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From monitoring to management: a standardized indicator-based assessment of long-term dredge disposal impact across three benthic habitats for macrobenthos, epibenthos and fish

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Abstract

Purpose Dredging of harbours and navigation channels and the associated disposal of dredged material in designated areas may impact the marine environment. Disposal at sea is regulated by the OSPAR convention and several EU environmental regulations. To comply with these regulations, managers need a simple presentation of potential environmental impacts, based on standardized assessments of available monitoring data. This long-term observational study (2005–2019) provides a standardized indicator-based assessment of dredge disposal effects on three ecosystem components (macrobenthos, epibenthos and demersal fish) in three benthic soft-sediment habitats in the Belgian part of the North Sea.

Methods Ecosystem impact was determined by a Benthic Ecosystem Quality Indicator (BEQI) analysis. This approach measures differences in species composition, species richness, density and biomass between an affected and a non-affected set of samples. The relationship between amount of dumping (pressure), benthic habitat type, and ecosystem impact (BEQI) was investigated using linear mixed-effects models.

Results Environmental impact differed slightly among benthic habitats, but increased with increasing pressure in all three habitats, with the strongest effects observed for the macrobenthos ecosystem component due to their restricted mobility.

Conclusions Our findings demonstrate the added value of a standardized indicator-based approach to assess dredge disposal impact across benthic habitats and ecosystem components in soft sediments. It provides managers with a standardized environmental evaluation tool and a simplified traffic light representation of potential dredge disposal impact. The results based on long-term observational monitoring data confirm the importance of contiguous periodic monitoring, with a focus on the macrobenthos community.

Keywords Dredging · Macrobenthos · Epibenthos · Demersal fish · Benthic indicator · North Sea

1 Introduction

Approximately 41% of our marine ecosystems are strongly affected by anthropogenic stressors, while the remainder experiences mild impacts, leaving no part of the oceans untouched (Halpern et al. 2008; Elliott et al. 2020). Dredging is a daily practice in ports and navigation channels to prevent siltation. The dredged material is normally disposed of in designated marine areas, and currently the only activity licensed for disposal of solid waste at sea (Bolam et al. 2021). In European waters, like the Belgian part of the North Sea (BPNS), dredge disposal is regulated through the OSPAR convention and several national and EU environmental regulations, such as the Marine Strategy Framework Directive (MSFD, 2008/56/EC). These require appropriate

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monitoring and environmental impact assessments to follow up on potential negative effects on the marine ecosystem, and to organise the license regulation that allows for dredge disposal.

Disposal of dredged sediments can affect the marine ecosystem both directly (e.g. burying or smothering of benthic organisms, increased turbidity, release of nutrients and contaminants), and indirectly (e.g. changes to sediment composition and habitat modifications) (Newell et al. 1998; Bolam and Rees 2003; Manap and Voulvoulis 2015). The ecological consequences of dredge disposal on the seafloor ecosystem are well studied in European waters and beyond through control-impact monitoring (Bolam et al. 2006; Blake et al. 2009; Vivan et al. 2009; Katsiaras et al. 2015; Donazar-Aramendia et al. 2018; Baux et al. 2020; Dauvin et al. 2022). The main effects are a decline in diversity and abundance, and a shift in community structure. However, the impact of dredge disposal was mostly inconsistent due to the high variability in the amount and frequency of the disposed material, as well as differences between the nature of the disposed material and the receiving habitat (Bolam and Rees 2003; Bolam et al. 2006, 2021). Also, the ecosystem resilience (i.e. its ability to recover after a disturbance) plays an important role. For instance, in highly dynamic systems that are exposed to currents, waves and wind, the impact of dredge disposal is expected to be minimal, as recovery may be facilitated by larval movement from surrounding environments and the dispersal of disposed material to the surrounding area (Simonini et al. 2005; Blake et al. 2009; Vivan et al. 2009; Dauvin et al. 2022). Other studies performed in highly dynamic systems, found an increase in benthic diversity within the disposal site, related to an increase in sediment complexity (De Backer et al. 2014; Donazar-Aramendia et al. 2018). In more stable systems, adverse effects were observed, including the burial of benthic organisms, changes in grain size composition and increasing contaminant concentrations (Katsiaras et al. 2015). This wide range of effects indicates the need for a case-by-case assessment of the environmental impact of dredge disposal (Simonini et al. 2005).

Most studies on dredge disposal effects focus on the macrobenthic community, because of their restricted mobility, relatively long life span, high diversity and the wide range of trait modalities. As such, macrobenthos is considered a good measure for the overall health of marine ecosystems (Rakocinski and Zapfe 2004; Dauvin et al. 2007). In contrast, only few studies have been conducted on the direct and indirect impact of dredge disposal on mobile epibenthic and demersal fish communities. Additionally, most studies are based on a rather short-term period (2–5 years) (Vivan et al. 2009; Katsiaras et al. 2015; Donazar-Aramendia et al. 2018; Dauvin et al. 2022) and on specific environments (Blake et al. 2009). The number of long-term studies on dredge disposal (> 10 contiguous years) is limited, due to lack of financial support (Sukhotin and Berger 2013) and the lack of uniform sampling designs (Wolfe et al. 1987). Nevertheless, long-term impact studies are relevant to investigate whether environmental changes persist over time, and to differentiate between changes related to human activities and pressures vs. natural variability in the ecosystem (Wolfe et al. 1987; Sukhotin and Berger 2013; Borja et al. 2016). Moreover, only few studies report on research conducted at disposal sites that are continuously used (Bolam et al. 2006, 2011; Donazar-Aramendia et al. 2018). The current study addresses the above-mentioned shortcomings, as it is based on observational data for three ecosystem components (macrobenthos, epibenthos and demersal fish) from 15-years monitoring (2005–2019) in the BPNS at five disposal sites, located in three benthic habitat types (Fig. 1). The Belgian dredge disposal sites are already in use for several decades, however, robust monitoring data are only available after the Belgian regulation for the protection of the marine environment came into force in 1999 (Maes 1999). As there are no data before the dredge disposal activity started, no Before-After assessment could be performed. Inevitably, the long-term observational monitoring data that are used in this study show some inconsistencies in sampling effort (e.g., missing years, various amount and location of samples), challenging the impact assessment for each year. However, our uniform and standardized Control-Impact approach proves that data variation and data complexity can be converted to plain interpretations for policymakers.

Generally, indicators provide easily accessible and essential information by defining critical boundaries between healthy and degraded ecosystems (Van Hoey et al. 2010). A wide variety of biotic indices exists (e.g. AMBI, B02A, BQI, BENTIX, D²SI, D²SI2), for which the applicability depends on several factors, e.g., the human activity, the degree and intention of the human pressure, the environment, data availability and choice of reference zones (Borja et al. 2000; Dauvin et al. 2007; Baux et al. 2020). Several indicators are less appropriate for physical disturbances or require specific data or standardized sampling effort. Therefore, we chose the Benthic Ecosystem Quality Index (BEQI), as this indicator is applicable for physical disturbances, able to handle variation in sampling effort over time, and invokes a statistics-based algorithm approach to assess differences between two datasets (Van Hoey et al. 2007). This method considers diversity (number of species), community structure (Bray-Curtis similarity) and species

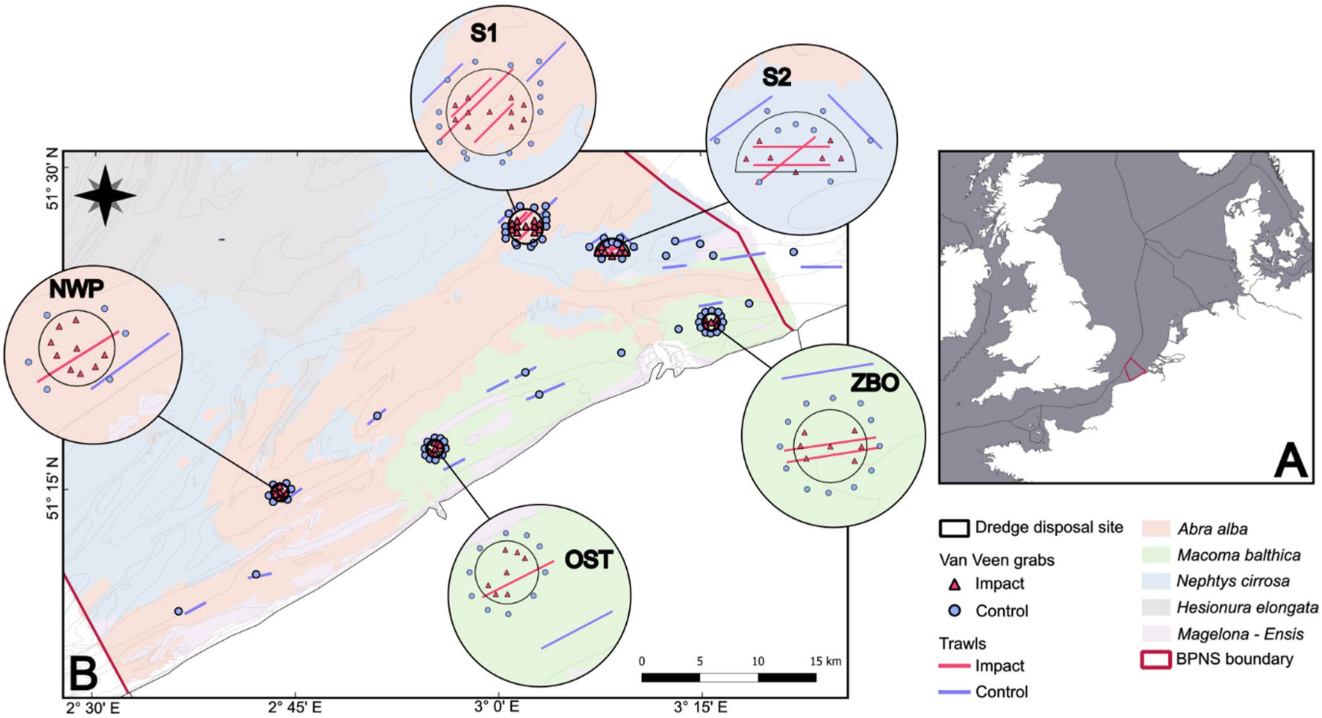

Fig. 1 Overview map of the study area, where (**A**) situates the BPNS (marked by the red border) within the North Sea region and (**B**) shows the dredge disposal sites OST, ZBO, S2, NWP and S1, and the sampling locations (impact and control) over the entire study period (2005-2019). The five disposal sites are situated within three different benthic habitats. Points represent Van Veen sampling locations (macrobenthos), while the lines show beam trawl sampling locations (epibenthos and fish)

density as important parameters, similar to other indicators. An added value of the BEQI indicator is the inclusion of biomass as a fourth parameter (Van Hoey et al. 2007). The large long-term dataset allows to investigate the correlation between the BEQI parameters and dredge disposal intensities. The fact that the disposal sites are located in different benthic habitat types gives the opportunity to investigate whether habitat type plays a role in the response of the benthic communities to dredge disposal.

Policymakers need straightforward information on the impact of dredge disposal to identify actions concerning changes in disposal practices, and simple indicators to allow for ecological assessments under the EU nature directives (e.g. MSFD). The present study covers three main research questions:

a. What is the impact of dredge disposal for three ecosystem components (macrobenthos, epibenthos and demersal fish) in the BPNS over 15 years of observational monitoring?
b. What is the correlation between dredge disposal intensities and the response of the three ecosystem components? Is there a difference across the three benthic habitat types?
c. What insights does the standardized indicator-based assessment (BEQI) provide for monitoring and management purposes?

2 Material & methods

2.1 Study area

The BPNS is a small and shallow part of the southern North Sea (3457 km², average depth 20 m) (Vandepitte et al. 2010). The seabed mainly consists of soft sediments (mud, silt, sand, coarse sand and gravel). Many human activities occur in the BPNS as represented in the Belgian Marine Spatial Plan (MSP) (Douvere et al. 2007; Verhalle and Van de Velde 2020). Recurrent dumping of dredged material is only allowed in five areas in the BPNS, located in three different benthic habitat types (Fig. 1). Benthic habitat types differ in sediment composition and are represented by typical macrobenthic species (Breine et al. 2018; Pecceu et al. 2021). Two sites are located close to the coasts of Oostende (OST) and Zeebrugge (ZBO) in the muddy *Macoma balthica* habitat. A third disposal site (S2) is located approximately 12 km offshore on the Vlakte van de Raan in the sandy sediment

Nephtys cirrosa habitat. The sites in front of Nieuwpoort (NWP) and Zeebrugge (S1) harbour, are located at 9 km and 17 km from the coastline in the fine muddy sand *Abra alba* habitat.

2.2 Data collection

2.2.1 Biological and environmental data

To determine the impact of dredge disposal on the benthic ecosystem, three ecosystem components (macrobenthos, epibenthos and demersal fish) were sampled at the five dredge disposal sites in the BPNS during the period 2005–2019. Macrobenthos was collected once a year in autumn with a Van Veen grab (0.1 m²). All macrobenthos samples were sieved fixed (before 2010) or alive (after 2010) on a 1 mm sieve. An 8%-formaldehyde-seawater solution was used for fixation and the samples were stained with eosin to ease the sorting process in the lab. In addition to the biological data, sediment samples were taken from each Van Veen sample. These were analysed afterwards via laser diffraction (Malvern Mastersizer 2000 G hydro version 5.40). Especially the median grain size (µm) and clay/silt fraction (< 63 µm) (%) are relevant environmental variables for delineating the benthic habitats (Van Hoey et al. 2004; Degraer et al. 2008). Therefore, the average median grain sizes (µm) and clay/silt fractions (< 63 µm) (%) (+ standard deviation) are summarized for the different dredge disposal sites and the control sites during the period 2006–2019 in Table 1.

Epibenthos and demersal fish were collected twice a year during spring and autumn using an 8-meter beam trawl equipped with a bolder chain and shrimp net (22 mm mesh size in the cod end). Before 2010, the fishing net was towed for 30 min at a speed of 4 knots over approximately 3500 m with the currents. From 2010 onwards the fishing distance and duration were halved to 15 min over approximately 1750 m, to ensure the impact samples were exclusively located within the dumping site (Van Hoey et al. 2012).

A Control/Impact design was implemented with several macrobenthos samples (7 to 11) and epibenthos-fish tracks (1 to 2) within the dumping area or impact site, and several samples (10 to 27) and epibenthos fish tracks (1 to 5) outside the area but within a similar habitat (control sites) (Supplementary Material Table A1). Due to several

Table 1 Average grain size fraction < 63 µm (%) (+ standard deviation SD) measured in the different dredging areas during the OSPAR survey in 2018, and the average grain size fraction < 63 µm (%) and median grain size (µm) (+ SD) measured at the different dredge disposal sites (impact/IMP) and at the control sites (REF) in the Belgian part of the North Sea during the period 2006-2019.

Dredging area	Grain size < 63 µm (%) (mean + SD)	Disposal site	Grain size < 63 µm (%) (mean + SD)		Median grain size (µm) (mean + SD)	
			IMP	REF	IMP	REF
Access channel Nieuwpoort	19.70 ± 11.10 (*N* = 2)	NWP	4.01 ± 10.13 (*N* = 45)	13.18 ± 17.83 (*N* = 120)	295.77 ± 62.48 (*N* = 45)	221.51 ± 81.56 (*N* = 120)
Harbour Nieuwpoort	58.08 ± 30.16 (*N* = 4)					
Marina Nieuwpoort	95.37 ± 5.64 (*N* = 3)					
Scheur West	41.72 ± 27.88 (*N* = 5)	S1	10.17 ± 20.17 (*N* = 130)	11.54 ± 16.63 (*N* = 271)	296.43 ± 130.57 (*N* = 130)	230.51 ± 93.58 (*N* = 271)
Scheur Oost	37.30 ± 21.60 (*N* = 4)					
Central part harbour Zeebrugge	99.13 ± 0.09 (*N* = 3)					
Access channel harbour Oostende	87.63 ± 6.46 (*N* = 6)	OST	26.08 ± 27.83 (*N* = 59)	39.83 ± 31.74 (*N* = 152)	197.39 ± 105.87 (*N* = 59)	135.36 ± 95.05 (*N* = 152)
Harbour Oostende	93.84 ± 7.61 (*N* = 14)					
Central part harbour Zeebrugge	99.13 ± 0.09 (*N* = 3)	ZBO	38.75 ± 24.90 (*N* = 84)	42.95 ± 27.16 (*N* = 244)	155.52 ± 115.76 (*N* = 84)	133.08 ± 130.62 (*N* = 244)
Harbour Zeebrugge	92.11 ± 14.64 (*N* = 20)					
Scheur West	41.72 ± 27.88 (*N* = 5)	S2	2.98 ± 9.72 (*N* = 82)	0.94 ± 4.22 (*N* = 170)	216.08 ± 37.33 (*N* = 82)	216.90 ± 41.26 (*N* = 170)
Scheur Oost	37.30 ± 21.60 (*N* = 4)					
Central part harbour Zeebrugge	99.13 ± 0.09 (*N* = 3)					

operational circumstances (research vessel availability, weather conditions, net ruptures, changed sampling and processing protocols), the number of macrobenthos samples and epibenthos and fish tracks varied throughout the years (Supplementary Material Table A1). For the epibenthos/fish assessments, the study period was subdivided in three periods of 4–5 years, according to the reporting period correlated to the disposal license periods (2005–2009; 2010–2015; 2016–2019) (Van Hoey et al. 2012; Hoey et al. 2022). The sampling protocol was consistent within each of these shorter intervals.

2.2.2 Dredge disposal data obtained from the Maritime Access Division

Dredging of harbours and navigation channels and dredge disposal activities within the BPNS are regulated by the Maritime Access Division and the Coastal Department of the Flemish Government. During the period 2008–2020, 128 to 155 million tonnes (dry matter) of dredged material has been yearly disposed by all OSPAR member states (Cronin et al. 2023). In the BPNS and the adjacent Scheldt estuary, the volume of disposed material ranged between 29 and 52 million tonnes of dry matter per year during that period (Cronin et al. 2023). Yearly dredge disposal intensities at the different disposal sites are listed in Table 3. The grain size distribution was measured at the different dredging areas along the Belgian coast during the OSPAR survey in 2018. The average mud fractions (0–63 µm) (%) (+standard deviation) obtained from this survey are summarized in Table 1.

2.3 Data analysis

2.3.1 Data processing

Two datasets were created: one for grab data (macrobenthos) and one for trawl data (epibenthos and fish). For macrobenthos, a total of 1256 Van Veen samples were analysed. The animals were identified to the lowest possible taxonomic level, counted and weighed as outlined in our Taxonomic Discrimination Protocol. Biomass was assessed by quantifying the wet weight on an analytical balance. Over the entire period, 17,454 macrobenthic taxa records were observed; 226 of these were unique. The macrobenthos sampling and analysing protocol follows the ISO 16665:2014 standard ("Water quality - Guidelines for quantitative and sample processing of marine soft-bottom macrofauna"). This methodology has been under accreditation since 24 May 2011 conform the NBN EN ISO/IEC 17025:2017 standard (cfr. BELAC nr. 315-TEST: issue date 06 March 2025, valid until 05 March 2030). Macrobenthos density and biomass were standardized to a surface unit of 1 m² and expressed as individuals/m² and wet weight (WW) g/m^2, respectively.

For epibenthos and fish, 431 trawls were processed onboard the research vessel immediately after catch. All fish and epibenthic species were identified to the lowest possible taxonomic level, counted and weighed. Wet weight was defined for epibenthos, not for fish. All mobile benthic animals living on top of the seabed were included in the epibenthos fraction. The fish fraction consists of demersal and benthopelagic fish, equally living on top of or in close association with the seabed. There is an ambiguous line and some overlap between the macrobenthos and epibenthos fraction, but we considered mobile epibenthic organisms to be more effectively sampled with a beam trawl. For all three ecosystem components some taxonomic groups were excluded from further analyses (summarized in Table A2 in Supplementary material). A total of 3728 epibenthos and 4915 fish taxa records were observed, including 116 unique taxa. Density and biomass data were standardized to a surface unit of 1000 m² and expressed as individuals/1000 m² and wet weight (WW) g/1000 m², respectively.

2.3.2 Indicator analysis

The Benthic Ecosystem Quality Index (BEQI, www.BEQI.eu) was evaluated at community level (level 3) (Van Hoey et al. 2007) to assess the ecological quality of the seafloor at five dredge disposal sites in the BPNS. For each benthic community parameter, the BEQI measures the deviation between the samples at the disposal sites and the respective control sites. Based on the four benthic community parameters: species richness, species composition (Bray-Curtis similarity), density and biomass (Van Hoey et al. 2007), BEQI is expressed as an Ecological Quality Ratio (EQR), ranging between 0 and 1, similar to other biotic indices (Borja et al. 2003). No specific indicator assessment exists for epibenthos and fish in literature, however, the BEQI was developed to compare any type of dataset with a similar sampling design. Therefore, EQR scores could be calculated for each parameter, each year or period, and for each ecosystem component.

Each BEQI assessment requires an impact and control dataset with sufficient spatial and temporal coverage to minimize natural variability. The control dataset is used to calculate the reference value for each status class boundary, based on a permutation test, in which 2000 random samples are selected with replacement from the reference dataset until the corresponding assessment sampling surface is reached. A probability distribution for each parameter is then created based on these 2000 values for a given sampling surface. The different status classes and related boundary values for each parameter from the probability distribution are listed in Table 2. To execute a valid BEQI calculation, the number of samples in the control dataset need to be higher than in the impact dataset, to allow for a proper randomisation estimate of the reference boundary values. The calculated EQR scores are then linked to the ecological status classes. More detailed information on the BEQI methodology can be found in Supplementary Material A3.

The overall EQR was calculated by averaging the individual scores from the four parameters. For macrobenthos, the EQR values could be computed per year (autumn values). For the epibenthos and fish assessments, the spring and autumn samples needed to be pooled within the three predefined 'licensing' periods (2005–2009; 2010–2015; 2016–2019), to achieve the required minimum number of samples (minimum three) to execute the BEQI analyses. Also, the confidence level for each parameter, i.e. the power to detect deviations from the reference area, was defined, with high (good/moderate) levels meaning a high reliability compared to low (low/very poor) confidence levels. The average EQR scores were considered confident when the confidence levels were high for at least three parameters.

2.3.3 Linear mixed modelling

To quantify the relationship between pressure and impact, i.e. the amount of dredge disposal and the impact on the three ecosystem components, two types of linear mixed-effects models (LMM) (Gaussian family) were developed within R version 4.5.1 (R Core Team 2025). The LMM analyses are based on the EQR scores for each parameter, i.e. species similarity, species richness, density and biomass, and on the overall EQR scores. Normality of the residuals was validated by a Shapiro-Wilk test and Q-Q plots with

Table 2 Reference boundary values related to the ecological status classes, with 0.6 predetermined as critical boundary value (bold) (Van Hoey et al. 2007)

Ecological status class	Reference boundary values for species richness & species composition	Reference boundary values for density & biomass
0.8 (high/good)	Median	25th and 75th percentile
0.6 (good/moderate)	**5th percentile**	**2.5th and 97.5th percentile**
0.4 (moderate/poor)	2/3 of good/moderate value	2/3 and 4/3 of good/moderate value
0.2 (poor/bad)	1/3 of good/moderate value	1/3 and 5/3 of good/moderate value

95% confidence intervals. The EQR scores were only linked to the amount of dredge disposal within the same year, as the disposal intensity (frequency) was consistent throughout the years for each disposal site. The models were developed for the macrobenthos, epibenthos and fish ecosystem components separately, with following equations:

Linear mixed-effects models (no interaction):

$$BEQI_{macrobenthos} \sim Pressure + (1\,|Year) + (1\,|Disosal\ site) \tag{1a}$$

$$BEQI_{epibenthos+fish} \sim Pressure + (1\,|Period) + (1\,|Disposal\ site) + (1\,|Season) \tag{1b}$$

Linear mixed-effects model (interaction):

$$BEQI_{epibenthos} \sim Pressure * habitat + (1\,|Year) + (1\,|Disposal\ site) \tag{2}$$

Within the first model type, only pressure (dredge disposal) was considered as fixed effect on the response variable (EQR scores) (Eq. 1a and 1b). A second model demonstrates whether the impact differed between the three habitat types. Here, both pressure and habitat were defined as fixed effects and an interaction between both factors was assumed (Eq. 2). This second model type could only be produced for macrobenthos, as the number of observations for epibenthos and demersal fish per habitat were too limited.

Year (or period) and disposal site are included as random effects within both model types to correct for repeated measures within year and site. Habitat type was not considered as random effect as this is strongly linked with disposal site (disposal site is nested within a habitat). In the case of epibenthos and fish, where the BEQI scores had to be calculated for periods of 4–5 years, the amount of dredge disposal was averaged for the respective periods (2005–2009; 2010–2015; 2016–2019). For these models, period instead of year was considered as random effect (Eq. 1b). Since epibenthos and fish samples were collected in spring and autumn, season was also considered as random factor (Eq. 1b). To ensure correct statistical inferences, some observations were excluded: as the current study is focused on the impact of dredge disposal from maintenance dredging (not capital dredging), we removed extremely high disposal values (> 10 million tonnes) from further analyses (Supplementary Material Table A1). Secondly, EQR scores with low confidence levels (low/very poor) were not considered within the models, while for the overall EQR scores, these were only included if a lower confidence level was observed for maximally one parameter. For both types of models, F-values and p-values were calculated using the Anova function in R (car package). P-values lower than 0.05 were considered statistically significant.

3 Results

3.1 Dumping of dredged material

The dredged material was mainly composed of clay/silt (Table 1). The five dredge disposal sites received a yearly average of 11 million tonnes dry matter during the study period 2005–2019. The largest amount of dredge disposal was registered at site S1, with an average of 5.3 million tonnes dry matter per year (Table 3a). The sites near the coast of Zeebrugge (ZBO) and at the Vlakte van de Raan (S2) received on average 3.1 million and 1.8 million tonnes dry matter per year, respectively (Table 3b, c). All three sites were exposed to continuous year-round dumping. Dredge disposal was lowest at NWP and OST, with 0.15 and 0.5 million tonnes dry matter per year, respectively (Table 3a, b).

3.2 BEQI evaluation

3.2.1 *Abra alba* habitat

At site S1, with highest dredge disposal, the EQR scores were consistently lower than 0.6 for all parameters for almost the entire study period, indicating a moderate, poor or bad ecological status (Table 3a). The detailed BEQI evaluation shows that macrobenthos density and biomass were significantly lower at the impact site (on average 1039 ind./m² and 32 g WW/m² per year) compared to the control site (on average 5429 ind./m²; 363 g WW/m² per year) (Supplementary Material Table A4b). The epibenthic and demersal fish communities also show some less pronounced deviation between control and impact at site S1 (Table 3). Mainly from 2010 onwards, a lower density of epibenthos and fish was observed in the impact area compared to the control area (on average 577 vs. 3674 ind./1000 m² for epibenthos, and 23 vs. 69 ind./1000 m² for fish) (Supplementary Material Table A5b and A6b). Overall, lower densities were recorded in spring, but the EQR patterns (deviation impact site vs. control site in relation to the amount of dredge disposal) for the epibenthic and fish community were similar in spring and autumn (Supplementary Material Table A5b and A6b).

At dredge disposal site NWP, all ecosystem components (macrobenthos, epibenthos and demersal fish) were generally unaffected by dredge disposal activities with overall good to high EQR scores (Table 3a). For macrobenthos, a moderate/poor similarity was occasionally observed for density and biomass, confirming a lower abundance at the impact site compared to the reference area (Table 2a, Supplementary Material Table A4a). For both disposal sites S1 and NWP, the confidence levels of the EQR scores were generally high, with a few exceptions for density and biomass (Table 3a).

3.2.2 *Macoma balthica* habitat

Within the *Macoma balthica* habitat, the overall EQR scores for macrobenthos were generally good to high (>0.6 or 0.8), indicating a good status and a high similarity between the impact and reference samples (Table 3b). Still, the macrobenthic community at disposal site ZBO seemed to be more impacted, with lower EQR scores in comparison to disposal site OST (Table 3b). The higher EQR scores (>0.8) at OST suggest a rather minimal impact. For both ZBO and OST, the confidence levels of the EQR scores for density and biomass were low (Table 3b) but remained mostly high for at least three parameters (Table 3b), especially from 2010 onwards.

For epibenthos, a generally good to high similarity was observed between the impact and reference data of both OST and ZBO in spring as well as autumn. Some differences in density and biomass were observed throughout the years and periods, but the confidence levels for density and biomass were generally low (Table 3b). Good to high EQR scores show that the fish community also shows similar behaviour in the impact and control sites for OST and ZBO in both spring and autumn during the entire study period, although some lower or higher densities were observed at the impact site, resulting in lower EQR scores (Table 2b; Supplementary Material Table A6c and A6d). The overall confidence level was high.

3.2.3 *Nephtys cirrosa* habitat

At disposal site S2, the overall EQR scores for macrobenthos were mainly higher than 0.6 or 0.8, suggesting a good or high similarity between the impact and control samples (Table 3c). Some differences were observed for density and biomass, resulting in slightly lower overall EQR scores. The confidence level for the overall EQR scores was generally high, although the density and biomass scores were less confident, especially during the period 2006–2012.

The epibenthic and fish community was in general very similar between impact and control for both seasons at S2 (good or high EQR scores, see Table 3c). For fish, there was a dissimilarity between the impact and control areas during the period 2016–2019, especially in autumn (Table 3c). This was related to higher fish densities in the impact area (174 individuals/1000 m² in S2 compared to 41 individuals/1000 m² in the control area) (Supplementary material Table A6e). The confidence level of the EQR scores was generally high, with a few exceptions in the case of density and biomass, especially for epibenthos during spring.

Table 3 EQR scores for each parameter and overall EQR scores for the three ecosystem components and for the five disposal sites within the BPNS during the period 2005-2019. a*)* shows the *Abra alba* habitat (NWP and S1), b) *Macoma balthica* habitat (OST and ZBO) and c) *Nephtys cirrosa* habitat (S2). Values above 0.6 represent a good (green; 0.6 – 0.8) or high (blue; 0.8 – 1) similarity with the reference area; scores below 0.6 imply a moderate (yellow; 0.4 – 0.6), poor (orange; 0.2 – 0.4) or bad (red; 0 – 0.2) similarity. Values without colour have a low confidence level. Empty cells mean no data. Within NWP, the exact amount of dumping was not available for 2005

a)

				Macrobenthos							Epibenthos					Demersal and benthopelagic fish			
Disposal site	Year	Amount of dredge disposal (tonnes dry matter/ year)	Season	Species similarity	Species richness	Density	Biomass	Average	Period	Season	Species similarity	Species richness	Density	Biomass	Average	Species similarity	Species richness	Density	Average
	2005			No data						Spring	0.75	0.90	0.69	0.52	0.71	0.77	0.70	0.78	0.75
	2006	178,269		0.79	0.83	0.96	0.89	0.87											
	2007	118,100	Autumn	0.77	0.80	0.45	0.63	0.66	2005-2009										
	2008	103,541		0.75	0.77	0.48	0.48	0.62		Autumn	0.76	0.85	0.75	0.73	0.77	0.82	0.87	0.99	0.89
	2009	156,456		0.78	0.78	0.77	0.80	0.78											
	2010	179,186		0.82	0.83	0.82	0.99	0.87											
	2011	64,234		0.81	0.73	0.78	0.80	0.78		Spring	0.73	1.00	0.79	0.80	0.83	0.80	0.84	0.85	0.83
NWP	2012	175,121	Autumn	Not sampled					2010-2015										
	2013	211,722																	
	2014	121,361		0.78	0.70	0.74	0.78	0.75		Autumn	0.78	1.00	0.69	0.74	0.80	0.78	0.95	0.78	0.84
	2015	162,128		No campaign															
	2016	177,248		Not sampled						Spring	0.83	1.00	0.80	0.86	0.87	0.87	0.89	0.33	0.69
	2017	111,235	Autumn	0.80	0.81	0.83	0.92	0.84	2016-2019										
	2018	214,675		Not sampled						Autumn	0.82	0.70	0.67	0.94	0.78	0.83	0.80	0.42	0.68
	2019	230,638																	
	2005	3,017,123		No data						Spring	0.84	0.73	0.83	0.88	0.82	0.89	0.77	0.96	0.87
	2006	11,722,690		0.36	0.41	0.08	0.18	0.26											
	2007	5,592,676	Autumn	0.45	0.45	0.12	0.05	0.27	2005-2009										
	2008	4,589,589		0.33	0.34	0.05	0.01	0.18		Autumn	0.85	0.83	0.74	0.71	0.78	0.85	0.83	0.92	0.87
	2009	6,144,522		0.40	0.54	0.15	0.07	0.29											
	2010	3,642,577		0.46	0.43	0.05	0.04	0.24											
	2011	5,290,142		0.55	0.47	0.22	0.60	0.46		Spring	0.59	0.53	0.18	0.23	0.38	0.75	0.93	0.64	0.77
S1	2012	4,320,751	Autumn	0.52	0.42	0.51	0.13	0.39	2010-2015										
	2013	5,988,596		0.59	0.50	0.37	0.17	0.41											
	2014	3,806,194		0.52	0.48	0.35	0.41	0.44		Autumn	0.59	0.70	0.25	0.27	0.45	0.65	1.00	0.28	0.64
	2015	5,538,995		No campaign															
	2016	5,658,408		0.47	0.47	0.33	0.10	0.34		Spring	0.48	0.50	0.09	0.06	0.28	0.60	1.00	0.35	0.65
	2017	5,690,034	Autumn	0.46	0.48	0.72	0.11	0.44	2016-2019										
	2018	4,192,492		0.42	0.39	0.31	0.08	0.30		Autumn	0.70	0.80	0.31	0.31	0.53	0.71	0.70	0.34	0.58
	2019	4,890,011		0.48	0.44	0.44	0.11	0.37											

b)

				Macrobenthos							Epibenthos					Demersal and benthopelagic fish			
Disposal site	Year	Amount of dredge disposal (tonnes dry matter/ year)	Season	Species similarity	Species richness	Density	Biomass	Average	Period	Season	Species similarity	Species richness	Density	Biomass	Average	Species similarity	Species richness	Density	Average
	2005	599,905		No data						Spring	0.79	0.90	0.64	0.67	0.75	0.85	0.89	0.68	0.80
	2006	819,665		0.79	1.00	0.98	0.77	0.88											
	2007	460,167	Autumn	0.82	1.00	0.78	0.97	0.89	2005-2009										
	2008	864,863		0.85	0.90	0.70	0.39	0.71		Autumn	0.86	0.83	0.71	0.66	0.76	0.86	0.84	0.97	0.89
	2009	241,544		0.87	0.76	0.77	0.81	0.80											
	2010	304,235		0.86	0.87	1.00	0.95	0.92											
	2011	562,690		0.91	0.85	0.79	0.96	0.88		Spring	0.76	0.70	0.10	0.05	0.40	0.81	0.97	0.27	0.68
OST	2012	359,997	Autumn	Not sampled					2010-2015										
	2013	654,488		0.81	0.95	0.89	0.94	0.90											
	2014	407,767		Not sampled						Autumn	0.80	0.96	0.81	0.81	0.84	0.78	1.00	0.72	0.83
	2015	504,944		No campaign															
	2016	1,196,719		0.81	0.89	0.89	0.67	0.82		Spring	0.82	0.87	0.38	0.61	0.67	0.87	0.80	0.68	0.78
	2017	284,015	Autumn	0.86	0.80	0.85	0.66	0.79	2016-2019										
	2018	599,360		0.85	0.81	0.68	0.63	0.74		Autumn	0.80	1.00	0.48	0.65	0.73	0.88	0.83	0.76	0.82
	2019	416,630		Not sampled															
	2005	2,973,545		No data						Spring	0.89	0.88	0.78	0.87	0.85	0.79	0.82	0.43	0.68
	2006	2,796,772		0.53	0.73	0.19	0.71	0.54											
	2007	2,219,780	Autumn	0.67	0.83	0.22	0.52	0.56	2005-2009										
	2008	4,667,225		0.71	0.70	0.57	0.63	0.65		Autumn	0.77	0.67	0.99	1.00	0.86	0.74	0.60	0.52	0.62
	2009	3,776,038		0.34	0.67	0.31	0.16	0.37											
	2010	3,342,526		0.81	0.80	0.89	0.72	0.81											
	2011	2,062,762		0.84	0.86	0.88	0.82	0.85		Spring	0.78	1.00	0.81	0.82	0.85	0.78	0.75	0.85	0.79
ZBO	2012	2,843,505	Autumn	0.81	0.70	0.68	0.99	0.79	2010-2015										
	2013	3,021,397		0.71	0.46	0.83	0.73	0.68											
	2014	4,226,341		0.64	0.39	0.73	0.94	0.67		Autumn	0.76	0.60	0.65	0.69	0.68	0.78	0.72	0.70	0.73
	2015	3,945,216		No campaign															
	2016	3,185,295		0.78	0.67	0.87	1.00	0.83		Spring	0.64	0.87	0.10	0.13	0.43	0.73	0.65	0.20	0.53
	2017	2,832,670	Autumn	0.68	0.40	0.74	0.73	0.64	2016-2019										
	2018	2,759,644		0.66	0.52	0.68	0.72	0.64		Autumn	0.74	0.73	0.74	0.98	0.80	0.79	1.00	0.54	0.78
	2019	2,164,986		0.77	0.64	0.61	0.64	0.66											

Table 3 (continued)

c)

Disposal site	Year	Amount of dredge disposal (tonnes dry matter/ year)	Season	Macrobenthos Species similarity	Macrobenthos Species richness	Macrobenthos Density	Macrobenthos Biomass	Macrobenthos Average
S2	2005	1,234,640	Autumn	No data				
	2006	596,317		**0.65**	**0.83**	**0.51**	0.69	**0.67**
	2007	127,704		**0.68**	**0.80**	**0.51**	0.77	**0.69**
	2008	80,014		**0.75**	**0.96**	**0.91**	0.98	**0.90**
	2009	1,591,871		**0.70**	**1.00**	**0.52**	0.69	**0.73**
	2010	2,598,212	Autumn	**0.87**	**0.88**	**0.40**	**0.30**	**0.61**
	2011	2,946,850		**0.87**	**0.89**	**0.17**	0.17	**0.53**
	2012	2,650,587		**0.83**	**0.82**	0.80	0.94	0.85
	2013	1,969,370		**0.84**	**0.80**	**0.89**	**0.77**	**0.82**
	2014	2,267,889		**0.87**	**0.90**	**0.83**	**0.94**	**0.88**
	2015	2,913,203		No campaign				
	2016	2,764,075	Autumn	**0.83**	**0.92**	**0.83**	**0.45**	**0.76**
	2017	1,983,285		**0.85**	**0.81**	**0.89**	**0.83**	**0.84**
	2018	1,686,373		No data				
	2019	1,924,513		**0.84**	**0.73**	0.95	0.88	0.85

Period	Season	Epibenthos Species similarity	Epibenthos Species richness	Epibenthos Density	Epibenthos Biomass	Epibenthos Average	Demersal and benthopelagic fish Species similarity	Demersal and benthopelagic fish Species richness	Demersal and benthopelagic fish Density	Demersal and benthopelagic fish Average
2005-2009	Spring	**0.85**	**0.85**	0.76	0.79	0.81	**0.85**	**0.84**	**0.95**	**0.88**
	Autumn	**0.81**	**0.80**	**0.80**	0.94	**0.84**	**0.85**	**0.80**	**1.00**	**0.88**
2010-2015	Spring	**0.79**	**0.82**	**0.62**	0.61	**0.71**	**0.87**	**0.72**	**0.85**	**0.81**
	Autumn	**0.78**	**0.80**	**0.70**	**0.82**	**0.78**	**0.86**	**0.83**	**0.92**	**0.87**
2016-2019	Spring	**0.81**	**1.00**	0.99	0.99	0.95	**0.78**	**0.70**	**0.51**	**0.66**
	Autumn	**0.81**	**0.80**	**0.79**	**0.72**	**0.78**	**0.77**	**0.72**	**0.09**	**0.53**

3.3 Linear mixed modelling

3.3.1 Macrobenthic community

When considering only pressure as fixed effect (model type 1), a significant decreasing relationship was observed between the amount of dredge disposal (pressure) and the overall EQR scores and the EQR score for density for macrobenthos during the period 2006–2019 ($p<0.05$) (Table 4). For the other parameters a non-significant decreasing trend was observed ($p>0.05$) (Table 4).

Model type 2, in which both pressure and habitat were included, demonstrated that the overall impact-pressure correlation did not differ significantly between the different habitat types ($p=0.5620$) (Table 4). Also, the separate EQR scores were not significantly different between each habitat at a particular pressure ($p=0.6929$) (Table 4). Only for species similarity, a significant interaction between the amount of dredge disposal and habitat was observed, implying that the impact on species composition was significantly different between the three habitats ($p=0.0252$) (Table 4). Within the *Nephtys cirrosa* habitat, the scores remained rather high along the pressure gradient, while for the *Abra alba* and *Macoma balthica* habitat, the EQR scores decreased slightly with increasing pressure (Fig. 2a; Supplementary Material Fig. A7a).

3.3.1.1 3.3.2 Epibenthic and fish community The LMMs for epibenthos and demersal fish, with only pressure included as fixed effect (Eq. 2) generally did not show a significant relationship between pressure and the EQR scores

Table 4 P-values from the linear mixed model analysis for each parameter within the macrobenthic, epibenthic and fish community, with only pressure included as fixed effect (model 1) and with both pressure and habitat (and interaction) as fixed effects (model 2). For epibenthos and fish, the p-values from model 2 are not shown. No biomass data are available for fish. Significant values (<0.05) are shown in **bold**

		Macrobenthos		Epibenthos	Demersal and benthopelagic fish
Parameter	Fixed effect	Model 1 No interaction	Model 2 Interaction	Model 1 No interaction	Model 1 No interaction
Average	Pressure	**0.0147***	0.9864	0.4201	0.1408
	Habitat		0.6929		
	Interaction		0.5620		
Species similarity	Pressure	0.9676	0.7953	0.0782”	0.0574”
	Habitat		0.6013		
	Interaction		**0.0252***		
Species richness	Pressure	0.4030	0.6377	**0.0369***	0.5457
	Habitat		0.5976		
	Interaction		0.4430		
Density	Pressure	**0.0178***	0.8592	0.4497	0.2074
	Habitat		0.8572		
	Interaction		0.5997		
Biomass	Pressure	0.0568”	0.5323	0.6507	
	Habitat		0.6457		
	Interaction		0.3610		

Significance level: ¨ 0.05–0.1, *<0.05

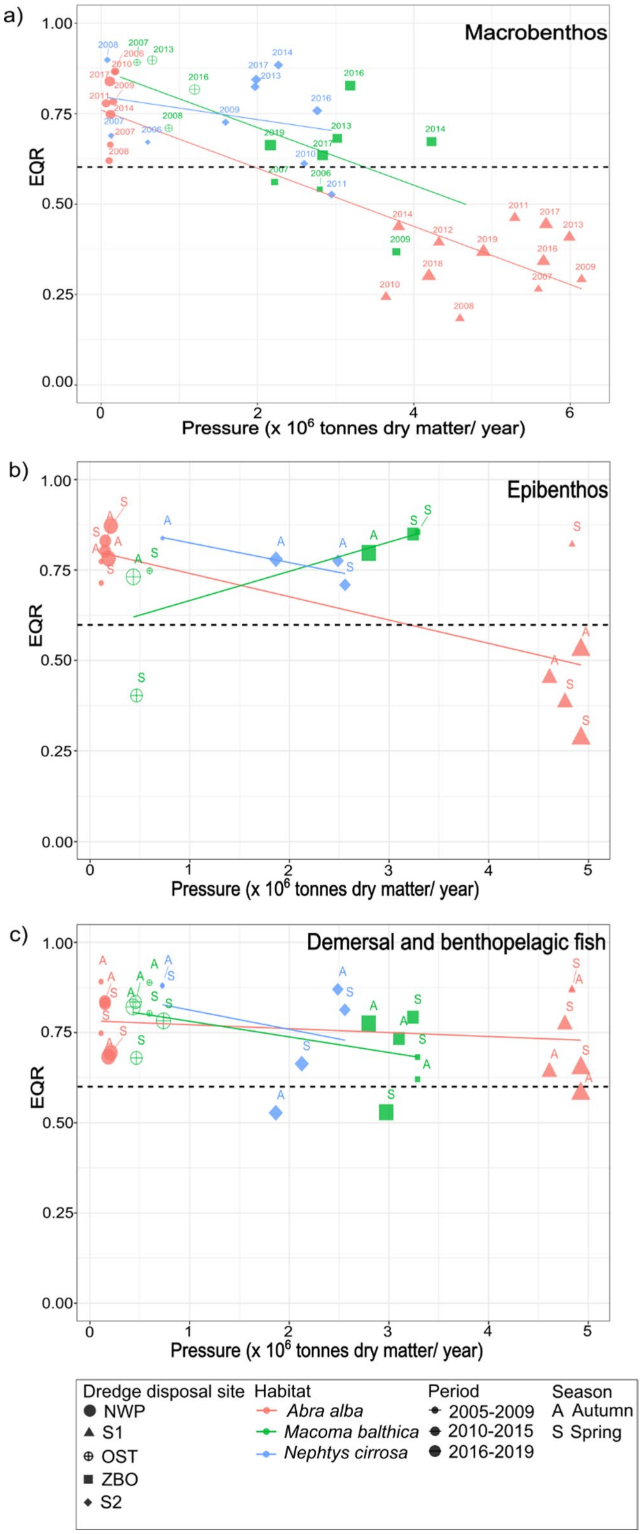

Fig. 2 Relationship between the average EQR scores and pressure (dredge disposal) for macrobenthos (**a**), epibenthos (**b**) and demersal and benthopelagic fish (**c**) within the three benthic habitat types, based on a linear model of the EQR scores in function of the pressure (including the interaction between pressure and habitat). Pressure is expressed as 10^6tonnes dry matter/ year. The dashed line represents the threshold value 0.6

($p > 0.05$; Table 3). In the case of epibenthos, only a pressure effect on species richness was observed as reflected in the lower EQR scores at high dredge disposal (Table 4; Supplementary Material Fig. A7b). Within the *Macoma balthica* habitat, the EQR scores for epibenthos seemed to increase with increasing pressure (Fig. 2b).

4 Discussion

This is the first contiguous long-term observational study (15 years) that investigated the ecological status of multiple areas in the BPNS and multiple ecosystem components exposed to ongoing dredge disposal activities by means of a standardized common indicator approach (BEQI). The observations for the epibenthic and fish communities are unique, as literature has mainly focused on macrofauna (Bolam et al. 2006; Whomersley et al. 2008; Dauvin et al. 2022). Moreover, the ecological effects on the benthic ecosystem could be linked to the disposal activity itself (amount of dredge disposal), allowing for the identification of the benthic parameters affected most by dredge disposal, as up to now only few studies have quantitatively investigated this aspect.

4.1 Macrobenthic, epibenthic and demersal fish community

The benthic indicator (BEQI) analysis demonstrated different responses of the benthic community at the different disposal areas within the BPNS, mainly related to the amount of dredge disposal (disposal intensity). For instance within the disposal site S1, characterized by the highest dredge disposal in all years, the benthic community is highly degraded in the disposal site compared to the control area over the entire study period. In accordance with previous research (Bolam et al. 2006; Katsiaras et al. 2015), a reduced diversity and density of macrobenthos was observed at S1. The higher similarity between the impact and the reference area for the other disposal sites can be related to a lower amount of dredge disposal (at NWP and OST), the highly dynamic conditions in the area and the sediment characteristics of both the habitat type and the disposed material (at ZBO and S2). Sediment properties within the disposal sites OST and ZBO (clay/silt fractions of >25%) are comparable with those within the dredging areas (Table 1).

Organisms living within the *Macoma balthica* habitat are already adapted to muddy sediments (Breine et al. 2018) and therefore less affected by the disposal of muddy sediments. This observation is comparable to other studies, suggesting a minimal impact in this type of sediments (Flemer et al.

1997; Bolam and Rees 2003; Simonini et al. 2005; Blake et al. 2009; Vivan et al. 2009; Dauvin et al. 2022). A previous study used another indicator assessment on the same dataset, namely the General Purpose Biotic Index method (GPBI), applicable for dredge disposal activities related to species loss, and came up with a similar conclusion (Labrune et al. 2021). Both disposal sites located in *Macoma balthica* habitat have a good ecological status (EQR > 0.6). Still, the intensity and frequency of dredge disposal seems to affect this habitat type as reflected in the lower EQR values at the ZBO site compared to the OST site. Previous studies demonstrated that a dumped layer of up to 30 cm still allows vertical migration of macrobenthic species (Maurer et al. 1981a, b, 1982; Miller et al. 2002). It is clear that dredge disposal in the BPNS does not create permanently thick layers, due to the shallowness of all disposal sites and the hydrodynamic conditions within the BPNS. As such the potential negative effects of burial related to dredge disposal are reduced and a quick recolonization from the surrounding areas is possible. This explains why macrobenthos at most disposal sites is minimally impacted or at least still present despite the high amounts of dredge disposal (e.g. at S1).

Within the epibenthic and fish community, the impact of dredge disposal was generally less pronounced compared to macrobenthos, most probably because these animals are more mobile and thus can move in and out of the disposal site more easily. The few effects recorded are mainly indirect, related to less favourable conditions within the disposal site, such as changed sediment composition, habitat modifications and/or reduced macrobenthic prey availability. Other studies also suggested that mobile epifaunal species are influenced rather indirectly by extraction activities as a response to their food sources (Smith et al. 2006). Significantly lower macrobenthic and epibenthic densities were observed within the disposal site S1, in comparison to the reference area (Supplementary material Table A4b and A5b), which may indicate a reduced food source availability. Some differences in epibenthos and fish densities can be explained by natural variability and an occasional rise or drop in abundance of certain dominating species, e.g. brittle stars (epibenthos) or gobies (fish). Such short-term variability effects may be better explained when assessed over a really long-term evaluation period.

In summary, our study confirms previous research, i.e. that the dredge disposal activity itself, the amount and frequency of dredge disposal and the nature of the receiving environment are important factors to determine disposal impacts (Bolam et al. 2006), but adds that this is not only the case for macrobenthos, but also for epibenthos and demersal fish. The lesser effect on the more mobile epibenthic and demersal fish species is probably related to a reduced occurrence of macrobenthic prey in the highly impacted disposal sites (e.g. S1). More detailed yearly trends have been published in a recent licensing report (Van Hoey et al. 2022), which are in line with the present BEQI observations.

4.2 Pressure and habitat effects

A significant pressure effect was confirmed for the macrobenthos community, with a general increase in impact with increasing pressure, at least when regarding all observations independent of habitat type (model type 1). Other studies also found a pressure effect on macrobenthos, with possible differences between habitats (Bolam et al. 2006; Dauvin et al. 2022). In the current study, the overall interaction between pressure and habitat was not significant (model type 2), although this does not imply that the impact of dredge disposal was equal in the three benthic habitat types. The EQR scores indeed decreased with increasing pressure within the *Abra alba* habitat, an effect that was less explicit in the *Macoma balthica* habitat where the overall impact remained rather low. In the *Nephtys cirrosa* habitat (disposal site S2), a contradictory pressure-response pattern was observed for macrobenthos (Supplementary Material Fig. A7a). The assessment gives high EQR scores for species similarity, indicating no significant differences in species composition between impact and control. However, it cannot be ruled out that the reference locations close to the impact site influenced the outcomes, as indicated by separate analyses of impact compared to nearby and far control sites (not shown here) (Van Hoey et al. 2022). Contrastingly, high EQR scores for species richness, approaching 1 in some cases, imply an increased diversity (Van Hoey et al. 2007), which suggests a possible enrichment within the originally sandy site S2 through the attraction of macrobenthic species associated with muddy sediments. An increased heterogeneity in sediment composition indeed might change the species composition as has been shown by other studies (De Backer et al. 2014; Donazar-Aramendia et al. 2018).

The non-significant but slightly different responses across benthic habitats most probably reduced the overall pressure effect. A likely explanation for the lack of statistical evidence is the limited number of observations in certain pressure classes. Within the *Abra alba* habitat, no observations were available for the intermediate pressure (1–3 million tonnes dry matter), while for the *Macoma balthica* and *Nephtys cirrosa* habitats data were lacking at higher pressures (> 4 million tonnes dry matter). The amount of dredge disposal at each site is related to the dredging needs close by, making it difficult to address this issue in an observational study. Nevertheless, the trend in the EQR scores was better visible when habitat type was considered. Of course, other factors may have caused the differences observed in this study which is solely based on observational data (Vivan et al. 2009; Bolam et al. 2021).

Due to the smaller number of samples and the grouping into 'licensing' periods of 4–5 years, we had to adapt the method for the epibenthic and fish communities. As such, the statistical power was only high enough to proof that the overall impact on the epibenthic and fish community did not really change with increasing pressure. Nevertheless, some benthic habitat differences in relation to dredge disposal pressure could be noted for epibenthos as well. For example, the EQR scores remained above 0.6 even when the pressure increased, but the number of species seemed to be slightly lower within the disposal site itself compared to the reference area (Supplementary Material Fig. A7b). Additionally, Fig. 2b and Supplementary Material Fig. A7b indicated a difference in response of several epibenthic community parameters within the *Macoma balthica* habitat compared to the other habitats. Although this result can only be seen as indicative because too few confident observations could be included (Fig. 2b; Supplementary Material Fig. A7b), such observations do show the advantage of contiguous long-term monitoring.

4.3 Insights for future monitoring and management

A standardized indicator-based assessment provides relevant information for policymakers and has already been used in numerous studies to evaluate human-induced changes in ecosystems, although a universal indicator applicable for multiple types of pressure does not exist (Dauvin et al. 2007). A benefit of indicators is the definition of critical boundaries of change or the ecological state of an ecosystem, comparable with a traffic light. In the case of BEQI, scores below 0.6 (red, orange, yellow) indicate a degraded ecosystem, while higher values (green, blue) imply a healthier system. The disposal site S1 was clearly adversely affected throughout the whole study period. Based on the simple traffic light system, a reconsideration of this dredge disposal location and the disposal volumes seems to be relevant. Partly based on our results and advice in our last licensing report (Van Hoey et al. 2022), the disposal volumes at S1 have been lowered (information provided by the Maritime Access Division). This measure may allow for some recovery of the benthic communities within the important *Abra alba* benthic habitat. These BEQI assessments for dredge disposal also feed directly into the MSFD D6 'seafloor integrity' assessment, where the level of habitat disturbance (status) and the extent (size of dumping areas adversely affected) is determined. The present results that are used in the MSFD assessment reveal that the spatial extent of the adverse impact of dredge disposal is very small (0.015 km² of the infralittoral sand and 0.003 km² of the *Abra alba* habitat). Based on our results, it is recommended that dredge disposal mainly occurs within areas with similar sediment properties as the dredged material.

The BEQI method allows for a standardized comparison between two types of datasets and provides a confidence level, indicating whether the data are appropriate to detect changes between impact and control zones. This confidence is influenced by the variability of the parameters (e.g. biomass is more susceptible to natural variability). To improve the confidence levels, the monitoring of the disposal sites in the BPNS has been subjected to changes over the course of the study period. There were changes in the amount of samples for certain sites and periods to increase the analytical power. Also, before 2010, the impact sampling tracks for epibenthos and fish fell partly outside the dumping site (meaning partly non-impacted areas) due to a sampling artifact (Van Hoey et al. 2012). From 2010 onwards, the sampling strategy was therefore optimized and the track length was halved to fit completely within the impact site. The fact that lower EQR scores were observed for the epibenthos and fish communities from 2010 onwards (at least for site S1) (Table 3) implies a possible masking of the impact in the first period (2005–2009) due to the inclusion of less disturbed areas as impacted area prior to 2010. Such shortcomings and solutions provide insights in the lessons learnt on how to optimize contiguous monitoring in long-term studies.

Also, the number of control samples differed and increased throughout the years, as the BEQI approach requires a bigger reference dataset compared to the assessment dataset in order to execute the randomization calculations in a confident way (Supplementary Material Table A1). The BEQI approach can handle this variation in sampling design better than other indicators (Van Hoey et al. 2007). Consequently, BEQI assessments cannot be performed at sample level, only on a set of (minimally 3) samples per timeframe (Van Hoey et al. 2007). That way, the epibenthic and fish data needed to be clustered in periods to fulfil the minimum number of samples rule, as such limiting the variation and number of observations.

An important issue for all biotic indices is the selection of suitable reference areas, as inappropriate control zones can lead to wrong interpretations (Halpern et al. 2008; Borja et al. 2012; Elliott et al. 2020). Nowadays, defining a proper reference area is rather challenging due to the scarcity of undisturbed or minimally impacted regions that share similar properties with an impacted area (Halpern et al. 2008; Borja et al. 2012; Elliott et al. 2020). In the present study the control areas were selected by considering the benthic habitat aspect and by including both nearby and further away locations from the impact site. Our results showed that the choice of reference areas might need some reconsideration for the future licensing periods, as for example for some (nearby) control areas in the *Nephtys cirrosa* habitat, wherefore our analysis indicates that they have been influenced by dredge disposal over time. In the *Macoma* *balthica* habitat, some control locations are not appropriate anymore, which probably resulted in the lower confidence levels related to the higher natural variability in that habitat (Breine et al. 2018).

The current study mainly aimed to define dredge disposal impacts on different ecosystem components, and the relationship with dredge disposal intensities in multiple benthic habitat types in function of future monitoring and management. Due to the lack of baseline information on the impact sites in the BPNS (dredge disposal already ongoing prior to the first monitoring data gathered), it was not possible to investigate cumulative dredge disposal effects over time. Still, future research may focus on other aspects based on the current long-term dataset, such as temporal trend analyses, sensitivity of species, or the systematic comparison of different benthic indicators.

5 Conclusion

Within the BPNS, the impact of dredge disposal generally increases with increasing pressure. Since the impact on macrobenthos was most pronounced, future studies should keep focussing on this ecosystem component. Nevertheless, additional monitoring of other ecosystem components allows for detecting cascading effects across the ecosystem. As expected, the macrobenthic community is clearly affected by dredge disposal. The observed impact on epibenthos and fish, although less pronounced, seems to be indirectly related to changes in sediment (habitat) composition and changes in the macrobenthic community as important prey for both epibenthos and demersal fish species.

The impact of dredge disposal varies slightly among the three benthic habitat types (*Abra alba*, *Macoma balthica*, *Nephtys cirrosa*). Although the trend is not statistically significant based on the current long-term dataset, this does not imply an equal impact across all benthic habitats. Despite the extensive and long-term dataset, the number of observations per year was still limited, a major drawback of an observational study, in which it is not possible to define the number of samples per pressure class beforehand. As such, it was not possible to draw conclusions about the impact in areas with intermediate pressure in the *Abra alba* habitat or in areas with high pressure for the *Macoma balthica* and *Nephtys cirrosa* habitat. Still, this study proofs that the BEQI method provides a standardized technique to determine changes in areas that have been subjected to dredge disposal in comparison to control locations. The resulting traffic light EQR scores provide relevant and easy to grab information for policymakers, creating the possibility to identify measures and actions that are required to reach a good ecological status in the frame of the MSFD directive.

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Declarations

Declaration of generative AI and AI-assisted technologies During the preparation of this work the authors used generative AI-tools in order to improve language and readability of the manuscript. After using this tool/service, the authors reviewed and edited the content as needed and take full responsibility for the content of the publication.

Competing Interest The authors certify that they have no affiliations with or involvement in any organization or entity with any financial interest or non-financial interest in the subject matter or materials discussed in this manuscript.

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