

CHAPTER 6

TURBINE-RELATED IMPACTS ON MACROBENTHIC COMMUNITIES: AN ANALYSIS OF SPATIAL AND TEMPORAL VARIABILITY

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Abstract

We investigated spatiotemporal variability in sediment properties and in macrobenthos communities ‘very close’ (*i.e.* at a distance of 37.5 m) to, and ‘far’ (*i.e.* at a distance of 350-500 m) from jacket foundations in the C-Power offshore wind farm on the Thornton Bank over a time span of three consecutive years. We anticipated that the locally modified water currents around wind turbines, as well as the depositional flow of faecal pellets and other detrital material produced by filter-feeding epifauna living on the foundations, could both contribute to a process of sediment fining and organic matter enrichment close to, and in the wake of, wind turbines. Such effects would in turn be expected to cause shifts in macrobenthos community composition, diversity and abundance. Our three-year analysis provided equivocal support for this hypothesis. Sediments ‘very close’ to jacket foundations had a significantly higher proportion of fine sand compared to samples collected at further distance, but this was not accompanied by a higher organic matter content in 2018 and 2019, rather the contrary. Average

macrobenthos abundance and diversity were always higher ‘very close’ to the turbines, but these differences were not consistently statistically significant. Macrobenthos species composition, however, did consistently differ between ‘very close’ and ‘far’ stations. It also changed significantly with time in the ‘very close’ stations, a difference which was largely attributable to a pronounced decline in the abundances of three otherwise dominant species of permeable sediments on the BPNS, and to an increase of several other species, in particular of the bio-engineering, small reef-building polychaete *Lanice conchilega*. This evolution should be carefully monitored in following years, the more so since feedback loops can be expected from the activity of *L. conchilega* on sediment fining and enrichment, because *L. conchilega* enhances the deposition and retention of fine particulate material from the benthic boundary layer. Overall, there appears to be a clear trend for sediments to become finer and organically enriched ‘very close’ to jacket foundations, with concomitant effects on the abundance, diversity and species composition of macrofauna. However, there is a large variability between turbines, and an

increased small-scale heterogeneity around individual turbines as a result of the emergence of small biogenic reefs further adds to an overall prominent variability, which renders statistically robust conclusions on the exact evolutions of sediments and macrobenthos under the influence of turbines difficult to draw. Future monitoring should therefore try to better incorporate small-scale variability in its sampling design, whereas targeted monitoring efforts should be directed at a better elucidation of the spatial scale over which fining and enrichment effects are being manifested.

1. Introduction

The 2020 Belgian targets for renewable energy depend in part on an expansion of offshore wind farms (OWFs) in the Belgian part of the North Sea (BPNS; Rumes *et al.* 2017). Within the eastern part of the BPNS, three parks are already fully operational (C-Power, Northwind and Belwind), a fourth one (Norther) having recently been constructed and having entered the operational phase as of 27 May 2020 (hence after the 2019 autumn monitoring campaign). On top of the 2020 targets, the Belgian government has decided to double the capacity of electricity outputs from wind energy, with a major contribution coming from offshore installations. In order to reach this target, a new concession area in the western part of the BPNS, near the border with France, has been designated for possible exploitation after 2020 (Rumes & Brabant 2018). Because wind farms introduce hard substrate in otherwise soft sediment environments, and because every stage (pre-construction, construction, operational and decommissioning phase) of the development of an OWF can potentially impact these soft sediments and their communities of living organisms (*i.e.*, benthos), consistent monitoring of impacts, as well as of (sometimes subtle) changes in environmental factors which could lead to impacts, remains of utmost importance (Gill *et al.* 2018).

The current OWFs are situated in generally medium to coarse sandy sediments with a low organic matter content (Van Hoey *et al.* 2004; Byers *et al.* 2013). Such sediments tend to be characterised by relatively poor macrobenthic communities in terms of both density and species diversity (Van Hoey *et al.* 2004; Reubens *et al.* 2009; Coates *et al.* 2014). However, the operational phase of an OWF may result in a modification of the habitat as a result of at least two possible mechanisms. First, wind turbines may affect local hydrological conditions (in terms of both currents and water column stratification; hydrology, sediment type, water column stratification) and infaunal community structures. At the same time, wind turbines rapidly become colonized by a dense community of epifaunal fauna, which to a large extent obtains its food from the water column while depositing both suspended sediment and organic matter particles to the seafloor surrounding the turbines. Both processes can affect sediment granulometry and organic matter content, both of which are extremely important environmental factors that structure the abundance, diversity and species composition of benthic communities (De Backer *et al.* 2014; Maar *et al.* 2009; Dannheim *et al.* 2019; Gill *et al.* 2018; Coates *et al.* 2014).

The prediction that the sediments surrounding wind turbines may progressively become enriched in fine sediment particles as well as organic matter were supported by the results of a targeted study that focused on a single gravity-based foundation in the C-Power OWF (Coates *et al.* 2014). The researchers hypothesised that such sediment fining and organic matter enrichment would likely result in changes in macrobenthic assemblages surrounding wind turbines, which would be expected to become more abundant and diverse, as is usually the case in similar finer sediments offshore (Wilding *et al.* 2012). However, tests of this hypothesis in the framework of the

yearly large-scale monitoring from 2015 onwards failed to discover any significant local enrichment effects when looking at multiple turbines (Colson *et al.* 2017). Because sampling in those campaigns focused at two specific distances from turbines, *i.e.* 50 m ('close') and 350-500 m ('far'), we revised the sampling strategy based on the consideration that turbine effects might be spatially more limited and therefore not generally observable at a distance of 50 m. Hence, from 2017 onwards, we shifted the 'close' samples as closely as possible (in terms of sampling) to the turbines, *i.e.* at 37.5 m from the center of a turbine (Lefaible *et al.* 2018). We also considered the possibility of differences in turbine effects depending on turbine type. Indeed, the study by Coates *et al.* (2014) was done at a gravity-based turbine, but the majority of turbines at the C-Power OWF have jacket foundations, whereas those at Belwind, and indeed most turbines that have recently been deployed or are being planned, are monopiles.

Results of the 2017 monitoring campaign revealed significant sediment fining, organic enrichment and changes in macrobenthic communities (higher densities, diversity and different compositions) at 37.5 m from jacket-based foundations in C-Power (Lefaible *et al.* 2018). Impacts around the monopiles in Belwind were less pronounced, with differences in sediment composition, macrobenthos abundance and diversity not being significant between the 'very close' and 'far' stations, whereas species composition did exhibit significant differences between both distances from the turbines (Lefaible *et al.* 2018). Still, a recurrent observation is that inter-turbine variability within an OWF is high, both in terms of epifaunal and macrobenthic communities (Jak & Glorius 2017). Hence, generalisations about turbine-related impacts should not be based on the results of a single

monitoring campaign, but should ideally be firmly rooted in patterns that are robust over space and time.

When the same sampling strategy was used in 2018, the higher fine sediment fraction 'very close' to jacket foundations was again observed, yet the organic enrichment was not (Lefaible *et al.* 2019). Such absence of an organic enrichment could obviously imply that there is no clear turbine effect. Alternatively, since the presence or absence of an effect is assessed from a comparison between 'very close' and 'far' stations, and since turbine effects may, with time, expand over a larger area, we cannot automatically rule out that an increase in sediment organic content occurred at locations well beyond the 'very close' stations, which could also erase significant differences between 'very close' and 'far' samples (Lefaible *et al.* 2019). This aspect is currently under investigation in the framework of a targeted monitoring that has been performed in August 2020 (Lefaible *et al.* in prep.), but would also be expected to show up in a multi-year comparison of sediment characteristics of 'far' stations. At the same time, to ascertain that any organic matter enrichment effects are local effects that are probably caused by the presence of turbines, it remains important to also monitor the evolution of the nearby reference area, which is characterised by a very similar sediment type and macrobenthos community, yet is not influenced by wind turbines. Due to the work restrictions imposed by UGent in response to the SARS-CoV-2 epidemic, these samples of the 2019 campaign have not yet been processed and analysed, but they will be done as soon as possible, enabling a comparative analysis of spatiotemporal trends inside and outside an offshore wind farm.

In this report, we focus on macrobenthic communities and their sedimentary habitat around selected jacket foundations in the C-Power OWF. More specifically, we investigate patterns of sediment fining, organic matter content and macrobenthic community abundance, species richness

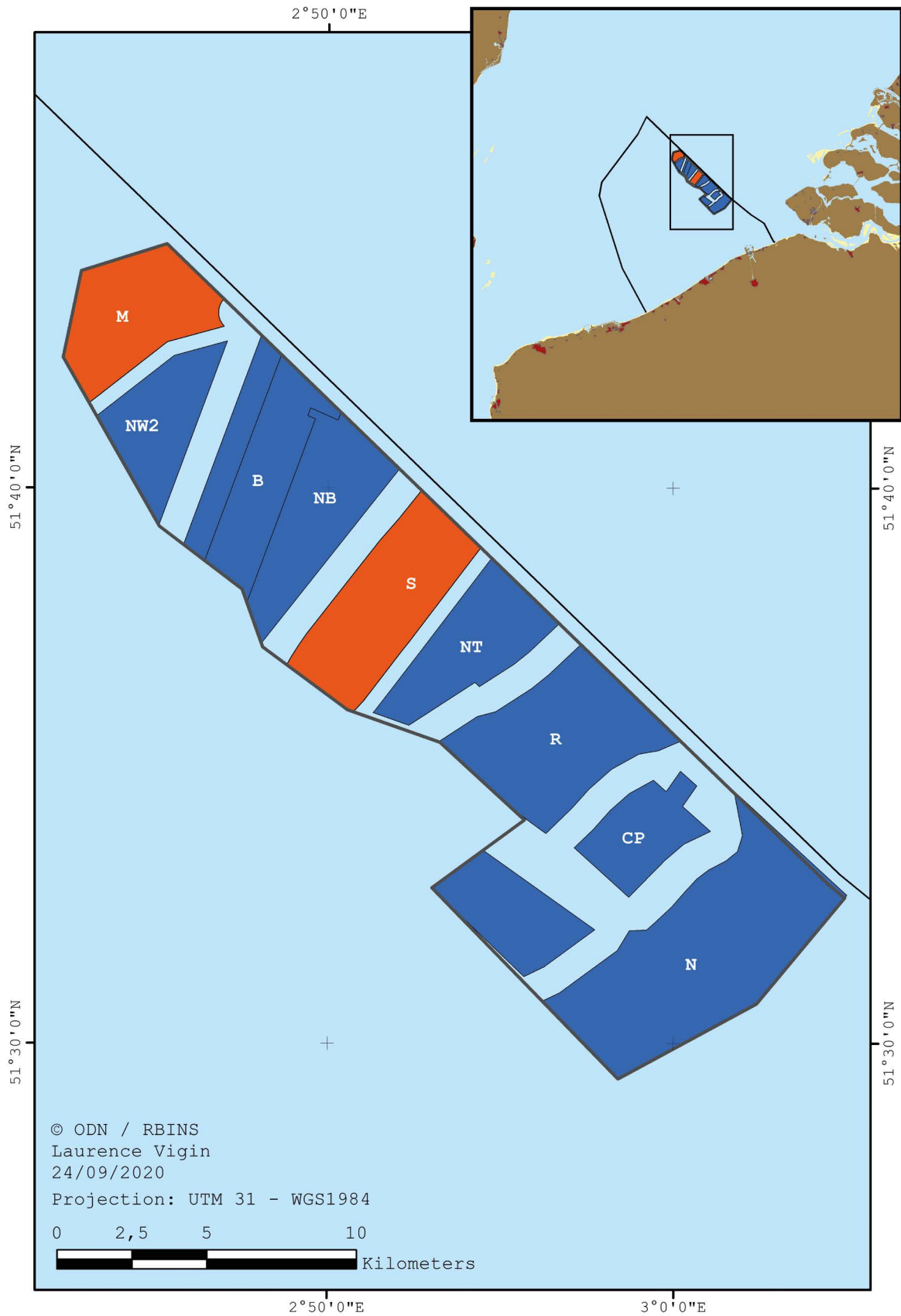


Figure 1. Wind farm concession area in the Belgian part of the North Sea. Blue areas represent the currently operational offshore wind farms (Norther, C-Power, Rentel, Northwind, Nobelwind, Belwind and Northwester 2), while orange areas are wind farms that are under construction (Seastar and Mermaid).

and taxonomic composition around selected turbines based over a 3-year series of consecutive monitoring campaigns. We also zoom in on the temporal patterns in abundance of fine sand-associated macrobenthic species that proliferate close to the turbines and compare them to the temporal patterns in species that naturally occur in the permeable sediments of the Thornton Bank.

2. Materials and methods

2.1. Study area

Sampling in 2019 was conducted in the concession area of the C-Power and Belwind OWFs (fig. 1), but in this report, we will focus on results of the C-Power monitoring only. This OWF is located on the Thornton Bank (TB), approximately 30 km offshore from the Belgian coast. C-Power became fully operational in 2013, and this OWF is composed of 6 gravity-based foundations and 48 jacket foundations (Rumes *et al.* 2017).

Because of unfavourable weather conditions, which precluded the use of the lighter vessel Aquatrot which is needed to approach the turbines until a distance of 37.5 m, the

autumn 2019 campaign had to be split up over several subcampaigns. This is not unusual and has happened before, but the rough weather conditions caused a larger-than-usual time gap of nearly two months between the collection of the ‘far’ and ‘very close’ samples on the Thornton Bank.

On top of the later-than-planned collection of the ‘very close’ samples came the SARS-CoV-2 epidemic which imposed strict measures, including the complete closure of our laboratory facilities at UGent for a period of more than two months. As a consequence, not all samples of the 2019 monitoring campaign have been processed and analysed (see table 1 for overview). However, a representative set of ‘very close’ and ‘far’ samples from the C-Power OWF (jacket foundations) had been analysed in time, and since they matched sampling stations from the previous years, we were able to use a set of ‘very close’ and ‘far’ samples from 12 turbines (see table 1, Thornton Bank samples, and fig. 2: 12 ‘very close’ stations and their corresponding ‘far’ stations were fully processed and included in this analysis) in a spatiotemporal data analysis on both

Table 1. Overview of the sampling moments, stations, vessels, sample numbers and already processed samples (in light blue) collected during the autumn 2019 monitoring. All collected samples are available at the Marine Biology Research Group, UGent

	Date of sampling	Vessel	Station	Number of samples taken	Number of processed samples
Bligh Bank	October 2019	RV Belgica	BB_FAR	36	7
	January 2020	Aquatrot	BB_VC	12	6
			BB_FAR_EXTRA	6	2
Thornton Bank	October 2019	RV Belgica	TB_FAR	32	16
	December 2019	Aquatrot	TB_VC	16	12
Bligh Bank Reference	October 2019	RV Belgica	BB_REF	16	0
Thornton Bank Reference (Goote Bank)	October 2019	RV Belgica	BGR	25	0

BB = Bligh Bank (Belwind OWF), TB = Thornton Bank (C-Power OWF). VC = ‘very close’ samples, taken at a distance of 37.5 m from the centre of a turbine; FAR = samples taken further away from the turbines, at distances between 350 and 500 m.

sediment characteristics and macrobenthos communities.

2.2. Sample design, collection and treatment

The potential effects of turbine presence (C-Power) on macrobenthos communities were tested by conducting spatial and temporal comparisons. For this purpose, samples were collected at two distances from the turbines during autumn 2017 through 2019 on board the vessels Aquatrot (for ‘very close’ samples) and RV Simon Stevin and RV Belgica. ‘Very close’ samples (TB: 16) were taken at approximately 37.5 m from the centre of the turbine, which is the closest distance from the turbine which can be relatively easily sampled without the help of divers, whereas ‘far’ samples (TB: 32) were collected in the middle between four surrounding wind turbines (*i.e.*, at the farthest

possible distance from the nearest turbines), *i.e.* at distances in between 350 and 500 m from any wind turbine (fig. 2).

The samples were collected from the vessels by means of a 0.1 m² Van Veen grab. A plexiglass core (Ø 3.6 cm) was taken from each Van Veen grab sample to collect the environmental data, which included sediment grain size distribution and total sediment organic matter content (TOM). After drying at 60°C, the grain size distribution was measured using laser diffraction on a malvern mastersizer 2000G, hydro version 5.40. Grain size distributions were used to determine the fine sand fraction (125-250 µm) in each sample. Total organic matter (TOM) content was calculated per sample from the difference between dry weight (determined after drying for 48 h at 60°C) and ash-free dry weight (2 h at 500°C).

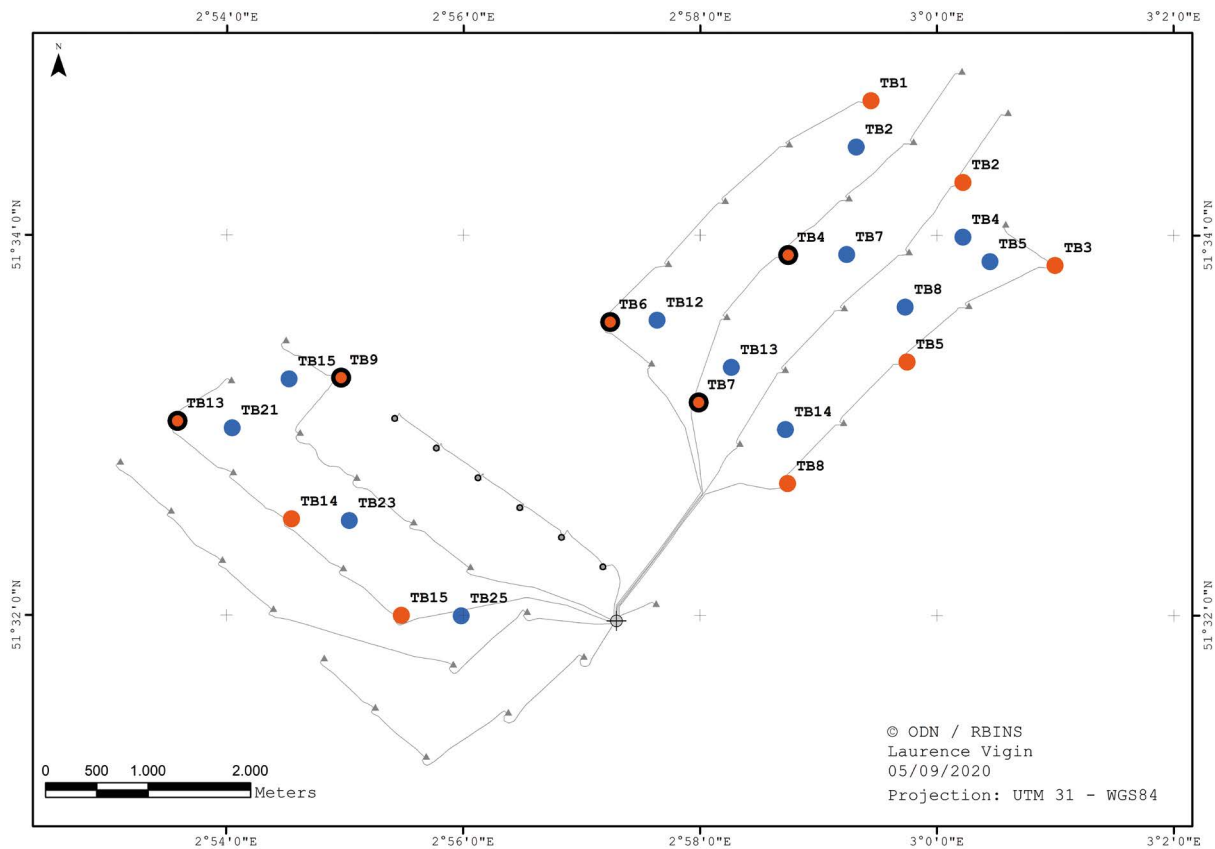


Figure 2. Overview of the selected ‘far’ (350-500 m; blue dots) and ‘very close’ (37.5 m; orange dots) locations at the Thornton Bank used in the present study of spatial and temporal variability. Very close locations highlighted in figs 3 and 4 are presented in this figure as grey dots with orange fill.

The rest of the sample was sieved on board the ship over a 1 mm mesh-sized sieve, and the macrofauna was sorted and preserved in a 4% formaldehyde-seawater solution and stained with Rose Bengal. In the laboratory, organisms were sorted, counted and identified to the lowest possible taxonomic level. Biomass was also determined for each taxon as blotted wet weight (mg). In this report, these taxa are further referred to as species.

2.3. Data analysis

Prior to statistical analysis, the total abundance (ind. m⁻²) and number of species (species richness, S) were calculated from the dataset. Univariate analysis (two-way ANOVA) was performed to assess differences between distances from the turbines ('far' vs 'very close') and years in terms of the above-mentioned biological variables and of the sediment fine sand fraction and TOM content. We used the paired 'very close' and 'far' samples of twelve selected turbines for which the data of three consecutive years were available. Assumptions of normality and homogeneity of variances were tested with a Shapiro-Wilk – and a Levene test, respectively, and log transformations were performed if these assumptions were not met. Tukey HSD post-hoc tests were performed to investigate significant differences in the interaction term Position x Year and/or in the factor Year.

Potential effects of distance and year on macrobenthic community structure were investigated with a Permutational multivariate Analysis of Variance (Permanova), based on a Bray-Curtis resemblance matrix of 4th root transformed data, with a fully crossed fixed two-factor design with factors distance (two levels: 'far' vs 'very close') and year (three levels: 2017 – 2018 – 2019). Because a significant result of a Permanova can be caused by truly significant differences between levels of a factor, but also by a strong heterogeneity among replicate data, homogeneity of multivariate

dispersions was verified using the permutest routine (distances among centroids; Anderson 2006; Anderson *et al.* 2006; in case this test indicated that variances were significantly heterogeneous, we present cautious interpretations of the Permanova results. Similarity Percentages (SIMPER) routine analysis was subsequently performed to identify the contributions of individual species to the dissimilarity between groups of samples (Clarke & Gorley 2006).

Additionally, linear mixed models were used to predict patterns in the densities of selected macrobenthic species in the 'very close' samples based on year, fine sand fraction and TOM content. Potential multicollinearity was verified using a Variance Inflation Factor (VIF). The final model was obtained by stepwise removal of non-significant parameters until all remaining partial regression coefficients were significant. The residuals were then inspected to detect outliers, which were subsequently removed from the models. A Shapiro-Wilk test was used to verify whether residuals were normally distributed.

All analyses were performed in R (version 3.5.2) with the packages *car* (Fox & Weisberg 2019), *vegan* (Oksanen *et al.* 2013) and *RVAideMemoire* (Hervé 2020).

3. Results

The fine fraction of the sediment (125-250 µm) did not vary significantly over the years, but was consistently higher in 'very close' sediments compared to sediments 'far' from the turbine (two-way ANOVA, factor position: $F_{1,66} = 21.383$, $p < 0.001$). While not statistically significant, fig. 3 suggests a trend of increasing fine sand fraction and organic matter content with time at the 'far' stations. The fraction of organic matter in the sediment (TOM) varied significantly among years and distances from turbines (two-way ANOVA, interaction Year x Position: $F_{2,66} = 3.54$, $p = 0.03$), but when investigated in more

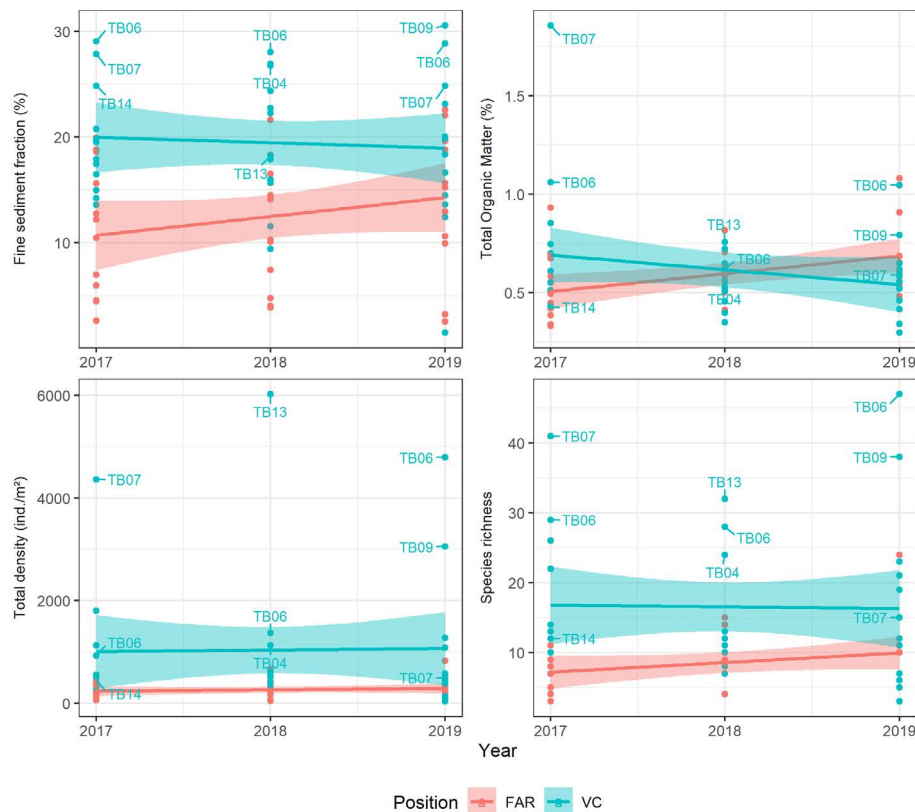


Figure 3. Temporal patterns in the fine sand fraction (125-250 μm ; top-left panel) and the total organic matter content (top-right panel) of the sediment, macrobenthic total densities (bottom-left) and species richness (bottom-right) in stations ‘very close’ to (VC, *i.e.* at 37.5 m) and ‘far’ from (FAR, *i.e.* at 350-500 m) wind turbines. Positive outliers were visually identified and related to the corresponding turbine at which they were recorded.

detail, no clear differences between pairs of data were apparent (all pairwise comparisons within this interaction factor had $p > 0.05$; fig. 3).

Irrespective of the year in which they were sampled, macrobenthic densities and species richness were significantly higher ‘very close’ to the turbines compared to at further distances (two-way ANOVA, factor position: $F_{1,66} = 25.22$, $p < 0.001$ for density and $F_{1,66} = 20.28$, $p < 0.001$ for species richness; fig. 3).

A large scatter was apparent from the patterns in these univariate variables (fig. 3). Throughout the three years of this investigation, consistently higher fractions of fine sediment, higher macrobenthic densities and species richness were recorded ‘very close’ to six out of twelve turbines (see fig. 3).

Whereas macrobenthos density and species richness remained stable over the years, significant temporal patterns were apparent at the level of community structure: the communities ‘very close’ to the jacket foundations were significantly different from those ‘far’ from the turbines, but this difference was dependent on the year (two-way Permanova, interaction Position x Year: $F_{1,71} = 2.03$ $p = 0.03$). The communities ‘very close’ to the turbines were significantly different from their ‘far’ counterparts in 2017 and 2018 (pairwise comparisons: $p < 0.05$), but not in 2019 (pairwise comparison: $p = 0.08$). In contrast to communities ‘far’ from the turbines, those ‘very close’ to the turbines evolved over time: those of 2019 were different from those of 2017 and 2018 (pairwise comparisons: $p < 0.05$). SIMPER analysis

Table 2. Results of SIMPER analysis, showing the species that contributed most (up to a cumulative value of approximately 70%) to the difference in community composition in the sediments ‘very close’ to the turbines on the Thornton Bank between 2017 and 2019. Species highlighted in bold were selected for further temporal analysis; species in black are typically associated with natural permeable sediments in BPNS sandbanks, while those in blue tend to be associated with fine sandy sediments in the BPNS. Untransformed abundance data are shown and cumulative contributions are reported as proportions of 1

	Av. abundance 2017 (ind.m ⁻²)	Av. abundance 2018 (ind.m ⁻²)	Av. abundance 2019 (ind.m ⁻²)	Cumulative sum 2017-2019
<i>Urothoe brevicornis</i>	164.17	182.50	67.50	0.21
<i>Nephtys</i> juv.	22.50	45.00	81.67	0.30
<i>Nephtys cirrosa</i>	65.83	78.33	48.33	0.35
<i>Actinaria</i> sp.	32.50	20.00	26.67	0.39
<i>Terebellidae</i> juv.	12.50	38.33	73.33	0.42
<i>Nototropis swammerdamei</i>	10.83	0.00	47.50	0.46
<i>Bathyporeia elegans</i>	16.67	27.50	20.83	0.49
<i>Spiophanes bombyx</i>	27.50	51.67	12.50	0.52
<i>Nototropis falcatus</i>	0.00	0.00	38.33	0.55
<i>Echinocardium cordatum</i>	10.00	16.67	20.00	0.58
<i>Lanice conchilega</i>	0.00	17.50	75.83	0.60
<i>Abludomelita obtusata</i>	23.33	2.50	32.50	0.62
<i>Eumida sanguinea</i>	6.67	2.50	60.00	0.65
<i>Edwardsia</i> sp.	1.67	0.83	47.50	0.67
<i>Spio</i> sp.	13.33	1.67	18.33	0.69

showed that the abundances of most species encountered in the 2019 samples collected ‘very close’ to the turbines were higher than in 2017 and 2018 (table 2). The exceptions to this rule were the burrowing amphipod species *Urothoe brevicornis*, the interstitial polychaete *Spiophanes bombyx*, and the errant polychaete *Nephtys cirrosa*, all three of which had declined in abundance. The abundance of unidentified *Actinaria* sp. was variable between 2017 to 2019, but this could have resulted from an identifier effect (cf. abundance of the actinarian *Edwardsia* sp. increased in 2019). Also the abundance of the amphipod *Bathyporeia elegans* was variable between the years.

We further investigated patterns in the abundance of the above three species that are typically associated with permeable sandbank sediments on the BPNS, and that declined between 2017 and 2019. We then compared these with patterns in

the abundances of the tube-building polychaete species *Lanice conchilega* and of juvenile Terebellidae, which are probably (mainly) juveniles of *L. conchilega*, as well as of the associated polychaete species *Eumida sanguinea*. *Lanice conchilega* and *E. sanguinea* are typical representatives of low-dynamic habitat and increased markedly in abundance in the sediments close to the turbines between 2017 and 2019 (fig. 4). This increase in Terebellidae and in *E. sanguinea* was mainly driven by local effects at four specific turbines; at these turbines, the increase was statistically significant for *L. conchilega* (linear regression, $\log[Lanice\ conchilega\ density + 1] = -4188 + 2 * Year$; $n = 12$; $R^2_{adj} = 0.6$; $p = 0.002$), but not for *Eumida sanguinea* nor for the juvenile Terebellidae. At the same time, it is important to note the extremely low abundances of these three species in the ‘far’ samples away from the turbines. Except for one turbine, lower abundances of the

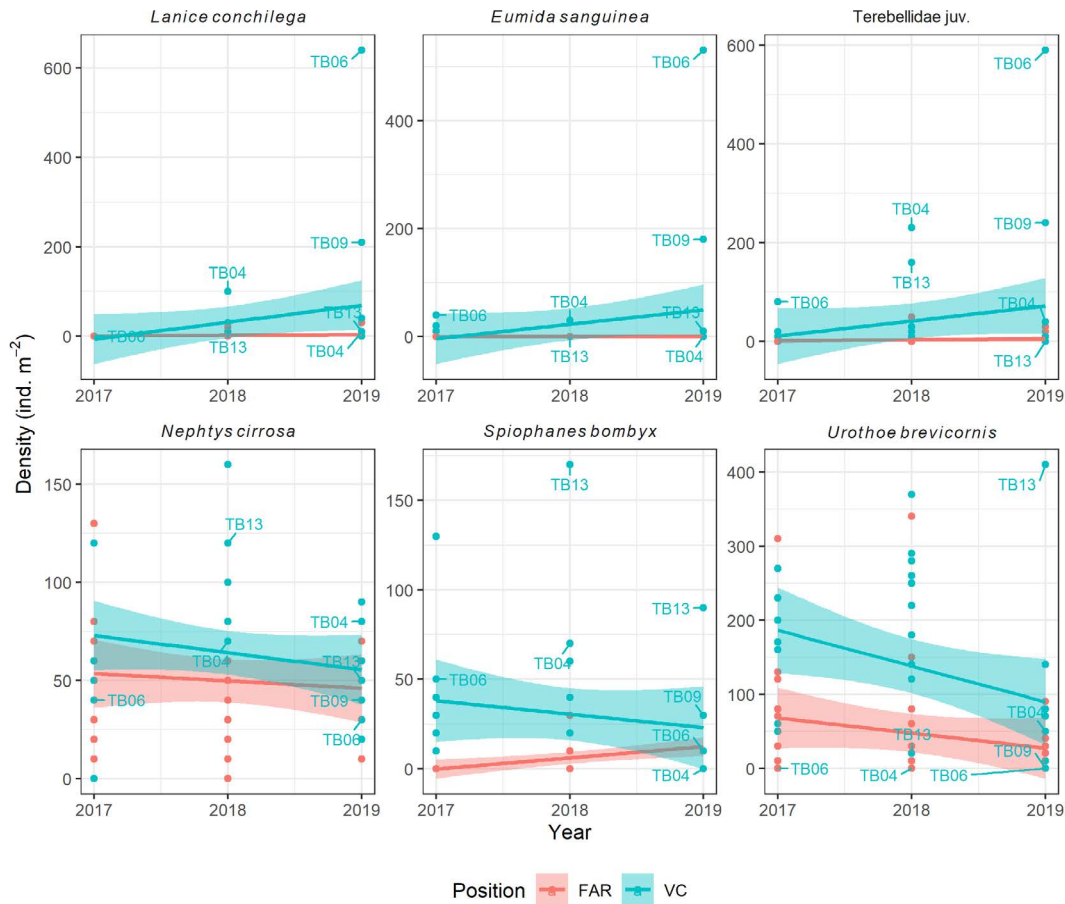


Figure 4. Temporal patterns in the abundances of *Lanice conchilega*, *Eumida sanguinea* and juvenile Terebellidae, and of *Nephtys cirrosa*, *Spiophanes bombyx* and *Urothoe brevicornis* in stations ‘very close’ to (VC, i.e. at 37.5 m) and ‘far’ from (FAR, i.e. at 350 – 500 m) wind turbines. Positive outliers have been visually identified and related to the corresponding turbine at which they were recorded.

species associated with permeable sands were observed at the very same turbines where the species from low-dynamic habitats became more abundant.

The temporal patterns in the abundances of the selected species ‘very close’ to the turbines were primarily dependent on the fraction of fine sediment. The abundances of polychaete species associated with low-dynamic habitats were positively correlated with the fraction of fine sediment. In addition, *L. conchilega* abundances increased over time, whereas abundances of *E. sanguinea* were positively correlated with higher TOM content of the sediment. These models were highly significant and nearly 40% of the variance in the models

was explained by the predictor variables (table 3).

In contrast, only very little variability in the abundances of the species associated with permeable sandbank sediments was explained by the predictor variables used in these models. The densities of the polychaete *N. cirrosa* did not show any dependence on year, fine sediment fraction or TOM content, whereas abundances of the amphipod *U. brevicornis* decreased significantly over the years. The densities of the interstitial polychaete *S. bombyx* slightly increased with the fine sand fraction: 10% of the variance in its abundance was explained by this variable, but the model was only just significant ($p = 0.04$; table 3).

Table 3. Results of linear mixed models on log-transformed abundances of selected macrobenthic species in the sediments very close to the turbines. The partial regression coefficients of the intercept and different predictor variables are shown when significantly different from zero, as well as the model adjusted R² and global p-value

Log-transformed species abundances	Intercept	Year	Fine sediment fraction	TOM	R ² _{adj}	Global p-value
<i>Lanice conchilega</i>	-0.002**	0.9**	0.2**	0	0.38	< 0.001
<i>Eumida sanguinea</i>	0	0	0.09*	2.7**	0.37	< 0.001
<i>Terebellidae</i> juv.	0	0	0.2***	0	0.38	< 0.001
<i>Urothoe brevicornis</i>	2177**	-1*	0	0	0.16	0.01
<i>Spiophanes bombyx</i>	0	0	0.1*	0	0.10	0.04
<i>Nephtys cirrosa</i>	0	0	0	0	0	> 0.05

*** p < 0.001; ** p < 0.01; * p < 0.05

4. Discussion

Artificial hard substrates such as wind turbines can alter local hydrodynamics by modifying local currents: they may increase the current speed on one side of the turbine while reducing it on the other, creating a mixture of more dynamic and more sheltered areas (Coates *et al.* 2014; Rivier *et al.* 2016) and hence a higher spatial heterogeneity. Erosion and resuspension of sediment may occur on that side of the turbine foundation where the current speed increases (Rivier *et al.* 2016), resulting in an increased bed shear stress, in mobilized fine sediment, and hence in an enhanced turbidity in the water column (Baeye & Fettweis 2015). In the wake of the turbine, reduced currents allow the deposition of this fine sediment and a lower seabed shear (Rivier *et al.* 2016), resulting in a sediment matrix that contains a higher proportion of fine particles, a concomitantly lower permeability, and therefore a higher retention of organic matter (Coates *et al.* 2014; De Backer *et al.* 2014). This phenomenon is referred to as the sediment fining hypothesis. The tidal dynamics add a further level of complexity, in that the described pattern is not ‘stationary’ but travels around a turbine with the movement of the tidal ellipses.

In addition to a higher retention of deposited organic matter by less permeable sediments, sedimentary organic matter content can be further enhanced by the depositional flow of the faecal pellets and organic detritus shed by the abundant epifaunal filter feeders that rapidly and massively colonise the turbines (Coates *et al.* 2014; van Deurs 2012). This in turn causes an increased food availability for the benthos near the foundations, and is expected to enhance local macrobenthos biodiversity. This suite of events is collectively referred to as the enrichment hypothesis (Coates *et al.* 2014; De Backer *et al.* 2014; Inger *et al.* 2009).

4.1. Sediment granulometry

and organic matter content provide only partial support for the sediment fining and enrichment hypotheses

The 2019 results show partial evidence of the ‘sediment fining theory’, with a significantly higher fine sand fraction at the ‘very close’ stations on the Thornton Bank compared to the ‘far’ stations (Bruns 2020). This corroborates results from 2017 and 2018 (Lefaible *et al.* 2018, 2019). By contrast, sediment TOM – while indeed being significantly higher in ‘very close’ compared to ‘far’ samples in 2017 – exhibited a different pattern in 2018 and

2019, being even a little higher in the ‘far’ samples. While not shown in this report, this trend was found on both the Thornton and Bligh Bank (Bruns 2020). This trend does not follow from a decreased TOM content in ‘very close’ samples in the two most recent monitoring years, but rather from a tendency towards an increasing TOM content – albeit as yet not statistically significant – in the ‘far’ stations. This trend definitely requires a close follow-up in the coming years; if sustained or even strengthened, it might indicate a spatial expansion of what has hitherto generally been considered to be a very localised impact of turbines on the seafloor.

The operational phase of an OWF entails a complex interplay between abiotic and biotic components, which may also act at partly different spatial scales, rendering a correct interpretation of observed patterns, and hence an accurate prediction of future ones, particularly challenging (Dannheim *et al.* 2020). In addition to changes in hydrodynamics, the epifaunal (mainly filter-feeding) fauna that colonises the turbine foundations (De mesel *et al.* 2015) is likely a key factor in any observed shifts in sediment organic matter content in the immediate vicinity of the turbines, and as such likely contributes to shifts in abundance, diversity and species composition of macrobenthic communities. In turn, shifts in the functional composition of the macrobenthos may further stimulate sediment fining and organic matter retention if species are favoured that enhance the deposition and retention of fine particles from the benthic boundary layer (De Backer *et al.* 2014; Dannheim *et al.* 2020).

It is therefore paramount that the current monitoring is continued in the next years, so that trends in sediment fining and enrichment over time and space can be properly documented. It would be

advisable, though, to better characterise the organic matter in the sediments at different distances from turbines in order to have a clearer picture of its main origin(s), and to trace the fate of particulate organic matter produced by fouling fauna: is it deposited onto the sediment in the wake of the turbines, and if so, within what distance range? To what extent is it consumed by benthic macrofauna and/or decomposed by microbenthos, and to what extent does it eventually end up being buried into these sediments? In August 2020, a targeted monitoring was performed around a single turbine on the Bligh Bank and, in 2021, a similar monitoring is planned around a jacket foundation on the Thornton Bank to sample for sediment granulometry, organic matter and macrobenthos along a more detailed distance gradient from the turbine (*i.e.*, including additional distances on top of the ‘very close’ and ‘far’ from the present monitoring design), in order to obtain a sharper picture of the extent of any sediment fining or enrichment effects. The results of that targeted monitoring may affect the design of future monitoring campaigns. In the meantime, initiatives are taken by Royal Belgian Institute of Natural Sciences and Ghent University to obtain funding to trace the fate of faecal pellets of the dominant epifaunal species into the water column, sediment and fauna around wind turbines.

4.2. Macrobenthos communities around wind turbines exhibit a shift in species composition and diversity

While the data from the past two years hence do not clearly corroborate the enrichment hypothesis in terms of sediment TOM content, enrichment of macrobenthos communities in the ‘very close’ sediments in terms of a higher species richness and overall abundance appear to be two fairly consistent trends over the past three years, although the values fluctuate substantially. In 2017

and 2018, significantly higher macrobenthic densities and species richness were observed in the ‘very close’ compared to the ‘far’ stations (Lefaible *et al.* 2018, 2019). While average species richness in the ‘very close’ stations was even slightly higher in 2019 than in 2018, the high variability among replicate samples rendered the difference with the ‘far’ stations statistically not significant in 2019 (Bruns 2020). Likewise, while very similar macrofaunal abundances in ‘very close’ samples were obtained in 2017 and 2019, being roughly threefold higher than in the ‘far’ samples, this difference was not statistically significant in 2019 as a consequence of high variability among turbines (Bruns 2020).

Similar to abundance and diversity, macrobenthos community composition differed substantially between ‘very close’ and ‘far’ stations, these differences being statistically significant in 2017 and 2018, but not in 2019. Unlike the results on abundance and diversity, macrobenthos species composition in the ‘very close’ samples also differed significantly between years, more specifically between 2019 on the one hand and 2017 and 2018 on the other. SIMPER analysis demonstrated the contributions of the different taxa to these among-year differences in community composition. A closer inspection of table 2 highlights that most of the taxa from the ‘very close’ samples were equally or more abundant in 2019 than in the two previous years. Notable exceptions to this pattern were *Urothoe brevicornis* (Amphipoda), *Spiophanes bombyx* and *Nephtys cirrosa* (both Polychaeta), all three of which declined in 2019. These three species are not just any species; they were the dominant taxa in sediments of the Thornton Bank during a baseline study in 2005, prior to any wind farm construction activity in the area (De Maerschalck *et al.* 2006). *Urothoe brevicornis* consistently remained the most abundant species in 2017 and 2018, whereas *N. cirrosa* ranked among the top-three of most abundant taxa until

2018. In 2019, however, neither of these species still ranked among the three most common taxa in ‘very close’ stations, although some caution is due for *N. cirrosa*, since a relatively high abundance of unidentified juveniles of the genus *Nephtys* was observed in 2019. All three species, *N. cirrosa*, *U. brevicornis* and *S. bombyx*, are key species of a *Nephtys cirrosa* community (Van Hoey *et al.* 2004; De Maerschalck *et al.* 2006; Breine *et al.* 2018), which is typical of well-sorted, medium to coarse, permeable sands in the BPNS. Such community is characterised by a moderate to low species richness (3-12 species per sample) and a rather low abundance (Van Hoey *et al.* 2004; Breine *et al.* 2018). Abundance-wise, the values we observed for total macrofauna were near the upper limit of what has been described for the *Nephtys cirrosa* community, whereas diversity-wise, the number of taxa consistently exceeded typical values for this community over the past three years (Lefaible *et al.* 2018, 2019; this report). It is tempting to suggest that the lower abundances of these three typical inhabitants of permeable sediments in 2019 could be a logical consequence of a sediment fining process. However, the measured environmental variables (*i.e.* fine sand fraction and total organic matter content) explained very little (up to 10% in *S. bombyx*) to no (*N. cirrosa* and *U. brevicornis*) spatiotemporal variation in the abundances of these key species. Hence, other drivers of the population abundances of these species are likely to be at play, perhaps including interactions with other macrobenthos species.

Indeed, the majority of the other macrobenthos species had very similar or (sometimes much) higher abundances in 2019 compared to the previous two years, which, along with the decline in two out of the three previously most common species, resulted in a significantly higher evenness in the 2019 macrobenthos communities ‘very close’ to the turbines (Bruns 2020). Among the taxa that exhibited a marked increase in abundance in 2019 were

Lanice conchilega, juvenile Terebellidae and *Eumida sanguinea* (Polychaeta). Given that juvenile Terebellidae are particularly difficult to identify to species level, and that *L. conchilega* is a terebellid polychaete, it is plausible that part or most of the juvenile Terebellidae belonged to this species. *Lanice conchilega* is a bio-irrigator which contributes to sediment bioturbation and a bio-engineer which, when present in sufficiently high densities, can alter the water flow in the benthic boundary layer, thus affecting bottom shear stress and the deposition and retention of fine particulates (De Backer *et al.* 2014; Alves *et al.* 2017), thereby contributing to both a fining of the sediment texture and an increasing organic matter content in surficial sediment. Indeed, sufficiently dense *L. conchilega* patches can function as small reefs which offer both food and shelter to a variety of benthic fauna (Callaway 2006; Rabaut *et al.* 2009). Their tubes also create habitat which provides a living space for other fauna. *Eumida sanguinea*, for instance, often hides inside *L. conchilega* tubes, provided these are of sufficient size. Hence, its increase in 2019 probably reflects its close association with *L. conchilega* (Callaway 2006; Rabaut *et al.* 2008). Both *L. conchilega* and *E. sanguinea* being typical representatives of fairly fine-grained, low-dynamic habitats, it is not surprising that variability in their abundances could be well explained by the fraction of fine sediment and – for *E. sanguinea* – TOM content. Indeed, models with these two predictor variables significantly explained nearly 40% of the variance in the population abundances of these two species. Any relationship between TOM content and *L. conchilega* abundance could be difficult to interpret, since both options – not mutually exclusive – exist: the species may respond positively to finer sediments with higher TOM content, but may in turn contribute to an increase in TOM in the sediments. The high patchiness – see further – of sediments with *L. conchilega* reefs further adds to the difficulty in finding

statistically significant relationships at the scale and design of our sampling.

The development of *L. conchilega* aggregations represents a strong deviation from the baseline assessment in 2005 (De Maerschalck *et al.* 2006)– together with the declining abundance of previously dominant species typical of permeable sediments, and with the increased macrobenthos diversity – suggests that the macrobenthos communities ‘very close’ to wind turbines exhibit intermediate characteristics between a *Nephtys cirrosa* and an *Abra alba* community, the latter being characteristic for fine to medium sandy sediments in the BPNS (Breine *et al.* 2018). Given that an *Abra alba* community is further characterised by high overall macrofauna abundances and a prominent presence of bivalves (Breine *et al.* 2018; Van Hoey *et al.* 2004), two features which are not (yet) met in the ‘very close’ stations, we can at present not claim that an overall shift is taking place from a *Nephtys cirrosa* to an *Abra alba* community. The development of *Lanice* reefs may in turn negatively affect species like *Urothoe brevicornis* and *Spiophanes bombyx*, two interstitial species which may suffer from space and/or interference competition with tube-building Terebellidae.

However, as for the sediment granulometry and organic matter results, the spatial and temporal variability in macrobenthos communities in terms of abundance, richness and species composition calls for a cautionary interpretation at this stage. Indeed, the macrobenthos of coastal waters is typically characterised by a large year-to-year variability as a consequence of, among other things, climatic conditions (particularly peak winter temperatures and storms). Hence, the current trends need to be closely monitored in future years in order for real patterns to be more conclusively discriminated from largely stochastic variability. In that respect, we cannot exclude that the late sampling of the ‘very close’ stations due to harsh (stormy) weather

conditions in October-November could have affected the 2019 results. Also, while the establishment of a population of a bio-engineering species like *L. conchilega* seems to fit a broader picture in which fining of the sediment facilitates settlement of *L. conchilega*, which in turn contributes to further (fine) sediment accumulation and habitat diversification, an unequivocal elucidation of cause-and-effect relationships and of feedback loops (here for instance between sedimentation and tube building; Alves *et al.* 2017b) in this chain of processes will require dedicated research efforts, as well as a more detailed investigation of the spatial patterns around wind turbines. Moreover, since ecological engineering effects of *L. conchilega* can vary strongly with population dynamics (Alves *et al.* 2017a), evaluating the importance of such feedback loops may prove yet more difficult.

4.3. Macrobenthic communities and sediment granulometry + organic matter content 'very close' to wind turbines exhibit a high degree of patchiness

The above-described development of high abundances of Terebellidae and of *E. sanguinea* 'very close' to wind turbines in 2019 was not a general phenomenon, but essentially occurred near roughly half of the investigated turbines, with particularly high abundances near turbines TB04, TB06, TB09 and TB13. The sediments 'very close' to these turbines were also characterised by some of the highest fine sand fractions, along with those of turbines TB07 and TB14, where *L. conchilega* was not abundantly present. When we focus on the first four turbines, there was a statistically significant increase in *L. conchilega* abundance with time (linear regression, $\log[\text{Lanice conchilega density} + 1] = -4188 + 2 * \text{Year}$; $n = 12$; $R^2_{\text{adj}} = 0.6$; $p = 0.002$). Sediments near three out of these four turbines were also characterised by low abundances of the species associated with permeable sands, supporting the idea that a

shift is taking place from a community typical of permeable, well-sorted sands to one of fine to medium sediments. At the 'far' stations away from the turbines, these same three species were extremely rare or absent, suggesting that if any expansion of the sediment fining and enrichment effects is taking place, it has not yet reached 'far' stations and/or has not attained a level where it attracts species that are characteristic of more sheltered sediments and which may enhance community shifts through their bio-engineering capacity.

The high abundances of *E. sanguinea*, a species which tends to associate with *L. conchilega*, indicate that *L. conchilega* were already forming small reefs near some of the turbines. Such reefs affect the deposition and retention of fine particulate matter from the benthic boundary layer and can therefore have reverberating effects on the shifts in sediment properties and associated macrobenthic communities. At the same time, given their typically small size ($\leq 10 \text{ m}^2$; Rabaut *et al.* 2008), *Lanice* reefs increase the small-scale patchiness of benthic communities and of sediment characteristics. Indeed, the reefs typically occupy $\leq 18\%$ of a suitable habitat/sediment (Rabaut *et al.* 2009). This may in part explain why during the last three years, *L. conchilega* and *E. sanguinea* seem not to consistently occur at the same wind turbines: given their small size and relatively restricted space occupation, *Lanice* reefs may be easily missed by Van Veen grab sampling. The high heterogeneity of sediments around turbines as caused by hydrodynamic properties (Dannheim *et al.* 2020; see section 4.1) can be strongly enhanced at an even smaller spatial scale by the development of bio-engineering fauna.

An additional factor which may contribute to the high variability among 'very close' stations of replicate turbines are mussel clumps which get detached from the foundations and settle onto the sediment, where they can create patches of locally

enriched (with organic matter) sediments that can be attractive to macrobenthic deposit feeders and scavengers, as well as create small-scale habitat heterogeneity (Lefaible *et al.* 2019; Bruns 2020). This phenomenon has repeatedly been observed over the past three years of monitoring.

Given that these sources of small-scale heterogeneity cause large variability in sedimentary and biological properties

around different turbines, they also hamper the detection of robust, statistically underpinned trends. It may therefore be necessary to reconsider the current monitoring design, where a reduction of the number of turbines included could compensate for a higher effort in sampling the local spatial variability at the turbine level, explicitly encompassing *Lanice* patches, *Mytilus* clumps and ‘bare’ sediments.

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