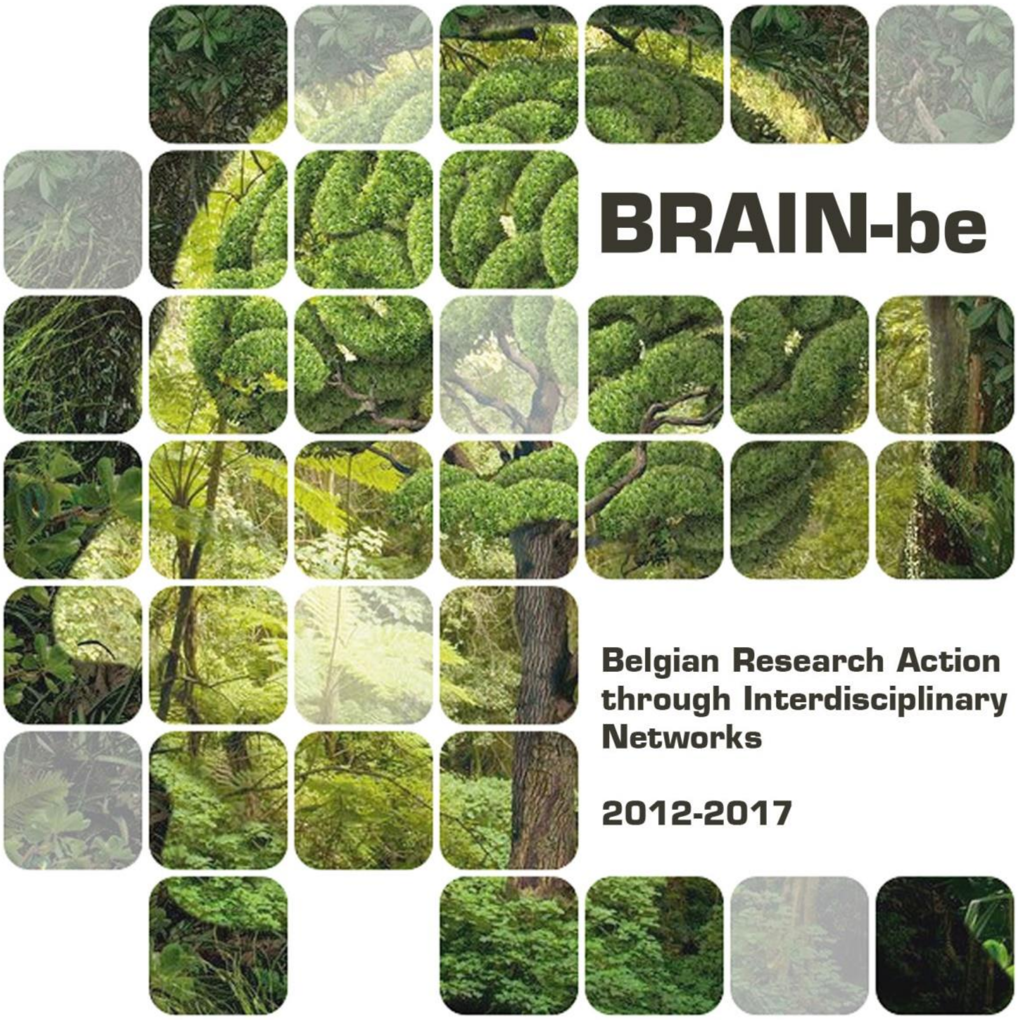

FaCE-IT

Functional biodiversity in a Changing sedimentary Environment: Implications for biogeochemistry and food webs in a managerial setting

Jan Vanaverbeke (RBINS) – Ulrike Braeckman (Ghent University) – Naomi Breine (ILVO) - Arthur Capet (University of Liège) - Emil De Borger (Royal NIOZ) – Steven Degraer (RBINS) – Felien Festjens (ILVO) - Marilaure Grégoire (University of Liège) – Evgeny Ivanov (University of Liège) – Ninon Mavraki (RBINS) – Tom Moens (Ghent University) – Karline Soetaert (Royal NIOZ) – Elise Toussaint (RBINS) – Gert Van Hoey (ILVO)

NETWORK PROJECT

FaCE-IT

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Contract – BR/154/A1/FaCE-It

FINAL REPORT

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ABSTRACT

As the introduction of offshore wind farms in the marine environment is increasing, an understanding of the effects of multiple wind farms on larger geographical areas is needed to guide the management of the marine space. In FaCE-It, we used a combination of detailed biogeochemical experiments, method development and measurements with oceanographic and diagenetic modelling. Our results show the strong local effects of offshore wind farms on the food web and highlighted the importance of the scour protection layer for increasing food availability and food diversity. FaCE-It highlighted the functional role of marine organisms, both for the benthic diagenetic processes as for filtering organic material from the water column. As such, the development of indices reflecting the functional effects of the presence of faunal groups needs increased attention. This change in functioning extends towards the larger geographical scale, beyond offshore windfarm locations, and across borders. At those larger scales, organic matter is redistributed and is retained within offshore wind farms, while the flux of organic matter to the seafloor is decreased outside the windfarms. This redistribution of organic matter triggers biogeochemical shifts, and results in an increased organic carbon content in the sediments as a consequence of the increasing importance of anoxic mineralisation pathways.

Keywords: offshore wind farms, North Sea, food web, spatial upscaling

1. INTRODUCTION

Marine coastal sediments are important habitats. Although they cover a relatively small surface worldwide, they are functionally very important. They underlie very productive waters, and hence are a sink for a large quantity of organic matter produced in the water column, and therefor areas with high mineralisation rates. At the same time, coastal sediments are subjected to a multitude of human pressures because of a wide variety of human activities. In European waters, impacts on the seafloor sediments resulting from traditional activities (fishing, aggregate extraction…) are now combined with pressures resulting from the ever-increasing installation and exploitation of offshore wind farms (OWFs). Given the proliferation of these OWFs, we need urgent knowledge on the effects of multiple OWFs on the marine ecosystem, and an increased understanding of the mechanisms. FaCE-It approached these changes as 'fining and hardening' of marine sediments, with fining of sediments being caused by additional deposition of organic matter to the environment, as observed in OWFs (Coates et al. 2014) but also considered an effect of multiple human activities (Debacker et al. 2014). Hardening encompasses the introduction of artificial hard substrates (turbines and associated scour protection layers (SPL) and the removal of the fine fraction in sand extraction sites. These physical changes in the natural environment will affect certain ecosystem functions and have consequences for the communities inhabiting the environment. Given the widely accepted biodiversity-ecosystem functioning link, an additional effect on ecosystem functioning is to be expected both on the local and the wider geographical scale, finally affecting the provisioning of ecosystem services to society. Science-based management of marine space therefore urgently needs tools to support decision making processes, based on a mechanistic understanding of how abiotic (e.g. sediment structure, hydrodynamics) and biotic (e.g. faunal activities) factors interact in driving ecosystem functions. FaCE-It aimed to contribute to this field of research, through a combination of detailed field measurements, lab experiments, food-web and ecological modelling, and the development of an integrated model allowing to assess the effects of human activities at a policy relevant geographical scale.

2. STATE OF THE ART AND OBJECTIVES

It is now clear that society's concern about climate change is leading to an increasing installation of renewable energy devices. Because of the competition for space on land, and the need for large windy areas, wind turbines are increasingly built in the marine environment, where they appear clustered as offshore wind farms. Europe is currently the stronghold of this technique, however there is growing interest in the US and China (https://www.4coffshore.com/offshorewind/). The installation of offshore wind farms is subject to monitoring the consequences of the introduction of these structures in the marine realm. Generally, the introduction of these devices leads to a fast colonisation of the structures by large densities of fouling fauna, which attract higher trophic levels (mobile crustaceans, fish and sea mammals) and results in changes in the sedimentary environment (summarised in Degraer et al. 2020). However, this knowledge is often gained through short-term monitoring programmes, executed at the local scale (within an OWF, very often at the level of a limited number of turbines). These monitoring programmes do not provide insight in the mechanism causing the changes, nor the consequences of these changes for the ecosystem and society. In addition, upscaling the monitoring results gathered at the local scale towards a larger scale, where the presence of multiple wind farms interacts with a plethora of other human activities, is not possible.

The FaCE-It objectives are therefore to understand the effect of the introduction of offshore wind farms in the marine environment on benthic ecosystem functioning (food web and organic matter cycling) from the local scale to the larger scale relevant for managers and policy makers. More specifically, FaCE-It aimed at providing **a benthic biogeochemical model** at the scale of the Southern Bight of the North Sea and increasing understanding of the **food-web structure** associated with offshore wind farms to produce a food-web model. This knowledge underpins the **refinement of indicators** on Sea Floor Integrity and food webs in the implementation of the Marine Strategy Framework Directive.

The FaCE-It strategy to reach these goals can be summarised in three important steps. A first step comprised a **detailed investigation of local effects** of the introduction of offshore wind farms on the functioning of the ecosystem. We mainly focused on collecting all components of the food web associated with the structures, comprising of the fauna colonising various parts of the turbine, the natural benthic environment, and the water column. After stable isotope analysis of the collected organisms, we analysed the data to (1) understand the structure of the food-web associated with a turbine, (2) explain the success of dominant colonising organisms and (3) contribute to answering the question whether fish attraction to turbines also leads to increased production. We further performed detailed biogeochemical measurements around a turbine, and developed necessary techniques to allow a sound interpretation of the data. A first **spatial upscaling** consisted of performing additional experiments to quantify the contribution of the colonising fauna to the possible depletion of the phytoplankton pool in the water column. In addition, we performed detailed biogeochemical measurements along a gradient from fine to coarse sediments, including locations affected by human disturbances to provide the environmental context for the measurements within the wind farm. All these models were **integrated in the final FaCE-It model** which integrated oceanographic modelling with newly collected data, which allowed to perform scenario runs after consultation with the FaCE-It stakeholder community. A final step was then the **refining and testing indicators for ecosystem**

functioning. Both the integrated model and refined indicators can serve policy directly to predict (by means of the model) and evaluate (with help of the indicators) the effects of human activities in the marine environment at relevant geographical scales.

3. METHODOLOGY

Activities within the FaCE-It project where arranged along three spatial scales. The most detailed scale included sampling at close distance to an offshore wind turbine to sample the components of the food web associated with the introduced hard substrates and was complemented with experimental food-web related research. A second spatial scale included sampling along a gradient in sediment properties, ranging from muddy, cohesive sediments to coarse permeable sediments in the Belgian part of the North Sea (BPNS. The largest geographical scale included multiple wind farms across jurisdictional borders, and was addressed through a modelling approach. To valorise our results, the FaCE-It models were applied to scenario's specifically requested by the members of the stakeholder committee of the project. In addition, FaCE-IT reviewed a suite of existing indicators for their capacity to reflect functional changes in the benthic environment as a result of human activities.

3.1 Detailed sampling of food-web components

To investigate the detailed food-web component of offshore wind turbines, sampling was conducted at the gravity-based wind turbine D6 (coordinates: 51°33.04′N – 0.2°55.42′E), located in the C-Power offshore wind farm on the Thornton Bank (Fig. 1) in August 2016 and 2017. This was the first offshore wind farm that was introduced in the BPNS, where the first six gravity-based foundations were built in spring 2008 (Degraer et al. 2010). The water depth ranges between 18 and 24 m and the total surface area of the wind farm is 19.84 km^2. The gravity-based foundations in C-Power have a diameter of 6 m at the sea surface, which increases to 14 m at the seabed (Reubens et al. 2011). Their base is surrounded by a scour protection layer (SPL), which consists of a filter and an armour layer of rock materials (Peire et al. 2009). The diameters of the armour and the filter layers are 58 and 62.6 m, respectively, while the total surface area provided by one gravity-based foundation and its SPL reaches the 2043 m^2 (Peire et al. 2009). The wind turbines are built on a naturally soft-sediment environment, with a median grain size of 312 – 427 μm (Coates et al. 2014).

The location and the wind turbine were selected based on the data availability on the assemblages of colonising fauna, fish species, the SPL and the surrounding soft sediments (Reubens et al. 2011, Coates et al. 2014, De Mesel et al. 2015). Sampling was conducted only in one season due to the logistic effort associated with sampling within the offshore wind farm in highly dynamic conditions. Moreover, similar colonisation patterns have been identified throughout the year on all gravity-based foundations in the area (De Mesel et al. 2013), and hence, we were confident that our samples represented the typical colonising assemblages occurring on such turbine foundations.

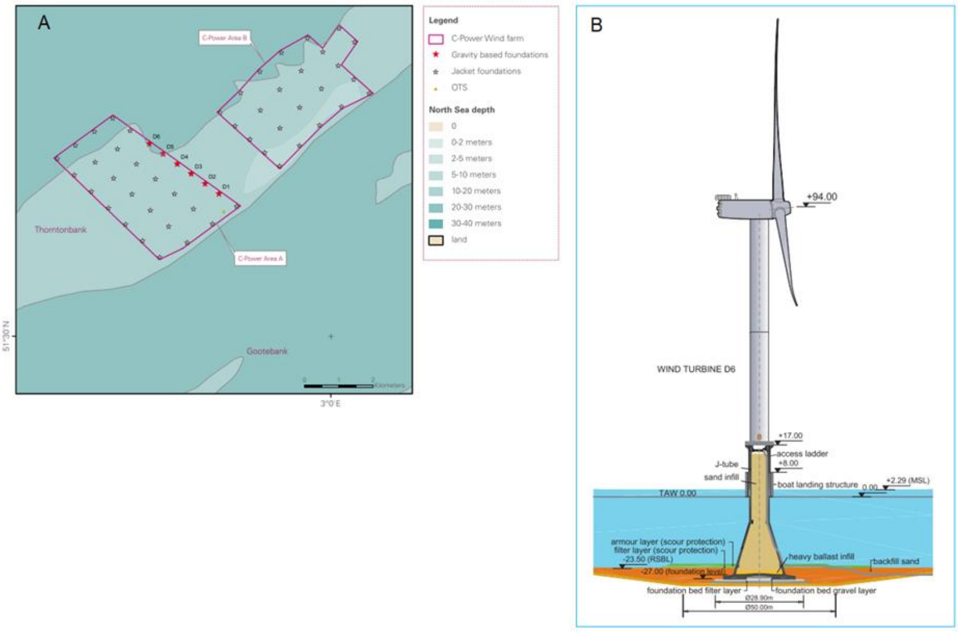


Figure 1 A: Map of C-Power wind farm indicating the position of the six gravity-based foundations (D1-D6 – source: Coates et al. 2013). B: Schematic representation of the gravity-based foundation D6 (source: Peire et al. 2009).

Organisms living along the entire depth gradient of D6, on its SPL and inhabiting the surrounding sediments, as well as benthopelagic, benthic and pelagic fish species near the turbine were collected. Specifically, six sampling zones (Fig. 2) based on macrofaunal zonation patterns (Lindeboom et al. 2011, Krone et al. 2013, De Mesel et al. 2015) and on structural/habitat differences (Baeye and Fettweis 2015) were selected for the investigation of the detailed food-web components: (a) the intertidal, (b) the *Mytilus edulis* zone (from now on mentioned as *Mytilus* zone which is found at 5 m depth), (c) the *Jassa herdmani* zone (from now on called *Jassa* zone and occurring at ~8 m depth), (d) the *Metridium senile* zone (from now on mentioned as *Metridium* zone and occurring at 15-25 m depth), (e) the scour protection layer (SPL), and (f) the surrounding soft sediment (soft substrate).

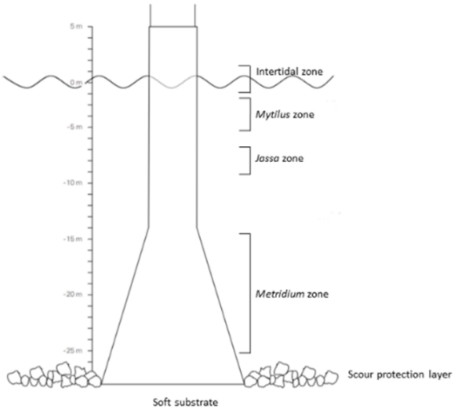


Figure 2 The six sampling zones along the depth gradient of the gravity-based foundation D6 in the Belgian part of the North Sea (modified from De Mesel et al. 2013).

The zonation pattern as defined by the macrofauna is not clearly determined, i.e. individuals of *M. senile* can be found in shallower zones and individuals of *J. herdmani* may occur at different zones. However, the abundance of these three species is higher in their zones of dominance compared to other zones.

Scientific divers collected colonising assemblages from the *Mytilus*, the *Jassa* and the *Metridium* zones by scraping the turbine (25 × 25 cm frame), from the SPL by collecting small rocks and from the soft substrate using an airlift suction device (surface sampled: 25 × 25 cm frame, sediment depth: 5 cm). Assemblages from the intertidal were scraped off by scientists aboard an inflatable boat at low tide. Benthopelagic and benthic fish species were collected with line-fishing, while spearfishing was conducted to catch benthic fish species at and in the vicinity of the SPL. Large crustaceans, i.e., lobsters and crabs, were hand-picked by scientific divers along the entire depth gradient. Finally, food sources, i.e. zooplankton, particulate organic matter, sediment organic matter and macroalgae, were collected as well.

All samples were processed for stable isotope analysis (SIA), a method to measure naturally occurring isotopic elements, such as carbon and nitrogen, in body tissue and identify trophic linkages between food sources and consumers (DeNiro and Epstein 1981). This method allowed us to identify the most important (in terms of food-web components) zones along the depth gradient of offshore wind turbines. Apart from muscle tissue for SIA, fish stomachs were also collected, and stomach contents were identified for the most abundant fish species occurring in close proximity to the gravity-based foundation, i.e. the benthic *Myoxocephalus scorpioides* (sculpin), the benthopelagic *Gadus morhua* (cod – only juvenile individuals collected) and *Trisopterus luscus* (pouting), and the pelagic *Trachurus trachurus* (Atlantic horse mackerel) and *Scomber scombrus* (mackerel).

3.2 Experimental food-web research

In a next step, we aimed at investigating the influence of colonising fauna on the local primary producers, and estimate the reduction of the primary producer standing stock due to the grazing activities of colonising organisms. For this purpose, a tripod with attached PVC panels (15 × 15 cm, Fig. 3) was placed within the C-Power wind farm (coordinates: 51°54.08′N - 2°91.68′E).

Figure 3 Tripod with PVC colonisation panels before deployment

The panels were roughened on one side to facilitate colonisation (Beermann and Franke 2012) and the entire structure remained in the water for one year before the collection of the PVC panels by scientific divers.

Upon recovery, the panels were fully colonised (Fig. 4) and were immediately placed in buckets with filtered seawater until they were transferred to the laboratory, where mesocosm pulse-chase experiments took place. Such labelling experiments are used to quantify the rates and pathways of short-term organic matter consumption within macrofaunal communities (Middelburg et al. 2000, Witte et al. 2003). Thus, an isotopic tracer in the form of food is added within the experimental environment (mesocosm), the organisms in the mesocosm feed on it and at the end of the experimental period, the tracer is chased back with the use of SIA.

Figure 4 Fully colonised PVC panel after one year deployment

In the laboratory, five fully colonised PVC panels were incubated in five separate experimental tanks that contained filtered seawater and an air supply. An experimental tank consisted of two chambers (upper and lower) connected with two water pipes that enabled water circulation (Fig. 5).

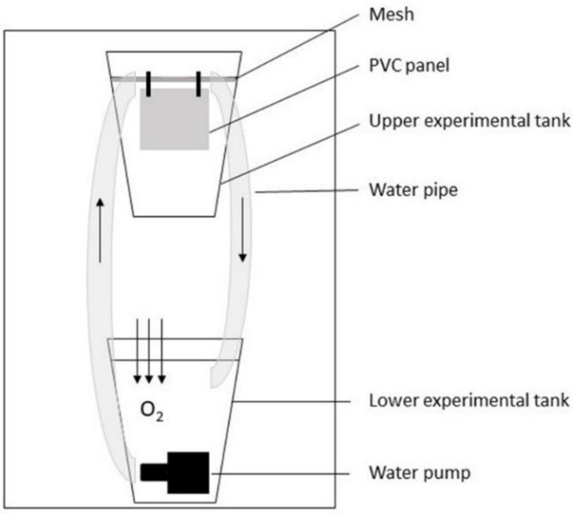


Figure 5 Schematic representation of the experimental set-up of the pulse-chase experiment (Mavraki et al. 2020)

Since most of the colonising fauna occurring on offshore wind turbines are suspension feeders (i.e. fauna that uses the water circulation in order to feed (Gili and Coma 1998), properly circulating the water within the experimental tanks was highly important. The upper chamber contained one hanging colonised PVC panel and a mesh that prevented the mobile and hemi-sessile organisms to flow to the lower chamber. The lower chamber contained an air pump providing oxygen to the system and a water pump circulating the water between the two tanks.

After 24h of acclimation, food with tracer was added homogenously to the upper chamber of experimental tanks. We provided food *ad libitum* (i.e. in excess) to make sure that the organisms had constant access to food during the three days of incubation. At the end of the incubation period, the PVC panels of all the experimental tanks were collected and scraped completely. At least five individuals per species per PVC panel were isolated and processed for SIA. The remaining organisms were kept for species identification, abundance and biomass measurements. The results of this experiment were upscaled to the total number of offshore wind turbines installed in the BPNS in the beginning of 2020 to estimate the effect of colonising fauna on the local primary producer standing stock. Calculations were performed by combining these results and previous studies (Table I). Finally, we compared the amount of carbon that is assimilated in the tissues of colonising fauna to that of a natural soft sediment macrofauna that occurs in the same surface area as the footprint of a monopile, a jacket and a gravity-based foundation.

Table I: Data used for the upscaling calculations for the two different colonising species, the turbine foundations in the BPNS, the annual primary production in the area and the carbon assimilation of macrofaunal organisms typically occurring on soft sediments in the area.

DATA					**References**
	Jassa herdmani	***Mytilus edulis* (<1cm)**	***Mytilus edulis* (1-3cm)**	***Mytilus edulis* (>3cm)**	*J. herdmani* and *M. edulis* (1-3cm): this study; *M. edulis* weight (>3cm): Bouma and Lengkeek, 2012; *M. edulis* weight (<1cm): Mallet *et al.*, 1987
Mean individual biomass (µg C)	285	51	238	394	

Mean individual biomass-specific C assimilation (µg C/µg C ind^{-1})	0.30	1.11	1.11	1.11	this study
Total surface area (m^2) on MONOPILES per species	384	192	192	192	Rumes *et al.*, 2013
Total surface area (m^2) on JACKETS per species	887	444	444	444	Rumes *et al.*, 2013
Total surface area (m^2) on GRAVITY-BASED per species	173	123	123	123	Ivanov *et al.*, pers. comm.
Total density (individuals m^{-2}) per MONOPILE	24339	251	1368	224	Kerckhof, pers. comm.
Total density (individuals m^{-2}) per JACKET	68848	22208	7800	0	Kerckhof, pers. comm.
Total density (individuals m^{-2}) per GRAVITY-BASED	47765	3268	5196	304	Mavraki *et al.*, unpublished data
Number of MONOPILES in BPNS	264				
Number of JACKETS in BPNS	48				
Number of GRAVITY-BASED foundations in BPNS	6				
Footprint per MONOPILE (m^2)	573				Rumes *et al.*, 2013
Footprint per JACKET (m^2)	10.5				Rumes *et al.*, 2013
Footprint per GRAVITY-BASED (m^2)	2227				Rumes *et al.*, 2013
Respiration fraction	0.28	0.36	0.36	0.36	*Mytilus edulis*: Hawkins and Bayne, 1985 *Jassa herdmani*: Yamada and Ikeda, 2006
Moulting fraction	0.04				
Defaecation fraction		0.48	0.48	0.48	
Assimilation fraction	0.32	0.16	0.16	0.16	
Surface area OWFs in BPNS (km^2)	238				
Total production in BPNS (g C m^{-2} y^{-1})	213				Lancelot *et al.*, 2005
Carbon assimilation (mg C m^{-2} d^{-1}) by macrofauna in sediment	Min: 3.3	Max: 42			Bühring *et al.*, 2006b

3.3 Sampling along a sediment gradient on the BPNS

We sampled along a gradient of biotic (the species community) and abiotic (sediment type and anthropogenic disturbance such as aggregate extraction and OWF construction) variables (Fig.6). These ranged from coarse grained, sandy sediments (stations 330, D6, BBEG), to fine sandy sediments (sts. 780, BRN11), to very fine-grained silty sediments (sts. 120, 130). This range of sediment types firstly represents a range of habitats, leading to differing species communities in the sampling sites, and secondly, coincides with almost the full possible gradient of permeability of marine sediments, i.e., how easily a fluid (seawater) can move through the sediment.

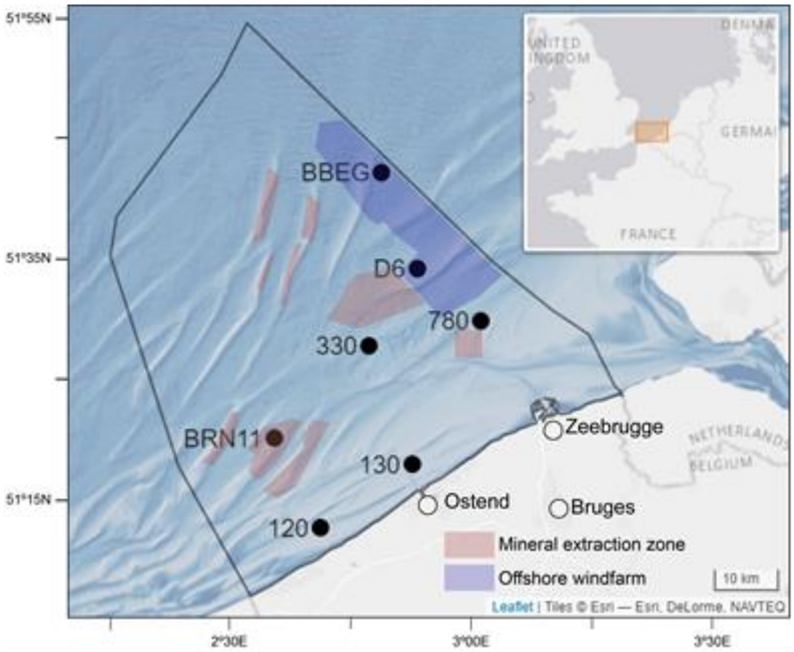


Figure 6. Locations of sampling sites in the Belgian Part of the North Sea, visited in the years 2016 and 2017.

Most sampling sites were visited both in 2016 and 2017 with the RV Simon Stevin (https://www.vliz.be/nl/rv-simon-stevin). Station BBEG was only visited in 2016, as in 2017 monopile foundations were being installed in the area during the sampling period. So, in 2017 additional samples were taken 200 m north and south of turbine D6 (C-Power), resulting in two additional sampling stations (sts. D6S, D6N).

On each sampling site multiple samples of the seafloor were taken to study different areas of interest. (1) Boxcore samples were used to study sediment biogeochemistry in detail (Fig. 7 A). From boxcore samples, a set of subsamples was taken with different purposes: incubations were performed with subsamples in laboratory conditions to measure nutrient exchange rates, sediment community oxygen consumption (SCOC), and bioirrigation rates; porewater was extracted from a set of subcores to construct porewater nutrient profiles; further subsamples were taken to analyse the physical and organic properties of the sediment.

Figure 7 Devices used to sample sediments during FaCE-It fieldwork. A. Boxcorer, B. Van Veen grab, C. Sediment Profile Imager (SPI).

(2) Van Veen grabs were collected to identify the organisms present in the sediment (Fig. 7 B), and to calculate the bioturbation- and bioirrigation potential of the species community (resp. BP_c and IP_c) in the sediment. In addition, functional diversity (functional richness, eveness, divergence and RaOQ) indices were calculated and biological trait analysis was performed. These so-called functional indices were included to establish links between functional traits of organisms and ecosystem processes and functions of the seafloor. Finding practical applications of functional traits is a rapidly developing field in need of this type of ground-data, to feed into models of species and trait distribution (e.g. Beauchard et al., 2017; Mestdagh et al., 2018; Schenone et al., 2019).

(3) In-situ pictures of the sediment were collected using a Sediment Profile Imager (SPI) camera (Fig 7, C). Whereas the boxcorers and the Van Veen grab are established methods in the field, the development of SPI protocols is still ongoing, and was specifically included in FaCE-It as a novel attempt to link features visible on the images to functional aspects of the seafloor.

The coupling between sediment structure, sediment biogeochemistry, and benthic fauna was investigated through extensive statistical analysis of the collected data.

First, mineralization process rates were derived from incubation measurements using a mass balance model (Soetaert et al., 2001). This method has previously been used by members of the consortium (Braeckman et al., 2010, 2014), but was extended for FaCE-It with additional processes: anoxic mineralization and dissimilatory nitrate reduction to ammonium (DNRA). With this, we were able to go beyond describing patterns of nutrient exchange rates, to specific biogeochemical processes related to ecosystem functions. These process rates were subsequently used in the statistical analyses. We used both linear modelling techniques described by Zuur et al. (2007, 2009) as well as variance partitioning (Borcard et al., 2011). Linear modelling was used to find statistical relations between explanatory variables and biogeochemical processes. This approach was based on Braeckman et al. (2014), with the addition of new predictors and processes. Variance partitioning was performed to investigate whether variations in biogeochemical process rates throughout the sampling domain were due to physical sediment characteristics, or the presence of certain macrofauna. The tight coupling between benthic fauna and their habitat necessitates a technique such as variance partitioning when looking for significant statistical links between biotic and/or abiotic factors and ecosystem processes. However, FaCE-It is one of the few examples that have applied this technique in a marine context (Ysebaert and Herman, 2002; Godbold and Solan, 2009; Belley and Snelgrove, 2016; Mestdagh et al., 2020)

3.4 Other data included in the FaCE-It analyses

To evaluate changes in ecosystem functioning within a policy context, indicator approaches are required. Therefore, we investigated several indicators to evaluate functional aspects of the benthic community. To test their applicability, a wider set of benthic and Sediment Profiling Imaging (SPI) data was used in addition to the gradient data collected within FaCE-It. The additional data were collected through other projects (Figure 8), such as ILVO dredge & sand extraction monitoring program, UGent Winmon monitoring program and the fishery project “Impact assessment of pulse trawl fishery”, a co-operation between NIOZ and ILVO. For each project, an impact-control sampling strategy is followed, where the control locations (CTRL) are located in the vicinity of the impact sites (IMP) in a similar physical environment.

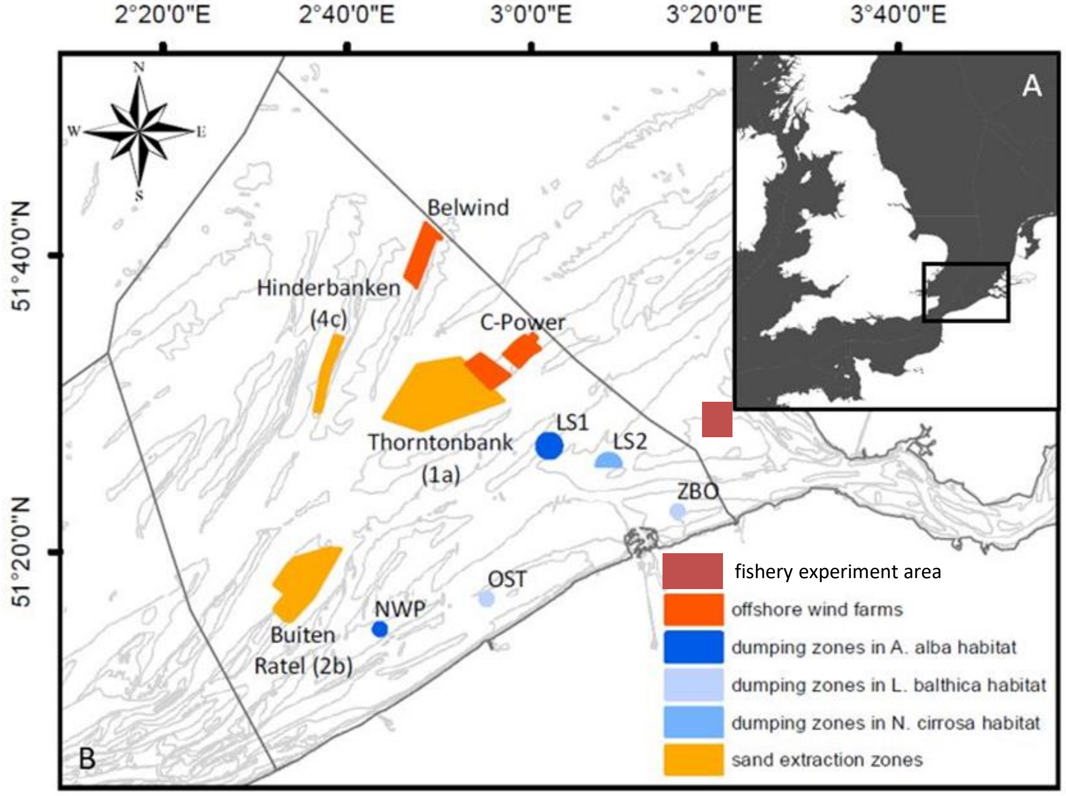


Figure 8. Overview map with A) location of the Belgian part of the North Sea (BPNS), B) location of areas with ongoing human activities from which additional data was included.

For the dredge disposal case, a long-term time benthic data series (2007-2016, 635 stations sampled in total) of macrobenthic data was available for the five disposal sites. The number of sampling stations per dumping area changed over the years. The SPI data was collected at two dredged disposal sites. The design at LS1 consists of 42 different stations along two transects and additional locations within quadrant 1 (western half of the site) and 2 (eastern half of the site) (Lauwaert et al., 2016), which were sampled in autumn 2014, summer 2015, and summer and autumn 2018. It was not possibble to sample all stations at every sampling occasion. All 16 stations were sampled in autumn 2018 for LS2.

For the sand extraction case, benthic data from 442 stations from Buitenratel (BR, zone 2b), Thorntonbank (TB, zone 1a) and Hinderbanken (HB; zone 4c)) (Liam et al., submitted) were used. The SPI data was collected in augustus 2014, with respectively 17, 3 and 13 impact stations and 6, 7 and 8 control stations in areas BR high impact, BR medium impact and HB.

Benthic data for wind energy exploitation activities was obtained from the yearly, large-scale monitoring performed by Ghent University within C-Power and Belwind OWFs for the years 2017-2018. In total, 185 stations were sampled and impact samples were taken at 37.5 m distance from the turbine center, whereas the control samples were collected at 350-500 m distance from any surrounding turbine. In 2017, SPI pictures were taken 200m North and South from the wind turbine D6 (D6N and D6S; gravity based foundation). In 2018, SPI pictures were collected in 4 directions (NW, NE, SE, SW) around the wind turbine, at 100m from D5 (gravity-based foundation). In 2019, those were located at 150, 200 and 250m from D6 and A7 (jacket foundation).

To assess the benthic effects from fisheries activity, SPI data was collected during an in-situ BACI (before-after control-impact) design experiment carried out in June 2018. We collaborated with professional beam trawl and electric pulse trawl fishermen using 4 m wide bottom trawl gears to create 3 different trawl treatments: pulse trawling with electricity turned on (Pulse On), pulse trawling with electricity turned off (Pulse off), and standard beam trawling (Tickler). The experimental design consisted of 3 plots for each of the treatments (Pulse On, Pulse Off, and Tickler), where the designated fishing vessels passed 6 times to disturb the entire plot. SPI information was then collected from 3 stations located inside each of the 9 experimental plots before and directly after fishing (T0 versus T1) and also in 3 untrawled control areas.). SPI information was collected from 3 stations located inside the experimental plots before and directly after fishing (T0 versus T1) and also in 3 untrawled control areas.

3.5 Developing a novel method to measure bioirrigation

We developed a novel method to accurately measure bioirrigation rates in sediment cores. Bioirrigation is usually measured by adding some type of tracer substance to the water column of a core containing sediment with organisms. The decrease of the concentration of the tracer is then equated to a bioirrigation rate. Issues with traditional methods are that they can depend strongly on the duration and the setup of a measurement experiment. Therefore, it is impossible to compare results of different studies, which in addition often produce results in units that cannot be used in diagenetic models. We performed experiments using uranine (Na-Fluorescein, $C_{20}H_{10}NaO_5^-$) as a tracer substance, and a continuously measuring fluorometer to track the concentration of uranine in the water (Fig 9).

Figure 9 Cyclops c6 spectrofluorometer used in the novel method to measure bioirrigation rates, and the fluorescent tracer uranine

These measurements were then used to parametrize a mechanistic model of bioirrigation. This combination allowed us to estimate the bioirrigation rate, and the depth over which water is exchanged in the sediment. Initial work with uranine was done by Na et al. (2008) and Meysman et al. (2006), who used uranine as a tracer for bioirrigation. We made this technique applicable to fresh samples (e.g. cores collected in the field and transported to the laboratory), and found a way to effectively deal with issues such as adsorption of uranine on sediment particles. We used this technique to study bioirrigation in field samples, collected at six sites in and around the Oosterschelde estuary, to track changes in bioirrigation behaviour throughout the year. For this, sediment cores were collected monthly and transported to the laboratory, where the tracer was added during an incubation and its exchange with sediment porewater was monitored using a fluorometer (Turner Designs Cyclops 6). After collecting the tracer data, sediment cores were sieved over a 0.5 mm sieve to collect and identify macrofauna present in the sediment.

3.6 Food web sampling

At station 330 and 780 (Fig. 6), and within the C-Power OWF, additional samples were collected to sample components of the entire food web. Van Veen grabs were used to collect macrofaunal organisms; a hyperbenthic sledge was deployed to collect hyperbenthos and beam trawl and otter trawl sampling collected demersal and epibenthic organism. Organisms were processed for stable isotope analysis according to the same protocol as followed for the detailed food web study in the vicinity of the D6 turbine (section 3.1).

3.7 Spatial upscaling

Model description

To parameterize the local impact of fouling fauna and upscale it over the Southern Bight of the North Sea (SBNS), a numerical modelling tool was chosen. The hydrodynamic model ROMS (Regional Ocean Modeling System; Shchepetkin and McWilliams, 2005) was coupled with the wave model SWAN (Simulating WAves Nearshore; Booij et al., 1997) and with the sediment model CSTMS (Community Sediment Transport Modeling Systems) within the COAWST framework (Coupled Ocean Atmosphere Wave Sediment Transport model; Warner et al., 2010). The coupled model solves equations for temperature, salinity, horizontal velocities, and various tracers (e.g. sediment classes) on two model grids (Fig. 10): a coarse grid covering the SBNS and the part of the English Channel and a high resolution grid focusing specifically on the BPNS. The coarse resolution model serves primarily to smoothen boundary conditions for the high resolution model. The two models are coupled interactively in two-way nesting, exchanging information every 10 minutes of the simulation time. Both models have 15 vertical layers, following the bottom bathymetry (so-called sigma-layers).

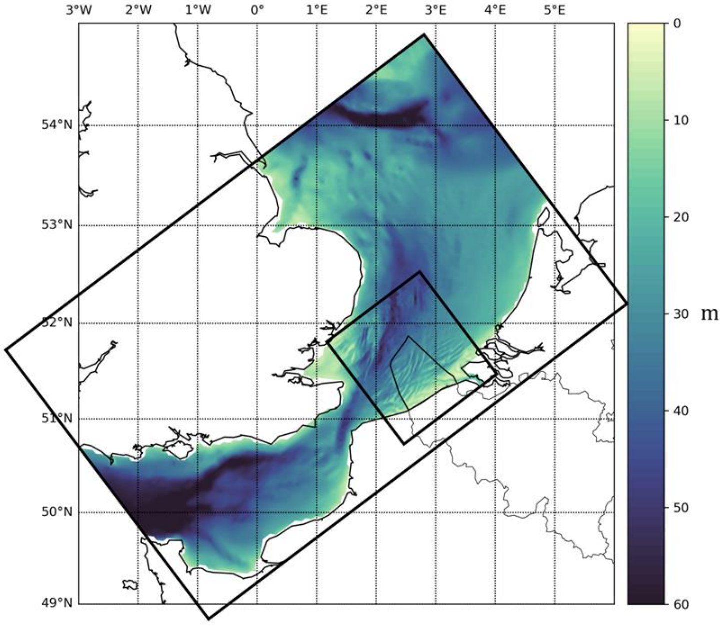


Figure 10 Extension of the coarse (large rectangular) and high (small rectangular) resolution model grids. The coarse resolution model covers the SBNS and the English Channel with a 5x5 km^2 grid, while the BPNS is covered by the 1x1 km^2 grid of the high resolution

The coupled model is forced on the sea-air interface and at the open boundaries (Figure 11). At the sea-air interface, the model is supplied with meteorological variables acquired from the ECMWF, that are further used in the bulk atmospheric fluxes calculations (Fairall et al., 2003). In addition, the vertically integrated climatological series of the primary production of carbon, constructed from the CMEMS data are imposed at the open surface (Butenschön et al., 2016).

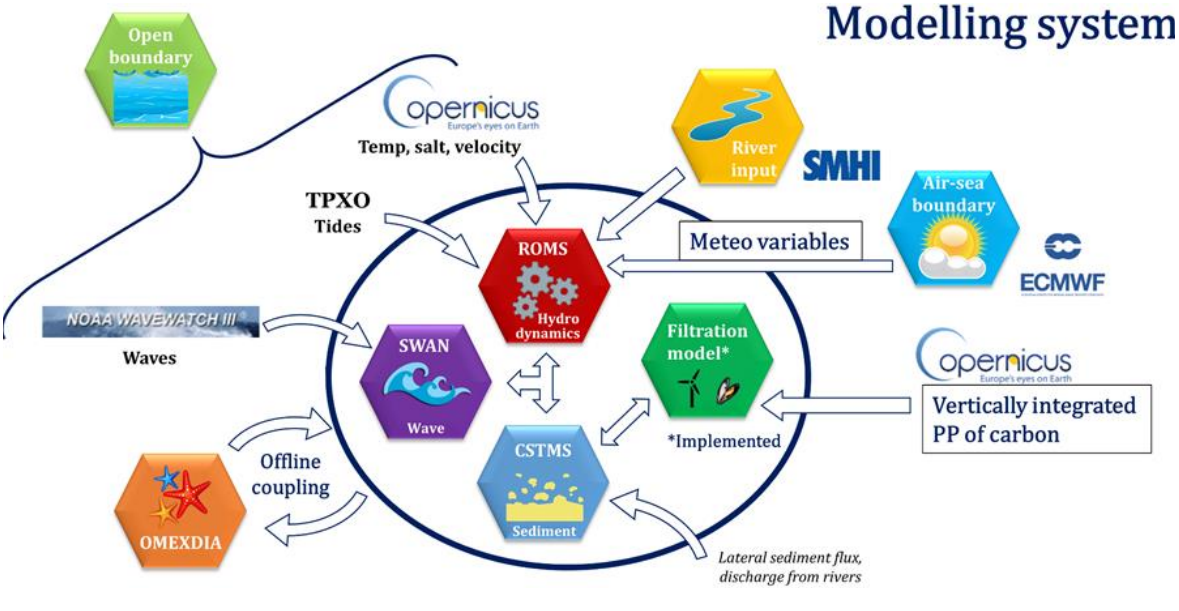


Figure 11 Model coupling and forcing. The dark-blue circle in the centre comprising ROMS, SWAN, CSTMS and a filtration model represents the COAWST coupling system. All the arrows pointing inside the circle are different forcings. The two arrows pointing in opposite directions and representing the exchange are referring to the coupling with the diagenetic model OMEXDIA.

At the open boundaries (the Channel in the south, the greater North Sea in the north), the model receives data for water temperature, salinity and water speed from a CMEMS regional product. The model also modulates a tidal signal using 11 tidal constituents available from a regional TPXO product (Egbert and Erofeeva, 2002). The wave signal, acquired from a global WAVEWATCH III product is also imposed at the open boundaries (Tolman, 1989). Besides, there are 4 major rivers (Thames, Seine, Rhine + Maas and Scheldt) in the domain, imposed as point sources of water discharge, temperature, salinity and sediments (Lindström et al., 2010).

We have also implemented a filtration model (Fig. 12) as a part of the sediment model CSTMS. The main principle of the filtration model was inspired by the paper of Slavik et al. (2019) who calculated the amount of filtered carbon proportionate to its concentration in the water column.

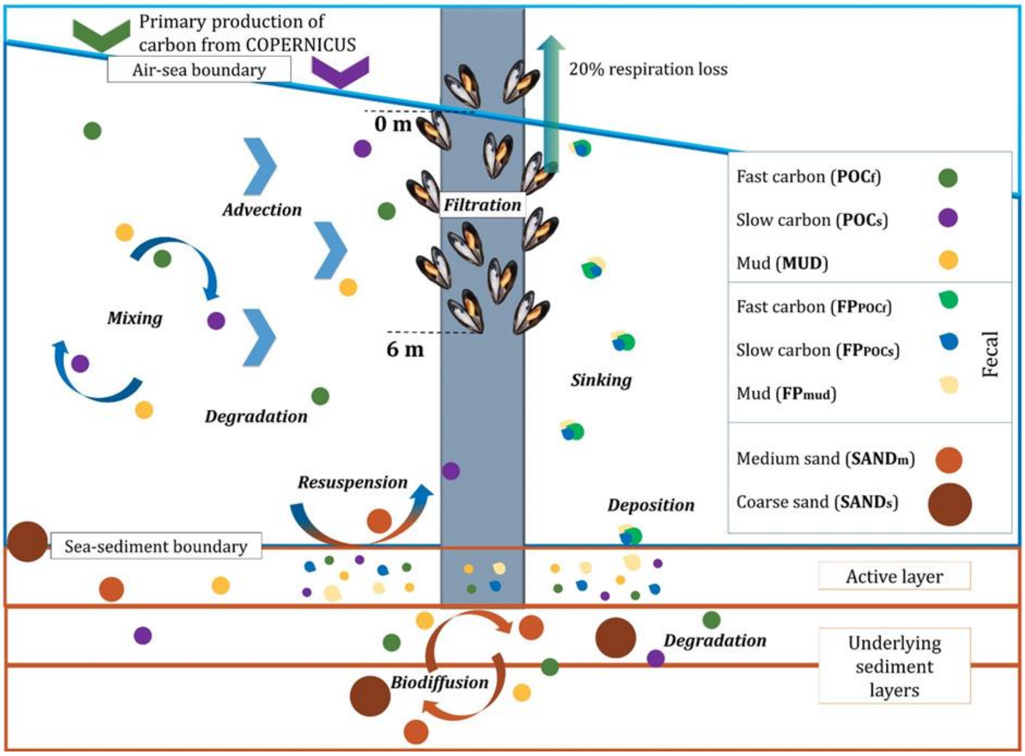


Figure12 Filtration model – schematic representation of all involved processes. The model accounts for the filter feeders colonizing turbine foundations, which lump together suspended organic and inorganic particles that sink into the bottom in a form of faecal pellets (From Ivanov et al. 2021).

Our model calculates the amount of organic carbon and inorganic fine materials filtered from the water by the fouling fauna (in this model only the blue mussel M. edulis, as their biodeposition is several orders of magnitude higher than deposition from the other local feeders). Blue mussels eject filtered materials as faecal pellets that rapidly sink to the bottom, where they fall apart into carbon and inorganic fine particles, and can again be subject to resuspension or burial due to bioturbation. The fouling fauna is assumed to live in the upper 6 m of the water column. The model does not account for its annual cycle, but it accounts for variations in primary productivity through the CMEMS forcing. Furthermore, we account for the tidal dynamics, as mussels exposed to the air during the low tide do not filter. The distribution of wind turbines in the model is defined via two variables: the mean monopile radius (defining the total number of colonizing blue mussels) and number of monopiles in the model's grid cell.

The model has undergone meticulous validation for temperature, salinity, residual and instant currents, tidal ellipses, tidal amplitudes and phases, water discharge through the straits, suspended particulate matter in the water column during high and low tides, steady-state of sediment classes in the sediment bed, significant wave periods and heights and the total organic carbon (TOC) deposition at the stations. The results of this validation are explained in more detail in Ivanov et al. (2020) and Ivanov et al. (2021). We also have tested the model's ability to resolve the variability of the spatial extent of the brackish Scheldt-Rhine plume and the intensity of the tidal currents during different tidal phases and under different meteorological conditions. The high resolution model has shown a wider range of variability from one season to another and from one tidal phase to another, therefore its results were analysed for simulation scenarios, while the coarse resolution model was given the role of a boundary condition smoother.

Coupling to sediment biogeochemistry

We subsequently fed output of the COAWST modelling setup, in particular the organic matter deposition fluxes, into a dynamic model of sediment biogeochemistry, adapted from OMEXDIA (Soetaert et al., 1996a, 1996b) (Fig. 13) In this numerical model, the recycling of organic matter to free inorganic nutrients is modelled on a 1D grid consisting of multiple layers. This implementation was performed for each cell of the COAWST model, resulting in a modelling result for the full BPNS and bordering regions of the same 1 x 1 km^2 resolution.

In OMEXDIA organic matter is degraded through microbial and faunal respiration processes, which are regulated by the availability, or inhibited by the presence of reactants (O_2, NO_3^-) in a given depth-layer. Specifically, the processes are oxic respiration, denitrification (nitrate respiration), and anoxic respiration (the lump sum of mineralization processes using electron acceptors other than O_2 and NO_3^-). To implement bioturbation in OMEXDIA, which differs throughout the modelling domain, we used information on species biomass collected in FaCE-It (Toussaint et al., 2021), and in 2011 (Braeckman et al., 2014), and coupled this to a description of faunal activity in relation to food input quantity and quality (Zhang and Wirtz, 2017).

To force realistic spatial variability in bottom water nutrients, bottom water concentrations of NO_3^-, NH_4^+, O_2 and DIC were extracted from the IBI MFC biogeochemical analysis and forecast system (Sotillo et al., 2015; E.U. Copernicus Marine Service Information, 2020). Other annually varying inputs such as sediment porosity, bottom water temperature, and organic matter freshness were passed on from the COAWST model.

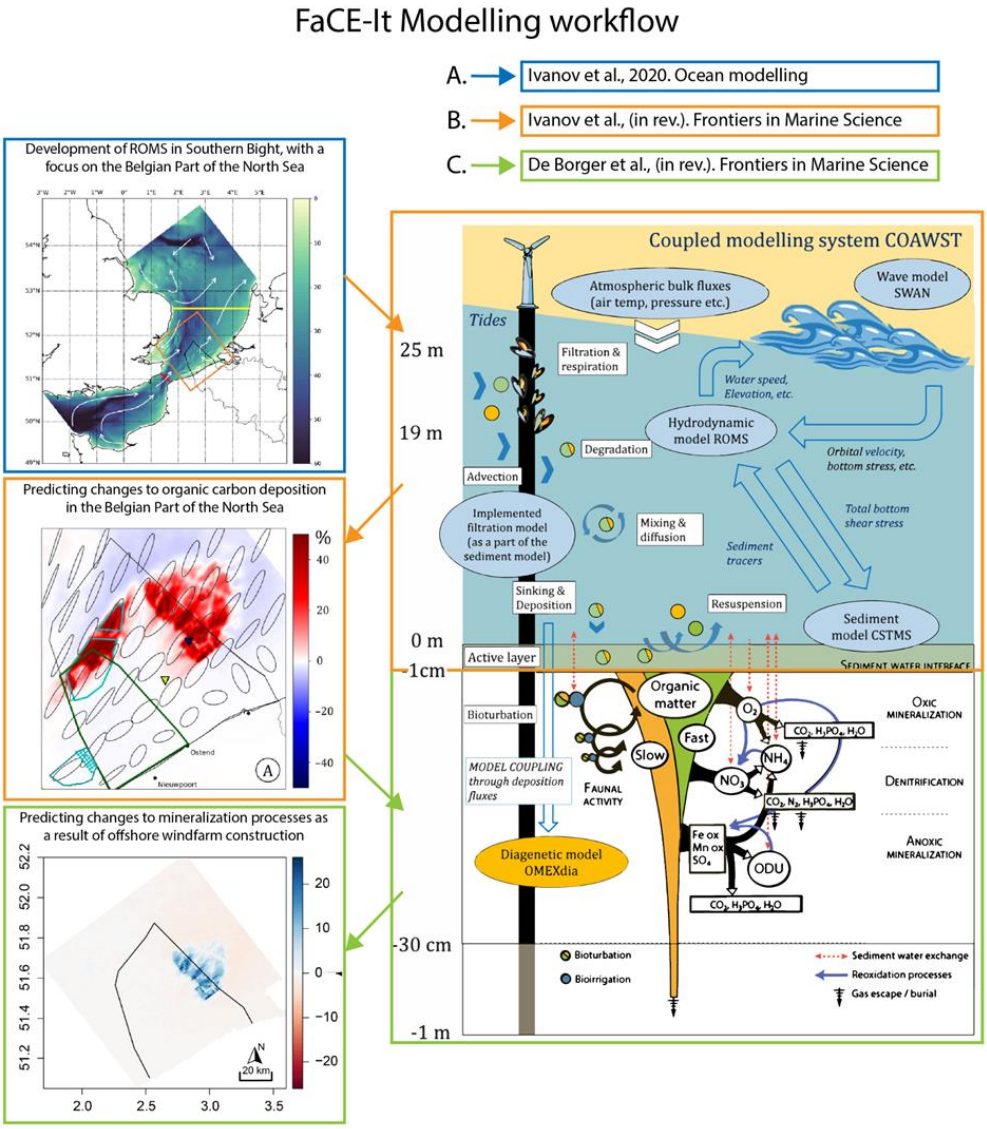


Figure 13 Outline of the spatial modelling work of the FaCE-It.

This one-way coupling between the COAWST and OMEXDIA was chosen after weighing other alternatives, either deriving empirical relations between diagenetic processes and environmental variables and inputs (meta-modelling, Capet et al., 2016), or implementing an extended biogeochemical component in COAWST. The advantage of either of these approaches would have been a two-way coupling, allowing for a feedback from the sediment back to the water column, which will be needed in the future to address the ecosystem effects of OWFs with added certainty. However, we chose the one-way coupling to represent the sediment in high resolution, and time considered, implementing and validating the (sediment) biogeochemistry in COAWST turned out to be beyond the scope of FaCE-It. A similar coupling of a ROMS model component to OMEXDIA through carbon deposition fluxes has previously been shown to work in the Louisiana shelf (Fennel et al., 2013; Laurent et al., 2016).

4. Stakeholder questions and scenarios definition

Different scenarios were developed to address concerns raised by a stakeholder during the FaCE-It annual meeting in 2019. The representative from WWF-Belgium has formulated the following issues:

1) How can new OWFs in the new concession area (NCA) be designed in terms of the number of turbines and their locations, in order to preserve the gravel beds in that Natura 2000 area?
2) What impact would the turbines cause if located close to the gravel beds in the Natura 2000?

Up until the end of October 2020, the only information available about a new OWF in the NCA was the expected total amount of energy produced (2100 MW, from https://www.4coffshore.com/offshorewind/, accessed on 14/10/2020). No information was available concerning the location and number of turbines, alongside the nameplate capacity. Therefore, we chose three nameplate capacities based on the used turbine techniques in the BPNS and in the Dutch and French coastal zones: 8.4 MW as in the recently operational SeaMade and Norther OWF; 10 MW as in the experimental Dutch Borssele-V OWF; and 13 MW as in the future Dunkirk OWF (derived based on the expected produced amount of energy and the total number of turbines). These choices resulted in the total number of 250, 210 and 162 turbines respectively. In addition, two principal placement scenarios were developed. In the Northern placement, all the turbines are located at least 3 km distance from the closest gravel bed; while in the Southern placement all the turbines are placed as close to the Belgian coast as possible, resulting in an overlap with gravel beds in the area. The position of turbines for all six resulting scenarios (3 turbine totals x 2 placement scenarios) can be seen on Figure 14. The density of monopiles was chosen as 2 monopiles per km^2, which would allow the wind to recover in the wake of the turbines (Choudhry et al., 2014). We note that we did not target neither the cost minimization (associated to the electrical cable extension), nor bathymetric constraints.

We also ran a reference (R-0) scenario, with no OWFs in the domain. This scenario represents the natural state for total organic carbon (TOC) deposition without human alterations (i.e. turbine foundations substructures), and allows to compare any scenario in terms of its relative impact to the reference scenario. The C-0 or Current scenario considers only the current concession area (CCA, including the Borssele OWFs), without the NCA or Dunkirk OWF. The D-210 or Dunkirk scenario is an additional scenario that features all the current and future OWFs in the BPNS and its surroundings, and was developed to respond to the question: what would happen if all the Dunkirk turbines are located as close as possible to the border of the BPNS? The D-210 scenario is identical to the N-210 scenario in terms of the number and placement of turbines in the NCA.

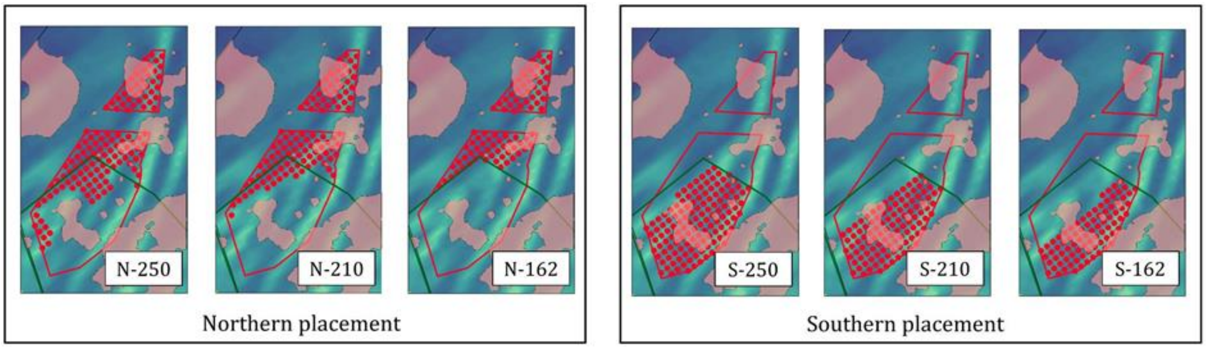


Figure 14 Location of turbine foundations in the NCA under two principal scenarios: Northern placement and Southern placement. In case of the Northern placement the turbines are located at least 3 km away from the closest gravel bed. In case of the Southern placement the turbines are placed as close to the coast as possible, covering several Natura 2000 gravel beds. Total number of turbines (250, 210 or 162) referring to a chosen nameplate capacity of a single turbine: 8.4, 10 or 13 MW.

Alterations to biogeochemical seafloor functioning were modelled for three scenarios described above: a first scenario contained no OWFs (pre-2008 in the BPNS; R-0 in Table II), a second scenario contained all currently (2021) installed turbine foundations in the BPNS, and also in the Dutch Coastal zone bordering the BNPS - the Borssele OWFs ('Current' scenario, derived from scenario C-0 in Table II). In a third scenario we included turbines in the western concession zone in the BPNS, and we included the Dunkirk OWF in the French Coastal Zone bordering the BPNS ('Future' scenario, D-210 in Table II). For these scenarios of OWF development, we looked at the results after 20 years of operation.

Table II. Scenarios for model runs. Reference (R-0) scenario refers to the year prior to 2008, when no turbines were yet installed. Current scenario refers to the modern day scenarios (all turbines are already installed in Belwind, C-Power, Northwester 2, Northwester 2, Northwind, Norther, Rentel, SeaMaid and Dutch Borssele) but no turbines yet in the NCA and Dunkirk). Northern placement and Southern placement consider several configurations of turbines for the NCA, differing in their number and placement. Dunkirk scenario has N-210 on its base plus turbines at the Dunkirk OWF.

Scenario	CCA	NCA	Monopile number at the NCA	Dunkirk	Abbreviation (where the number indicates the quantity of monopiles in the NCA)
Reference / Baseline	No	No	0	No	R-0
Current	Yes	No	0	No	C-0
Northern placement	Yes	Yes	250, 210, 162	No	N-250, N-210, N-162

Southern Placement	Yes	Yes	250, 210, 162	No	S-250, S-210, S-162
Dunkirk / Future	Yes	Yes	210	Yes	D-210

4. SCIENTIFIC RESULTS AND RECOMMENDATIONS

4.1. A detailed investigation on the food-web structure in OWFs

4.1.1 Food-web components along the depth gradient

The detailed food-web study along the depth gradient of a gravity-based foundation indicated differences in food-web complexity between the different depth zones (Fig. 15, Table III). The lowest trophic complexity (indicated as the Standard Ellipse Area in an isotopic biplot) was found in the *Mytilus* zone and the intertidal, followed by the *Metridium* zone. This is explained by the composition of the assemblages occurring on these zones. All these zones mainly consist of sessile suspension feeders that are constrained in their access to food (Richoux et al. 2013). Sessile suspension feeders generally have a low capacity to exploit different food sources and our results indicate similar diet compositions (Dubois and Colombo 2014, Catry et al. 2016, Kaymak et al. 2018).

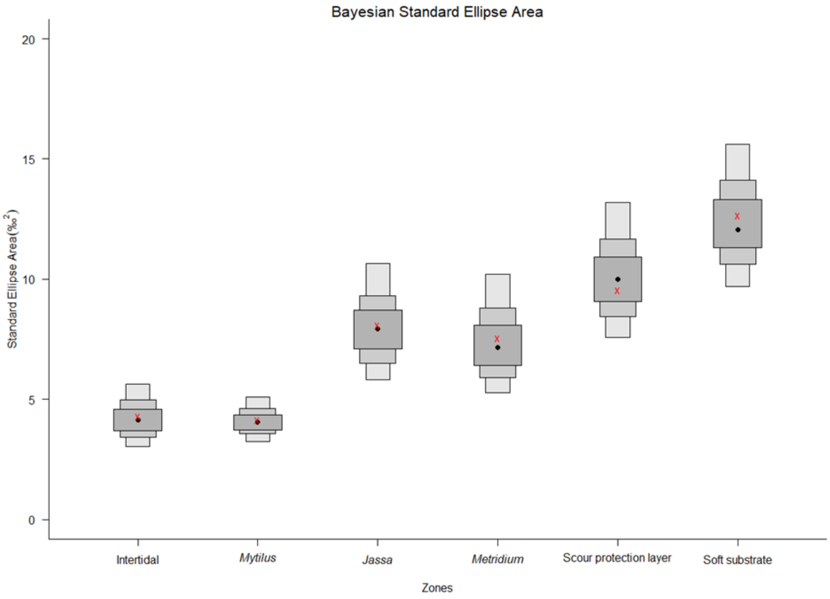


Figure 15 Food-web complexity at the six different zones along the depth gradient of a gravity-based foundation.

In contrast, an increased food-web complexity was observed in areas where organic matter accumulation occurs, i.e. SPL, soft substrate and *Jassa* zone. The largest complexity was observed for the assemblages in soft substrates, followed by the SPL, suggesting that these two zones can accommodate a variety of organisms. The *Jassa* zone showed the third highest food-web complexity, which was not significantly higher than that of the *Metridium* zone. However, the dominant species in the *Jassa* zone (the amphipod *Jassa herdmani*) has a unique behaviour. *Jassa herdmani* is a tube-building amphipod that produces large mats mainly consisting of organic matter that derives from the water column (Beermann and Franke 2012). This amphipod performs grooming motions, which loosen the organic matter that is accumulated between these mats (Dixon and Moore 1997). Thus, part of the organic matter becomes available for other organisms, resulting in an increased range of available food sources in the *Jassa* zone.

Soft sediments near turbine foundations showed the highest food-web complexity. They have a low median grain size compared to sediments further away from the turbines, and a high organic carbon content, which is largely a consequence of the biodeposition of organic matter by the colonising organisms occurring on the turbine (Coates et al. 2014). This causes a shift in the macrofaunal composition and increases species richness close to the turbines (Coates et al. 2014, Lefaible et al. 2018). The alteration in biodiversity leads to more diverse local food-webs with a high number of trophic groups in soft sediments close to turbine foundations. Furthermore, the soft substrate assemblages near the turbines consist of a combination of soft sediment and colonising species. Indeed, some colonising species are capable of (temporarily) surviving in the soft substrate, after having been dislocated from the turbines and/or the SPL (Bouma and Lengkeek 2012, Fernandez-Gonzalez et al. 2016).

Table III Size of trophic niches of the six macrofaunal zones occurring along the depth gradient of a gravity-based foundation

Zone	**Trophic niche size**
Intertidal	4.31
Mytilus	4.14
Jassa	8.09
Metridium	7.53
SPL	9.54
Soft substrate	12.63

The assemblage at the SPL showed the second largest food-web complexity and diversity. SPLs are made of rocks, forming a collar around the foundations (Baeye and Fettweis 2015), adding complexity and providing additional habitat (Causon and Gill 2018). However, this study revealed that SPLs have a significant role not only for increasing habitat heterogeneity but also for promoting food-web complexity at the base of the turbine foundations. SPLs can accommodate species of multiple trophic levels (ranging from small deposit feeders to large predatory fish species), which results in increased food-web complexity. This was further confirmed by the feeding habits of the species occurring in this area, which exploited a wide range of food sources found at and in between the rocks of the SPL. The pivotal role of SPLs for the food-webs around offshore wind turbines was further highlighted by the feeding ecology of fish species associated with it (benthopelagic and benthic), which feed on colonising organisms for a prolonged period (see 4.1.2). The combination of these results reveals the significant function of SPLs which operate more efficiently as artificial reefs than the piles themselves. Furthermore, our results provide evidence that SPLs increase the available food quantity for both sessile and mobile fauna. However, it should be highlighted that foundations without SPLs, i.e. jackets, were not investigated, and an objective comparison between turbines with and without SPL is currently lacking. Nevertheless, the results of this study suggest that SPLs could act as 'trophic-diversity hot spots' at the base of gravity-based and possibly also monopile foundations.

4.1.2 Attraction and production of fish in OWF

The stomach content analysis of the five fish species that abundantly occur in close proximity to the D6 gravity-based foundations indicated that cod juveniles, pouting and horse mackerels mainly feed on the amphipod *Jassa herdmani*. The stomach contents of mackerels were dominated by zooplankton, while the stomachs of sculpin mainly contained *Pisidia longicornis* and fish (Fig.16, Table IV).

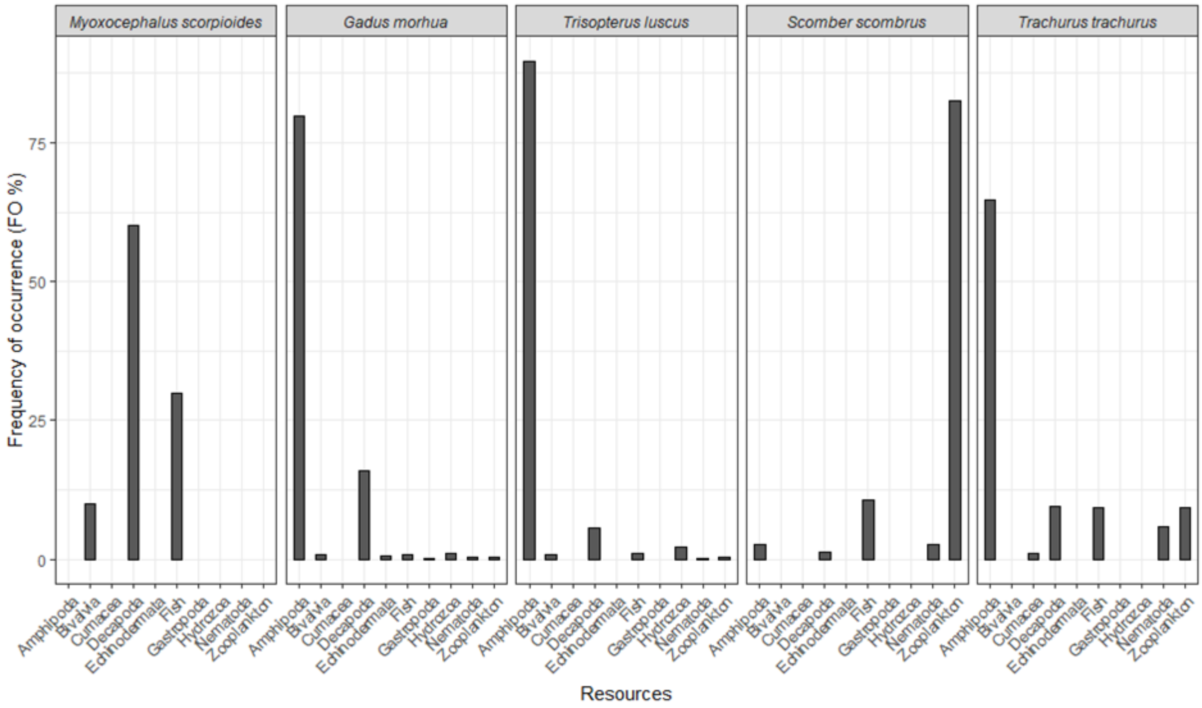


Figure 16 Percentage of abundance (%) of the prey items in the stomach contents of each of the five fish species, the pelagic Scomber scombrus *and* Trachurus trachurus, *the benthopelagic* Gadus morhua *and* Trisopterus luscus, *and the benthic* Myoxocephalus scorpioides.

Table IV List of prey items. Percentage (%) of abundance of consumed items in the diets of the five different fish species

	Pelagic		Benthopelagic		Benthic
	Scomber scombrus	***Trachurus trachurus***	***Gadus morhua***	***Trisopterus luscus***	***Myoxocephalus scorpioides***
Amphipoda					
Jassa herdmani	2.7	61.2	76.2	87.6	-
Monocorophium acherusicum	-	1.2	0.6	0.5	-
Phtisica marina	-	-	0.3	1.2	-
Stenothoe valida	-	-	2.5	0.2	-
Unidentified sp.	-	2.2	-	0.1	-
Bivalvia					
Aequipecten opercularis	-	-	-	0.3	-
Crepidula fornicata	-	-	-	0.1	-
Mytilus edulis	-	-	0.4	0.3	10
Unidentified sp.	-	-	0.4	-	-

Cumacea					
Unidentified sp.	-	1.2	-	-	-
Decapoda					
Astacidea	-	-	-	0.1	-
***Liocarcinus* sp.**	-	1.2	-	0.2	-
***Macropodia* sp.**	-	-	-	0.4	-
***Megalopa* sp.**	-	1.2	-	-	-
***Pagurus* sp.**	-	-	0.1	-	-
Pilumnus hirtellus	-	-	0.1	-	-
Pisidia longicornis	-	5.9	15.3	4.7	40
Processa modica	-	-	-	0.1	10
Unidentified sp.	1.3	1.2	0.4	-	10
Echinodermata					
Ophiothrix fragilis	-	-	0.7	-	-
Gastropoda					
Unidentified sp.	-	-	0.1	-	-
Hydrozoa					
Sertularia cupressina	-	-	-	0.1	-
***Tubularia* sp.**	-	-	1.2	2.1	-
Nematoda					
Unidentified sp.	2.6	5.9	0.4	0.1	-
Pisces					
Engraulidae	4.0	-	-	-	-
Unidentified sp.	6.7	9.4	0.8	1.2	30
Zooplankton					
Unidentified sp.	82.6	9.4	0.4	0.1	-

However, SIA did not always show similar results as the stomach content analysis. Specifically, SIA indicated that mackerel exploit a variety of prey items and has the least specialized diet, while juvenile cod, pouting and sculpin have a more specialized diet, exploiting one or a few prey items (Fig. 17).

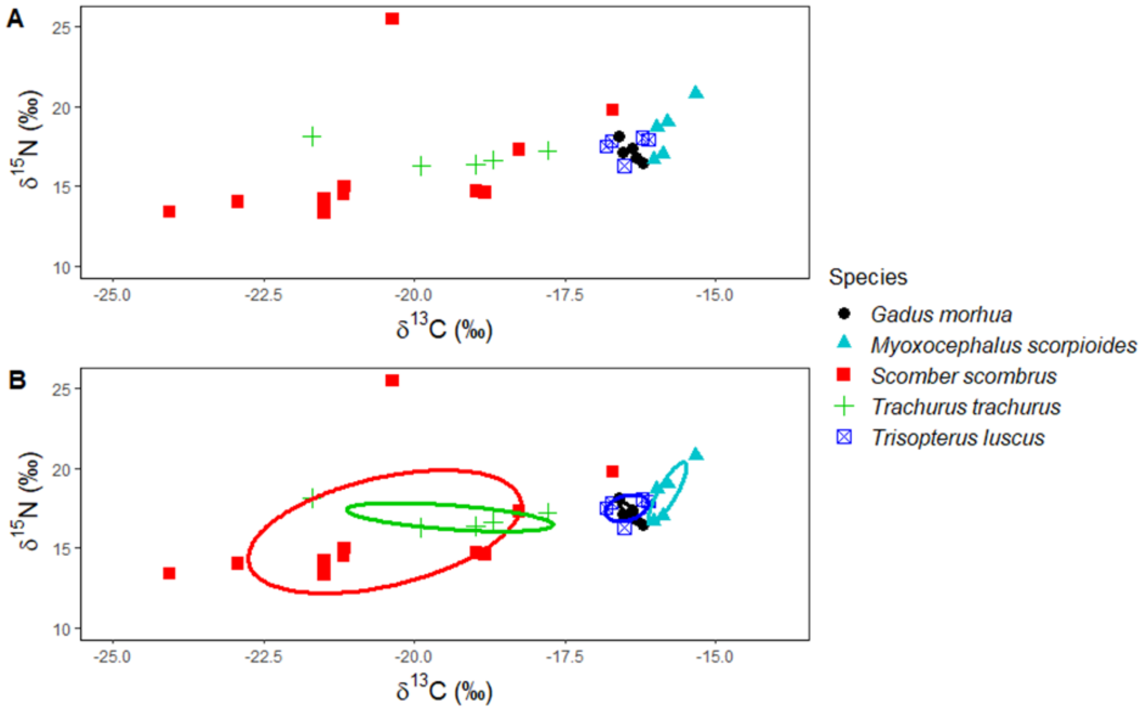


Figure 17 A: Individual stable isotope values of the abundantly present fish species. B: Trophic niche areas of the five targeted fish species. Different colours represent different species.

The combination of these two methods (SIA and stomach content analysis) indicated that juvenile cod mainly feeds on colonising amphipods (i.e. *Jassa herdmani*), while colonising crustaceans (amphipod Jassa herdmani and crab *Pisidia longicornis*) is the main prey item of pouting (Fig. 18). Small crustaceans (i.e. *Jassa herdmani* and zooplankton) was the main prey item of horse mackerel, and the diet of sculpin was mainly based on decapods (i.e. *Pisidia longicornis*) and fish. Mackerel was the only species that exploited equal shares of most of the prey items included in this dual analysis.

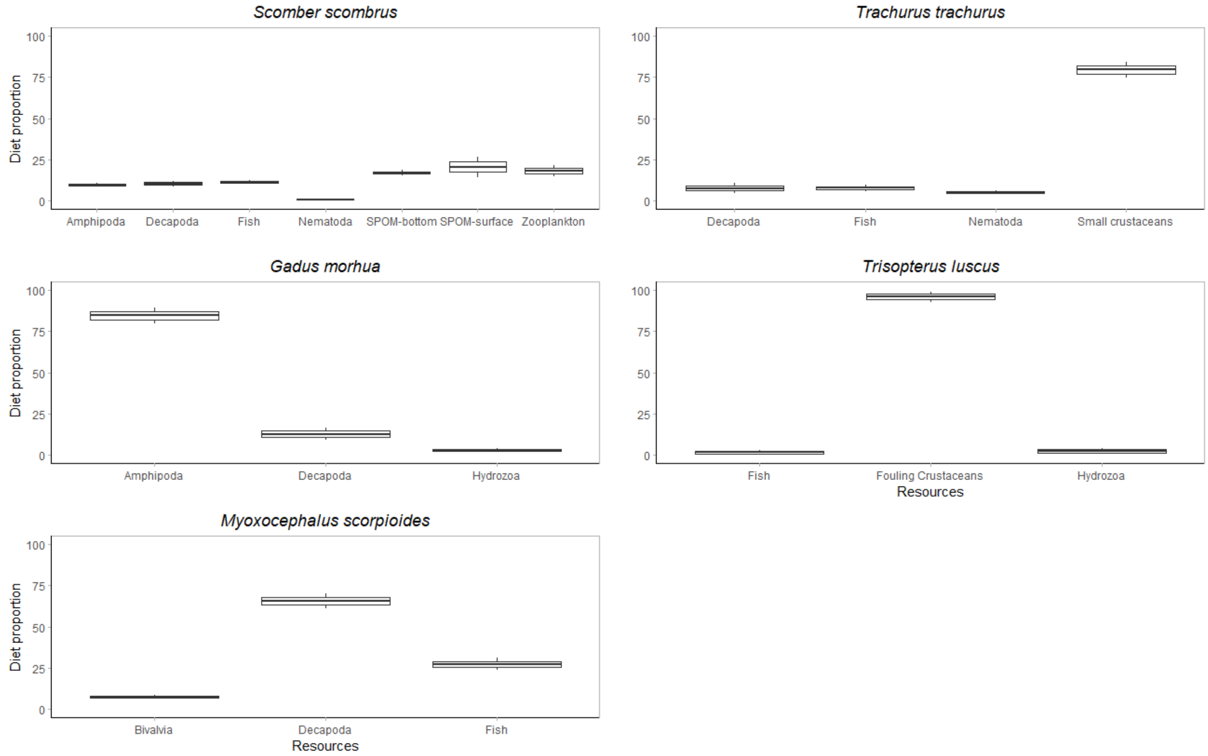


Figure18 Contribution of prey items in the diet of the five investigated fish species as were estimated by the combination of stomach content and stable isotope analyses.

The combination of stomach content analysis (SCA) and SIA is a powerful approach on determining feeding habits of species. SCA provides a snapshot of recently ingested (up to 10 h) prey items, but it can provide a high taxonomic resolution of the stomach contents that may be difficult to obtain with other methods (Lin et al. 2007). However, it has multiple limitations, such as the requirement of high sampling frequencies (Parkyn et al. 2001, Daly et al. 2013). Stomach contents mainly consisting of colonising organisms can be an indication of attraction towards artificial structures to forage.

The SCA limitations can be overcome by combining it with SIA, which provides a time-integrated (up to 6 months) and space-integrated dietary estimation (Bearhop et al. 2004, Newsome et al. 2007). Thus, SIA gives long-term information about the feeding ecology of a species. A diet-based study on colonising organisms for a prolonged period of time indicates a long-term residency close to artificial structures, which is a clear sign of attraction that could potentially lead to increased local production. SIA provides information on the truly assimilated food items (Boecklen et al. 2011), but it does not offer the taxonomic resolution that SCA provides (Daly et al. 2013). Therefore, we combined these two methods to acquire a complete picture of the feeding ecology of the five fish species. We hypothesized that when both SCA and SIA show that the fish' diet relies on colonising fauna, then fish species utilize the artificial hard structures as feeding grounds for a prolonged period of time. When SCA shows a diet based on colonising fauna, which is not confirmed by SIA, fish species occasionally use the artificial hard habitats as feeding grounds. Finally, fish probably do not exploit artificial hard structures as feeding grounds when both SCA and SIA indicate a diet based on prey items that are not associated with these structures.

Our study showed that two benthopelagic (pouting and juvenile cod) and one benthic (sculpin) fish species use artificial hard substrates, such as OWFs, as feeding grounds for a prolonged period since both the short- (stomach content analysis) and the long-term (SIA) dietary analyses suggested that their diets are based on colonising organisms. The pelagic horse mackerel feeds only opportunistically in the OWF area since SCA and SIA indicated contradictory results. Finally, the pelagic mackerel mainly fed on zooplankton, and hence, it did not show any exploitation of the artificial hard substrates as feeding grounds. All these species are attracted towards the offshore wind turbines, but juvenile cod, pouting and sculpin seem to remain in the area to feed for a prolonged time. Overall, the results of this study corroborate the hypothesis that there are two main pathways supporting the feeding ecology of fish species associated with OWFs: the 'pelagic pathway' where fish species consume prey items of pelagic origin, such as zooplankton, and the 'benthic pathway' where fish species exploit colonising organisms, such as *J. herdmani* and *P. longicornis*. These findings could provide an indication of the co-existence of attraction and production on the same artificial reef system (Cresson et al. 2019).

OWFs and other artificial reefs could potentially act as production areas for fish species that exploit the abundant colonising organisms. Support for the production hypothesis has already been provided for pouting and juvenile cod individuals in the Belgian OWFs (De Troch et al. 2013, Reubens et al. 2013b). Juvenile cod individuals remain in close proximity to offshore wind foundations (Reubens et al. 2013a) and settle in the area suggesting increased production (Reubens et al. 2014). Pouting individuals occurring close to offshore wind turbines show better fitness proxies compared to individuals found in sandy areas (Reubens et al. 2013b). Our results further corroborate the previous findings, since both pouting and juvenile cod individuals seem to remain in the OWF area for a prolonged time to feed.

Our study provides evidence that sculpins could also increase their local production within the artificial reef area of OWFs since they were found to feed for a prolonged time on colonising fauna. Artificial habitats have been reported to have a positive effect on the production of benthic fish because of the increased food availability and the low fishing mortality (Cresson et al. 2019). It is, therefore, possible that OWFs could promote the production of benthic species.

This research cannot provide unambiguous answers on whether there is enhanced production of benthic and benthopelagic fish species or not, but it supports the hypothesis that their local production could potentially increase. Furthermore, the results clearly show that OWFs provide increased feeding activity and efficiency. Fish production in the Belgian OWFs is also promoted by the fisheries prohibition. This makes OWFs act as *de facto* marine protected areas with potential spill-over benefits to commercial fish stocks in the wider region (Ashley et al. 2014). In the BPNS, future turbines will be more powerful and more efficient than the currently installed ones, requiring more space between each turbine and possibly (partially) permitting fisheries within the OWF areas. However, the regulation of IMO in Belgium prohibit shipping within 500 m safety zone around wind turbines (Mehdi et al. 2017), which might have an effect on fish production. The increased number of OWFs in the southern North Sea together with the fish attraction towards them, the prolonged residency and the (partial) fisheries restrictions around these structures could result in enhanced production with spill-over benefits in the region. Further research on the possible increased production is necessary and collaboration between countries is needed to identify the cumulative effects of the OWF establishment on fish stocks.

RECOMMENDATION BOX: SPLs are important parts of OWFs from an ecological point of view. They do not only provide food and shelter, but they also increase diversity of food items, providing support for the basis of the food web, potentially cascading into increased local production of commercially important species. However, ecological considerations have never been considered in SPL design up until now. We recommend investigating how SPLs can be optimised (material choice, different gradings, adding habitat complexity) to structures that both counteract scouring and at the same time promote structure and function of marine ecosystems. Furthermore, properly designed SPLs could potentially be exploited as nature restoration techniques, promoting the attachment of species under pressure, such as species belonging in the OSPAR red list.

A second point that became clear from this research is the fact that the restriction of fishing activities can be a significant reason for the observed patterns. Fisheries restrictions allowed fish not only to approach offshore wind turbines and their SPLs but also to remain in the area for a prolonged period of time. Restricting human activities, such as commercial fishing, from the (newly introduced) OWF zones may eventually lead to fish production in the area, accompanied by a spill-over effect and promoting fish stocks in the BPNS and beyond.

4.2 Upscaling the effect of colonising fauna on turbines on pelagic primary production

The organisms that were attached on the PVC panels after 1 year deployment and were exposed to the labelled food showed different degrees of carbon assimilation. *Jassa herdmani* and *Mytilus edulis* indicated the highest total carbon assimilation (Fig. 19). All the other taxa showed significantly lower carbon assimilation. This is explained by the feeding preferences of the taxa included in the experiment. *Jassa herdmani* is a competitive suspension feeder that may restrict other organisms from feeding when it occurs in high abundances (Mavraki et al. 2020a). The juveniles of *M. edulis* that were attached on the PVC panels assimilate carbon at a faster rate (Hentschel 1998) and they can filter small food items more rapidly than adults (Jacobs et al. 2015). The other taxa that were included in the experiment were either suspension feeders that prefer to feed on particles completely different than the food provided (e.g. *Metridium senile* mainly consumes zooplankton – Östman et al. 2010) or were predators (e.g. nudibranchs) and, thus, did not feed on the provided food.

Since *J. herdmani* and *M. edulis* assimilated by far the largest amount of carbon compared to the other colonising taxa, we only used these two species for the upscaling calculations. Our results indicated that these two species occurring on all offshore wind foundations in the BPNS assimilate in their tissues in total 200 ton C y^{-1}, which is ~0.40 % of the primary producer standing stock in the OWF area of the BPNS (Table V).

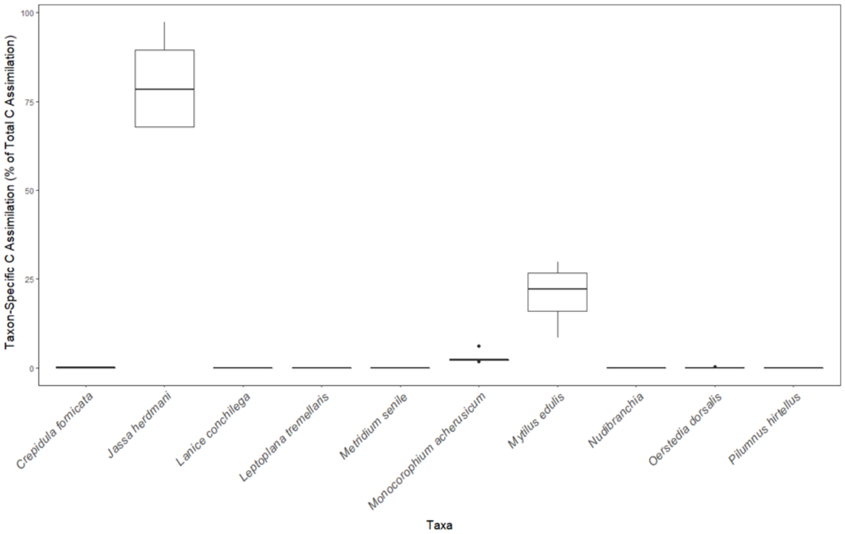


Figure 19 Taxon-specific carbon assimilation (expressed as % of the total carbon assimilation by all the consumers) of the different taxa incubated in the experimental tanks.

Table V Assimilation of the primary producer standing stock in the tissues of the species Jassa herdmani *and* Mytilus edulis*.*

Assimilation of primary productivity	***Jassa herdmani***	***Mytilus edulis* (<1cm)**	***Mytilus edulis* (1-3cm)**	***Mytilus edulis* (>3cm)**
Total Biomass C per MONOPILE (µg C m^{-2} d^{-1})	2309790	4243	108708	29420
Total Biomass C per JACKET (µg C m^{-2} d^{-1})	6533675	375416	619825	0
Total Biomass C per GRAVITY-BASED (µg C m^{-2} d^{-1})	4532899	55244	412899	39927
Total C assimilation per MONOPILE (µg C m^{-2} d^{-1})	701736	4699	120399	32584
Total C assimilation per JACKET (µg C m^{-2} d^{-1})	1984993	415791	686485	0
Total C assimilation per GRAVITY-BASED (µg C m^{-2} d^{-1})	1377138	61185	457305	44222
Total C assimilation per MONOPILE (mg C turbine^{-1} d^{-1})	886959	902	23117	6256
Total C assimilation per JACKET (mg C turbine^{-1} d^{-1})	5797548	184486	304571	0
Total C assimilation per GRAVITY-BASED (mg C turbine^{-1} d^{-1})	782498	7495	56002	5417

Total assimilation for all MONOPILES in BPNS (g C d^{-1})	234157	238	6103	1652
Total assimilation for all JACKETS in BPNS (g C d^{-1})	278282	8855	14619	0
Total assimilation for all GRAVITY-BASED foundations in BPNS (g C d^{-1})	4695	45	336	33
Total assimilation (kg C d^{-1}) for *M. edulis* and *J. herdmani* on all OWF foundations	549			
Total assimilation (kg C y^{-1}) for *M. edulis* and *J. herdmani* on all OWF foundations	200391	in 365 days		
Total assimilation (ton C y^{-1}) for *M. edulis* and *J. herdmani* on all OWF foundations	200			
Total production in OWFs BPNS (ton C y^{-1})	50694	assuming equal primary productivity throughout OWFs since enhanced primary productivity hasn't been quantified yet		
		0.40	% of primary productivity in BPNS OWFs is assimilated by *Jassa herdmani* and *Mytilus edulis*	

These species occurring on all the Belgian OWFs ingest carbon (which is available in nature in the primary producers, such as phytoplankton) that amounts at 657 tonC y^{-1} (Table VI). This suggests that 1.3 % of the annual local primary producers are grazed upon by *J. herdmani* and *M. edulis* occurring on the offshore wind turbines that were installed in the BPNS as of March 2020. This amount is not remarkable given that ~ 25 % of the annual primary production in the BPNS is deposited in the sediment (Provoost et al. 2013). Even though this decrease is currently considered small, the installation of more OWFs in the future will naturally lead to an increased reduction of the primary producers, which are the major energy source for ecosystems globally (Imhoff et al. 2004).

Table VI Ingestion of local primary producers by the species Jassa herdmani *and* Mytilus edulis

Processing of primary productivity	***Jassa herdmani***	***Mytilus edulis* (<1cm)**	***Mytilus edulis* (1-3cm)**	***Mytilus edulis* (>3cm)**
SUMMED assimilation (kg C d^{-1}) for all OWF foundations	517	9.14	21	1.68
SUMMED PP processing (kg C d^{-1}) for all OWF foundations	1600	57	132	11
Total processing (kg C d^{-1}) for *M. edulis* and *J. herdmani* on all OWF foundations	1799			
Total processing (kg C y^{-1}) for *M. edulis* and *J. herdmani* on all OWF foundations	656654	in 365 days		
Total processing (ton C y^{-1}) for *M. edulis* and *J. herdmani* on all OWF foundations	657			
Total production in OWFs BPNS (ton C y^{-1})	50694	assuming equal primary productivity throughout OWFs since enhanced primary productivity hasn't been quantified yet		
		1.30	% of primary productivity in BPNS OWFs is grazed upon by *Jassa herdmani* and *Mytilus edulis*	

The comparison of carbon assimilation between different types of foundations and the natural soft sediment indicated that the introduction of jacket foundations results in the highest increase in carbon assimilation compared to the otherwise bare sediments (ratio turbine/sediment: min: 14242 – max: 181259 – Table VII). The establishment of monopile foundations leads to the second highest carbon assimilation (ratio turbine/sediment: min: 38 – max 485), while the presence of gravity-based foundations causes the smallest increase (ratio turbine/sediment: min: 9 – max: 116). Therefore, the introduction of offshore wind turbines significantly increases the local carbon assimilation in the tissues of colonising macrofauna compared to the natural soft sediment communities. Thus, the natural carbon cycle changes due to the establishment of these structures (Mangi et al. 2013). This research clearly indicated that the presence of jacket foundations leads to the most significant increase in carbon assimilation compared to natural soft sediment macrofauna. This is caused by the high densities of *J. herdmani* and *M. edulis* on this type of foundation (Krone et al. 2013). In constrast, the presence of monopiles and gravity-based foundations has a smaller effect on the local carbon assimilation, suggesting that their installation has a smaller effect on the local carbon cycling.

Table VII Comparison of the carbon assimilation between the hard substrate and the natural soft sediment macrofauna related to the surface of each type of foundation

CALCULATIONS FOR COMPARISON WITH PERMEABLE SEDIMENTS	**min. assimilation**	**max. assimilation**	**References**
Carbon assimilation (mg C m^{-2} d^{-1}) by macrofauna in sediment	3.3	42	Bühring *et al.*, 2006b
Total C assimilation per MONOPILE (mg C $turbine^{-1}$ d^{-1})	917234		
Total C assimilation per JACKET (mg C $turbine^{-1}$ d^{-1})	6286605		
Total C assimilation per GRAVITY-BASED (mg C $turbine^{-1}$ d^{-1})	851412		
Amount of C that is NOT assimilated by soft sediment macrofauna due to the construction of 1 MONOPILE (mg C)	1889	24048	
Amount of C that is NOT assimilated by soft sediment macrofauna due to the construction of 1 JACKET (mg C)	35	441	
Amount of C that is NOT assimilated by soft sediment macrofauna due to the construction of 1 GRAVITY-BASED (mg C)	7349	93536	
Ratio C assimilation per MONOPILE (mg C m^{-2}) / C not assimilated in permeable sediments (mg C m^{-2})	485	38	
Ratio C assimilation per JACKET (mg C m^{-2}) / C not assimilated in permeable sediments (mg C m^{-2})	181259	14242	
Ratio C assimilation per GRAVITY-BASED (mg C m^{-2}) / C not assimilated in permeable sediments (mg C m^{-2})	116	9	

The analysis of the food-web properties of the benthic compartment of permeable station 330 and cohesive sediment station 780 is in an early phase. The covid-19 pandemic prevented sending of samples to stable isotope facilities, which only opened their doors near the end of 2020. Given the complexity of the data, a full analysis cannot be provided here, but will serve the construction of a Linear Inverse Model (LIM) that will help understanding the effect of OWFs on the carbon flow through the benthic compartment of marine coastal ecosystems. A preliminary analysis (SIMPROF routine) resulted in the delineation of 15 significantly different clusters of organisms. One cluster consisted of the sediment organic matter of the three stations, all organisms were grouped in the other clusters. Generally, organisms were clustered according to their habitat, regardless of sampling location. The organisms inhabiting the SPL of turbine D6 were generally clustered with the demersal organisms of the OWFs and sandy and cohesive sediments. The organisms inhabiting the higher parts of the turbines were either clustered in a separate group, or belonged to a cluster with pelagic food sources. While this analysis is rather exploratory, it does suggest that the presence of OWFs results in a locally altered food web, where fouling organisms benefit from pelagic food sources.

RECOMMENDATION: The FaCE-It food-web related research clearly showed that food-web structure and functioning is affected, both at the local and wider geographical scale. This is clearly documented at the structural level; however, this needs to be translated into a functional understanding of flow of energy through the coastal ecosystem ultimately leading to production of harvestable biomass. We recommend to fund and conduct scientific research along our pioneering lines in a context of the introduction of multiple OWFs in the marine environment, to enable informing future decisions about decommissioning scenarios for OWFs.

4.3 Benthic ecosystem functioning along a permeability gradient in the BPNS

4.3.1 Methodological developments: Uranine as tracer for bioirrigation

Biogeochemical processes in the sediment are determined by the presence of electron acceptors for the oxidation of organic matter. The presence of these substances largely depends on exchange processes with the water column, which can be diffusive (slow) or faunal-induced, i.e. bioirrigation (fast). Measuring bioirrigation was generally done by measuring the exchange of Br^- between water and sediment, which is methodologically challenging and time consuming. Therefore, we designed a novel technique for easier measurements of bioirrigation.

By fitting fluorescent tracer measurements in a mechanistic model, we were able to infer both the bioiririgation exchange flux (pumping rate, mL cm^{-2} h^{-1}), and the estimated distribution of this exchanged volume in the sediment (attenuation coefficient, cm^{-1}). Figure 19 captures why this technique is an improvement over traditional tracer-based method of deriving bioirrigation. The model was able to fit measured data points with high confidence (Fig. 19 A), producing a best-fit line surrounded by a fairly narrow envelope of alternative model outcomes (distributions of other possible fits considering the data). The comparison with a more traditional discrete sampling strategy is also made. The black dots, and dotted linear decrease fitted through them on Fig. 20 A shows that a linear decrease fits the data less well and is dependent on the timing of the sampling events (the location of the black dots).

The meaning (and relevance) of both parameters is explained in the right-hand panel of Figure 20. The rate of bioirrigation (pumping rate) determines and is mostly visible in the initial decrease of the tracer, and how quickly the steady state is achieved. The attenuation coefficient determines

the sediment volume over which bioirrigation mostly occurs, as well as the difference between initial and ultimate water column concentrations near steady state. This adds important nuance to the interpretation of bioirrigation rates, as similar irrigation rates may have divergent effects on sediment biogeochemistry when the depth over which solutes are exchanged differs.
The application of this novel technique to field cores collected in the Oosterschelde estuary shows what this added nuance means in the field. In total, 70 and 47 successful incubations were performed in the course of 16 months with cores from the intertidal and the subtidal respectively, distributed over 6 sampling stations.

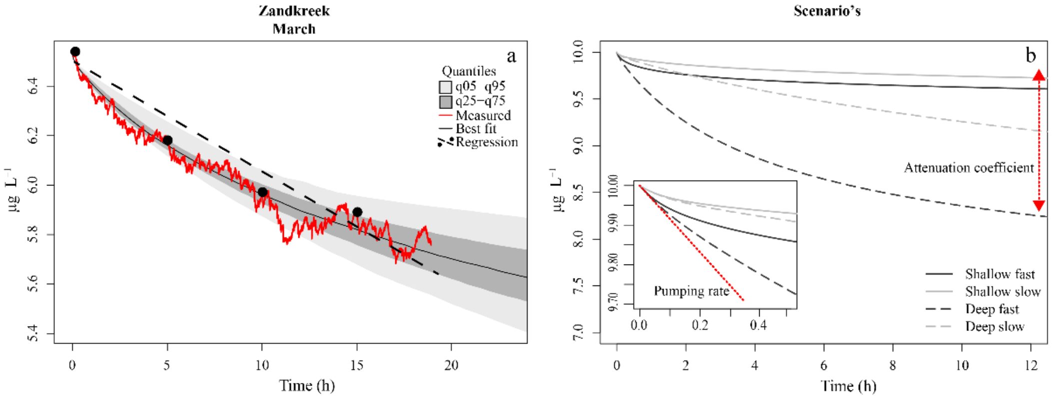


Figure 20 (a) Model fit to data (red line) from a core at Zandkreek in March 2017. The best-fit tracer profile (solid black line) is shown along with the range of model outputs as quantiles (light and dark grey). An example of a linear fit (dashed line) through ((fictitious) samples taken every 5 h (dots) is also shown. (b) Example model output for different combinations of pumping rate (slow = 0.15 ml cm^{-2} h^{-1}, fast = 0.8 mL cm^{-2} h^{-1}), and attenuation coefficients (shallow = 5 cm^{-1}, deep = 0.5 cm^{-1}). The inset shows a close-up from the first half hour of the simulation. The red line illustrates the effect of the pumping rate, which has the strongest initial effect; the red arrow illustrates the effect of the attenuation coefficient which determines the depth of the irrigation. From De Borger et al. (2020).

The macrofauna communities differed between both habitats, and individual sampling locations (Fig. 21; Table 3, De Borger et al., 2020). Species abundances in the intertidal areas were generally 1-to-2 orders of magnitude higher than in the subtidal areas (Fig. 21 a), with peak abundances observed in the Dortsman site in autumn and spring (16054 ± 13939 ind m^{-2}). Biomass, on the other hand, was higher in the subtidal areas (22.31 ± 26.42 g AFDW m^{-2}), as opposed to intertidal areas (10.51 ± 8.59 g AFDW m^{-2}) (Fig. 21 b).

Our results showed that the pumping rate was not significantly different between subtidal and intertidal areas, but that bioirrigation was shallower in the subtidal sediments, as indicated by the higher attenuation coefficient (Fig. 21 c, d). The species community in the subtidal sediments responsible for pumping was less dense, but (on average) the biomass was higher than in the intertidal sediments.

These results can be explained with the differences between intertidal and subtidal areas in mind. Intertidal areas experience stronger variations in physical stressors such as waves, temperature, light, salinity and precipitation than subtidal areas do (Herman et al., 2001) and are more subject to biological stressors such as predation by birds (Fleischer, 1983; Granadeiro et al., 2006; Ponsero et al., 2016). Burrowing more deeply or simply residing in

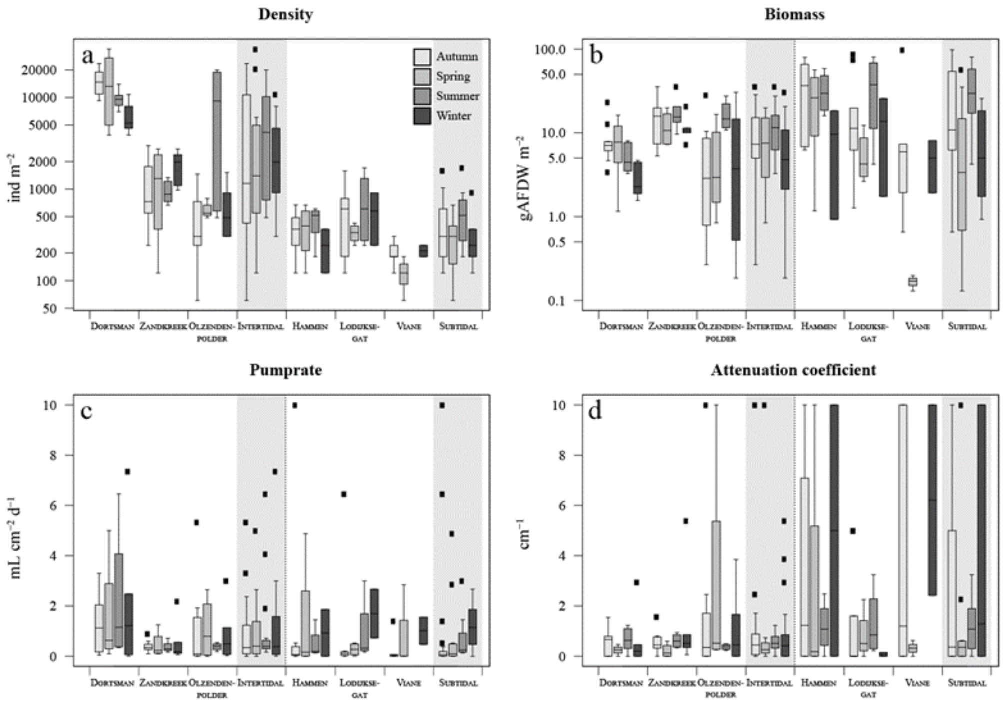


Figure 21: (a) Organism densities (ind m^{-2}); (b) organism biomass as ash-free dry weight (g AFDW m^{-2}); (c) model derived pumping rate (mL cm^{-2} d^{-1}); (d) model derived attenuation coefficient (cm^{-1}). Data arranged per station (white areas) and per habitat type, intertidal and subtidal (grey shaded areas). Black squares = outliers. From De Borger et al. (2020).

deeper sediment layers for a longer time are valid strategies for species in the intertidal areas to combat these pressures (Koo et al., 2007; MacDonald et al., 2014).

RECOMMENDATION: With this technique, measurements of bioirrigation characteristics can be collected independent of the experiment duration and physical setup. The rates produced by this model are also more directly applicable in diagenetic models. We therefore recommend a uniform application of this technique to expand the dataset on bioirrigation.

4.3.2 Organism – sediment interactions governing benthic ecosystem functioning.

With our methods we were able to describe new, and confirm existing interactions between sediment structure, macrofauna communities, and biogeochemistry. Key highlights are represented in the summary figure of the published article concerning this study (Fig. 22; Toussaint et al., 2021) (left panel Fig. 22).

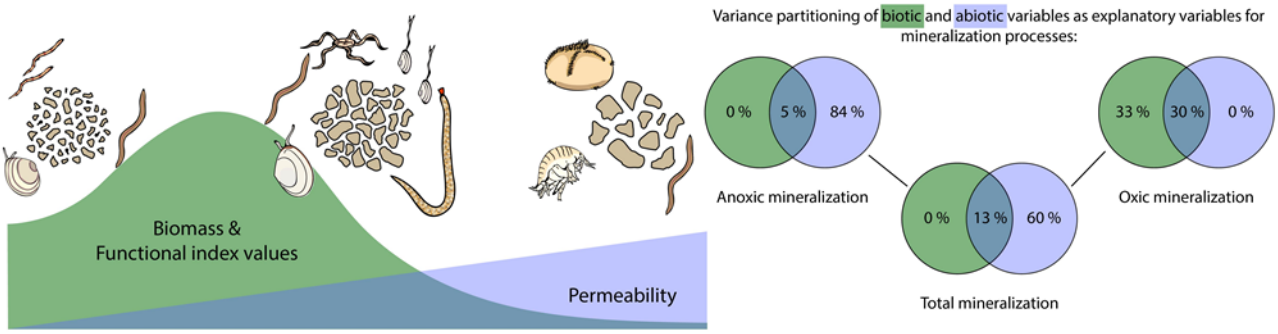


Figure 22 Graphical abstract of the FaCE-It results concerning the coupling of biotic and abiotic drivers of sediment biogeochemistry in the BPNS. From Toussaint et al. (2021).

Across the sampling gradient we found a confirmation of ecological theory: the distribution of macrofaunal biomass along a eutrophication gradient (Pearson and Rosenberg, 1978). Highest biomass was found in fine-sandy sediments of intermediate permeability (sts. 780,120), whereas lower biomasses were found in coarse sand and silty anoxic sediments of highest and lowest permeability respectively (resp. sts. BRN11, BBEG, 330, D6; and st. 130). The species contributing to the biomass corresponded mostly to those expected based on previous works in the area (Degraer et al., 2003, 2008; Van Hoey et al., 2004), with the *Abra alba* community represented in stations 120 and 780 (e.g. *Abra alba*, *Kurtiella bidentata*, *Fabulina fabula*, *Owenia fusiformis*). Station 120 was representative of the *Limecola balthica* community (Degraer et al., 2003), and species in permeable sediments (330, BBEG, D6) generally belonged to the *Nephtys cirrosa* community. We found an interesting difference between the two offshore wind stations (D6N, D6S). Whereas species richness and biomass in D6S was low, these were remarkably higher in D6N, where species such as *Echinocardium cordatum*, *Diogenes pugilator*, *Eteone* sp., and *Spiophanes bombyx* were found and also biomass density was much higher (23 ± 14 vs 3 ± 5. g WW m^{-2}, mean ± standard deviation). These stations also differed in grainsize characteristics, with D6S the coarser sediment of the two (median grain size D6N: 353 ± 12 μm, D6S: 524 ± 203 μm).

With these last observations we concluded that spatial heterogeneity is a feature of anthropogenic disturbance. Spatial differences observed around wind turbine D6 could be due to altered local hydrodynamics linked to the presence of the turbine (Rivier et al., 2016; Legrand et al., 2018), or dynamics of sandwaves in the area (Cheng et al., 2020) in combination with deposition of (pseudo)faeces by filter feeders living abundantly on the turbine (Coates et al., 2014; Baeye and Fettweis, 2015).

Changes in major mineralization pathways in the area were closely related to the sampled permeability gradient (Fig. 23). Total C mineralization was highest in least permeable samples (st. 130), and decreased towards the most permeable samples (st. D6N; Fig. 23 A, with total C mineralization varying from 0.21 ± 0.19 to 168 ± 74 mmol C m^{-2} d^{-1}). Similar trends were seen for sediment community oxygen consumption (SCOC, range: 13.7 ± 2.5 - 57 ± 15.1 mmol O_2 m^{-2} d^{-1}; Fig. 23 B), oxic mineralization (range: 0.1 ± 0.1 – 43.1 ± 19.3 mmol C m^{-2} d^{-1}; Fig. 22 C), and anoxic mineralization (0.1 ± 0.1 - 187.8 ± 23.8 mmol C m^{-2} d^{-1} Fig. 23 D). For denitrification, nor the other nitrogen cycling processes nitrification and DNRA, this link with permeability was present (Fig. 23 E). In the predictive models (ANNEX 1), the link to permeability was confirmed as this variable was included as a significant explanatory variable in the models for SCOC, oxic mineralization, anoxic mineralization, and N mineralization.

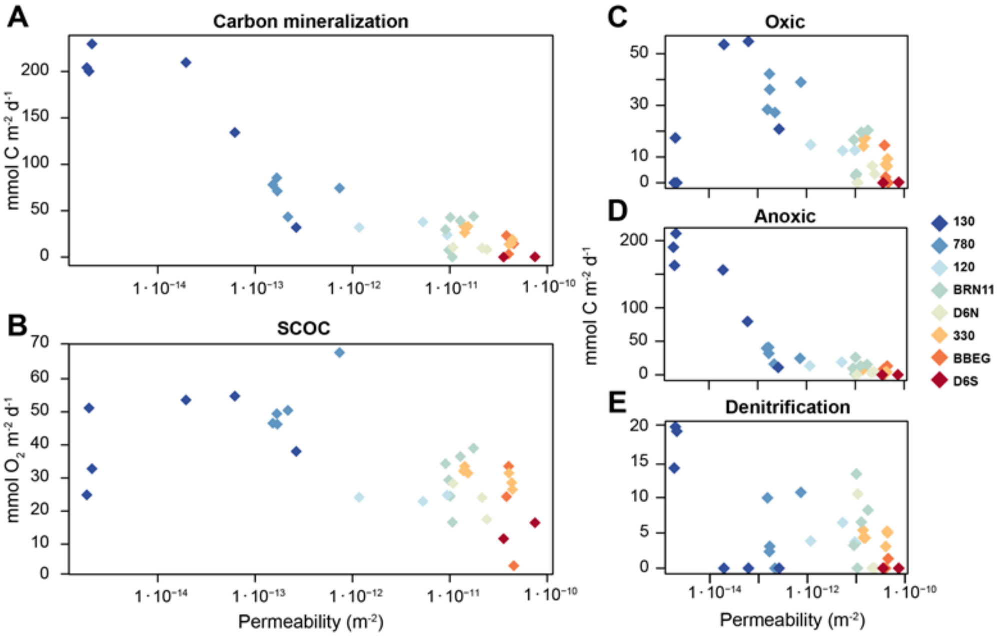


Figure 23 Carbon mineralization processes: (A) Total organic carbon mineralization (mmol C m^{-2} d^{-1}); (B) Sediment community oxygen consumption (mmol O_2 m^{-2} d^{-1}); (C)– (E) Rate of carbon mineralization allocated to oxic or anoxic mineralization, or denitrification. From Toussaint et al. (2021).

Additionally, our results add to the growing understanding of the importance of DNRA (and its drivers) in the nitrogen cycle of coastal sediments. DNRA was previously thought to strictly occur in anoxic sediments, but recent evidence has shown that bacteria and eukaryotes capable of DNRA may thrive near the oxic-anoxic interface within the sediment (Kamp et al., 2011, 2015; Behrendt et al., 2013; Hellemann et al., 2020). By including this process in the mass balance model, we were able to observe a shift from denitrification to DNRA as the main nitrate (NO_3^-) consuming process towards more (bio-)irrigated sediments. We hypothesized that DNRA increases in importance relative to denitrification when high nitrate concentrations are available (e.g. by ventilation activity, which favours both DNRA and denitrification), but where concentrations of organic matter to mineralize are lower (disfavouring denitrification, since this is a strictly heterotrophic process). Since the mass balance approach has limitations, we recommend adding dinitrogen (N_2) measurements to the methodology in order to better constrain N_2 fixation (and production) processes in the mass balance model.

RECOMMENDATION: This study clearly showed the importance of permeability when investigating biogeochemical processes in coastal sediments. Given its importance and the fact that permeability is cheap to measure, we recommend to consider it a standard variable to measure in biogeochemical studies. In addition, because of its tight link with ecosystem functioning (biogeochemical cycling), we recommend to explore permeability as an indicator for MSFD Descriptor 6. A second recommendation is to further investigate the 'stability' of trait scores across different environmental conditions. We further recommend to use the statistical models resulting from FaCE-It as tools to make decisions on the level of change that can be allowed by human activities.

4.4 Functional indicators to assess changes in benthic ecosystem functioning

FaCE-It has contributed considerably to a better understanding of the functioning of the marine ecosystem (biogeochemical, food web) under changing natural and anthropogenic pressures. To evaluate such changes in functioning within a policy context, indicator approaches are generally used. Indicators are measuring tools, which consist of an algorithm combining measured variables. This results in an assessment score, which is scaled along a yardstick to evaluate the direction of change (bad or good). In FaCE-It, we focused on exploring functional indicator approaches based on already existing benthic impact datasets of various human activities, coupled with the data obtained during FaCE-It. Therefore, monitoring data from dredge disposal, sand extraction, fishery activities and offshore wind energy exploitation within the BPNS were analysed. Regarding functional indicator approaches, we focused on (1) biological trait-based approaches, assessing changes in functional diversity (e.g bioturbation/bio-irrigation potential indicator, functional diversity indicators) and shifts in functional groups caused by human pressures (section 4.4.1), and (2) sediment profile imaging based indicators, reflecting on the physico-chemical state of the sediment (section 4.4.2).

4.4.1 Biological trait-based approaches

The use of biological traits provides a framework to gain insights in the relative presence of certain functional properties related to life history, behavioral or morphological characteristics of benthic communities (Bremner et al., 2003; Bremner, 2008; Beauchard et al., 2017). Such trait-based analysis can also be used to identify impact-driven alterations to ecological functioning (Bremner et al., 2003; Tillin et al., 2006; Gogina et al., 2014; Shojaei et al., 2015; Villnas et al., 2018). Furthermore, functional indices can be used to facilitate comparison and evaluate ecosystem functioning across environmental gradients. Therefore, we performed a detailed description of the species-specific biological traits of the soft-sediment communities within the BPNS that can support ecosystem-based management (Breine et al., 2018). We investigated the performance of the bioturbation and bio-irrigation indicator and their linkage with the biogeochemical measurements (Toussaint et al. 2021). We further explored the utility of functional diversity indices (section 1.1.3) and a multivariate visual tool to judge on their applicability in assessing changes in benthic functioning for environmental impact assessments (EIA) (Festjens et al., in prep.).

Trait modality baselines

Breine et al. (2018) present a revision of the structural characteristics of the soft-bottom benthic communities in the BPNS and provides, for the first time, an insight into their functional characteristics. Compared to previous research (Van Hoey et al., 2004 ; Degraer et al., 2008), the dataset was expanded considerably on a spatial and temporal scale (1994-2012). Five communities were identified, each with their own structural characteristics, indicator species, sediment properties and spatial distribution. The offshore area was dominated by the *Hesionura elongata* community (formerly *Ophelia borealis*), and although previously characterised by low diversity and abundance, this study shows that this community has the second highest species diversity and moderate densities. Also, a new community, the *Magelona-Ensis leei* community, was found in very shallow water and was characterized by low diversity and the dominance of the non-indigenous species *E. leei*. The structural characteristics and spatial distribution of the previously described *Limecola balthica*, *Abra alba* and *Nephtys cirrosa* community remained largely unchanged. Though structurally distinct, the communities overlap in some of their functional attributes. Analysis of biological traits revealed that in the coarser permeable sands, both the *N. cirrosa* and *H. elongata* community harbor more free-living mobile individuals, causing diffusive mixing (Fig. 24).

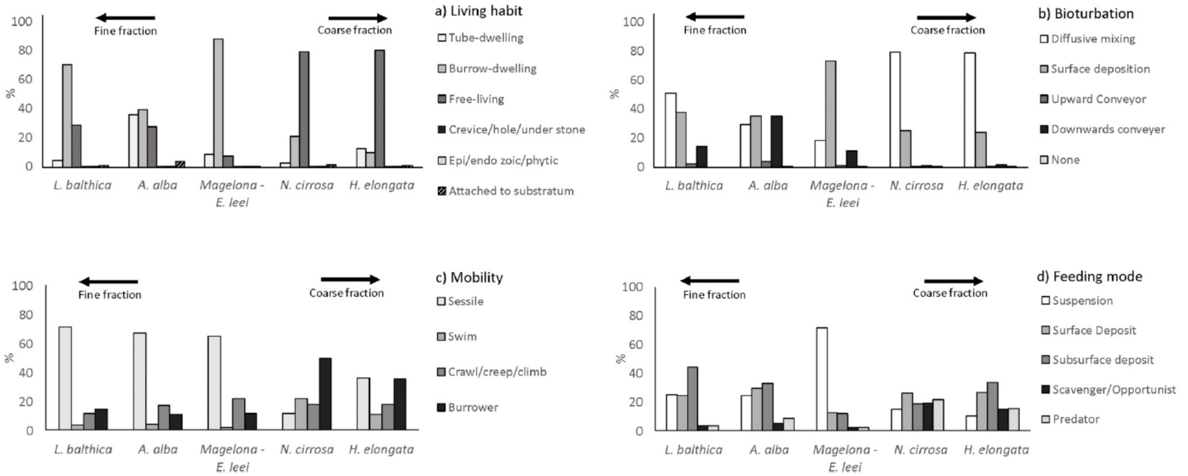


Figure 9. Bar plots of the behavioral traits (living habit, bioturbation, mobility, feeding mode) showing -per community-the importance of each modality within the trait, based on species abundances. Arrows are indicative of general median grain-size characteristics.

The finer sand (*A. alba, Magelona- Ensis leei*) and muddy (*L. balthica*) communities harbour more sessile, tube building and burrow dwelling species. Apart from diffusive mixing, sediment reworking in these finer sands occurs through surface deposition and downward convection. In combination with higher densities and biomass, this results in a higher bioturbation potential (BPc) for the *A. alba* and *Magelona-Ensis leei* community. Finally, measures of functional diversity and redundancy were calculated, and were also highest in the *Abra alba* and *Hesionura elongata* community, thus demonstrating the importance of both communities for conservation purposes from a structural as well as a functional perspective. The analyses revealed that even in well-studied areas, new data provide new insights, with important consequences for management. As such, the compiled data and findings can be used for indicator testing (see next sections), the optimization of monitoring designs in the study area, and the delineation of baseline values (thresholds) (E.g. for BPc, see section 2) to determine the quality status of the benthic ecosystem in the framework of international and national management guidelines such as the European MSFD, the Habitats Directive and the Marine Spatial Planning Directive (Shephard et al., 2015; Van Hoey et al., 2013; Cooper and Barry, 2017).

Bioturbation / bio-irrigation potential indicator

BOX 1. Bioturbation / bio-irrigation indicator

The community bioturbation potential (BPc) is a proxy for the biogenic modification capacity of a benthic community through particle reworking (Solan et al., 2004; Queiros et al., 2013). For each species i in a sample, the abundance and biomass (resp. Ai and Bi) were scaled, and multiplied with either its mobility score Mi and reworking score Ri (for the BPc; Queirós et al., 2013), or its burrow type BTi, feeding type FTi, and injection pocket depth IDi (for IPc; Wrede et al., 2018).

$$BP_C = \sum_{i=1}^{n} \left(\frac{B_i}{A_i}\right)^{0.5} \cdot A_i \cdot M_i \cdot R_i$$

$$IP_C = \sum_{i=1}^{n} \left(\frac{B_i}{A_i}\right)^{0.75} \cdot A_i \cdot BT_i \cdot FT_i \cdot ID_i$$

In our study, we used the Bioturbation Potential (BPc) and the Irrigation Potential (IPc) (see Box 1) of the community as proxies for two types of faunal activity, respectively particle reworking and burrow ventilation. The indices were tested as possible predictors of individual biogeochemical processes: oxic mineralization, nitrogen mineralization and anoxic mineralization (Toussaint et al., 2021). Our hypothesis was that oxic mineralization processes would be more associated with biological features of the environment, most specifically with aspects of burrow ventilation, whereas anoxic mineralization processes would be stronger related to physical aspects of the sediment. This hypothesis was investigated based on the FaCE-It samples, collected along the sediment gradient.

Table VIII Variables used for describing the biological functioning in the different stations, for successive years. Values are expressed as average ± sd, for three replicates (n = 3). (from Toussaint et al., 2021).

Year	Station	Irrigation rate $L\ m^{-2}\ d^{-1}$	IP_c $gWW^{0.75}\ m^{-2}$	BP_c $gWW^{0.5}\ m^{-2}$	Abundance $Ind.\ m^{-2}$	Biomass $gWW\ m^{-2}$
2016	130	3.31 ± 2.79	125 ± 130	212 ± 186	297 ± 194	10 ± 11
	780	1.17 ± 0.09	6014 ± 1981	4017 ± 1190	1443 ± 482	1035 ± 130
	BRN11	24.92	112 ± 81	269 ± 78	270 ± 178	6 ± 2
	330	2.12 ± 0.8	282 ± 132	390 ± 110	788 ± 489	9 ± 7
	BBEG	6.7 ± 3.21	5 ± 6	21 ± 25	24 ± 20	1 ± 1
2017	130	15.64 ± 2.31	2341 ± 1845	2834 ± 1678	2716 ± 2617	387 ± 267
	780	6.42 ± 0.82	4106 ± 2381	6299 ± 2891	6196 ± 3184	772 ± 440
	120	0.62 ± 0.65	1340 ± 1279	2021 ± 1192	4923 ± 701	451 ± 593
	BRN11	7.76 ± 0.33	2760 ± 3980	1294 ± 689	811 ± 106	157 ± 207
	D6N	4.98 ± 2.08	715 ± 547	831 ± 205	1317 ± 283	23 ± 14
	330	10.91 ± 8.93	24 ± 12	163 ± 119	200 ± 81	2 ± 1
	D6S	3.95 ± 1.04	40 ± 55	175 ± 215	435 ± 181	3 ± 5

We found high values for both BPc and IPc in fine sandy – muddy sediments (station 130, 780, 120), and lowest values in the most permeable, coarse sediments (station 330, BBEG, D6N, D6S), except BRN11 in 2017 (**Error! Reference source not found.**). While the higher values of the functional indices in finer sediment are partly attributed to higher faunal biomass, the functional traits of only few species in permeable sediments (e.g. the sea urchins *Echinocardium* sp. and the polychaete *Nephtys* sp.,) attain high values which contribute markedly to the index scores.

Toussaint et al. (2021) showed that BPc explained some of the variance in oxic mineralization, SCOC, total N mineralization and nitrification. The particle reworking activity and fragmentation and dispersal of organic matter by bioturbators makes it more accessible for micro-organisms and results in increased mineralization rates, both oxic and anoxic. Therefore, BPc can be considered as good proxy for evaluating these processes.

In contrast, the IPc index was not selected as an explanatory variable for the burrow ventilation activity (measured ventilation rate) (Toussant et al., 2021). This seemingly contradicts previous findings that stressed the importance of burrow ventilation (Braeckman et al., 2010; Wrede et al., 2018). There are two possible reasons for this. First, the IPc index might not be an accurate estimation of the burrow ventilation rate, as it seems more strongly correlating with the burrow ventilation depth (De Borger et al., 2020). Second, the index reflects the potential to express an activity, but it does not include the temporal dynamics of the activity. Temporal variation is strongly pronounced for ventilation as animals do not ventilate constantly (Volkenborn et al., 2016). Therefore, we believe that IPc could be more useful in predicting biogeochemical processes if a metric for temporal dynamics were included in the index.

Functional trait diversity indicators

According to Villegér et al. (2008), functional diversity cannot be summarized in a single number as it has to include components of richness, evenness and divergence which take into account the trait values and their abundance. In Festjens et al. (in prep), we therefore investigated the entire set of functional diversity indices to evaluate changes in the benthic ecosystem due to anthropogenic activities. We tested 5 indices, which were functional richness (FRic), functional evenness (FEve) and functional divergence (FDiv) (Villéger et al. 2008), functional dispersion (FDis; Laliberté and Legendre, 2010) and Rao’s quadratic entropy (RaoQ; Botta-Dukkat, 2005)) (definitions are provided in BOX 2). To test for an effect of different human activities on the functional diversity indices, linear mixed-effect models (lmer, from the ‘lme4’ package in R, Bates et al., 2015) were used.

BOX 2. Functional diversity indices – definitions and conceptual diagrams

FRic: represents the total amount of functional space filled by the community (minimum convex hull volume) and has no upper limit.

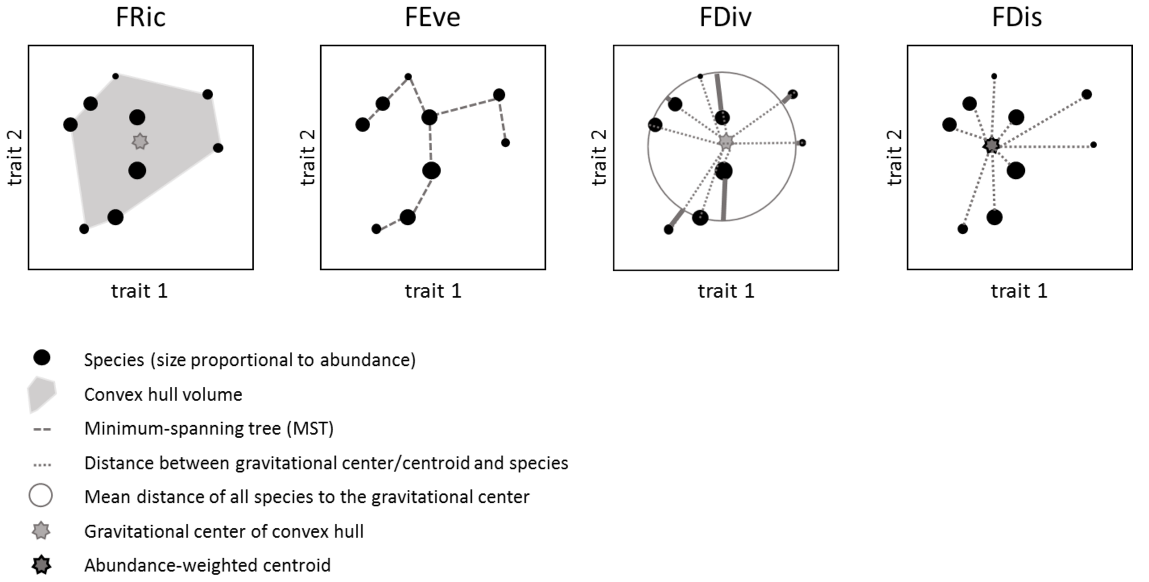
FEve: represents the evenness of species abundance distributions in functional trait space (average branch length of minimum-spanning tree, weighted by relative abundance), and ranges between 0-1.

FDiv: measures distribution within trait space (individual species’ deviation from the mean distance of all species to the gravitational center of the convex hull) and ranges between 0-1.

FDis: measures distribution within trait space (mean distance of each species to the abundance-weighted centroid of the community and has no upper limit.

RaoQ: generalized form of the Simpson index, which measures the amount of trait dissimilarity between two random individuals in the community (conceptually similar to FDis but more widely used in marine impact assessment research), has no upper limit.

Each of these indices is an independent measure of functional trait space, and the way species are dispersed within this trait space. Apart from FRic, all indices take into account species abundance in the quantification of functional diversity, i.e. by weighting the pair-wise species dissimilarity in the trait space by the product of relative abundances of the species. This gives a differential weight to the traits of more dominant and/or less abundant species, as species abundance affects various components of ecosystem functioning (Petchey & Gaston 2006).



Graphs designed after Villéger et al. (2008) and Laliberté et al. (2010).

Whereas sand extraction and dredge disposal resulted in a similar structural change in the benthic assemblage according to De Backer et al. (2014b), there were some differences when looking at the functional diversity indices. A larger pressure of sand extraction leads to higher values in nearly all functional indices and locations (except at Hinderbanken, where functional evenness was significantly lower at higher pressure), whereas a higher amount of dredge dumping leads to significantly lower index values for functional richness, functional dispersion and Rao's quadratic entropy (Table IX). Functional evenness however is also higher at higher dredging pressure, which means that there is a higher redundancy of less, more similar functional modalities present in the dumping areas. As the dredge disposal occurs in different habitat types (*A. alba*, *L. balthica* and *N. cirrosa*), we observed also a slightly different response by the functional diversity indices. For example, the functional richness was significantly lower under high dumping intensity in the *A. alba* and *L. balthica* habitat, whereas not significant within the *N. cirrosa* habitat.

Table IX Summary table representing the observed changes in the three cases studies for the univariate (functional diversity indices) and multivariate (FCA analyses) analyses. '+' indicates an increase, '-' indicates a decrease and '0' indicates no significant difference.

		FRic	FEve	FDiv	FDis	RaoQ	FCA (separation)	Effect/response traits determine difference between none-high impact	indicative trait modalities
Dredge disposal	*A. alba*	-	+	0	-	-	yes	response	l<1, sr10, edAsex, edBen, lhFree
	L. balthica	-	+	0	-	-	no	effect	/
	N. cirrosa	0	0	0	-	-	yes	effect	btUp, btDown, lhEpi, lhCrevice, sr201-500
Sand extraction	BR	0/+	+	0	+	+	yes	response	sr201-500, l>10, btDown, mExoskeleton, sp10, lhAttach
	HB	0/+	-	0	0	0	yes	response	l>10, mExoskeleton, edBrood, ldDir, mbSwim
	TB	0/+	0	0	0	0	no	response	/
OWF	TB	+	0	0	0	0	yes	equal	sr10, l < 1, edBrood, mExoskeleton, ldDirect, lhEpi, lhBurrow, spSurf
	BB	0	0	0	0	0	yes	equal	

As was the case in previous OWF research (Lefaible et al., 2018, Coates at al., 2013), no sign of impact could be found in the Belwind OWF, where construction started later than at the C-Power OWF, and which consists of monopile foundations instead of jacket foundations. At C-Power, functional richness is significantly higher in the impact area as a result of an influx of new species. De Backer et al. (2014b) suggested a shift occurred towards a dynamic, transitional assemblage due to attraction of species characteristic for muddy sands. This effect is probably restricted to very close distances (<50 m) from the turbine foundation (Coates et al., 2013; Colson et al., 2017; Reubens et al., 2016). Another possible reason for this higher functional richness is the impact of turbine-associated epifaunal communities on the very close proximity of those turbines, the so-called 'artificial reef-effect', which creates favorable conditions for a more diverse community (Degraer et al., 2020). The other functional diversity indices were not significantly different between impact and control in the Belwind and C-power area.

To conclude, the functional diversity indices were useful in detecting changes in the benthic ecosystem due to anthropogenic activities. Nevertheless, in our case, functional divergence (FDiv) seems not sensitive enough (no significant changes in any case), whereas functional richness (FRic) and functional eveness (Fev) are very relevant. FDis and RaoQ followed the same trends across all cases (using one of the two is fine), as they are mathematically related indices (FDis is based on the amount of trait similarity between species in a community, whereas RaoQ between individuals in a community).

Shifts in functional groups

To identify shifts in trait composition due to human activities, Fuzzy Correspondence analysis (FCA) was performed (R package Ade4, using the dudi.fca function (Dray & Dufour, 2007)). FCA creates ordination biplots, whereby points represent the abundance-weighted trait composition of stations, and the trait modalities that drive (effect traits) or explain (response traits) those patterns are also plotted. Points in closer proximity are indicative of stations with a functional similarity. The dredging pressure categories ('none', 'low', 'medium' and 'high'), or control/impact regimes for the windfarm case, were then superimposed on the reduced two-dimensional ordination output and the pairwise distances between the centroids were calculated and used as a proxy for the relative similarity between those groups (cfr. Bolam et al., 2016). These FCA plots were produced for each case and location, both using all the traits and for response and effect traits separately. In case the impact regimes are separated on the first or second axis without too much overlap, the arrangement of the traits on the that FCA axis is extrapolated and put on a gradient ranging from impact to control. This way traits associated with the impact stations could be identified.

The outcome for each case is summarized in Table IX and only the FCA for the wind farm case with all traits is visualized in Figure 25. We generally observed (except for the case of dredge disposal in *L. balthica* habitat and the case of sand extraction on the Thornton bank) a separation in trait composition between the control and high impact samples (highest pairwise distances). For the sand extraction case, we observe that the response trait types are mostly determining the difference between impact and control. In the dredge disposal case, these are the effect trait types for the *N. cirrosa* and *L. balthica* habitat. In the OWF case, we see equal contribution of response and effect traits in discriminating control and impact. There are no traits or trait-modalities that were consequently more associated with the impact samples for each case. The indicative traits for the impact samples are listed in Table IX. The trait modalities associated most with the higher dumping pressure in the *A. alba* habitat are a short lifespan (l<1) and small body size (sr10), asexual (edAsex) or benthic egg development (edBen) and a free-living lifestyle (lhFree). Those traits are indeed indicative of species occurring in disturbed environments. A contradictory observation is made for the high impact area at the Buiten ratel sand extraction area, where we observed that the community associated with the 'high' pressure consists of large (sr201-500), long-living (l>10) bioturbators (btDown) with an exoskeleton that live deeper within the sediment (sp10) or have an attached lifestyle (lhAttach). Those trait characteristics for species living in a high impact area is not entirely logic, but is related to the fact that the disturbance there has led to the occurrence of more coarse material (cobbles), attracting for example attached lifestyle species. In the OWF areas, the five modalities most associated with the impact stations are the same: small maximum body size (sr10) , short lifespan (l < 1), active brood care (edBrood), exoskeleton (mExoskeleton) and no larval stage (ldDirect) (Table IX; Figure 25).

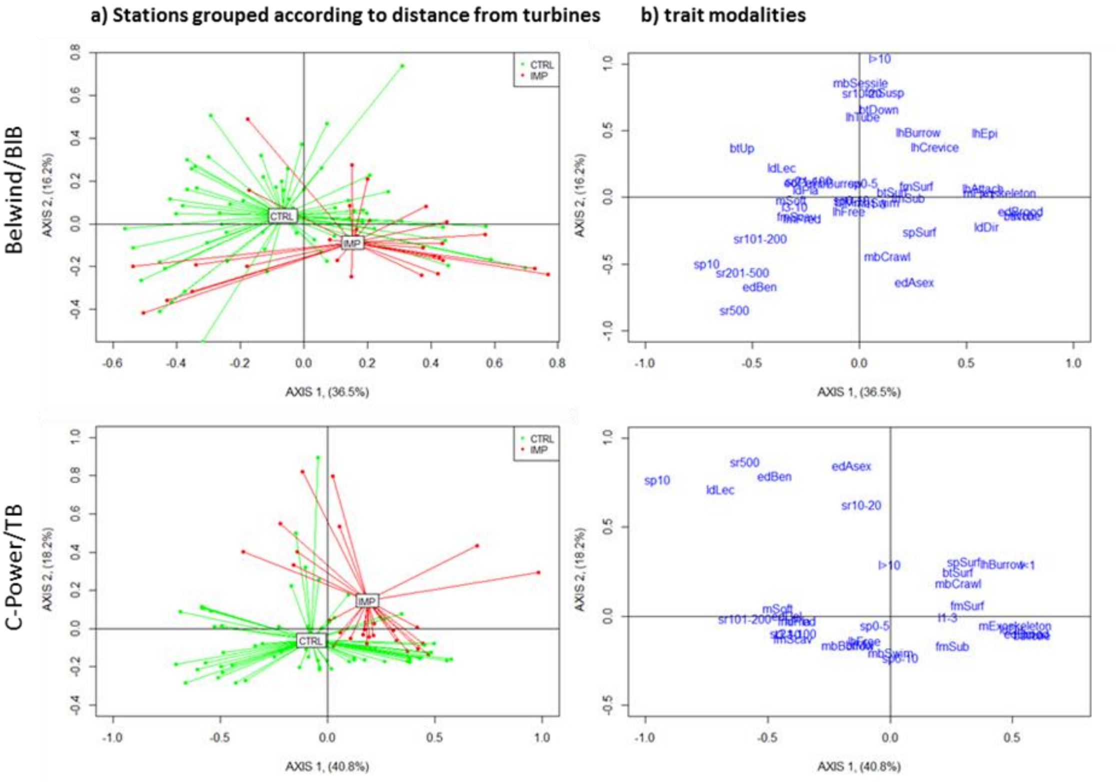


Figure 25 FCA ordination biplots for the offshore windfarm case with the plotted stations allocated to IMP or CTRL and with the trait modalities plotted on the same graph.

Additionally, epiphytic or burrow-dwelling living habit (lhepi, lhBurrow) and sediment position at the surface (spSurf) also scored very high in both OWFs. This confirms the shift to a slightly disturbed community in the vicinity of the turbines, and is in agreement with the sediment fining within those areas (Lefaible et al. 2018).

To conclude, functional correspondence analyses are a very useful tool to visualize shifts in traits composition between control and different impact categories. It delivers descriptive insights in which functional traits are responding to a certain impact. However, a quantitative parameter suited for indicator-based assessments cannot be derived. Our analyses further reveal that there are no traits or trait-modalities that were consequently more associated certain impacts..

Sediment Profile Imaging

Sediment Profile Imaging (SPI) allows operators and analysts to conduct a quick examination of benthic habitat quality. The SPI consists of a wedge-shaped prism with an inverted periscope that penetrates the seafloor to reflect the sediment-water interface (SWI). This results in an image of the first ~20 cm of the sediment (maximum), which can be analyzed in detail for biological, physical and chemical parameters (Table 2). The SPI is a well-known tool that is undergoing a recent revival. In this study (Van Hoey et al., in prep.), we tested the performance and applicability of the SPI and the derived parameters for EIA in a dynamic coastal environment (shallow part Southern Bight of the North Sea) affected by sand extraction, trawl fisheries, dredge disposal and wind farms. All aforementioned activities can cause specific changes to the sediment-water interface and benthic community structure. The optical SPI technique can provide a suitable tool to detect and assess these alterations.

The utility of the SPI to analyse sediment structures and to assess disturbances is proven for less dynamic, muddy habitats (Rhoads and Germano, 1986; Rumohr and Schomann, 1992; Nilsson and Rosenberg, 2000). The naturally, more stable sediment layering patterns in such areas, allows straightforward detection of disturbances with optical techniques such as SPI. In shallow, dynamic coastal areas such as our sampling sites, the continuous sediment reworking by physical forces can hamper the detection of human disturbance.

Table X Overview of the investigated SPI characteristics

Parameter	**Description**
Sediment Type	Determination of the sediment class (muddy sand, very fine sand, fine sand, medium sand, coarse sand and very coarse sand). This was visually determined by comparing it with reference pictures for each sediment class.
Mud clasts	This parameter register the presence or absence of disposal material in the form of clumps or layered sediment (mud-sand).
Surface Relief & bedforms	This is a measure of bed roughness, which is visually determined by classifying the pictures in two classes (even or uneven surface). The uneven surface can be driven by physical process (then the presence of bedforms is noted) or biological (than the presence of tubes, mounds, feeding pits, epifauna is noted) or a combination.
Prism Penetration Depth	Gives an estimate of sediment compaction, using the average penetration depth (cm), measured via the SpiArcBase software, from the surface-water interface. The further the prism entered into the sediment the softer the sediments, and likely the higher the water content.
Sediment apparent Redox Potential Discontinuity depth (aRPD)	It gives an estimate of the depth to which sediments appear to be oxidized. This depth is determined where the sediment colour changed from brown-light grey (oxidized) to black (reduce sediments). The aRPD depth parameter is an essential estimator of the benthic habitat condition, with higher quality when the reduced layer is deeper (Nilsson & Rosenberg, 1997; Rhoads & Germano, 1987).
% of anoxic surface area	The surface area of the dark grey to black parts of the sediment are calculated. This is quantified as percentage of the total sediment surface on the SPI picture.
Surface faunal features	The presence of tubes (in three classes: few, some, a lot), Epifauna (real count, species level), Faecal pellets (present-absent), were registered.
Subsurface faunal features	The presence of infaunal organisms (real count), burrows (more or less than 3) and oxic voids (present-absent).
Indicators	The Organism Sediment Index (OSI) (Rhoads and Germano 1987) assign a numerical score to several evaluation aspects, as the depth of the aRPD (6 classes), the successional stage (predefined in 3 classes), the presence of methane bubbles (not present in our study area) or gas voids in the sediment, and visual signs of low dissolved oxygen concentrations (anaerobic sediment at the interface). The benthic habitat quality Index (BHQ), primarily developed for Scandinavian fjords (Nilsson and Rosenberg 1997), was adapted for this study (Table 2). The BHQ is calculated from the sum of scores of three basic groups of data: Surface structures (max score of 5), subsurface structures (max score of 4) and aRPD (score between 0 – 5). In the adapted BHQ, the surface structure group of data is based on the presence of faecal pellets, epifauna, or the tubeworm presence, with three classes according to their tube density (a few [1-5 tube], some [5-20 tubes], numerous [>20 tubes]). The subsurface structure group is evaluated based on the presence of infauna (score 1) and the quantification of burrows, within three classes (a few, some, numerous) according to their

	amount. The aRPD score system is based on the anoxic area percentage divided into 5 classes.

Our results confirm that the SPI is indeed a good tool for a rapid assessment of the seafloor status, especially in habitats with abundant fauna (e.g. fishery case) or for activities directly disturbing the sediment (e.g. dredge disposal) (Table XI). For sandy, permeable sediments and activities taking place therein, the applicability is lower, as less SPI derived parameters seem relevant to use for EIA assessment.

Table XI Overview of the relevance of the SPI derived parameters for EIA purpose. Green means relevant, red means not-relevant, whereas grey means informative

	Fishery	Dredge disposal	Sand extraction	Wind farms
Habitat	Fine muddy sand	Fine muddy sand to medium sand	Medium to coarse sand	Medium sand
Sediment class	▨	🟩	🟩	▨
Bedforms	▨	▨	▨	▨
Surface relief	🟥	🟥	🟥	▨
Mud clasts	🟥	🟩	🟥	🟥
Prism penetration	🟩	🟩	🟩	▨
a-RPD	▨	▨	🟥	🟥
% anoxic sediment surface	🟩	🟩	🟩	🟩
Surface fauna	🟩	🟩	🟩	🟩
Sub-surface fauna	🟩	🟩	🟥	🟥
OSI indicator	🟥	🟥	🟥	🟥
BHQ indicator	▨	▨	▨	▨
Adapted BHQ indicator	🟩	🟩	🟩	🟩

In general, the SPI delivers valuable information in characterizing the study area with regards to sediment type, which sometimes clearly changed due to the activity. Bedforms and surface relief are not easy to use in dynamic environments as their occurrence is mainly driven by the natural hydrodynamics, and therefore the impact of a human activity on it is not detectable. The mud clasts, resulting from dredge disposal are easy to detect and assess as SPI parameter. This pressure is not relevant for the other cases, so this parameter is judged as not-relevant. Prism-penetration, a proxy for sediment compaction is a relevant parameter, which can be influenced by the human activities as fishery, dredge disposal and sand extraction, but not in the OWF case. Due to the heterogeneity of the sediment, the a-RPD depth cannot be measured in a straightforward way, therefore we calculated the % of anoxic sediment surface within the SPI image. This parameter was measurable in all cases and is relevant to evaluate differences between control and impact data (Fig 26).

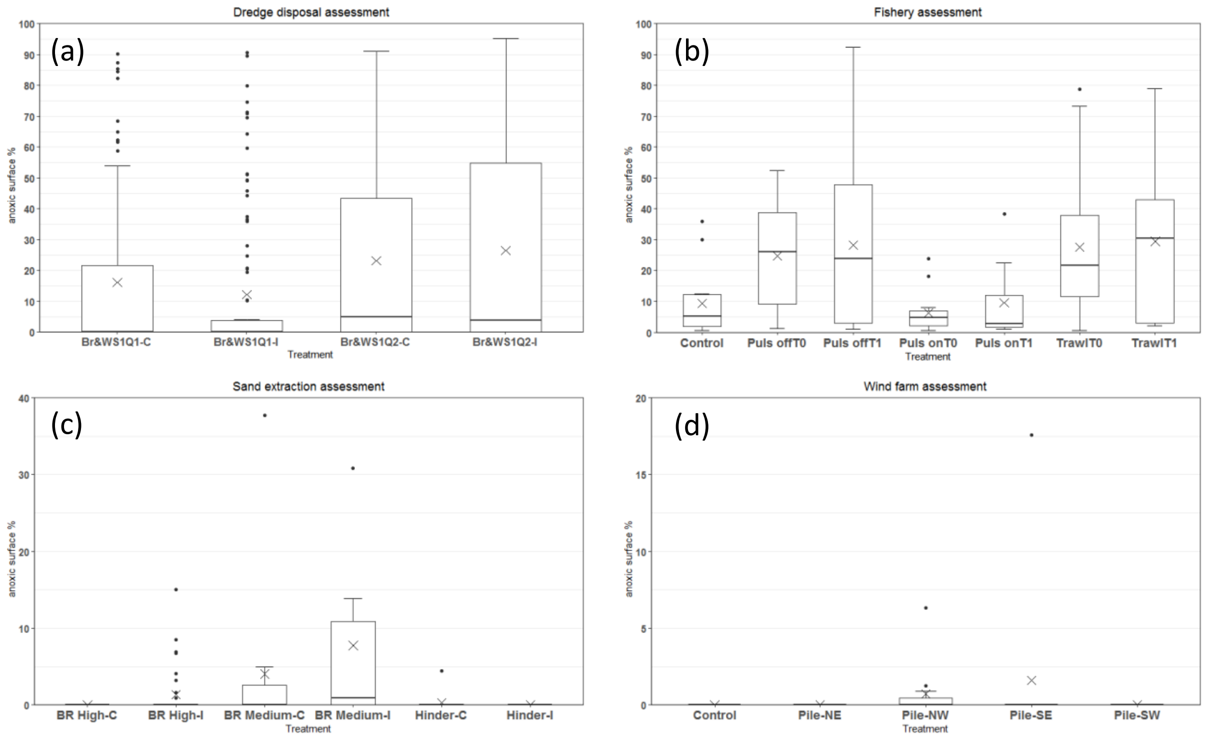


Figure 26 The anoxic sediment surface percentage (%) for the different case studies (a) control-impact evaluation dredge disposal site Br&WS1; (b) fishery assessment; (c) sand extraction assessment and (d) wind farm assessment. The black dots represent outliers and "x" the mean and the line the median.

The work of FaCE-It confirms the applicability of SPI derived data for EIA assessment in shallow, dynamic coastal areas. Independently, the SPI-derived characteristics provide important information, while in combination, they provide an integrative view for the characterization of seafloor integrity complimentary to the information derived from grabbing, coring or other traditional sediment extraction devices. SPI easily enables scientists to observe certain seabed characteristics and processes in situ. Therefore, the use of the image (SPI) in combination with infaunal sampling techniques forms the perfect tool for a proper EIA or scientific research (Germano et al., 2011), even in dynamic coastal systems, as illustrated by this study. For example, in the OWF case, very small anoxic surface areas (<0.6%) were detected in 5 out of 39 samples (anoxic sediment surface % lower than 0.6). One picture from the SE direction revealed a surprisingly high (17.6%) percentage of anoxic sediment. Lower values (0.3 to 6.3%) were observed at the NW side of the turbine. At the control sites, both in NE and SW direction, no anoxic sediment was detected. Differences in occurrence of surface fauna were detectable in each case, whereas sub-surface fauna (burrows, infauna) was not really detectable in permeable sediments. From the SPI derived indicators (OSI, BHQ), we can conclude that the OSI indicator is not applicable and that the BHQ indicator parameters need to be adapted slightly to make it relevant for assessments in shallow dynamic coastal areas. An adapted BHQ indicator is in development.

From the SPI derived indicators (OSI, BHQ), we can conclude that the OSI indicator is not applicable and that the BHQ indicator parameters need to be adapted slightly to make it relevant for assessments in shallow dynamic coastal areas. An adapted BHQ indicator is developed and tested (Table X).

4.4.2 Functional indicator use for policy

This work is of direct relevance for the Belgian MSFD implementation process, where the ambition is to include indicators that assess the functional status of the ecosystem. For the moment, the BPc index is already part of this MSFD assessment framework, and functions as well as an conservation objective for the assessment under the Habitats Directive. Here, we outline how additional FaCE-It knowledge serves this BPc indicator definition for both directives.

Under the MSFD, the indicator is defined as: "the bioturbation potential (BPc), an indicator for the functioning of the benthic ecosystem, maintains a minimum value of 0.60 (as determined via BEQI procedure) for the *Abra alba* habitat type in the fall".

For the Habitats Directive, this was reformulated as: "the bioturbation potential (BPc), an indicator for the functioning of the benthic ecosystem, maintains a minimum value of 331 for the *Abra alba* habitat type in the fall".

Both BPc indicator definitions contain a threshold value for obtaining a good status. This value is determined based on a pre-defined reference dataset (Van Hoey et al., 2014; Breine et al., 2018), reflecting the known benthic characteristics of the benthic communities in the BPNS. The baseline study (Breine et al., 2018; section 1.1.1), shows us that the BPc (average of 430.6 ± 23.3 [St error]) is highest for the *Abra alba* habitat, which is a reason to assess this indicator only within the *Abra alba* habitat. For obtaining a good environmental status within the MSFD assessment, the BPc value of a set of samples has to fall within the 2.5 and 97.5 statistical percentile as defined through a bootstrapping procedure within the reference dataset for the same amount of samples. For example, for a sampling effort of 50 samples within the *Abra alba* habitat, we expected that the BPc falls within the range of 322-551.5 to obtain a good environmental status outcome. Values outside this range, indicate that the habitat is not in good environmental status and it is therefore considered as impacted. For the Habitats Directive, the BPc threshold value is fixed, by choosing the value 331 as threshold for obtaining the conservation goal, irrespective of sampling effort. This corresponds with the 25 statistical percentile (high/good boundary in the MSFD assessment) as defined based on a bootstrapping procedure within the reference dataset for a sampling effort of 10 samples.

To conclude, the FaCE-It work shows that incorporation of functional characterizations (biological traits), alongside structural aspects (e.g. biomass, species diversity, abundance) in evaluating changes in the marine environment is achievable. On the one hand, biological trait-based approaches (BPc, IPc, functional diversity indices, FCA) are in some cases a relatively good proxy for evaluating changes in ecosystem functions and processes. Therefore, BPc is already included in the assessment procedure for the MSFD and Habitats Directive. IPc seems currently less applicable, and the correlation with measured irrigation rates is not clear. The functional diversity indices, functional richness, functional evenness and RaoQ are very relevant in detecting changes due to anthropogenic activities. Nevertheless, the complex patterns in biological trait shifts indicate that the changes are case- (pressure of human activity) and habitat dependent in our study area. This habitat dependency of trait expressions is also put forward based on the biogeochemical measurements themselves. From the SPI-derived indicators (OSI, BHQ), we can conclude that the OSI indicator is not applicable and that an adapted version of the BHQ indicator (as developed in Face-it) is needed for assessments in shallow dynamic coastal areas.

RECOMMMENDATION:
Functional indicators can be used in environmental impact assessment of human activities and in the EU environmental legislation (MSFD, Habitat Directive). Nevertheless, some steps need to be taken to improve the applicability of those functional indicators.
First, future research is needed to further unravel which response or effect traits are sensitive to certain human induced pressures as the indicator responses are depending on the type of pressure and the habitat characteristics.
Second, some indices, such as the IPc, are currently considered less applicable and need to be refined. We recommend to include a metric for temporal dynamics in calculation of the index.
Third, for several indicator approaches (e.g. functional diversity indices), threshold values and an accompanying measuring rule need to be defined. Such thresholds can be derived based on exploring the point of change in the pressure-response relationships (cf. results above), Another way is to extract baseline values from the MSFD reference dataset of the BPNS (cf. BEQI procedure).

4.5 Spatial upscaling

4.5.1 Changes in TOC and inorganic fine materials deposition flux: main patterns

Figure 27 features relative changes in the annual TOC deposition flux of the D-210 scenario compared to the R-0 scenario. It shows that the flux generally increases within 5 km to the closest turbine, with the highest increase (>15%) observed next to the turbine. Beyond 5 km from the closest turbine, the flux slightly (<2%) decreases, with the largest depletion observed at the distance of 9–15 km. Concerning the distribution of flux changes for different depths, all the depths within the range between 15 - 32 km experience a flux increase (up to 6% for the depth of 25 m), while shallower and deeper places experience flux decreases. The mismatch between the depth with the largest flux increase (observed at 25 m depth) and the highest number of built turbines (allocated at 29 m depth) shows that the OWF impact is larger on the shallower depths, because the sinking TOC has less time to degrade before reaching the sediment bed, compared to deeper places.

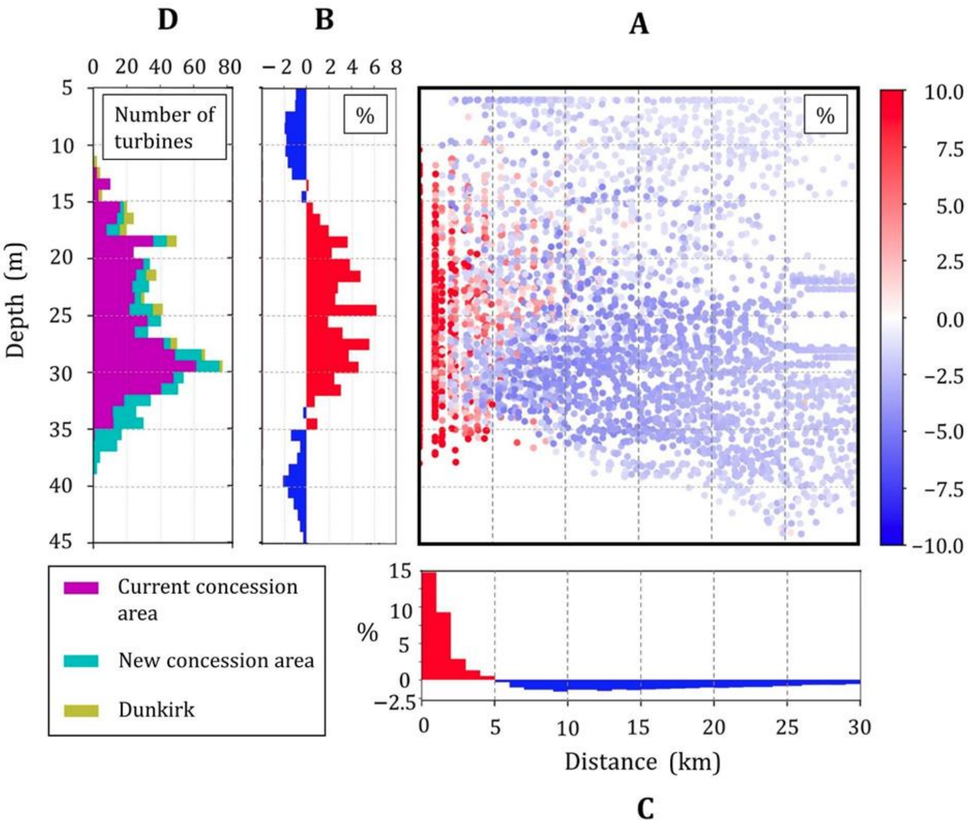


Figure 10 Annual TOC deposition flux change between the scenarios D-210 and R-0. (A) Scatter plot of these changes for each grid cell in 30 km proximity to the closest turbine, as a function of distance (to that turbine) and depth. (B) Area-weighted changes as a function of distance (to that turbine) and depth. (B) Area-weighted changes as a function of the water depth. (C) Area-weighted changes as a function of the proximity to the closest monopole. (D) Cumulative bar plot of number of turbines for each concession area (CCA, NCA, Dunkirk). (From Ivanov et al. 2021)

Figure 28 shows the change in the annual TOC fluxes for different scenarios compared to the R-0 scenario. In the N-250 scenario (subplots A and B), the positive changes in the TOC flux propagate beyond the OWF perimeters. The changes align with the direction of the tidal ellipses (A) and follow the residual currents (B). Because the major axis of the tidal ellipse stretches in SW--NE direction, the positive changes (up to 10%) can stretch for several km along this axis, however in the direction of the minor tidal axis (NW--SE), the positive changes propagate beyond the perimeter only if the local residual current aligns with it. The local residual counter-current associated with the combined Scheldt-Rhine brackish river plume (south to the gyre #1) cuts off the positive changes in the TOC flux, resulting in a sharp gradient in TOC flux change (from ~+40% down to ~-5% within 2-3 km). In several places of the domain, residual currents form enclosed contours, so-called "retention gyres", associated with the increased amounts of TOC deposition compared to the surroundings. The largest increase in the TOC deposition flux (> 50%) is observed inside the perimeters of the oldest OWFs in the CCA (e.g. Belwind), where the monopile density is high (exceeds 3 monopiles km^{-2}). The positive TOC flux changes between the CCA and the NCA are separated by a narrow zone (~2 km wide) of TOC flux decrease, meaning that the effects of the CCA and the NCA do not combine.

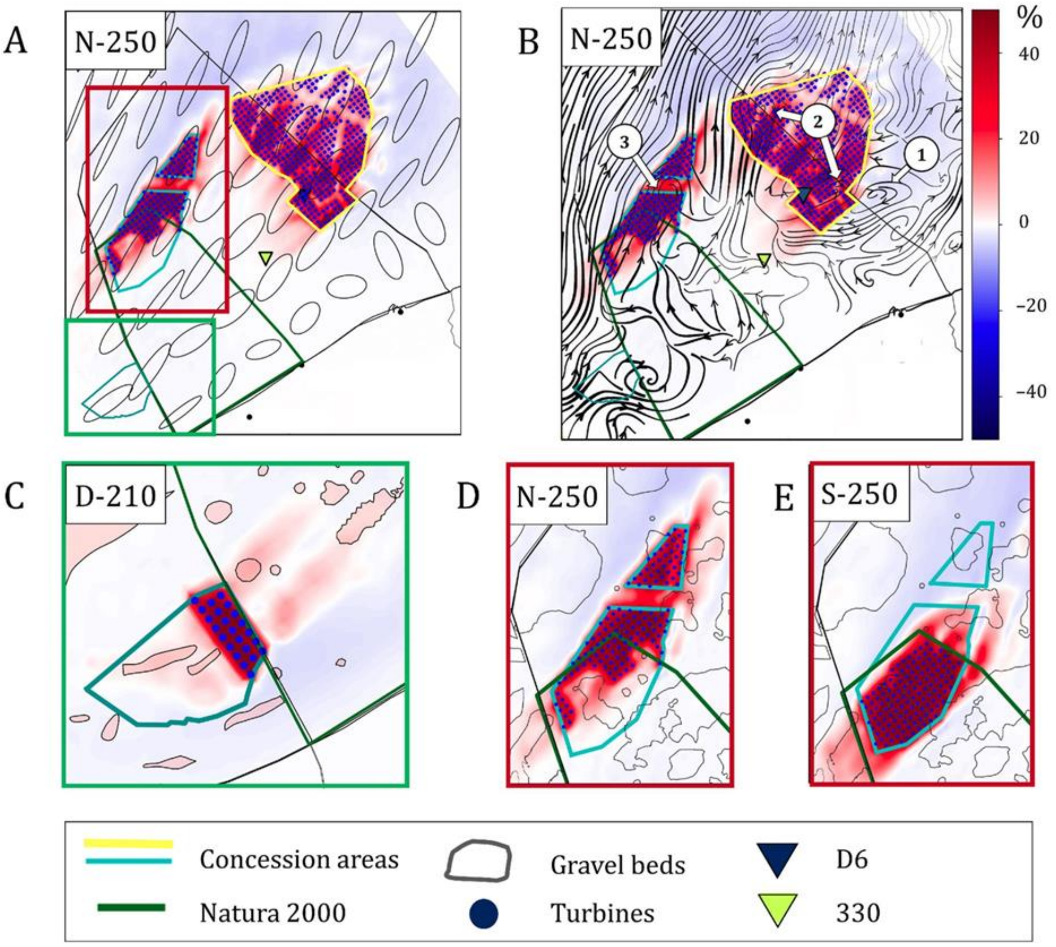


Figure 28 Maps of the annual change in the TOC deposition flux for different scenarios (indicated in the upper-left corner of each subplot) compare to the R-0 scenario, (A) with superimposition of tidal ellipses (M2 constituent); (B) with superimposition of residual currents. Enumerated white circles denote residual gyres acting as retention traps for the TOC deposition flux. (C), (D), (E) – close-ups on the Dunkirk, and on the NCA for the Northern and Southern placement scenarios respectively. (from Ivanov et al. 2021)

The biodeposition of inorganic particlesplays a marginal role in their total resuspension-deposition dynamics, which is mainly conditioned by the tidal cycle. The amount of biodeposited inorganic materials is three orders of magnitude smaller than their background tidal dynamics. This amount, however, might be slightly underestimated, due to the lack of parametrization of faecal pellets decomposition (in the model, they fall apart into organic and mineral components immediately), which would result in the easy removal and redeposition of particles away from the OWFs following the general direction of residual currents.

4.5.2 Changes in TOC deposition flux on the protected gravel beds

The Northern placement (Figure 28, subplot D) has a visibly smaller effect on the northern Natura 2000 gravel beds (∼10%), than the Southern placement (subplot E), resulting in a huge increase (>50%) in deposition on the underlying gravel beds. To better quantify the effect, the spatially-averaged annual TOC deposition flux change on the gravel beds was calculated for different scenarios within both the NCA and Natura 2000 area (magenta colour, Figure 29), and within the

Natura 2000 but outside the NCA (cyan colour). In all Northern placement scenarios the resulting TOC deposition flux on the northern gravel beds increased by about 10%, while in case of the Southern placement the flux change exceeded 50%. Interestingly, the difference between scenarios with different turbine numbers but the same turbine placement is small (less than 10% of difference comparing 250 and 162 monopiles for any of the placements).

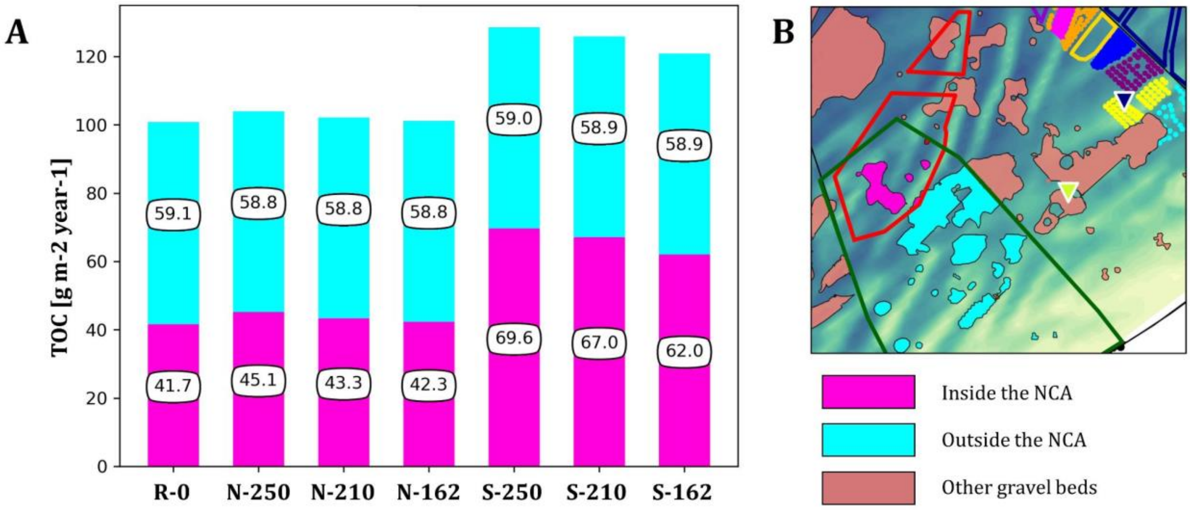


Figure 29 Bar plot comparing different scenarios in terms of the annual spatially-averaged TOC deposition flux on the Natura 2000 protected gravel beds inside the NCA (magenta colour) and outside the NCA (cyan colour). (B) Position of the gravel beds indicated indicated by the corresponding colours. (from Ivanov et al. 2021)

The turbines of the Dunkirk OWF (Figure 28, subplot C) produced two 'tongues' of increased TOC deposition, separated by a zone of TOC flux decrease. This pattern is associated with the bottom bathymetry: the Dunkirk OWF redistributed the TOC from shallow to deeper areas. However, its overall impact on the Natura 2000 southern gravel beds was not as large as the impact of the NCA turbines on the northern gravel beds (~10% vs. 50% respectively), because the Dunkirk OWF has fewer projected monopiles. In case if the turbines are located further away from the BPNS border, the impact is expected to be even smaller.

4.5.3 Offshore wind farm footprint on sediment biogeochemistry

The changes in organic matter distribution associated with biodeposition from the OWF, also had clear effects on mineralization processes in the sediment. In discussing the result,s we make a distinction between "Inside OWF" changes, "Outside OWF" changes, and "BPNS-wide" changes. "Inside OWF" means we look at changes only within the OWFs, whereas "outside" OWF means we only look at those outside OWFs. "BPNS-wide" means we average changes across the entire BPNS, both inside and outside of the OWFs.

The general observed pattern was an increased intensity of mineralization processes inside the OWFs, and a slight spill-over effect on processes outside the OWFs. When averaged across the BPNS, net effects were small.

Inside OWF

Total mineralization rates within OWFs were on average 27 and 30 % higher compared to the baseline scenario (no OWFs) for the 'current' and 'future' scenario. In certain places in the OWFs, rates increased by 69 % (Fig. 29 A, B), similar to the increase of the organic carbon flux. The increase in total mineralization was supported by increases in all individual processes, primarily by increased anoxic mineralization (Fig. 29 C, D), followed by oxic mineralization (Fig. 29 E, F). Denitrification increased by 2.3 to 2.5 % on average (Fig. 29 G, H), which resulted in additional removal of 7 to 10.7 kmol N y^{-1} from 'current' to 'future' scenario. These higher denitrification rates within OWFs originated from the increased organic matter influx. The relative importance of denitrification (percentage of total mineralization) did not increase, rather it decreased by 18 % and 20 %.

Within OWFs, the conversion of organic carbon deposited on the sediment to DIC returning to the water column also decreased, by -3.6 and -4.5 % in the 'current' and 'future' scenario respectively (Table XII). As a result, the carbon stock in the sediments of OWFs increased. The amount of organic carbon in the upper 10 cm increased by 9.6 and 11 % on average (Table XII), and up to 37 % in certain places. This constant input of fresher organic matter further increased the reactivity of the organic carbon pool within OWFs by similar values.

Outside OWFs

Within the BPNS, we noticed a slight spill-over to the regions outside of the OWFs (Table XII, 'Outside OWF BPNS'). Total mineralization rates increased only slightly (0.3 – 0.6 % compared to baseline), as a result of a spill-over of organic carbon deposition from within the OWFs (Table XII). Again, mineralization efficiency (DIC / OC_{input}) decreased slightly as a result of this increased deposition (-0.2 and -0.6 %). This was associated with an increase of the carbon stock when averaging regions outside of the OWFs (by 0.3 – 2 %, Table XII).

BPNS wide estimations

The net effects of these changes for the BPNS were an increased total mineralization of 2.3 % in the 'current' scenario, and 3.4 % in the 'future' scenario. This increase in mineralization was caused by the increased deposition within the OWFs, which spilled over slightly into neighbouring regions. Whereas deposition and mineralization rates further away from the OWFs decreased (Fig. 30), there was still a net increase when averaging values over the BPNS. With a decrease in mineralization efficiency by 0.4 and 1 % respectively (mainly as a result of the proportionate increase of anoxic mineralization), the carbon stock in the top 0.1 m of the sediment increased by 3.3 and 6.5 % in the 'current' and 'future' scenario, respectively.

Small net changes to the importance of denitrification in total mineralization had a minor effect on the amount of N removed from the BPNS. Total denitrification rates increased by 0.15 and 0.35 %, but these values represented a proportional decrease when compared to the total mineralization (decreases of 1.38 and 3.04 %). Still, the total N removed from the BPNS increased by 0.17 and o.37 % in the 'current' and 'future' scenario respectively.

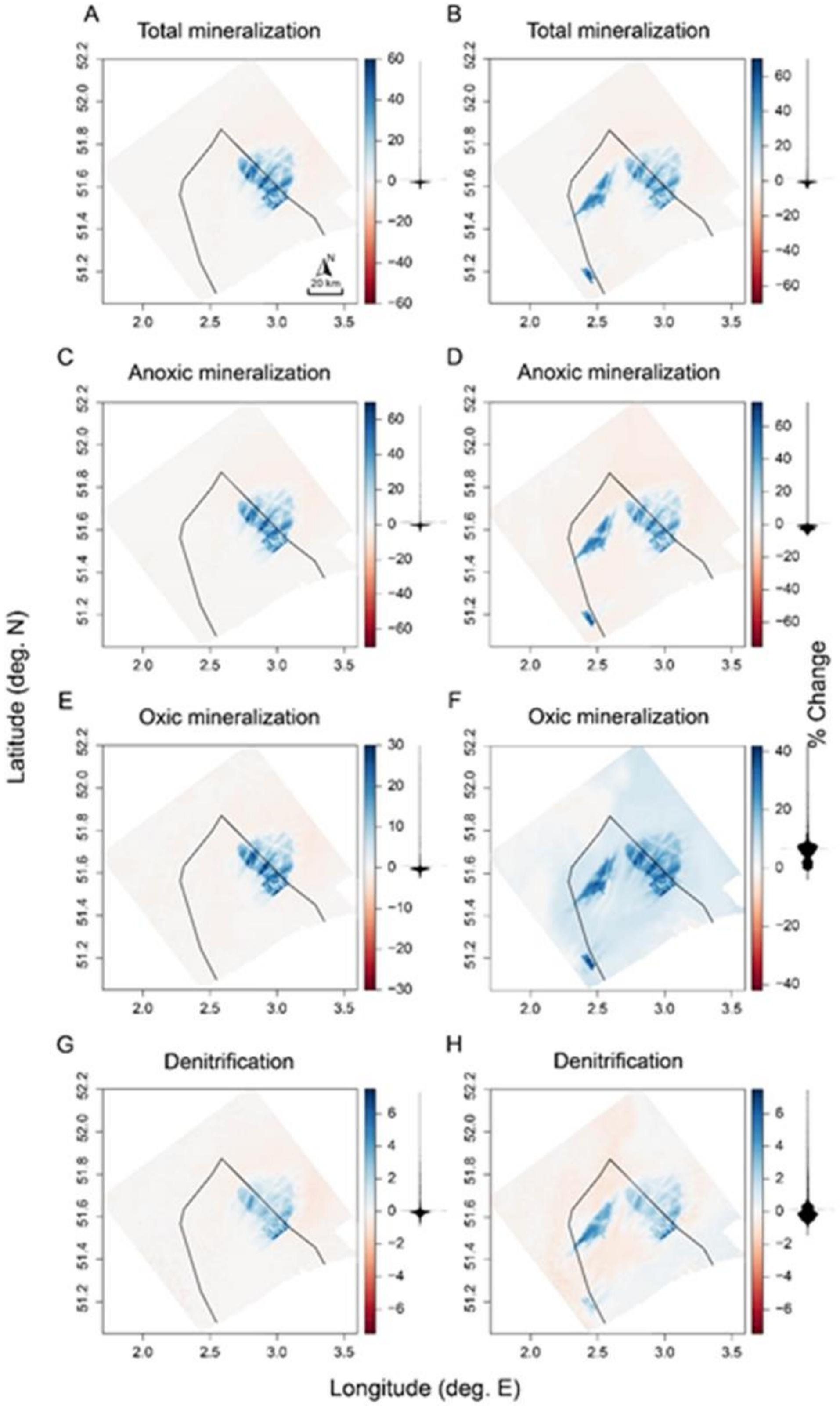


Figure 30. Changes in rates of total mineralization (A, B), oxic mineralization (C, D), anoxic mineralization (E, F), and denitrification (G, H) relative to the baseline scenarios (in %) for the "current" scenario (left column), and the "future" scenario (right column), with the Belgian EEZ outlined in black. Density distributions of the values are represented as black shapes corresponding to the axis from the color key. From De Borger et al. (in press).

Table XII Averaged values and relative changes between tested scenarios and the baseline of carbon mineralization processes for the current (C) and future (F) scenarios, subdivided in values over locations with OWFs, the BPNS excluding OWFs (Outside OWF BPNS), and the BPNS. Values for total mineralization, organic carbon flux, and DIC flux in mmol m^{-2} d^{-1}, the conversion represents the proportion of the incoming organic carbon flux released as DIC, and carbon storage is the total reactive carbon in the top 10 cm of the sediment (mol m^{-2}). From De Borger et al. (in press).

	Total mineralization		**Org. C flux**		**DIC flux**		**Conversion**		**Carbon storage**	
	mmol m^{-2} d^{-1}		*mmol m^{-2} d^{-1}*		*mmol m^{-2} d^{-1}*		-		*mol m^{-2}*	
Scenario	C	F	C	F	C	F	C	F	C	F
OWF locations										
Baseline	14.7	13.9	17.2	16.4	-14.6	-13.9	1..0	0.9	89.3	88.2
Scenario	18.6	18.0	22.0	21.4	-18.5	-17.9	0.9	0.9	97.9	97.9
% Change	26.9	29.5	28.1	31.0	26.6	29.4	-3.7	-4.6	9.7	11.0
Outside OWF BPNS										
Baseline	15.7	15.9	17.5	17.7	-15.6	-15.8	1.1	1.1	84.4	84.4
Scenario	15.7	16.0	17.6	17.8	-15.6	-15.9	1.1	1.1	84.4	84.5
% Change	0.3	0.6	0.3	0.4	0.3	0.6	-0.2	-0.6	0.3	2.0
BPNS										
Baseline	15.6	15.6	17.5	17.5	-15.5	-15.5	1.1	1.1	84.7	84.7
Scenario	16.0	16.1	17.9	18.1	-15.8	-16.0	1.0	1.0	85.4	85.8
% Change	2.3	3.4	2.5	3.6	2.2	3.5	-0.4	-1.0	3.3	6.5

Implications of the results

Our extensively validated tool that we have developed for the project - the coupled hydrodynamic-wave-sediment-filtration model, also coupled with the diagenetic model, allows for the upscaling of the local effect of deposition by fouling communities from a single offshore turbine foundation over the regional (BPNS and SBNS) scales. The model not only takes into account the biofouling impact of each individual turbine in the region, but also accounts for synergetic and antagonistic effects of neighbouring turbines. Our tool is available online (https://github.com/Ezhen/COAWST_setup.git) and it will play a role in future projects (e.g. CE2COAST). Implementation of such tools in context of European seas helps to support the EU policies, such as the Blue Growth strategy (Eikeset et al., 2018), the Marine Spatial Planning Directive, the Habitats Directive (Coffey and Richartz, 2003) and Marine Strategy Framework Directives (particularly for Descriptor 6 “Seafloor integrity”).

Main outcomes from studying the TOC biodeposition

1) The effect of blue mussel carbon deposition leads to a significant (up to 50% and more) increase in total carbon deposition within the perimeters of the OWFs. Around the OWF perimeter (2 km proximity), this increase is about 2 - 15%. Those changes may potentially lead to changes in local benthic communities.
2) At the distance of 5 - 30 km away from the closest monopile, the carbon deposition flux decreases (up to 2%), due to carbon consumption by the fouling mussels, with the maximum decrease observed at 9 - 13 km distance. The decrease in TOC deposition is significantly smaller than the increase within the 5 km proximity to the monopiles, but stretches over much greater distances.
3) The spatial pattern of TOC deposition flux changes is not uniform or symmetric, but is highly affected by the local tidal dynamics (direction of the major axis of the tidal ellipse of the dominant regional tidal constituent M2) and by the residual (thermohaline, overtidal) regional circulation (including residual gyres, acting as retention zones), as well as by the waves. Tides and waves induce the bottom stress and resuspend the organic carbon, which is then carried away by the tidal and residual currents.
4) Hydrodynamics (residual currents) are necessary to be considered in OWF impact assessment on sediment communities. Placing the monopiles at least 3 km downstream from the protected benthic communities will lead to a moderate increase (~10%) of TOC deposition to them. If placed upstream, the TOC deposition flux increase of ~5% will be achieved by placing the monopiles at least 7 km away from the bottom communities. In the orthogonal directions to the local residual currents and to the major axis of local tidal ellipse, the recommendation of turbine placement would be 2–4 km away from the protected communities.
5) The monopiles positioning plays a much more important role than their total number. If the NCA turbines are placed at least 3 km downstream from the gravel beds, the resulting impact would be an increase of TOC deposition by 10% compared to the situation without turbines. Monopile installation close by the gravel beds will result in 50% flux increase. However, changing the number of turbines by more than 50% (from 162 to 250) will result in only a ~10% flux increase between the two scenarios.

RECOMMENDATION: In order to limit carbon biodeposition on the protected Natura 2000 gravel beds by about 10% of increase, we recommend placing NCA turbine foundations at least 3 km NE or NW from the closest gravel bed, or at least 7 km in SW direction. Overall, the choice of proximity of turbine foundations to any protected sites, other than Northern Natura 2000 gravel beds, should be primarily made taking into account the local hydrodynamics (residual circulation and tidal currents), as well as the bottom bathymetry, and thus cannot be generalized as a strict set of recommendations for an ideal OWF. While the long-term effects of the biodeposition could be underestimated for inorganic particles, due to current model limitations, placing the turbines over or next to the gravel beds will certainly result in carbon biodeposition increase several times higher, when if turbines were placed at least several km away. Interestingly, the increase of number of turbines in the NCA by about 40% will result in a relative impact increase by only about 10%.

4.5.4 So What?

Our findings indicate that OWFs installed on the permeable sediments of the Southern North Sea become local sinks for carbon relative to the baseline scenario. The ‘current’ and ‘future’ wind farm developments increased the carbon stock in the upper 10 cm of the sediment by $28.7*10^3$ to $48.4*10^3$ tonnes respectively, which coincides with 0.014 – 0.025 % of Belgium’s greenhouse gas emissions (118.5 million tons in CO2 equivalents in 2018; VMM et al., 2020). In that sense, the carbon potentially stored in OWFs represents a small but significant carbon offset in carbon accounting, which can be added to the intrinsic carbon economy of OWFs. In Belgium, an energy production of 3391 GW was attributed to OWFs in 2018 (at 0 $tCO_2\ GW^{-1}$; operational cost only; Belgian Debt Agency, 2018). Compared to a gas turbine for example (380 $tCO_2\ GW^{-1}$), this corresponds to $1.29*10^6$ tCO_2 not emitted by using wind-generated power. When including the life cycle (material sourcing, construction…), this factor ranges between 14 and 111 (depending on the reference chosen; Dolan and Heath, 2012). This would reduce CO_2 emissions between $1.04*10^6$ and $2.86*10^6$ tCO_2. To this, our estimated quantities of organic carbon stored in the sediment contribute an additional 1 to 4.6 %.

The importance of this increased storage depends on different factors. Firstly, sediment disturbing activities such as bottom trawling may or may not be prohibited within OWFs (e.g. the UK and France allow trawling outside a 50 m radius around individual turbines, whereas in Belgium, the Netherlands and Germany it is prohibited completely). Bottom trawling disturbs the sediment and will likely prevent build-up of the carbon stock (De Borger et al., 2021). A second factor is that the expected lifespan of a wind turbine is 20–25 years (Nghiem and Pineda, 2017). After this, the concession zone needs to be restored to its original condition by the concession holder (Kruse et al., 2019). In case of such full decommissioning, the increased organic matter input will cease while the decommissioning activities themselves can trigger the release of accumulated carbon to the water column as they will most likely include sediment disturbing activities. However, when alternative partial decommissioning scenarios would be considered in which part of the subtidal structure would either remain in place, be repurposed, or translocated (Fowler et al., 2020); organic matter filtration by fouling fauna and subsequent local carbon storage in sediments should be considered an important decision criterion.

4.5.5. Recommendations for future research

To make this final upscaling exercise possible, FaCE-It bridged major knowledge gaps and developed new methodology. Still, we had to make several assumptions that have led to recommendations for future work, and improvements to model predictions. These are:

- We used the blue mussel (*Mytilus edulis*) as filter feeders, present as a constant biomass on the turbine foundations. This is the same approach as previous modelling efforts (Slavik et al., 2019), and mostly, it is because there are no datasets yet that show a constant seasonal pattern in biomass that can be used in a model. The same stands for the vertical distribution of the blue mussel -- we assume that it is uniform in the upper 6 m, however in reality it can be more complex. We recommend a better seasonal and vertical resolution in data collection on fauna living on turbines. This can be facilitated by developing semi-automated data collection technology (i.e. video footage by ROVs).

- In addition to the current model, the work of FaCE-It (Mavraki et al., 2020b) showed that other filter feeders such as the amphipod *J. herdmani* are of significant importance to the filtration capacity of offshore wind substructures (which was found after model simulations had been performed, and thus not included). Since the output of these types of models hinge on how the filtering community is described, an accurate description of the temporal dynamics, and the species composition of this community and its functional consequences for the environment will improve the realism of model outputs.

- In the model, deposited faecal pellets are immediately converted to their primary compartments, when they reach the sediment bed. In reality, decomposition takes time, but no studies were found or carried out on that. Thus, our model might overestimate the spread of deposited carbon and inorganic fine materials. Therefore, detailed research on the spatial and temporal fate of faecal pellets in water column and sediment will help assessing the spatial redistribution of organic matter.

- The collection of validation material: usually, models with a strong spatial and/or temporal component are forced with a time-series of data. For sediment biogeochemistry, these types of time series do not yet exist in the modelling domain (the Belgian Part of the North Sea). Promising technical developments aimed at facilitating the comparison between in-situ observations and model outputs include eddy covariance derivation of benthic nutrient and oxygen fluxes, which allows for an increase of the spatial footprint of measurements (Berg et al., 2003), and regular benthic sampling through automated benthic stations, which can better constrain the temporal variability of in-situ flux acquisition (e.g. Toussaint et al., 2014; Moriarty et al., 2017).

- For technical reasons, we omitted a dynamic feedback coupling of the seafloor and the water column. With this, we implicitly assumed that primary production in the water column would not be affected by potential changes in ambient nutrient concentrations resulting from the sediment, whereas in reality this will definitely be the case (Floeter et al., 2017). Specifically, a dynamic coupling with a water column biogeochemistry model (Luff and Moll, 2004; Slavik et al., 2019), would make it possible to investigate whether water column nutrient concentrations in OWFs can be expected to decrease, or increase as a result of the transfer of organic matter to the sediment, and whether this in turn affects phytoplankton concentrations. Especially in shallow shelf seas, nutrients regenerated in the sediment co-regulate primary production events in the water column (Ruardij and Van Raaphorst, 1995; Soetaert and Middelburg, 2009). A negative feedback on phytoplankton blooms for example, would decrease faecal pellet production and subsequent storage of organic carbon in the sediment, thus introducing self-regulation in the system. Future improvements to modelling efforts should include this feedback coupling to shed light on the importance.

- The existence of sediment wakes around the turbines that are visible via remote sensing and have been sampled during field campaigns (Baeye and Fettweis, 2015) leads to an assumption that the filtered materials can concentrate inside them and be carried away over longer distances. However, representation of those wakes are not possible in the current model resolution (1x1 km), because their widths do not exceed several dozens of meters. Going to higher resolutions via the second level of nesting would allow to model the wakes and assess their importance for redeposition of filtered materials.

- The monopile foundations and their rotating turbines also have a direct impact on the local and regional hydrodynamics: they perturb the circulation and increase bottom roughness through their scour protection layers, subsequently altering the shear stress, resuspension, mixing and stratification (Grashorn and Stanev, 2016: Rennau et al. 2012). They are also capable of generating local eddies (Simpson et al., 1982), by cutting the current into two streams that merge back together downstream. The consumption of wind by the rotors creates a “shadow zone” downstream, that forms an upwelling-downwelling system (Christiansen and Hasager, 2005; van der Molen et al., 2014). All these effects can be assessed through running a simulation with a second level of nesting and then parameterized on a coarser resolution.

RECOMMENDATION: We recommend to install monitoring devices in, and around OWFs to track the dynamics of nutrients, phytoplankton, and physical parameters (e.g. turbidity, stratification) in the water column, since this data is currently lacking, but at the same time of high importance for future predictions. Secondly, we stress that the marine ecosystem does not abide to geopolitical boundaries. When evaluating the ecosystem effects of offshore windfarms placed within a specific countries EEZ, changes to matter fluxes will affect other countries’ EEZ, and this needs to be accounted for.

4.6 Added value of FaCE-It research

Apart from its **scientific output** (3 PhD’s, 13 peer reviewed papers, 25 posters and presentations on scientific conferences), the FaCE-It project provided added value in many aspects.

First of all, the project resulted in a **strong Belgian ‘OWF research community’**, continuing the FaCE-It research line and obtaining funding from various sources, including Belspo and the Federal Public Service Public Health, Food Chains Safety and Environment, NWO (The Netherlands), INSITE (UK) and INTERREG. At least 4 currently funded research projects have strong roots in FaCE-It, and given the established research network, all these projects are cooperating through either knowledge and data exchange, and planning of joint cruises and experiments.

FaCE-It created **new methodology and tools** which are now available to the Belgian and international research community. Uptake of these methods and tools is facilitated on the one hand by advertising them in international networks (ICES BEWG, ICES WGMBRED), and through **training of the future generation of marine scientists**. FaCE-It has organised two international summer schools in Calvi (France) for PhD students and young post-doctoral researchers (summer 2017 and 2019), and reached a large group of international students during an annual training week in the framework of the MSc in Marine and Lacustrine Science and Management ("Oceans and Lakes").

From a **policy perspective**, FaCE-It scientists have been active in policy support in Belgium and abroad. FaCE-It science and scientists have an important role in an ongoing Belgian research programme that will provide recommendations for the installations of OWFs in the new concession zone. FaCE-It science and scientists contributed to an ICES advice requested by OSPAR, provided knowledge and expertise to the Dutch process towards providing guidelines on an ecological framework for offshore wind developments (HaSPRO), and provided guidance in a UK workshop on introducing artificial hard substrates in marine protected areas (Natural England/JNCC). Given the prominent place of FaCE-It at the international OWF research horizon, FaCE-It scientists were invited to contribute and/or coordinate to a series of activities in both the Atlantic (series of webinars on OWFs in the framework of the BAIRD Symposium, NYDSERDA State of Science workshop on Wind and Wildlife) and Pacific Ocean (US Offshore Wind Synthesis of Environmental Effects Research).

5. DISSEMINATION AND VALORISATION

FaCE-It scientists interacted with fellow scientists, students, the public at large, stakeholders and the national and international policy level throughout the project.

FaCE-It interaction with scientists was mainly through publication of papers in the peer-reviewed literature (13 papers published) and oral or poster presentations at scientific conferences (25) presentations. The final event of FaCE-It was organised as the 52nd edition of the Liège Colloquium, attended by 170 attendees from 15 countries. FaCE-It research was spotlighted in two invited key note talks, a regular presentation and two posters. Given the sanitary measurements at the moment of organising the event, it was organised as a virtual meeting, making use of the ZOOM and Gather platforms to allow for a maximum interaction between attendees. During the colloquium, a special event on the current status of Offshore Wind Farm monitoring in the Belgian part of the North Sea was organized. The main findings of FaCE-It were presented in one of the talks.

The FaCE-It Research Gate page was followed by 87 persons from many countries and resulted in almost 1400 reads, reflecting the international interest in the progress and results of the FaCE-It project.

FaCE-It research was also presented during regular meetings of international networks, including the ICES Benthos Ecology Working Group, the ICES Working Group on Marine Benthal Renewable Energy Developments, EGU, Nereis Park… and therefore reached a wide international scientific audience.

One of the FaCE-It goals was to organise a summer school for PhD students and young post-doctoral researchers. This summer school was organised in summer 2017 at the marine station of STARESO in Calvi (Corsica, France). Given the large interest and the positive response, a second edition was organised in summer 2019. In total, we trained 21 early career scientists in these summer schools. Additional future young scientists were trained in FaCE-It techniques and approaches during five annual field and experimental training weeks within the framework of the MSc Marine and Lacustrine Sciences and Management (Oceans an Lakes, about 20 students annually).

We reached out to the public at large through demonstrations of techniques during the 2017 Science Day hosted by Flanders Marine Institute, which attracted 771 visitors. Another way of sharing knowledge with the public at large was through our participation in the "ZeeUitzicht' programme, where scientists presented their results to the public at large in apartments with a sea view. An interview with FaCE-It scientists on biodiversity in offshore windfarms in the Belgian part of the North Sea appeared in Frontaal, the magazine of Gents Milieu Front, a local environmental organization. Finally, we coordinated a contribution to Natuur.Focus, a magazine published by Flanders largest nature conservation group Natuurpunt, focusing on biodiversity patterns in the Belgian part of the North Sea, with considerable focus on the effects of OWFs on the biodiversity of our coastal area.

Interaction with stakeholders was organised through our annual stakeholder meetings. During these meeting, we were very carefully in avoiding scientific jargon and spent considerable time and effort to place our results in a societal relevant context. Stakeholders were given the chance to bring forward specific topics of interest, and were invited to suggest scenarios to test the integrated final FaCE-It model, which actually lead to the scenario run and results as reported above.

Outside the project, stakeholder interaction was maintained at different levels. FaCE-It scientists contributed to the activities of the Think Tank North Sea (topics: Living with Nature, Living with Climate Change), and were consulted in formal and informal discussions during the process leading to the new Belgian Marine Spatial Plan. Following the acceptance of the plan, FaCE-It scientists were invited to participate in the dialogue between policy, industry and nature conservation, leading to prioritization of research needs and a new research programme.

At the international level, FaCE-It reached out to scientists, industry and other stakeholders (mainly fisheries and industry) through its participation in a series of webinars on the introduction of artificial hard substrates in the marine environment, as part of the online version of the BAIRD symposium (organised by University of Rhode Island and ICES WGMBRED). Along the same lines, FaCE-It scientists were invited to contribute FaCE-It knowledge to several processes preparing the installation of offshore wind farms on both the East and West Coast of the US (NYSERDA, BOEM, US Offshore Wind Synthesis of Environmental Effects Research). Closer to home, FaCE-It scientists were invited to share results in several discussion with representatives from industry, fisheries, tourism and nature conservation in the UK (organised by Natural England) and the Netherlands (HaSPRO, WOZEP) and Germany (upon invitation by the Thünen Institute).

Finally, FaCE-It allowed the participants to build the necessary expertise and skills to successfully compete for funding. FaCE-It partners are now active in various projects (OUTFLOW, FOOTPRINT, CE2COAST, WMP-N2000) funded be a variety of funding mechanisms in Belgium (Belspo, FPS Public Health, Food Chain Safety and Environment) and abroad (UK, the Netherlands, JPI).

6. PUBLICATIONS

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ANNEX I

Summary of the significant ($p < 0.05$) predictive models for SCOC, oxic mineralization, anoxic mineralization, total N mineralization, nitrification and the probability of nitrate being transformed into nitrogen gas through denitrification. For each model, the distribution and the variance structure when applicable are detailed. The AIC and the adjusted R-squared or explained deviance when available are provided, when available (From Toussaint et al., 2021)

Model	Distribution	Variance structure	AIC	Standard error
SCOC = 32.73*** -2.23 $\cdot 10^{11}$ Permeability* + 5.41 $\cdot 10^{-3}$ BP_c*** - 0.59 SpeciesRichness	Gaussian (LM)		AIC = 160 Adj R^2 = 58%	Intercept: 3.91 Perm.: 9.94 $\cdot 10^{10}$ BP_c: 1.12 $\cdot 10^{-3}$ Sp. richness: 0.32
Oxic mineralization = 4.08*** + 7 $\cdot 10^{-3}$ BP_c*** - 1.29 10^{11} Permeability – 0.42 SpeciesRichness	Gaussian (GLS)	Power of variance covariate ~ Permeability	AIC = 276	Intercept: 2.71 BP_c: 1 $\cdot 10^{-3}$
Anoxic mineralization = 7.72 *** - 3.37 $\cdot 10^{11}$ Permeability*** + 14.64 TOM*** - 0.16 IrrigationRate***	Gaussian (GLS)	Exponential of variance covariate ~ TOM | Station	AIC = 265	Intercept: 1.25 Perm.: 0.03 TOM: 1.88 Irr.: 0.02
N mineralization = 3.96*** - 7. 66 $\cdot 10^{10}$ Permeability*** + 3.01 TOM*** + 8.25 $\cdot 10^{-4}$ BP_c * - 7.89 $\cdot 10^{-2}$ IrrigationRate*	Gaussian (GLS)	Different standard deviation ~ Station	AIC = 214	Intercept: 0.53 Perm.: 1.66 $\cdot 10^{10}$ TOM: 0.49 BP_c: 3.34 $\cdot 10^{-4}$ Irr.: 3.39 $\cdot 10^{-2}$
Nitrification = 0.11*** + 3.13 $\cdot 10^{-5}$ BP_c*	Gamma (GLM)		AIC = 210 Explained Deviance = 6%	Intercept: 0.02 BP_c: 1.20 $\cdot 10^{-5}$
Denitrification = 2.71 + 1.40 TOM*** + 0.10 Fines* - 0.27 IrrigationRate* - 1.08 $\cdot 10^{-3}$ IP_c*	Gaussian (LM)		AIC = 105 Adj R^2 = 46%	Intercept: 1.54 TOM: 0.28 Fines: 0.04 Irr.: 0.12 IP_c: 0.41 $\cdot 10^{-3}$
Den / Den + DNRA = 1.25* - 0.17 IrrigationRate*	Binomial (GLM)		AIC = 46 Explained Deviance = 16%	Intercept: 0.5 Irr.: 0.07