishing effort of dredgers and potters was greater adjacent to marine aggregates sites than elsewhere, and also positively correlated to extraction intensity. This is consistent with omnivorous and scavenging benthic species, as well as fish such as common sole, black sea bream, and cod, being attracted to these areas (Desprez *et al*., 2014).

2.2.1 Conclusion

Recent studies suggest that fishing activity is generally not deterred by extraction activities. However, WGEXT recognizes that extraction of marine aggregates can potentially be a serious threat to commercially important fish species when functional impacts affect sensitive and threatened species (e.g. through loss of spawning areas).

The ICES guidelines for the management of marine sediment extraction (ICES, 2003), which have been adopted by OSPAR, recommend (i) a limited modification of the bathymetry and topography of the target area, to minimize the effects on fishing; and (ii) the adoption of appropriate extraction site locations, to prevent any harmful effect on habitats of prime importance for fish resources (see Table 2.3).

Table 2.3. Contribution to MSFD D3 according to the various impacts detailed in the ICES Guidelines for the management of marine sediment extraction (2003).

Effects of extraction	**Impact on**	**Potentially influenced MSFD descriptors**	**Level of contribution of WGEXT guidelines to MSFD descriptors**			
			Introduction	**Baseline survey**	**Impact assessment**	**Mitigation**
Seabed removal	Bathymetry/ topography	D3: Commercial fish and shellfish populations are within safe biological limits				"Modification of the depth to minimize the effects on fishing" and "Spatial and temporal zoning to protect sensitive fisheries or to protect access to traditional fisheries"
	Sediment composition	D3: Commercial fish and shellfish populations are within safe biological limits			Biological impact assessment: "effects on the fishery and shellfishery resources including spawning areas, nursery areas, overwintering grounds for ovigerous crustaceans, and known routes of migration"	
	Habitat and communities	D3: Commercial fish and shellfish populations are within safe biological limits	"Protecting the interests of other legitimate uses of the sea"	Biological setting: "information on the fishery and shellfishery resources"		"Agreeing exclusion areas to provide refuges for important habitats or species, or other sensitive areas" and "Selection of dredging equipment and timing of operations to limit impact upon the biota (birds, benthos, sensitive species and habitats, and fish resources"

Table 2.3 (continued)

Effects of extraction	Impact on	Potentially influenced MSFD descriptors	Level of contribution of WGEXT guidelines to MSFD descriptors			
			Introduction	Baseline survey	Impact assessment	Mitigation
Sediment plume	Turbidity	D3: Commercial fish and shellfish populations are within safe biological limits			Physical impact assessment: "transport of fine sediment from overflow" Biological impact assessment: "effects of aggregate dredging on pelagic biota", "effects on the fishery and shellfishery resources including spawning areas, nursery areas, overwintering grounds for ovigerous crustaceans, and known routes of migration" and "effects on any other legitimate use of the sea"	
	Deposition	D3: Commercial fish and shellfish populations are within safe biological limits			Physical impact assessment: "settlement of fine sediment from overflow". Biological impact assessment: "effects on the fishery and shellfishery resources including spawning areas, nursery areas, overwintering grounds for ovigerous crustaceans, and known routes of migration"	

2.3 Descriptor 4: Foodwebs

"All elements of the marine foodwebs, to the extent that they are known, occur at normal abundance, diversity, and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity." (EC, 2010)

D4 concerns important functional aspects of marine ecosystems, such as energy flows and the structure of foodwebs.

Thompson *et al*. (2012) emphasized that foodweb ecology will act as an underlying conceptual and analytical framework for studying biodiversity and ecosystem function, if some challenges are addressed, such as relating foodweb structure to ecosystem function or understanding the effects of biodiversity loss on ecosystem function. Trophic structure is an important driver of community functioning and many biological traits, particularly body size, which, in turn, determines which species interact (Nordström *et al.*, 2015).

2.3.1 Indirect effects of substrate loss

Depletion of benthic communities could, in theory, functionally affect higher trophic levels (e.g. fish and birds), through the loss of habitat and potential food sources (Birklund and Wijsman, 2005). Several fish species are more or less closely related to the seabed by their way of feeding. Plaice, sole, dab, gurnard (Triglidae), red mullet (*Mullus* spp.), haddock (*Melanogrammus aeglefinus*), whiting, and cod, feed primarily on benthic organisms like bivalves, worms, crustaceans, and sea urchins. Coastal seabeds, due to their high productivity, are also important feeding areas for diving birds such as ducks, terns, penguins, and northern gannets (*Morus bassanus*; Michel *et al.*, 2013). Top predators, such as seabirds and mammals, can be highly sensitive to changes in the abundance and diversity of their primary prey, although many bird species are able to switch to alternative prey (Rombouts *et al.*, 2013).

According to FishBase[2], more than 48 species of fish in the Northeast Atlantic are associated with sandy gravel seafloors for spawning (e.g. herring, black bream, and sole). In addition, about 40 other species are associated in other ways with these habitats, such as rays, dogfish (*Squalus acanthias*), plaice, sandeels (*Ammodytes marinus*), and sharks. However, most flatfish species of commercial interest develop and reproduce in fine and silty sands, which tend to be of less interest for extraction.

Shellfish are an important component of coastal foodwebs. For example, they are prey for shellfish-eating birds, such as the common scoter (*Melanitta nigra*), and demersal fish (Kaiser *et al.*, 2006; Tulp *et al.*, 2010). Due to this importance, the impacts of aggregate extraction on shellfish species are being investigated in The Netherlands, using the American razor shell (*Ensis directus*) as a model organism because of its high prevalence, in terms of biomass, in the Dutch coastal zone (ICES, 2016b).

Few studies have directly investigated disturbances to mobile fish species related to marine aggregate extraction (Hwang *et al.*, 2010; Desprez *et al.*, 2014; de Jong *et al.*, 2014), or have suggested that a significant impact will occur (Stelzenmüller *et al.*, 2010; Vanstaen *et al.*, 2010; Drabble, 2012; Marchal *et al.*, 2014). This makes general predictions on the impact of marine aggregate extraction on mobile fish difficult.

[2] https://www.fishbase.se/search.php

In Korea, significantly lower species richness (−60%), species diversity, and fish abundance (−90%) were associated with seabed disturbance caused by seabed sediment mining (Hwang *et al.*, 2010). At a French experimental site in the eastern English Channel (Baie de Seine; Desprez *et al.*, 2014), monitoring between 2007 and 2011 showed a strong negative impact of aggregate extraction on fish presence, both in the number of species (−50%) and in abundance and biomass (−92%). However, such a strong impact was not observed in the commercial extraction site at Dieppe (+50, −35, and +5% for number of species, abundance, and biomass, respectively). This difference could be explained by the difference in extraction strategy (zoning), with a low intensity in Dieppe (< 1 h ha^{-1} $year^{-1}$), and a medium to high intensity in the Baie de Seine (4–10 h ha^{-1} $year^{-1}$).

In a Dutch deep sand extraction site, significant differences in demersal fish species assemblages were associated with variables such as water depth, median grain size, fraction of very fine sand, biomass of shells, and time after the cessation of sand extraction (de Jong *et al.*, 2014). One and two years after cessation, a significant, 20-fold increase in demersal fish biomass, dominated by plaice, was observed in the deeper, muddy parts of the extraction site that were colonized by high densities of white furrow shell (*Abra alba*).

A study by Pearce (2008) investigated the importance of benthic communities within marine aggregate extraction areas as a food resource for higher trophic levels. The study noted that changes to the benthos due to dredging were likely to cause alterations in the diet of demersal fish, which may be unfavourable. However, given the natural levels of trophic adaptability observed, a change in dietary composition might not have been damaging to the fish populations, as the majority of species studied were likely to switch prey sources, provided sufficient biomass was available to support them.

Between 2004 and 2011, three combined studies (benthos, fish, and stomach contents monitoring) were undertaken at two French sites (Dieppe and Baie de Seine) in the eastern English Channel (Desprez *et al.*, 2014). Evidence of trophic adaptability was observed, with an increase in the abundance of sole within the extraction and, particularly, the deposition areas. In Dieppe, black sea bream, gurnards, and cod were absent from the sandy reference and deposition areas, but were attracted to dredging areas by the abundance of opportunistic benthic species (e.g. the decapod genera *Pisidia* and *Galathea*) that recolonize dredging areas between extraction periods (fallow areas) and after the cessation of activity.

2.3.2 Effects of the turbid plume

A direct consequence of the increased turbidity caused by aggregate extraction, is the reduction in light penetration into the water column, which can affect the whole trophic web. Indirect impacts through the creation of turbidity plumes include:

- reduction in phytoplankton primary production, which constitutes the basis of the foodweb;
- disruption of the feeding and respiration of zooplankton;
- reduction of phytoplankton intake by shellfish, and potential additional stress (i.e. higher energetic costs) to these organisms because they need to excrete silt in the form of pseudo-faeces (Michel *et al.*, 2013);
- avoidance behaviour in visual predatory fish, such as mackerel and turbot. For herring and cod, critical levels were demonstrated at very low silt concentrations (3 mg l^{-1});
- mortality of eggs and larvae. Herring and cod larvae mortality occurs at silt levels of 20 mg l^{-1}, while eggs can tolerate concentrations > 100 mg l^{-1} (Westerberg *et al.*, 1996);

- smothering/damage to sensitive benthic receptors caused by deposition of sediments (Last *et al.* 2011).

It should be noted that these potential effects will only occur in the case of very intensive extraction, with several dredging vessels working simultaneously over a wide area during a couple of years, such as occurred during the Port of Rotterdam harbour expansion (de Jong *et al.*, 2015). For a single vessel without screening, previous research suggests a rapid deposition of coarser particles in the first 10 min within 1 km around the extraction area, thus making most of the indirect impacts unlikely. Coarse particles show an immediate dilution of about 10–100%, from several g l^{-1} down to 20 mg l^{-1}, while the subsurface plume of silts decreases slowly down to the background level after 2 h (Duclos *et al.*, 2013).

Cook and Burton (2010) reviewed the potential impacts of aggregate extraction on seabirds. One direct effect was the potential for increased turbidity to affect seabirds ability to see prey. Vision is important for foraging for a number of species of seabirds, including terns, the common guillemot (*Uria aalge*), and the northern gannet. However, material generally falls out of suspension relatively quickly (mostly within 500 m), meaning this increased turbidity is short term and within a limited area. During spring tides in a macrotidal environment, Duclos *et al*. (2013) observed that the turbid plume disappeared in 2 h, with a maximal deposit extent of 800 m for sands and 6.5 km for silts.

In a review of the impacts of marine dredging activities on marine mammals, Todd *et al.* (2014) also concluded that sediment plumes are generally localized. As marine mammals often reside in turbid waters, significant impacts from turbidity are probably temporary, as observed with seals around extraction sites in the North Sea. However, entrainment, habitat degradation, noise, suspended sediments, and sedimentation, can affect benthic, epibenthic, and infaunal communities, which may impact marine mammals indirectly through changes to their prey.

2.3.3 Foodweb indicators

Many foodweb indicators are also relevant to other MSFD descriptors including D1 and D3 (groups/species targeted by human activities) and D6 (early warning indicators).

The existing suite of indicators gives variable focus to the three important foodweb properties, structure, functioning and dynamics, and more emphasis should be given to the latter two. Indicators based on the structure and processes of benthic groups can help to describe trophic functioning. However, the currently proposed indicator 4.3.1 (Abundance trends of functionally important selected groups/species) is based on the rationale that changes in population status of functionally important species or groups will affect food web structure and functioning (Rombouts *et al.*, 2013).

The proposed indicators, particularly those based on abundance and biomass, can inform on the structural properties of foodwebs, but they may provide only partial information about its functioning. Hence, the development of criteria for D4 should be directed towards more integrative and functional indicators that consider: (i) multiple trophic levels or a whole-system approach (i.e. ecosystem-based indicators), (ii) processes and linkages (e.g. trophic transfer efficiencies); and (iii) the relation of foodweb dynamics to specific anthropogenic pressures.

Our ability to predict community change is still impeded by a lack of knowledge of long-term functional dynamics that span several trophic levels. In a long-term dataseries spanning four decades (Törnroos *et al*., 2019), the linkage between fish and zoobenthic functional community change was weak, with the timing of the changes being area- and trophic-group specific. Therefore, the authors of this study recommended quantifying change in multiple functional measures, to help assessments of biodiversity change move beyond taxonomy and single trophic groups.

Table 2.4. Contribution to MSFD D4 according to the various impacts detailed in the ICES Guidelines for the management of marine sediment extraction (ICES, 2003).

Effects of extraction	**Impact on**	**Potentially influenced MSFD descriptors**	**Level of contribution of WGEXT guidelines to MSFD descriptors**			
			Introduction	**Baseline survey**	**Impact assessment**	**Mitigation**
Seabed removal	Habitat and communities	D4: All elements of the marine foodwebs occur at normal abundance and diversity (functional aspects) D4.3: Abundance/ distribution of groups/ species targeted by human activities		Biological setting: "trophic relationships (e.g. between benthos and demersal fish populations by stomach contents analysis)"	Biological impact assessment: "effects on the fishery and shellfishery resources" and "effects on trophic relationships (e.g. between the benthos and demersal fish populations)"	"Agreeing exclusion areas to provide refuges for important habitats or species, or other sensitive areas" and "Selection of dredging equipment and timing of operations to limit impact upon the biota (birds, benthos, sensitive species and habitats)"
Sediment plume	Turbidity	D4: All elements of the marine foodwebs occur at normal abundance and diversity (functional aspects)			Physical impact assessment: "transport of fine sediment from overflow" Biological impact assessment: "effects of aggregate dredging on pelagic biota", "effects on the fishery and shellfishery resources including spawning areas, nursery areas, overwintering grounds for ovigerous crustaceans, and known routes of migration", and "effects on any other legitimate use of the sea"	

2.3.4 Conclusion

With respect to D4, direct and indirect effects of marine aggregate extraction are proportional to the size of dredging areas, with limiting factors, such as the trophic adaptability of fish and bird species, the ability to move to avoid disturbed areas, and the tolerance of marine mammals to turbidity. Mitigation measures were included in the WGEXT guidelines (ICES, 2003) to protect sensitive species and habitats, and limit the impact upon the biota (see Table 2.4).

2.4 Descriptor 6: Seabed integrity

"Seabed integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected. Physical and biological damages, having regard to substrate characteristics." (EC, 2010)

The physical impact of extraction is site-specific and linked to many factors, such as hydrodynamics, sediment grain size, and dredging method and intensity. The action of extracting aggregates alters the seabed topography, creating isolated furrows (dredge tracks about 2–3 m wide and 0.5 m deep) in extensive sites (Cooper *et al.*, 2005; Le Bot *et al.*, 2010), and persistent depressions up to several meters deep after several years of localized extractions (Degrendele *et al.*, 2010; Gonçalvez *et al.*, 2014; de Jong *et al.*, 2015; Mielck *et al.*, 2021).

Aggregate removal can lead to a change in the seabed substrate by removing superficial layers of sediment and exposing coarser sediments (Cooper *et al.*, 2007; Le Bot *et al.*, 2010), or by altering the particle size distribution as a result of deposition from overflow (Boyd *et al.*, 2005; Krause *et al.*, 2010; Barrio-Froján *et al.*, 2011; Cooper *et al.*, 2011; Wan Hussin *et al.*, 2012; de Jong *et al.*, 2015). Aggregate extraction generally results in an increased variability of particle size composition within both high and low dredging intensity sites (Cooper *et al.*, 2007; Desprez *et al.*, 2014).

In the case of screening, intensive deposition of fine sands from overflow can affect the seabed substrate in a wide area outside the extraction area (Boyd and Rees, 2003; Newell *et al.*, 2004b; Barrio-Froján *et al.*, 2008; Last *et al.*, 2011; Tillin *et al.*, 2011; ICES, 2016a).

The distribution of marine organisms and communities is strongly related to hydrodynamic and morphological parameters, and sediment type (McLusky and Elliott, 2004; Baptist *et al.*, 2006; Degraer *et al.*, 2008; Pesch *et al.*, 2008). Thus, any physical changes in the seabed will lead to a response in the composition of its natural benthic assemblages. This will affect the habitat quality in a wider area, impacting the transport of fish larvae, and the abundance of food for fish, birds, and mammals.

The direct removal of surface aggregate sediments and associated fauna results in an immediate and local loss of the benthic fauna in the order of 60% for the number of species, and 80–90% for abundance and biomass (Newell *et al.*, 1998, 2004a, 2004b; Desprez, 2000; Boyd and Rees, 2003; ICES, 2009; Krause *et al.* 2010; Desprez *et al.*, 2014). The impact may range from almost total defaunation (Simonini *et al.*, 2007) to a more subtle and less significant change (e.g. van Dalfsen *et al.*, 2000; Robinson *et al.*, 2005).

Impacts of extensive dredging (frequency of disturbance < 10 h ha^{-1} year^{-1}, footprint < 50% of the licensed area), occurring mainly in areas with strong hydrodynamic conditions and mobile sediments, tend to be less pronounced, and have limited functional consequences (e.g. lower reduction in biomass) on the higher trophic levels (Bonvicini *et al.* 1985; Desprez *et al.*, 2014). In sandy areas of the North Sea and the Baltic Sea, the effects of sand extraction only became evident when the annual extractions affected 50% of the licensed area, causing a drop in biomass values (Birklund and Wijsman, 2005). Thus, extraction intensity is clearly an indicator for the sustainability of the impact on benthic communities, and a spatio-temporal indicator for managing marine resources (Bokuniewicz and Jang, 2018).

The cumulative impact, in time and/or space, of multiple extractions, results in a continuous disruption of benthic communities, which are reduced to their simplest form (few tolerant species, reduced abundance, and minimal biomass due to the elimination of long-living bivalves and echinoderms; Boyd and Rees, 2003; Newell *et al.*, 2004a; Robinson *et al.*, 2005; Cooper *et al.*, 2007; Barrio-Froján *et al.*, 2008).

Differences in impact, and subsequent recovery, also depend on local hydrodynamics (Mestre *et al.*, 2013), sediment characteristics, and the nature and type of stress to which the community is adapted to in its natural environment (ICES, 2009). In the sandy bottoms of the North Sea, small-scale disturbances in seabed morphology and sediment composition result in limited effects on the benthic community (van Dalfsen *et al.*, 2000), but large-scale and deep sand extractions can result in a net increase in sediment fines and a corresponding increase in the biomass of the white furrow shell (*Abra alba*; de Jong *et al.*, 2015).

In gravelly areas, the impact of aggregate extraction is higher, because the heterogeneity and stability of this type of sediment favours more diversified and abundant communities (Seiderer and Newell, 1999; Newell *et al.*, 2001; Cooper *et al.*, 2011).

The main indirect impact of dredging is linked to the deposition of sediment from the overflow or screening plume, which can cause smothering/damage to sensitive benthic receptors. The extensions of deposits have been calculated for spring tides conditions in the English Channel at 800 m for sand and 6.5 km for silt (Duclos *et al.*, 2013).

The majority of studies (Desprez, 2000; Boyd and Rees, 2003; Newell *et al.*, 2004b; Cooper *et al.*, 2007; Desprez *et al.*, 2010) suggest that adverse biological change is constrained to 100–200 m from the dredge area, even where sedimentary change has been detected at greater distances, of up to 2 km from the dredge site, following remobilization by strong local tidal currents (Newell *et al.*, 2002; Robinson *et al.*, 2005; Cooper *et al.*, 2007; Desprez *et al.*, 2010).

Several types of indirect effects have been observed depending on the intensity of oversanding and the nature of the seabed:

- On gravelly bottoms, the elimination of the benthic fauna can be almost complete, equivalent to the loss observed in the dredged area, because the original communities are unable to withstand a big deposition of fine sands (ICES, 2009; Desprez *et al.*, 2010). Under permanent extraction activities and remobilization in areas under strong hydrodynamic conditions, the original stable bottom will be replaced by a continuously remobilized substrate (Newell *et al.*, 2004b; Robinson *et al.*, 2005; Desprez *et al.*, 2010).
- Beyond a few hundred meters from the extraction site, there can be a rapid increase in the number and abundance of species consistent with the low dispersion of overflowing sediments. The average biological characteristics of samples were taken from within four predefined zones of deposition next to an aggregate extraction site off Dieppe, France (Table 2.5). Boyd and Rees (2003) also showed that faunal composition changed gradually with distance from the extraction site. This is mainly due to the fact that the distribution of species is correlated with the sedimentary characteristics of the deposition area (medium to fine sand).

- A transition from a sandy-gravelly bottom with a diverse epifauna to a sandy seabed with a less diverse infauna can occur as a result of overflow (Boyd *et al*., 2005; ICES, 2009; Desprez *et al*., 2010).
- On sandy bottoms, the benthic fauna are less affected in the deposition area than in the extraction site (Newell *et al*., 2004b). The benthic species that are least sensitive to overflow deposits are those able to move rapidly through the sediment, and free-swimming epifaunal species (e.g. crabs and shrimps).
- Species richness, abundance, and biomass can locally increase at the limits of overflow areas when sediment deposition is extensive and the available food is increased through organic enrichment (Newell *et al*., 2002; Desprez *et al*., 2010).

Generally, the creation of sediment plumes has the potential to adversely impact benthic organisms through an increase in sediment-induced scour and smothering, and through damage and blockage to respiratory and feeding organs (Tillin *et al*., 2011). The effects of suspended sediments and sedimentation are species-specific, but invertebrates, eggs, and larvae are most vulnerable.

Studies, such as Last *et al*. (2011), have investigated the impacts of increased suspended particulate matter (SPM) and smothering on a number of benthic species of commercial or conservation importance, under a range of environmental and depositional conditions. All species survived the higher SPM conditions. The ross worm (*Sabellaria spinulosa*) was highly tolerant to short-term burial (< 32 d), and its growth rate showed significantly higher tube growth under high SPM conditions. Szostek *et al*. (2013) showed that elevated SPM had no short-term effects on survival of the king scallop (*Pecten maximus*), but observed a reduction in its growth rate. This species appeared more tolerant of burial and elevated levels of SPM than the queen scallop (*Aequipecten opercularis*).

Table 2.5. Biological characteristics of samples taken from different zones of the Dieppe extraction site (From Desprez et al., 2010). Numbers in parentheses are standard error. AFDW: Ash free dry weight.

Predicted deposition zone	Number of species	Abundance (ind. m^{-2})	Biomass (g AFDW)
No deposition	50 (12)	2 394 (1 030)	12 (6.6)
Low deposition	80 (4)	3 906 (26)	16 (2.1)
Moderate deposition	28 (6)	585 (282)	5 (1.8)
High deposition	18 (8)	262 (107)	1.4 (1.6)

2.4.1 European Commission selected indicators for seabed integrity

Indicators that show the ecosystem response to human pressures form the basis of the tool kit with which we can describe environmental status (Borja *et al*., 2016). A number of such indicators have been selected by the European Commission to assess seabed integrity (see Rice *et al*., 2012).

2.4.1.1 Type, abundance, biomass, and areal extent of relevant biogenic substrates

Examples of the coastal ecosystems dominated by epibenthic engineers are *Sabellaria* spp. reefs, *Mytilus* spp. beds (Cooper *et al*., 2007; Pearce *et al*., 2007, 2014; Gibb *et al*., 2014), *Chaetopterus* spp. beds (Rees *et al*., 2005), *Lanice* spp. meadows (Braeckman *et al*., 2014), and other biogenic reefs (Farinas-Franco *et al*., 2014). These are some of the most valuable ecosystems in the world, but remain threatened and declining.

An example of a reverse in biodiversity decline, has been the return to extraction sites of the tubeworm *Sabellaria spinulosa* (Cooper *et al*., 2007; Pearce *et al*., 2007; Gibb *et al*., 2014; Desprez *et al*., 2014; key species of the Habitats Directive and the OSPAR list of endangered species), observed from the early stages of recolonization, and possibly facilitated by the deposit of sand overflow.

2.4.1.2 Extent of the seabed significantly affected by human activities for the different substrate types

Halpern *et al.* (2008) estimated that 41% of marine areas are already strongly affected by multiple anthropogenic perturbations, but did not allow for mitigation measures for these activities. In their assessment of six direct physical pressure types affecting the seabed of England and Wales (i.e. physical loss, physical damage, non-physical disturbance, toxic contamination, non-toxic contamination and biological disturbance), Eastwood *et al.* (2007) estimated that selective extraction caused by demersal trawling affected 5–21% of the total area, while the pressure arising from aggregate dredging affected only 0.1% through direct removal, and 1.2% through the siltation caused by screening plumes. Thus, in comparison to the relatively localised impacts of aggregate dredging, seabed disturbance by demersal fishing gear constitutes over 99% of the known footprint of all human pressures on the UK seabed (Foden *et al*., 2010).

In the study from Eastwood *et al*. (2007), 0.1% of the total area corresponds to the cumulative surface of all licensed areas which are potentially affected by the draghead. However, this percentage does not consider the undisturbed areas within the licensed site. Consequently, the direct footprint is probably even lower than 0.1%.

The relatively minor impact of seabed mining reflects its localized effects and relatively small footprint, together with the well-developed management measures for this industry sector (Elliott *et al*., 2014). Although the pressure of sediment extraction is several orders of magnitude lower than the pressure resulting from fishing, mandatory data collection is recommended from the dredgers electronic monitoring systems (EMS) or automatic identification systems (AIS), along with the collation of data on the licensed area. It should not be overlooked that an activity may have a disproportionate effect on a specific biological habitat (ICES, 2019a). It should also be noted that using licensed areas as a proxy overestimates the pressure from marine sediment extraction, because only a part of these areas is really extracted on a yearly basis. The ICES advice on D6 to the EU mentions that the information from EMS and AIS should be used to provide a more detailed understanding of the location of extraction areas within a site, and to use the recording of additional metrics such as volume of extracted aggregates, over and above the common metric of licensed area (km^2; ICES, 2019b).

2.4.1.3 Presence of particularly sensitive and/or tolerant species

Sensitivity measures the degree of the response to stress using indicators (species, communities, or habitats). Identifying the sensitivity of species and biotopes relies on accessing and interpreting available scientific data in a structured way, to disseminate suitably presented information to decision-makers (Hiscock and Tyler-Walters, 2006). Sensitivity information can be overlaid with the distribution of protected or threatened species and habitats, designated areas, and the location and intensity of specific activities considered damaging to the marine environment.

The mapping of different benthic habitat components is considered key information for the implementation of the MSFD, particularly for the identification of sensitive habitats. WGMHM is currently examining the managerial uses of habitat maps (e.g. in assessments of environmental status; ICES, 2016a).

The ICES Guidelines for the management of marine sediment extraction (ICES, 2003) point out the importance of this indicator in the selection process of extraction areas to protect threatened benthic communities, and to allow effective resource management. The most sensitive species/habitats are maërl beds (with high structural diversity), spawning areas (fundamental to functional diversity), and biogenic reefs (important for both structural and functional diversity), all of which have specific protection measures (OSPAR[3], Natura 2000).

The presence of particularly sensitive or tolerant species should inform on the condition of the benthic community but other characteristics of the benthic community, such as species composition, size composition, and functional traits, provide an important indication of the potential of the ecosystem to function well. However, Zettler *et al*. (2013) demonstrated that the use of static indicator species, which assumes that species have a similar sensitivity or tolerance to natural or human-induced stressors, does not account for possible shifts in tolerance along natural environmental gradients, and between biogeographic regions. Therefore, their indicative value is questionable.

The level of pressure on habitats and species will be different depending on the nature of the extraction-related impact. Table 2.6 details the level of impact observed at an extraction site in Dieppe (Desprez, 2011) on the different habitats and species identified in the major international conventions that regulate the management of the activities and the protection of the marine ecosystem.

Table 2.6. Sensitivity of key-species and habitats (identified by international conventions) to various levels of impact of marine aggregate extraction (E: extraction; T: turbidity; D: deposition) in Dieppe.

Sensitivity to extraction	Pressure levels				
Indicators of impact	**High**	**Mean**	**Low**	**Negligible**	**Positive**
OSPAR species					
Cod	T	D			E (zoning)
Rays			E/T	D	
OSPAR habitats					
Sabellaria reefs	E			T	D
Maerl banks	E/T/D				
Hard substrates with Modiolus	E/D		T		
ICES habitats					
Spawning areas	E/T/D				
Nurseries	E/D			T	
Shell beds	E	D		T	
NATURA 2000					
1110.2 (gravelly sands)		E/T/D			
1110.3 (medium sands)			E/T	D	

Ware et al. (2009) provided different options for aggregate indicators based on impacts to the physical and biological environment. These include the percentage of silt/sand and gravel, and benthic indices such as diversity and biomass (van Hoey *et al*., 2007, 2010). It should be noted

[3] https://www.ospar.org/work-areas/bdc/species-habitats/list-of-threatened-declining-species-habitats. Last accessed 10th March 2022

that the efficacy of both the Infaunal Quality Index and M-AMBI (multivariate-Aztiz´s marine biotic index) cannot currently be supported for inshore gravel (Fitch *et al*., 2014).

Other indicators have also been proposed, such as biological traits of the benthic community (Bremner *et al*., 2006a, 2006b, 2008), habitat heterogeneity (Hewitt *et al*., 2008), and functional diversity (Törnroos *et al*., 2014).

Hewitt *et al.* (2008) showed that biological trait analysis (BTA) was able to distinguish differences in sensitivity at a site to different stressors (extraction, sedimentation, and suspended sediments). Thus, BTA could be used as a first step in strategic prioritization of sensitive areas, and as an underlying layer for spatial planning.

Functional indices may provide a more detailed assessment of benthic communities than structural ones, but the overall outcome is broadly similar for both types of indices. This suggests that the measurement of functional indices may be unnecessary for routine monitoring purposes (Culhane *et al*., 2014; Strong *et al*., 2015), although they may provide some value by revealing more specific aspects of change in a system.

Between-habitat differences in functional traits are driven by differences in organism density, rather than by the presence/absence of individual traits, emphasizing the importance of density shifts in driving function (Hewitt *et al*., 2008).

Metrics which are closely associated with species number and density of individuals scored highest for sensitivity to aggregate extraction impacts. Similar findings can be found in the literature regarding a variety of activities that typically result in physical impacts on the seabed and its associated fauna (Ware *et al*., 2009, 2010). A benthic ecosystem quality index (BEQI) was developed by van Hoey *et al*. (2007) for *inter alia* monitoring windfarms, maintenance dredging deposits, and aggregate extraction, and has been used on the Belgian continental shelf (de Backer *et al*., 2014). However, while some indicators are used already to a certain extent, further work is required to develop approaches for assessing the physical impacts of aggregate extraction (Schleuter *et al*., 2010; Fitch *et al*., 2014).

The relative lack of sensitivity of traditional indices [(AMBI, M-AMBI, ITI (Infaunal trophic index[4]) and BENTIX (Biotic index[5])] may be attributed to their dependence on species responses to organic enrichment (Ware *et al*., 2009; Targusi *et al*., 2014), an impact not routinely associated with aggregate extraction activities (Salas *et al*., 2006).

Indices can be appealing, because they can be used to reduce complex data to single numbers, which seem easy to understand (Green, 2011). However, this is not representative of the biological or environmental reality, which is rarely one-dimensional. Green (2011) suggests that indicators should not be used because information can be lost, and misleading conclusions can be reached. He concludes that if indices must be used for a non-scientific reason, it is better to use them together with other statistical methods that retain more of the information in the biological dataset.

Impact indicators for major drivers of marine biodiversity loss are currently lacking (Woods *et al*., 2016). As knowledge and understanding increases regarding the strengths and weaknesses of competing index approaches, unified approaches must be developed that provide managers with the simple answers they need to use information on the ecological condition of an area effectively and efficiently (Borja *et al*., 2009a, 2009b).

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[4] Numerical representation of the distribution of dominant feeding groups of benthos.

[5] Index based on the concept of indicator groups.

2.4.1.4 Impact and natural variability (signal-to-noise ratio)

Ecological and environmental variability of natural ecosystems (see Gray and Elliott, 2009) precludes the widespread use of simplistic design and analysis tools to detect the effects of human activities (Frid, 2011; Frid and Caswell, 2014; Clare *et al*., 2015). Scale is one of the most important concepts in impact assessment (Hewitt *et al*., 2001). As the spatial or temporal scale increases, both the number of processes, and their importance in influencing local populations and communities, will change, thereby increasing the variability encompassed by the study.

The long-term variability in diversity, traits, and functions in benthic communities is largely unknown. How these changes affect ecosystem functioning and services is one of the most challenging current research questions, and not only in the field of benthic ecology research. An analysis of the taxonomic and trait-based macrofauna long-term community variability and diversity (Meyer and Kröncke, 2019), showed that taxonomic and trait-based diversity remained stable over time, while different regimes were found in taxonomic and trait-based community structure, correlated with climatic variables and epibenthic abundance as the most important environmental drivers

2.4.2 Recovery

Impact assessments should capture the physical and ecological recovery after cessation of the pressure. It should be noted that ecological recovery of biota can occur without a full physical recovery of the geomorphology of the seabed (ICES, 2019a).

The recovery time depends on, or is influenced by, the amount, intensity, and frequency of aggregate dredging. In some cases, dredging can lead to a complete defaunation of the sediment (Simonini *et al*., 2007), minimizing possibilities of recolonization from adjacent areas. Furthermore, the exposure of anoxic sediments can prevent recolonization (Krause *et al*., 2010).

The recovery time is also strongly related to environmental characteristics (Woods *et al*., 2016). The importance of hydrodynamics was observed around the UK (Foden *et al*., 2009, 2010), where 96% of extraction activity occurs in sand or coarse sediment. The mean period for biological recovery was 8.7 years in deeper targeted coarse sediments with moderate tidal stress, while for shallow coarse sediments with weak tidal stress a longer period of 10.75 years was estimated. Foden *et al*. (2009) observed that the mean period for physical recovery can be more than double that period (20 years) in deep coarse sediments.

Clean sand communities, adapted to high energy environments, have the most rapid recovery rate following disturbance (Dernie *et al*., 2003; Foden *et al*., 2009; Coates *et al*., 2014). Simonini *et al*. (2007) observed the end of the recovery phase (structure and community composition) after 30 months in sand seabeds where dredging operations did not change the physical characteristics of the sediment, but led to a complete defaunation at the dredged site.

To minimize recovery times following the cessation of dredging, it may be preferable to grant new aggregate extraction licences in sites of high natural disturbance (e.g. coarse sand dunes), where the macrofaunal communities present are poor (< 5 g m^2), adapted to regular bottom disturbance, less sensitive to the physical impacts caused by dredging, and able to rapidly recolonize exploited sites (Cooper *et al*., 2005, 2011).

Extraction intensity may also influence the rate of recovery (Boyd *et al*., 2003, 2004; Thrush *et al*., 2008; Birchenough *et al*., 2010; Wan Hussin *et al*., 2012; Waye-Barker *et al*., 2015). Recovery times of 7 years have been observed at sites with low dredging intensity (< 1 h ha^{-1}), and up to 15 years at sites with high dredging intensity (> 10 h ha^{-1}). The timing of the dredging in relation to the timing of recruitment, can be another means to enhancing recolonization.

Unless physical conditions can first be restored, impacted sites may not fully recover the pristine biological community (Cooper *et al*., 2011). Fifteen years after the cessation of extraction at the site in Dieppe, pebble crests, and their associated benthic and fish communities, are still present in a natural environment of coarse sands (Desprez *et al*., 2014). This situation is similar to that observed for wind farms, which have introduced artificial hard substrates in the sandy sediments of the North Sea (de Troch *et al*., 2013; Wehkamp and Fischer, 2013; Vandendriessche *et al*., 2014; Stenberg *et al*., 2015; Dannheim *et al*., 2019), with a highly species-specific attraction effect for fish (adequate refuge in combination with additional food resources).

Achieving a functioning ecosystem is more important, and more relevant to the definitions of recovery, than merely achieving the presence of structural features (e.g. the presence of certain species; Verdonschot *et al*., 2012). The ecological-functionality concept focuses on the conservation and recovery of functionality at the species and ecosystem levels. It quantifies ecological function from the viewpoint of species recovery, and proposes a Green List for species that defines a fully recovered species through representation, viability, and functionality. This concept is designed to integrate conservation at population, species, and ecosystem levels, beyond the minimal requirement of maintaining species presence through extinction avoidance. Considering functionality may also be a good opportunity to catalogue the benefits of biodiversity for human wellbeing (Resit Akçakaya *et al*., 2020).

The rate of stabilization and recovery of ecological functioning appears to depend on the environmental context, but can be in the order of 5–10 years for marine benthos (Coates *et al*., 2014; Waye-Barker *et al*., 2015).

Physical disturbances of the seabed by fishing gears (trawling and dredging) can result in permanent community changes when the frequency and extent of disturbance outstrips the recovery potential (Thrush *et al*., 2008). For marine aggregate extraction, exact values for acceptable disturbance limits have yet to be developed (Cooper, 2012, 2013). However, different functional metrics used to investigate the rate of recovery in ecosystem function after dredging, indicate that the disturbed area is capable of a full recovery given enough time: 1–2 years at a low dredging intensity site, 2–4 years after short but intensive dredging events (Kenny *et al*., 1998; Sarda *et al*., 2000; van Dalfsen *et al*., 2000; van Dalfsen and Essink, 2001), and up to 15 years after a long period of commercial extraction (Wan Hussin *et al*., 2012; Waye-Barker *et al*., 2015). These time-scales were observed with traditional measures of abundance and biomass (Cooper *et al*., 2005). However, are there limits beyond which the capacity of impacted habitats to recover is compromised?

After many years of sustained dredging in the North Sea, it was seen that even when one of the measured variables departed significantly from an equitable state, the effect did not persist from one year to the next. The potential for short-term partial recovery of the assemblage had not been compromised, at least in terms of abundance and species richness (Barrio-Froján *et al*., 2008). The authors of this study mention the need for models specifically designed to assess the degree of acceptable disturbance from aggregate extraction.

Complete recovery is the return of an ecosystem to its original, pre-disturbance state, whereby the abundance, diversity, structure, and functioning of the biological community are the same as prior to the disturbance (Woods *et al*., 2016). However, a system recovery may not require a return to a similar biomass, biodiversity, or community composition. Frid and Caswell (2014) showed evidence that, during some periods, changes in function were linked to changes in several key, or rivet taxa, whereas, during other periods, function was maintained in the face of taxonomic change. Clare *et al*. (2015) confirmed that ecological functioning, in terms of trait composition, was statistically indistinguishable across periods that differed significantly in taxonomic composition.

Wan Hussin *et al*. (2012) stated that functional metrics are complementary to traditional environmental assessments metrics, for measuring the recovery of macrofaunal communities after marine aggregate dredging. Further, analysis suggests that ecological functioning can be sustained in communities undergoing long-term compositional change, as characteristically similar (redundant) taxa exhibit compensatory changes in population densities (Clare *et al*., 2015).

GES cannot be defined exclusively as pristine status, but is rather the status when the impacts from use are sustainable. For this, two conditions need to be met (Rice *et al*., 2012):

- The pressure does not hinder the ecosystem components from retaining their natural diversity, productivity, and dynamic ecological processes; and
- recovery from perturbation, such that attributes lie within their historical range of natural variation, must be rapid and secure.

For Borja (2014), recovering ecosystem structure and functioning is a grand challenge. Therefore, studies are needed for a deeper knowledge of recovery processes (Borja *et al*., 2010), and for promoting the ecological restoration of damaged ecosystems.

2.4.3 Restoration

In general, there are two types of restoration (Elliott *et al*., 2016): (i) type A, which consists of the modification of the environment to allow biota to recover; and (ii) Type B, which consists of the direct restocking or replanting of biota.

Few studies provide evidence of how ecological knowledge (ecoengineering) might enhance restoration success (Verdonschot *et al*., 2012). Type A restorations can be either active (seeding of shells and gravels) to remedy any critical damage caused (Collins and Mallinson, 2007; Cooper, 2012, 2013), or passive, by minimizing stressors to prevent any critical damage (timing of dredging in relation to timing of recruitment, extensive extraction, and landscaping; Desprez *et al*., 2014; de Jong *et al*., 2015).

Seabed landscaping aims to create diverse habitat conditions in sand extraction areas by leaving large-scale bed forms on the dredged seabed after completion of the work. In this way, landscaped mining areas are hypothesized to encourage recolonization and promote higher biodiversity and productivity after completion of the dredging work (de Jong *et al*., 2014, 2016; Rijks *et al*., 2015).

The limited number of restoration studies underlines several problems: (i) the effects mostly occur only in the short term and at a local scale, (ii) the organism group(s) selected to assess recovery do not always provide the most appropriate response, (iii) the recovery time-lag is highly variable, and (iv) most restoration projects incorporate restoration of abiotic conditions, and do not include abiotic extremes and biological processes.

2.4.4 Conclusion

ICES guidelines (ICES, 2003) recommend the use of mitigation methods to limit changes to seabed topography and sediment transport, and to sediment type and associated biota in sensitive areas, either directly or indirectly (Table 2.7).

With respect to D6, WGEXT recognizes that the extraction of marine sediments will result in direct changes to physical parameters, and the function and structure of ecosystems. The exploitation of marine aggregates should preferably take place in naturally unstable bottoms (e.g. coarse sand dunes), where benthic communities are poor, adapted to regular bottom disturbance, and able to rapidly recolonize exploited sites. Whereas such an approach is theoretically attractive, it is doubtful whether these locations can provide what the industry needs, both in terms of quantity and proximity to markets.

However, the group is content that, in the context of appropriate consent regimes that provide for rigorous environmental assessment and evaluation of each proposal to extract sediment, the impacts of marine aggregate extraction may be considered to be within environmentally acceptable limits and, therefore, not adverse (Cooper, 2013; Cooper and Barry, 2017).

WGEXT suggest that in defining what adverse impacts are, it should be accepted that direct changes to the physical structure of the seabed will result from the extraction of marine sediments. Defining non-adverse as connoting no environmental change from pre-dredge conditions would, in the opinion of the group, be inappropriate and detrimental to the continued ability of Member Countries to extract marine sediments from their seabed. The reason for this is that there can be environmental change independent from the extraction. Thus, a comparison would need to be made with a reference area, and not solely with pre-dredge conditions. In addition, benthic fauna can recover even without the recovery of the seabed morphology at the extraction location. Therefore, WGEXT recommends that the possibility of recovery after marine sediment extraction should be acknowledged by incorporating it as a D6 criteria, and by taking it into account when assessing GES (ICES, 2016c).

2.5 Descriptor 7: Hydrographical conditions

"Permanent alteration of hydrographical conditions does not adversely affect marine ecosystems." (EC, 2010).

Changes in seabed morphology and associated hydrodynamic effects have the potential to affect adjacent coastlines (Demir *et al*., 2004; Kortekaas *et al*., 2010). For many sandy beaches, the coast is eroded by storm waves and storm currents in winter, and rebuilt under calmer conditions in summer. The area where the exchange of sand is taking place on a yearly or multiyear time-scale is called the active coastal or active beach profile. If dredging is undertaken within the area of sediment movement known as the active beach profile, material can become trapped within depressions caused by dredging, preventing it from moving back onshore during calmer conditions (Brampton and Evans, 1998). As a consequence, extraction sites are rarely located in the nearshore area of sediment movement where they could affect the beach, either during summer by preventing its nourishment, or during winter by increasing its erosion.

In the North Sea, below the 20-m depth contour, no impacts from aggregate extraction were observed on wave regime, sediment transport, or the stability of the coastline, even in the case of a large-scale 16 km^2 extraction pit with a depth of 20 m (Rijkswaterstaat, 2014). Closer onshore, the removal of sediment during marine aggregate extraction may impact sediment transport pathways that replenish the coastline.

In southern Portugal, sand was dredged on the continental shelf for beach nourishment, and a research project (SANDEX) assessed its physical effect on the seabed and coastline. Around 370 000 m^3 of sand were extracted leaving a rectangular sandpit with dimensions of 900 m length, 150 m width, and, on average, 5 m depth, located 4000 m from the shore, at a depth of 15–20 m (Gonçalves *et al*., 2014). Numerical modelling showed that the tidal flow and the orbital wave velocities within the pit and neighbouring areas were modified by the presence of the pit. The excavation influenced the tidal flow in an area of approximately 9000 m^2 around it. In that area, the maximum flow velocity increase was 2%, observed near the pit, and the maximal decrease was 16%, seen in the deepest zone of the pit. The orbital velocities for the storm wave

Table 2.7. Bathymetry/topography effects, sediment change effect, and turbidity effects with relevance to MSFD D6, based on impacts detailed in the ICES Guidelines for the management of marine sediment extraction (2003).

Effects of extraction	Impact on	Potentially influenced MSFD descriptors	Level of contribution of WGEXT guidelines to MSFD descriptors			
			Introduction	Baseline survey	Impact assessment	Mitigation
Seabed removal	Bathymetry/ topography	D6: Seabed integrity. D6.1. Physical damage, having regard to substrate characteristics		Dredging activity: "spatial design and configuration of aggregate dredging (maximum depth of deposit removal, shape and area of resulting depression)". Physical setting: "bathymetry and topography of the general area"	Physical impact assessment: "changes to the seabed topography", "changes to the behaviour of bedforms within the extraction and adjacent areas" and "time-scale for potential physical recovery of the seabed"	"Modification of the depth to limit changes to sediment transport"
		D6.2. Condition of benthic community			Biological impact assessment: "changes to the benthic community structure"	
	Sediment composition	D6: Seabed integrity D6.1. Physical damage, having regard to substrate characteristics		Physical setting: "sediment particle size distribution", "stability and/or natural mobility of the deposit", and "estimate of bed-load sediment transport"	Physical impact assessment: "changes to sediment type", "exposure of different substrates", "transport and settlement of fine sediment from overflow"	
		D1: Biological diversity is maintained: quality and occurrence of habitats, and distribution and abundance of species		Biological setting: "fauna and flora within the area likely to be affected by aggregate" and "presence of any areas of special scientific or biological interest designated under local, national or international regulations"	Biological impact assessment: "changes to the benthic community structure and to any ecologically sensitive species or habitats" and "effects on sites designated under local, national or international regulaions"	"Agreeing exclusion areas to provide refuges for important habitats or species, or other sensitive areas" and "Selection of dredging equipment and timing of operations to limit impact upon the biota (birds, benthos, sensitive species and habitats"

Table 2.7 (cont.)

Effects of extraction	Impact on	Potentially influenced MSFD descriptors	Level of contribution of WGEXT guidelines to MSFD descriptors			
			Introduction	**Baseline survey**	**Impact assessment**	**Mitigation**
Sediment plume	Turbidity	D6.2.1. Presence of particularly sensitive species		Biological setting: "presence of any areas of special scientific or biological interest adjacent to the proposed extraction area, such as sites designated under local, national or international regulations"	Physical impact assessment: "effects on water quality through increases in the amount of fine material in suspension" Biological impact assessment: "effects on the fishery and shellfishery resources including spawning areas, nursery areas, overwintering grounds for ovigerous crustaceans, and known routes of migration"	"Preventing on-board screening or minimizing material passing through spillways when outside the dredging area to reduce the spread of the turbid plume"
	Deposition	D6: Seabed integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems are not adversely affected. D6.2.1. Presence of particularly sensitive species			Physical impact assessment: "settlement of fine sediment from overflow" Biological impact assessment: "effects on the fishery and shellfishery resources including spawning areas, nursery areas, overwintering grounds for ovigerous crustaceans, and known routes of migration"	Preventing on-board screening or minimizing material passing through spillways when outside the dredging area to reduce the spread of the turbid plume

conditions showed a decrease of 15% within the pit, and an influence extending up to the 4 m contour, but not reaching the shore (Lopes *et al.*, 2009). Bathymetric analysis, conducted between May 2006 and November 2008, showed an accretion of sediments of around 60 000 m^3, which would put the recovery time from the excavation at about 24 years, very similar to modelling results. Phillips (2008) investigated areas in the south of Wales where critical beach loss has been associated with dredging activities. However, after five years of beach monitoring, no qualitative or quantitative links between marine aggregate dredging and beach erosion could be found. Instead, natural changes, such as shifts in wind direction and an increase in easterly storms, were the most significant factors affecting beach formation processes.

The removal of a significant depth of sediment results in a localized drop in current strength, associated with the increase in water depth. This reduced strength in the bottom current can cause the deposition of fine sediments within the dredged depressions from overflow discharges (Duclos *et al.*, 2013; Krause *et al.*, 2010) and/or from natural sediment transport (Desprez, 2000; Cooper *et al.*, 2007; Le Bot *et al.*, 2010). For the seaward harbour extension of the Port of Rotterdam, which started 2009, the large-scale sand extraction, down to 20 m below the seabed, generated a rapid and local marked increase in the fraction of fine muddy sands in the troughs and deepest areas of the extraction site (de Jong *et al.*, 2014). At several other locations in the pit, silt concentrations increased up to 8% after a few fluctuations (van Tongeren, 2018). In 2018, a multibeam backscatter survey focusing on sediment classification showed that there was no indication of a significant rate of enrichment with mud in the pit. Rather, there had been a slow accumulation of fine sediments, particularly in the oldest parts of the pit (van Dijk *et al.*, 2019).

A dredging strategy favouring extensive extraction on larger surfaces with trailing dredges can limit the depth of pits, and their consequences on topography, hydrodynamics, and sediment. In addition, it will limit the impact on the biota, favouring benthic recolonization, and will respect access to traditional fisheries.

2.5.1 Conclusion

As a result of the influence of changing hydrodynamics on the benthos, ICES guidelines (ICES, 2003) proposed mitigation for extraction methods to limit the depth of extraction, the creation of topographic features, and associated changes to hydrodynamics (see Table 2.8).

In general, and in relative terms, the dimensions of dredged pits are so small that the deepened area has little influence on the macroscale current pattern. Furthermore, it was concluded that, in most cases, the current pattern would only be changed in the direct vicinity of the dredged area.

2.6 Descriptor 8: Contaminants

"Contaminants are at a level not giving rise to pollution effects." (EC, 2010)

Contaminants are generally associated with the silt/clay fraction of sediments. Given that aggregate dredging targets sand and gravel, this issue is not typically of concern. In an extraction site located near the mouth of the River Seine estuary, Ifremer studied the effect of marine aggregate extraction on water quality, due to the potential remobilization of contaminants from sediments (Menet-Nedelec *et al.*, 2015). The main results of this study were as follows:

- among contaminants associated with the turbid plume, only trace metals could be quantified;
- desorption in the dissolved phase concerned a very low fraction of these trace metals; and
- concentrations of trace metals in both the particulate and dissolved phases were back to the pre-dredge levels one hour after the end of extraction activity. The chemical impact was temporary and did not last longer than the turbid plume.

This study concluded that there was no need for a long-term monitoring of the water quality over the period covered by the mining license.

2.7 Descriptor 11: Energy including underwater noise

> "Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment." (EC, 2010)

In recent years, awareness has increased over issues concerning underwater noise resulting from dredging and sediment extraction. Many marine organisms use sound to sense the environment around them and to find prey. Consequently, an increase in anthropogenic low-frequency noise, such as that produced by dredging (Ainslie *et al*., 2009; Dreschler *et al*., 2009; Robinson *et al*., 2011; Thomsen *et al*., 2009), has the potential to cause adverse effects. The value of 200 kHz for sonar sources is an accepted threshold above which no marine species are believed to be functionally sensitive (D 11.2).

The extent to which these effects disseminate through the foodweb to marine mammals is unknown, but some speculations can be made based on available data. For instance, large variability exists in hearing sensitivity between different fish species. However, in general, fish are sensitive to low frequencies (Popper and Fay, 2011), which puts them at risk from dredging noise. Few studies have looked at dredging noise specifically (Robinson *et al*., 2012), but avoidance of low-frequency vessel noise has been reported for some fish species (de Robertis and Handegard, 2013; and Handegard *et al*., 2003) noted vertical and horizontal avoidance by cod of a bottom-trawling vessel. Dredging noise is unlikely to result in direct mortality or permanent hearing damage for fish, but long-term exposure could theoretically affect the fitness of some individuals.

Responses to the particle motion of low-frequency sound have also been recorded in cephalopods (Mooney *et al*., 2010), which can form an important part of the diet of some marine mammals. Low-frequency noise, in the 1–10 kHz band, altered cephalopod breathing rhythms and movement.

The sound level radiated by a dredger undertaking full dredging activities is normally in line with that expected for a cargo ship travelling at moderate speed (de Robertis and Handegard, 2013; Robinson *et al*., 2011). However, extracting gravel does cause additional noise impact (Dreschler *et al*., 2009; Robinson *et al*., 2011). In the UK, underwater noise from aggregate extraction has been largely discounted as a significant impact (Thomsen *et al*., 2009). Similarly, in The Netherlands, noise levels from dredgers were not in the top seven major underwater sound sources (Ainslie *et al*., 2009). During the reclamation works for the enlargement of the harbour of Rotterdam, a monitoring programme on underwater sound measured the noise from a large range of trailer suction hopper dredgers (in power and in volume, 2000–22 000 m^3). For all frequencies, the noise of dredging and dumping was less than the noise of transit (Heinis, 2013).

Table 2.8. Contribution to MSFD D7 according to the various impacts detailed in the ICES Guidelines for the management of marine sediment extraction (ICES, 2003).

Effects of extraction	**Impact on**	**Potentially influenced MSFD descriptors**	**Level of contribution of WGEXT guidelines to MSFD descriptors**			
			Introduction	**Baseline survey**	**Impact assessment**	**Mitigation**
Seabed removal	Bathymetry/ topography	D7: Permanent alteration of hydrographical conditions does not adversely affect marine ecosystems	"Ensuring that methods of extraction minimize the adverse effects on the environment and preserve its overall quality once extraction has ceased".	Physical setting: "local hydrography including tidal and residual water movements"	Physical impact assessment: "changes to the behaviour of bedforms within the extraction and adjacent areas", "time-scale for potential physical recovery of the seabed", and "implications for local water circulation resulting from removal or creation of topographic features on the seabed"	"Modification of the depth to limit changes to hydrodynamics"
Sediment plume	Turbidity	D7: Permanent alteration of hydrographical conditions does not adversely affect marine ecosystems		Dredging activity: "number of days per year on which aggregate dredging will occur", "whether on-board screening will be carried out", and "whether aggregate dredging will be restricted to particular times of the year"	Physical impact assessment: "effects on water quality through increases in the amount of fine material in suspension"	

Table 2.9. Contribution to MSFD D11 according to the various impacts detailed in the ICES Guidelines for the management of marine sediment extraction (2003).

Effects of extraction	Impact on	Potentially influenced MSFD descriptors	Level of contribution of WGEXT guidelines to MSFD descriptors			
			Introduction	Baseline survey	Impact assessment	Mitigation
Ship activity	Underwater noise	D11: Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment	"Ensuring that methods of extraction minimize the adverse effects on the environment and preserve its overall quality once extraction has ceased"	Dredging activity: "information on noise emission"	Biological impact assessment: "noise emission"	

Dredging has the potential to impact marine mammals, but effects are species- and location-specific, and also vary with the dredging equipment type. In general, evidence suggests that if management procedures are implemented, effects are most likely to be masking (i.e. simply contributing to background noise), and causing only short-term behavioural alterations due to shifts in prey availability (Todd *et al*., 2014). The exclusion of prey from foraging areas has the potential to impact marine mammals negatively. However, the extent of this negative impact is species- and context-specific, and will depend on the significance of the feeding ground, the ability to switch prey species, and the availability of alternative foraging areas.

2.7.1 Conclusion

ICES guidelines (ICES, 2003) mention the necessity to minimize noise emissions down to levels that do not affect the marine environment (Table 2.9).

With respect to this D11, WGEXT recognizes that extraction of marine sediment does generate underwater noise. However, aggregate extraction is only contributing to the noise of shipping, and introduces no negative effects from the extraction itself. Therefore, the biological impacts of noise generated by dredging activity are more possible than probable.

3 Discussion

A method for assessing by activity categories the vulnerability of marine ecosystems to various anthropogenic threats has been proposed by Halpern *et al*. (2008). The authors showed that extraction of marine aggregates exerts a lower pressure than, in decreasing order of perturbation, invasive species, pollution, development, toxic blooms, demersal fisheries (Blyth *et al*., 2004; Lambert *et al*., 2014), and the phenomena of hypoxia.

This review of existing research contributes to a permanent WGEXT objective of identifying the most appropriate ways of ensuring that marine aggregate extraction does not affect the GES of the marine ecosystem. An accurate quantification of dredging activity is the prime objective of the annual WGEXT meetings. For pressures and effects footprints, site-specific examples are reviewed every year, along with cumulative effects assessments (ICES, 2016c, 2019a), taking into consideration that combined effects from all types of physical disturbance are spread to 86% of the European coastal area (Korpinen and Andersen, 2016; Stelzenmüller *et al*. 2018).

3.1 Prevention

Environmental impact assessments should take into account the ICES Guidelines for the management of marine sediment extraction (ICES, 2003, which provide guidance for the selection of appropriate extraction site locations, and the implementation of mitigation and monitoring programmes, by:

- encouraging an ecosystem approach to the management of extraction activities, and the identification of areas suitable for extraction;
- protecting sensitive areas, important habitats (such as marine conservation areas) and industries (including fisheries), and the interests of other legitimate uses of the sea; and
- ensuring that extraction methods minimize adverse effects and preserve the overall quality of the environment once extraction has ceased.

A revision of the guidelines is currently underway, and will address issues including changes in policy, spatial planning, legitimate use of the sea, MSFD, underwater noise, overflow, cumulative impacts, mitigation measures, and databases.

3.2 Impact

Monitoring programmes have to provide sufficient information to allow a confident assessment of GES (van Hoey *et al*., 2010). However, it is necessary to consider that the geographical scale on which the MSFD operates is much larger than that of single projects. For instance, extraction activity is often taking place in a relatively small area (from a few km^2 for the direct pressure of the draghead, to several km^2 for the footprint of overspilling) and often only for a limited amount of time. Therefore, the spatial and temporal components of the activity and related pressures and impacts are also limited (ICES, 2015), and it is difficult to conceive how a judgment can be made regarding the likely significance of a local dredging project for GES. The appropriate scale at which management measures are taken is likely to be a key issue for various descriptors, and the cost of the monitoring must consequently also be taken into account (Cooper *et al*., 2019).

3.3 Recovery

The possibility of recovery after sediment extraction should be acknowledged by incorporating it into the criteria for assessing GES (Borja *et al*., 2010). This should be aided by joint monitoring programmes (Shephard *et al*., 2015) and expert judgement in cases where there is a paucity of information or data on which to base management decisions (Elliott *et al*., 2018b).

It is important to realize that biological/ecological recovery can be reached without recovery of the physical state (ICES, 2019a). Following recovery, the benthic community may be somewhat different from the original, and more adapted to the new seabed morphological state. The functions of the ecosystem, in the sense of the MSFD, will only be altered when these changes happen over a very large area. Even in the case of permanent loss of the original morphological state of the seabed, the benthic fauna may recover, and, therefore, the structure and functions of the ecosystems can be safeguarded and the benthic ecosystems may not be adversely affected.

The time and spatial scale at which a specific activity, pressure, and impact should be assessed is an issue that needs to be investigated. Nature itself is continuously changing, and trends, whether or not human induced, are not easy to assess (ICES, 2016b).

3.4 Mitigation

To allow sustainable use of marine resources (Birchenough *et al*., 2010), there is a clear need for enforcing management measures such as:

- Seasonal closures for specific areas (i.e. during recruitment seasons). Such seasonal restrictions exist in a few countries (UK, France, and Finland) to protect the spawning periods of vulnerable fish species, such as herring during winter or sole during spring (ICES, 2017).
- Rotation of dredging intensity (creation of fallow areas) to allow recolonization and recovery of macrobenthos. In a local context, controlling the area and intensity of dredging, and allowing undisturbed deposits to act as refuges between dredged furrows, may be effective measures for enhancing the rehabilitation of the seabed. There may also be environmental benefits from rotating dredging operations across different zones, and leaving the fallow areas between the extraction areas to rehabilitate for several years before reworking. Future case studies are needed on the consequences of marine aggregate extraction on marine biota over sufficiently long time-scales, to underpin the derivation of reliable and scientifically credible models (Barry *et al*., 2010).
- Compensation (of the habitat, the resource, or the users) and mitigation (Elliott *et al*., 2016).
- Exploratory restoration techniques (ecoengineering) in areas where the seabed has been impoverished as a result of extraction activities.
- Limitation of screening to prevent damaging effects of intensive oversanding.

The implementation of the Directive (EC, 2016a) raises questions requiring increased scientific knowledge and understanding on aspects such as:

- Pressure levels, for multiple activities, that clearly equate to acceptable levels of environmental impact.
- Integrated assessment of cumulative effects.

- Specific scientific indicators and more quantitative reference conditions, particularly for benthic habitats.

3.5 Restoration and landscaping

EMU Ltd (2004) reviewed existing methods and approaches to marine habitat remediation, and their potential application to marine aggregate dredging sites following disturbance. Most restoration projects (e.g. artificial reefs or seeding of shells or gravels) incorporate the restoration of abiotic conditions, but do not include abiotic extremes and biological processes (Verdonschot *et al.*, 2012). Few studies provide evidence on how ecological knowledge might enhance restoration success (van Dalfsen and Aarninkhof, 2009; de Jong *et al.*, 2014, 2015, 2016).

The ecoengineering approach (Elliott *et al.*, 2016), developed and used in dredging and marine infrastructure projects, starts with a thorough understanding of the local environment, and aims to increase the overall value of the project both for nature and society. Traditionally, dredging operators were requested to leave a flat bed after mining. In contrast, seabed landscaping aims at creating diverse habitat conditions in sand extraction areas by leaving large-scale bed forms on the dredged seabed after completion of the work. In this way, landscaped mining areas are hypothesized to encourage recolonization, and promote higher biodiversity and productivity after completion of the dredging work (Rijks *et al.*, 2015).

To bring forward the interpretation of GES descriptors from the point of view of sediment extraction, the concept of switching to an approach based on functionality and recoverability should not be lost in future work (ICES, 2019b). Studies are still needed to increase knowledge of structure and function recovery processes through time (Cooper *et al.*, 2008; Foden *et al.*, 2009, 2010; Barrio-Froján et al., 2011; Wan Hussin et al., 2012), and to promote ecological restoration to repair damaged ecosystems (Collins and Mallinson, 2007; Cooper, 2012, 2013; de Jong *et al.*, 2014).

3.6 Gaps

This review also highlights the following knowledge gaps, related to sediment extraction, and needed in order to fulfil MSFD requirements:

D 1	Requirement of high-resolution maps of habitat types (Woods *et al.*, 2016).
D 3	Mapping of spawning areas (ICES, 2011).
D 4.2	Proportion of selected species at the top of foodwebs.
D 4.3.1	Abundance/distribution of groups with fast turnover; lack of primary production indicators.
D 6.2	Size composition of a community reflected by the proportion of small and large individuals.
D 6.2.3	Proportion of biomass or number of individuals in the macrobenthos above some specified length/size.
D 6.2.4	Parameters describing the characteristics of the size spectrum of the benthic community.
D 7	Permanent alterations of hydrographical conditions.

3.7 Limits of MSFD descriptors

The MSFD aims at GES in marine waters, following an ecosystem-based approach, and focused on 11 descriptors related to ecosystem features, human drivers, and pressures. Furthermore, 29 subordinate criteria and 56 attributes are detailed in an EU Commission Decision (EC, 2016c).

The analysis for the GES decision, and the associated operational indicators, revealed ambiguity in the use of terms, such as indicator, impact, and habitat, and considerable overlap of indicators assigned to various descriptors and criteria. Berg *et al*. (2015) suggested a rearrangement and elimination of redundant criteria and attributes, avoiding double counting in the subsequent indicator synthesis, a clear distinction between pressure and state descriptors, and the addition of criteria on ecosystem services and functioning.

In documents on D1, D3, and D4, marine sediment extraction is mostly not directly mentioned as a pressure.

The interconnection between D1 and D6 can be seen in the almost the identical wording for pelagic species in D1C5 and benthic species in D6C5.

In EC (2015a), pressures are not indicated, but it is mentioned that there are strong links with descriptors that do indicate pressures, such as D6 and D7.

Finally, in later documents, e.g. EC (2016b), the link between D1, D4, and D6 is present.

3.8 Improvements for MSFD descriptors

Frequently, D1 (biodiversity), D3 (commercial fish and fisheries products), D4 (foodwebs), and D6 (seabed integrity) are combined into one integrated descriptor: marine ecosystem (I and E and EA, 2015). However, in relation to sediment extraction, it is useful to deal with them as separate descriptors.

For D6, it is clear that marine sediment extraction can influence the integrity of the seabed. Dredging can also alter hydrographical conditions (D7), and, as a sound producing activity, it can influence D11.

3.8.1 Descriptor 1

The most important criteria for species are already formulated in the Habitats Directive, but in draft 4 of the proposal for a Commission decision on GES criteria (EC, 2016b), extra criteria under MSFD have been formulated:

D1C1 Species distribution range and, where relevant, distribution patterns, are in line with the prevailing physiography, geography, and climate.

D1C2 Population abundance of the species, in terms of numbers and/or biomass, is not adversely affected due to anthropogenic pressures, and its long-term viability is ensured.

D1C3 Population demographic and physiological characteristics of the species (e.g. body size, age-class structure, sex ratio, fecundity, survival, and mortality rates) are indicative of a natural population which is not adversely affected due to anthropogenic pressures.

D1C4 The habitat has the necessary extent and condition to support the different life stages of the species.

D1C5 The condition of the habitat type is not adversely affected, including its biotic (typical species composition and their relative abundance) and abiotic structure, and its functions. Such an approach was adopted by Cooper and Barry (2017) for sediment extraction.

3.8.2 Descriptor 4

In draft 4 of the proposal for a Commission decision on GES criteria (EC, 2016b), four criteria related to anthropogenic pressures are mentioned. They are focused on:

D4C1 Species distribution and the relative abundance and diversity of the tropic guild.

D4C2 Abundance, in terms of numbers or biomass, across trophic guilds.

D4C3 Size distribution of individuals across relevant species of the trophic guild.

D4C4 Productivity of the trophic guild.

In an ICES special request advice from the EU on revisions to MSFD manuals for D3, D4, and D6 (20/03/2015; ICES, 2015), it is noted that only a few EU countries mention pressures on foodweb components, particularly fisheries. Extraction as such is not mentioned.

Physical disturbance of the habitat and (benthic) fauna are currently the most determining factor for the status of the marine ecosystem, and, therefore, are also decisive for the functioning of foodwebs (I and E and EA, 2015).

3.8.3 Descriptor 6

At an ICES workshop held in February 2015 (ICES, 2015), aggregate extraction was highlighted as one of the activities causing physical habitat loss and damage, with potential consequences for the integrity of the seabed. To judge the pressure, considering the spatial and temporal scales of the impact is crucial. In the workshop, physical damage was mentioned as the main pressure resulting from aggregate extraction, but it was suggested that physico-chemical disturbances should be integrated (e.g. the impact on anoxic seabeds in the Baltic Sea).

The main topic of the workshop was the incorporation to D6 of the newly proposed criteria functionality and recoverability, in combination with the existing criteria of physical damage and benthic conditions. The adoption of a concept including three criteria themes (pressure, state, and impact) was proposed, linked to the existing and newly suggested criteria (Figure 3.1). From the point of view of marine sediment extraction, this is a good approach. The recovery criteria theme is important for marine sediment extraction, because, even when the benthos is completely removed, total recovery by recolonization is often possible.

The idea to incorporate recovery in the formulation of criteria has not been implemented so far. In the document on progress on article 8 of the MSFD assessment guidance (EC, 2016a), three criteria are mentioned, out of which, only the second gives room for the acknowledgement of recovery:

D6C1 Spatial extent and distribution of the physical disturbance.

D6C2 Spatial extent of the adverse effect of the physical disturbance per habitat type.

D6C3 Spatial extent and distribution of physical loss.

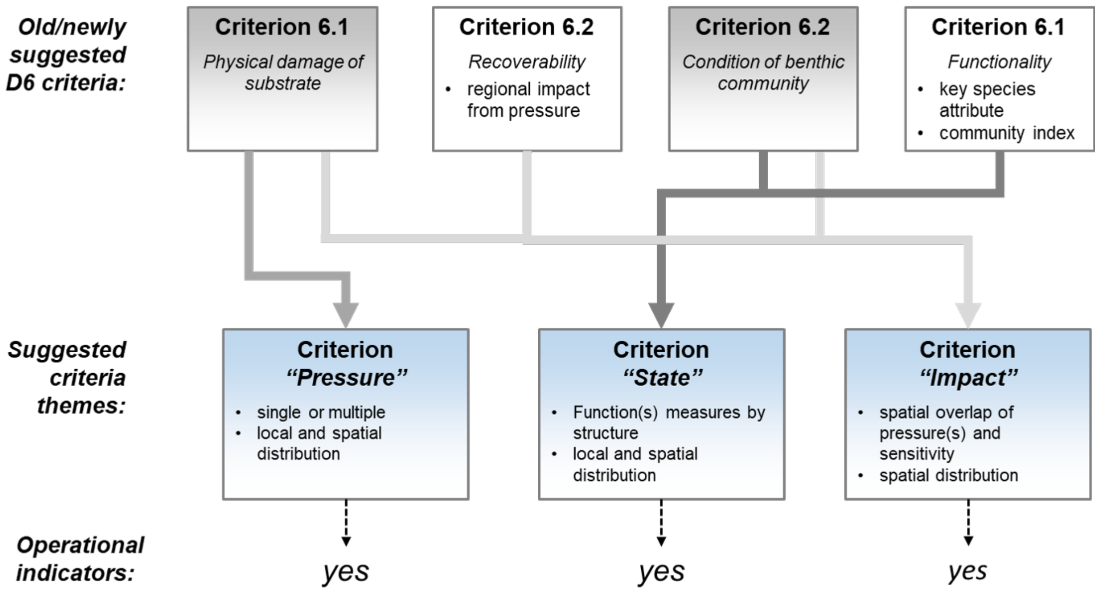

Figure 3.1. Conceptual diagram illustrating how work under both the old (2010) and the newly suggested (2014) criteria (grey and white boxes respectively) can be merged for a conceptually stronger assessment and use of existing indicators/data to measure progress towards GES for seabed integrity (ICES, 2015).

In draft 4 of the proposal for a Commission decision on GES criteria (EC, 2016c), the formulation and numbering are slightly different:

D6C1 Spatial extent and distribution of physical loss (permanent change) of the natural seabed. Marine sediment extraction is defined as a pressure on the seabed, and not as a source of seabed loss, because of the potential for recovery after cessation of the extraction (ICES, 2019b).

D6C2 Spatial extent and distribution of physical disturbance pressures affecting the seabed.

D6C3 Spatial extent of each habitat type that is adversely affected by physical disturbance through change in its structure and function (species composition and their relative abundance, size structure of species, or absence of particularly sensitive or fragile species, or species providing a key function). The areas must be expressed as a proportion (%) of the total area (D6C1, D6C2) or as proportion (%) per habitat type (D6C3).

In this proposal, physical loss is regarded as a permanent change to the seabed that has lasted, or is expected to last, for a period of two reporting cycles (12 years) or more. This seems to give room for recovery, but it should be mentioned that biological/ecological recolonization can be reached without recovery of the physical state, albeit perhaps with a different assemblage of taxa.

In the proposal for a Commission decision (EC, 2016c), two extra criteria regarding benthic habitats are mentioned that are related to both D1 and D6:

D6C4 The extent of loss of the habitat type that results from anthropogenic pressures, does not exceed a specified proportion of the natural extent of the habitat type in the assessment area.

D6C5 The condition of the habitat type is not adversely affected, including its biotic (typical species composition and their relative abundance, size structure of species, or absence of particularly sensitive or fragile species or species providing a key function) and abiotic structure, and its functions.

Although the formulation of these last two criteria, especially D6C5, sounds more like descriptors, the idea is to operationalize these criteria by setting values for the proportion (in %) of the extent of loss, and thresholds for the condition of habitats.

In the ICES special request advice from the EU on revisions to MSFD manuals for D3, D4, and D6 (20/03/2015; ICES, 2015), three actions are proposed:

- develop and test standards for human pressure on benthic habitats;
- address the role of scale and connectivity in setting boundaries for the seabed; and
- assess the recoverability of seabed integrity.

3.8.4 Descriptor 7

In draft 4 of the proposal for a Commission decision on GES criteria (EC, 2016c), the following criteria are formulated:

D7C1 Spatial extent and distribution of changes in hydrographical conditions to the seabed and water column (e.g. changes in wave action, currents, salinity, temperature, and/or oxygen concentration), associated, in particular, with physical losses or permanent changes to the seabed.

D7C2 Spatial extent of each benthic habitat type adversely affected (in terms of physical and hydrological characteristics, and associated biological communities) due to permanent alteration of hydrographical conditions.

In EC (2015b), changes to the morphology of the seabed are mentioned as a pressure. Sediment extraction will, at least temporarily, change the morphology of the seabed. However, an important point is the spatial and temporal scale of this change, and the scale of its effects. The document also mentions the ICES guidelines on marine sediment extraction (ICES, 2003), which recognized that sand and gravel extraction, if undertaken in an inappropriate way, may cause significant harm to the marine and coastal environment. However, sand extraction in The Netherlands, for the enlargement of the harbour of Rotterdam, showed that it is very possible to have a responsible deep extraction (20 m) when the right boundary conditions are applied (Rijkswaterstaat, 2014).

D7 is a pressure descriptor that focuses on permanently altered hydrographical conditions. The pressure results from changes in the morphology of the seabed/coast leading to shifts in waves, currents, or salinity, which can have a subsequent impact, such as changes in habitat conditions (e.g. from fine sediments to a hard substrate; EC, 2015c). In this sense, marine sediment extraction can be a pressure for D7, especially when it is a large-scale extraction, or an extraction in a specific vulnerable area.

A specific impact of note, related to D7C2, is the risk of oxygen depletion in the case of very deep extractions and/or extractions which are carried out in very low dynamic waters, when oxygen-rich water can no longer reach the seabed.

3.8.5 Descriptor 11

In draft 4 of the proposal for a Commission decision on GES criteria (EC, 2015d, 2016b), the following criteria are formulated:

D11C1 The spatial distribution, temporal extent (number of days and their distribution within a calendar year), and levels of anthropogenic sound sources, do not exceed values that are likely to adversely affect marine animals.

D11C2 Levels of anthropogenic continuous low-frequency sound in two 1/3-octave bands do not exceed values that are likely to adversely affect marine animals.

In 2017, the EC set up technical working groups to establish threshold values for the GES criteria of the 11 descriptors of the MSFD.

- The Topic Group (TG) Seabed is developing EU method standards for determining threshold values for adverse effects, and the maximum allowable extent per habitat of habitat loss and adverse effects. This work should be finalized by the end of 2021 (MSCG 27-2020 Minutes).
- TG Noise has focused on the development of a methodology that could be used for determining threshold values for the criteria intensity and frequency at an EU level (EC, 2015b).

There is a need to understand the possible implications of choosing different threshold values, and ICES has been asked to conduct a trade-off analysis between fisheries and the seabed (ICES, 2019b).

3.8.6 Conclusion

This review of existing research (221 references), provides information on research related to various effects of marine aggregate extraction on the marine environment and the connection with criteria for GES, which are relevant to several descriptors of the MSFD, as summarized in Table 3.1. In addition, this review highlights gaps in current knowledge necessary to fulfil MSFD requirements, and points out the role of best expert judgement in cases of paucity of information on which to base management decisions.

Finally, this review provides a tool to improve understanding on the impact of extraction activity on coastal and marine ecosystems, and to optimize the management of this activity and its sustainable development. This work is directly addressing policy and management needs, particularly in support of the MSFD (Austen *et al*., 2018; Cormier *et al*., 2019).

Table 3.1. Number of references contributing to the MSFD descriptors relevant to marine aggregate extraction.

MSFD descriptors	Number of references contributing to descriptors knowledge
D1: Biological diversity	65
D3: Fish resources	14
D4: Marine foodwebs	20
D6: Seabed integrity	120
D7: Hydrographical conditions	11
D8: Contaminants	1
D11: Underwater noise	13

4 Conclusion

This review of existing research contributes to a permanent objective of WGEXT to identify the most appropriate ways of ensuring that marine aggregate extraction does not affect the GES of the marine ecosystem.

The geographical scale on which the MSFD operates is much larger than that of single project assessments. Extractions often take place in a relatively small area, and only for a limited amount of time. Thus, the spatial and temporal components of the activity, and the related pressures and impacts, are limited.

With respect to:

D1 WGEXT recognizes that the extraction of marine aggregates can potentially be a serious threat to biodiversity when exploitation projects affect gravelly areas that are either small or underrepresented in the geographical area (loss of habitat).

D3 Recent studies suggest that fishing activity is only locally and temporarily deterred by extraction activities. However, WGEXT recognizes that the extraction of marine aggregates can potentially be a serious threat to commercially important fish species, when impacts affect sensitive and threatened species (e.g. through loss of spawning areas).

D4 The direct and indirect effects of marine aggregate extraction on foodwebs are proportional to the size of dredging areas, with limiting factors, such as the trophic adaptability of fish and bird species, and their ability to move to avoid disturbed areas, or the tolerance of marine mammals to turbidity.

D6 WGEXT recognizes that direct changes to the function and structure of ecosystems, particularly physical parameters, will occur as a result of the extraction of marine sediments. However, the group is content that, in the context of appropriate consent regimes that provide for rigorous environmental assessment and evaluation of each proposal to extract sediment, these impacts may be considered to be within environmentally acceptable limits, and, therefore, not adverse. Defining adverse as no environmental change from predredge conditions is, in the opinion of the group, inappropriate and detrimental to the continued ability of ICES Member Countries to extract marine sediments from their seabed. Therefore, WGEXT recommends that the possibility of recovery after marine sediment extraction should be acknowledged by incorporating it into the criteria of D6, and by taking it into account in GES assessments.

D7 WGEXT recognizes that the dimensions of dredged pits are so small that the deepened area have little influence on macroscale current patterns, and will only change the current pattern in the direct vicinity of the dredged area.

D11 WGEXT recognizes that the extraction of marine sediments generates underwater noise. However, aggregate extraction is only contributing to shipping noise, and introduces no negative effects from the extraction itself.

With all descriptors, it is important to recognize their interconnected nature. For instance, if D1 and D4 are okay, then others, such as D6, must similarly be okay.

The ICES Guidelines for the management of marine sediment extraction (2003) provide recommendations for the adoption of appropriate extraction site locations and the implementation of mitigation and monitoring programmes, with the aim of preventing any harmful effect on habitats of prime importance, and taking into account the ecosystem-based approach for the management of human activities.

Management measures and voluntary industry actions can reduce significant and unwanted impacts, helping to meet sustainability objectives for extraction, conservation, and environmental management. Best practices, and their likelihood of reducing impacts, depend on local, regional, and national management objectives and priorities.

The possibility of recovery after sediment extraction should be acknowledged by incorporating it into the relevant criteria, and by taking it into account in GES assessments.

These objectives point out a need for long-term datasets, to develop relevant monitoring tools and acceptable disturbance level models, in order to achieve a better management of the activities of the marine aggregate extraction industry.

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Annex 2: List of abbreviations

AIS	Automatic identification systems
AMBI	Aztiz´s marine biotic index https://ambi.azti.es/
BEQI	Benthic ecosystem quality index
BTA	Biological trait analysis
CHARM	`Eastern channel habitat atlas for marine resource management´ project
D1	Descriptor 1: biological diversity
D3	Descriptor 3: commercial fish and shellfish resources
D4	Descriptor 4: marine foodwebs
D6	Descriptor 6: seabed integrity
D7	Descriptor 7: hydrographical conditions
D8	Descriptor 8: contaminants
D11	Descriptor 11: underwater noise
DEVOTES	`Development of innovative tools for understanding marine biodiversity and assessing good environmental status´ project http://www.devotes-project.eu/
EC	European Comission
EMS	Electronic monitoring system
EU	European Union
GES	Good Environmental Status
ICES	International Council for the Exploration of the Sea
M-AMBI	Multivariate - AMBI
MarBEF	Marine Biodiversity and Ecosystem Functioning https://www.marbef.org/
MPA	Marine protected area
MSFD	EU Marine Strategy Framework Directive
SI	Sensitivity index
SPM	Suspended particulate matter
TG	Topic Group
WGEXT	ICES Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem https://www.ices.dk/community/groups/Pages/WGEXT.aspx
WGMHM	ICES Working Group for Marine Habitat Mapping https://www.ices.dk/community/groups/Pages/WGMHM.aspx

Annex 3: List of species names

Bivalves	*Ensis directus*	American razor shell
	Pecten maximus	King scallop
	Mytilus spp.	Mussels
	Modiolus spp.	Mussels
	Aequipecten opercularis	Queen scallop
	Abra alba	White furrow shell
Crustaceans	*Cancer pagurus*	Brown crabs
	Galathea spp.	Squat lobster
	Homarus gammarus	Lobster
	Pisidia spp.	Porcelain crab
Worms	*Sabellaria spinulosa*	Ross worm
	Chaetopterus spp.	
	Lanice spp.	Sand macon
Fish	*Spondyliosoma cantharus*	Black bream
	Gadus morhua	Cod
	Solea solea	Common sole
	Limanda limanda	Dab
	Squalus acanthias	Dogfish
	Triglidae	Gurnards
	Melanogrammus aeglefinus	Haddock
	Clupea harengus	Herring
	Scomber scrombrus	Mackerel
	Pleuronectes platessa	Plaice
	Mullus spp.	Red mullet
	Ammodytes marinus	Sandeels
	Raja clavata	Thornback ray
	Scophthalmus maximus	Turbot
	Merlangius merlangus	Whiting
Birds	*Uria aalge*	Common guillemot
	Melanitta nigra	Common scoter
	Morus bassanus	Northern gannet