The effects of offshore wind farms on macrobenthic communities in the North Sea

Het effect van offshore windmolenparken op macrobenthos gemeenschappen in de Noordzee

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Summary

For centuries, water and wind were used as main energy resources, rapidly enhancing human civilisation. In the eighteenth century, the introduction of steam engines boosted the industrialisation, soon creating power plants fuelled by coal, oil or natural gas in the late nineteenth century. The production of electricity in the twentieth century mainly relied on these fossil fuels together with the generation of heat from nuclear power plants. However, the creation of nuclear waste, the exhaustion of fossil fuels and the increased emissions of greenhouse gases (e.g. CO_2) into the atmosphere, combined with rising concerns of global warming, led to the pursuit of renewable energy resources. In order to reduce greenhouse gas emissions, Europe enforced renewable energy target figures upon its member states through the implementation of two main European Directives 2001/77/EC and 2009/28/EC. Offshore wind farms (OWFs) have become the most widespread offshore renewable energy developments in Europe with 69 operational wind farms at the end of 2013. In Belgium, three OWFs have been installed, introducing numerous hard substrate foundations to a naturally sandy habitat.

Effects to the macrobenthos (invertebrate fauna retained on a 1 mm sieve) inhabiting the sediments around OWFs in the Belgian part of the North Sea (BPNS) were assessed in this PhD thesis. Untill recently, sand extraction, dredging and beam trawl fisheries were the three main human activities affecting the soft sediment macrobenthos in the BPNS. The introduction of an additional anthropogenic impact, affecting the physical characteristics of the seabed in the BPNS, raises concerns on how the local marine biodiversity will be affected. A Before After Control Impact (BACI) design was applied to assess the situation of the environment before and after the construction of OWFs, in comparison to a selected reference site. Through a multi-scale approach, large-scale effects associated with the construction and operational phases of OWFs were identified together with small-scale, process related research in the direct environment of one constructed foundation.

The first OWF was constructed in 2008 on the Thorntonbank with six gravity based foundations (GBFs). To distinguish natural from anthropogenic related fluctuations in the macrofaunal community, a long term analysis was first carried out in **Chapter 2**. Data was gathered from 1980 – 2012 on the Thorntonbank and Goote Bank (reference site). Both sandbanks were characterised by coarse, sandy sediments (331 ± 20 µm – 410 ± 41 µm) with a low species abundance (180 – 812 ind m^{-2}) and diversity (6 - 15 species per sample). Together with a dominance of the polychaetes *Nephtys cirrosa* and the amphipod *Urothoe brevicornis*, the macrobenthic community was classified into the

transitional *N. cirrosa* and *Ophelia borealis – Glycera lapidum* community. Throughout 32 years, the macrobenthos mainly illustrated strong temporal variations, which were related to the variable weather conditions in the area (e.g. cold winters and severe storms). The communities on the Thorntonbank and Goote Bank always illustrated a similar fluctuation. However, this trend was interrupted with the construction of the six GBFs in 2008. A significant difference in macrobenthic community composition was observed between the Thorntonbank and Goote Bank, recovering from 2009 onwards. The impacts of the construction (dredging) activities were noticeable with short-term changes to the macrobenthos. However, a rapid recovery was observed thereafter confirming that the macrobenthic communities in these areas have created a high resilience to environmental and anthropogenic stress. Furthermore, the Benthic Ecosystem Quality Index (BEQI) was assessed as an indicator in this chapter and reflected similar results.

Within the OWF concession areas, an overall prohibition to shipping activities (including fishing vessels) has been implemented with a 500 m safety zone around every operational OWF. Beam trawling has a direct physical impact on the seabed by scraping the first 3 – 6 cm, re-suspending sediments and removing or damaging non-targeted benthos. With the installation of 55 monopile foundations on the Bligh Bank in 2009, a first large area (21 km²) was closed to fishing activities in the BPNS. To investigate the potential recovery of vulnerable species, the macrobenthic community within the fishery enclosed area was compared with a surrounding control area in **Chapter 3**, two to three years after the construction of the OWF. Regular fishing activities around the OWF were registered through Vessel Monitoring System (VMS) data by the Institute for Agricultural and Fisheries Research (ILVO). Three years after the construction of the OWF on the Bligh Bank, subtle changes to the species composition were observed in the No Fishery area. The benthic mysid shrimp *Gastrosaccus spinifer* (30 ± 15 ind m^{-2}), tube-building polychaetes Terebellidae sp. (196 ± 151 ind m^{-2}) and the echinoderm *Echinocyamus pusillus* (73 ± 71 ind m^{-2}), sensitive to trawling activities, showed an increased abundance in the No Fishery area. Regular illegal fishing activities within the OWF were registered through VMS data and visual observations, possibly slowing down the recovery rates of these fragile organisms. Nevertheless, there is a possibility that the area could develop into an ecologically important habitat.

After downsizing to a smaller scale around one GBF on the Thorntonbank, drastic changes to the seabed characteristics and macrobenthic community composition were rapidly observed and documented in **Chapter 4**. Along four gradients and five distances around the GBF, the macrobenthos and the main characteristics of the permeable, sandy sediments were investigated. Three to four years into the operational phase, changes were detected up to a 50 m distance away from the scour protection system around the GBF. Sediment grain size significantly reduced from 427

µm at 200 m to 312 ± 3 µm at 15 m from the foundation along the south-west and north-west gradients. The organic matter content increased from 0.4 ± 0.01 % at 100 m to 2.5 ± 0.9 % at 15 m from the foundation. The observed changes in seabed characteristics possibly caused the increased macrobenthic density from 1390 ± 129 ind m^{-2} at 200 m to 18583 ± 6713 ind m^{-2} at 15 m. The community was dominated by the juvenile starfish (*Asterias rubens*) together with the tube-building polychaetes *Lanice conchilega* and *Spiophanes bombyx,* evolving away from the naturally occurring transitional *N. cirrosa* and *O. limacina – G. lapidum* community. The occurrence of dense *L. conchilega* patches and an increased macrobenthic abundance can enhance the food availability for demersal fish species inside the OWF. Together with the additional shelter from strong currents around the foundations, the areas close to the turbines could develop into important refuge areas for juvenile fish species in the future.

The decreased grain size, measured around the GBF, will reduce the permeability of the seabed. In the coarse, permeable sediments, pressure driven advective pore-water flows at the sediment-water interface, accelerate the mineralisation of organic matter and the recycling of nutrients. To determine how a decreasing permeability would affect the benthic ecosystem functioning around OWFs, an experimental pilot study was carried out in **Chapter 5**. Two experiments were set up in benthic chambers containing three sediment types with a decreasing permeability (coarse, intermediate and fine). The filtration capacity of the three sediment types was assessed by measuring the water penetration depth after adding a dye and by counting the reduction of diatom cells (*Skeletonema costatum*) in the water column during a second experiment. The water penetration depth decreased from 6.5 ± 0.2 cm in the coarse sediment to 0.8 ± 0 cm in the fine sediment. A similar trend was observed after adding the chain-forming *S. costatum* cells. Additionally, sediment community oxygen consumption (SCOC) rates and nutrient fluxes were measured before and after the addition of the diatom cells (organic matter). The advective pore-water flow in the coarse sediment facilitated the mineralisation of added organic matter (diatom cells) as reflected by the high SCOC rates (23.7 ± 1.9 mmol O_2 m^{-2} d^{-1}). Lowest SCOC rates were measured in the fine sediment 4.8 ± 0.1 mmol O_2 m^{-2} d^{-1} where solute exchange mainly relies on molecular diffusion. Low mineralisation of organic matter was also observed in the refined sediments, with a reduction in the NH_4^+ (ammonium) flux (from 834 ± 100 µmol m^{-2} d^{-1} to 228 ± 782 µmol m^{-2} d^{-1}) and NO_x (nitrate + nitrite) flux (from 239 ± 156 µmol m^{-2} d^{-1} to 96 ± 31 µmol m^{-2} d^{-1}) into the water column. The results suggest that the reduction in sediment grain size around the foundation could alter the benthic ecosystem functioning and mineralisation processes of organic matter.

In conclusion, this PhD research has demonstrated that the construction and presence of new hard substrates in a sandy environment has the potential to cause environmental changes to the marine ecosystem, possibly creating a new habitat within the BPNS. With the expansion of OWFs to a concession area of 238 km² in the longer term, a large area of the BPNS will be subjected to the presence of many hard substrate foundations with associated hydrodynamic changes and organic matter enrichment. Together with the prohibition of beam trawl fisheries, an evolution to a large ecologically rich and complex habitat is realistic. However, the increasing anthropogenic activities in the BPNS could also lead to a biodiversity loss resulting in a homogenisation of the environment. A few studies have modelled the ecological evolution from changes to the macrobenthic community on a small-scale to abrupt, benthic regime shifts on a larger scale. The incorporation of benthic indicators (e.g. BEQI) in monitoring studies can be helpful to quickly detect these modelled regimes shifts in the future and discriminate them from natural fluctuations.

Furthermore, future monitoring programmes should contain cross-border collaboration between European countries with a research driven, adaptive approach in order to fully understand the ecosystem processes around OWFs at different spatial scales. This approach will further assist managers, policy makers and wind farm developers to take appropriate mitigating measures if needed and generate an adaptive spatial planning on a regional scale such as the North Sea.

Samenvatting

Eeuwenlang gebruikte de mens water en wind als voornaamste energie bronnen, wat de ontwikkeling van onze maatschappij versnelde. De uitvinding van de stoommachine in de achttiende eeuw markeerde het begin van de industriële evolutie, en aan het eind van de negentiende eeuw ontwikkelde men krachtcentrales die energie opwekten door middel van fossiele brandstoffen zoals steenkool, olie en natuurlijk gas. De elektriciteitsproductie in de twintigste eeuw was vooral afhankelijk van deze fossiele brandstoffen samen met het genereren van warmte uit nucleaire kerncentrales. De toename van nucleair afval, de uitputting van fossiele brandstoffen en de verhoging van atmosferische broeikasgassen (bv. CO_2) gecombineerd met de verontrustende klimaatsverandering resulteerde in een zoektocht naar hernieuwbare energiebronnen. Om de uitstoot van broeikasgassen terug te dringen stelde Europa nieuwe doelstellingen voor hernieuwbare energie aan de hand van twee Europese richtlijnen 2001/77/EC en 2009/28/EC. Offshore windmolenparken werden snel de meest wijdverspreide offshore hernieuwbare energie ontwikkeling in Europa met 69 operationele parken tegen eind 2013. In België werden drie offshore windmolenparken geïnstalleerd, wat gepaard ging met de introductie van tientallen harde substraat funderingen in een natuurlijk zanderig habitat.

Effecten op het macrobenthos (ongewervelde fauna groter dan 1mm), die de sedimenten rond de windmolenparken in het Belgisch deel van de Noordzee (BDNZ) bewonen, werden in deze doctoraatsthesis geëvalueerd. Tot nu toe waren zandextractie, baggerwerken en boomkorvisserij de drie voornaamste menselijke activiteiten die het zachte substraat macrobenthos in het BDNZ beïnvloedden. De invoering van een extra antropogeen impact en de invloed op de fysische kenmerken van de zeebodem, verhoogt de bezorgdheid over het effect op de lokale macrobenthische biodiversiteit. Een Before After Control Impact (BACI) ontwerp werd toegepast om de omgevingssituatie vóór en na de constructie van de offshore windmolenparken te bepalen, in combinatie met een vergelijking met een geselecteerde referentieplaats. Via een multi-schaal benadering werden grootschalige effecten van de constructie en operationele fase van de offshore windmolenparken bepaald, samen met een kleinschalige, meer proces gerelateerd onderzoek in de directe omgeving van één geïnstalleerde fundering.

Het eerste offshore windmolenpark werd geïnstalleerd in 2008 op de Thorntonbank met zes gravitaire funderingen. Om de natuurlijke fluctuaties in de macrobenthische gemeenschap van de antropogeen gerelateerde te onderscheiden werd een lange termijn analyse uitgevoerd in **Hoofdstuk**

2. Data werden verzameld van 1980 – 2012 voor de Thorntonbank en Goote Bank (Referentieplaats). Beide zandbanken werden gekenmerkt door grof, zanderig sediment (331 ± 20 µm – 410 ± 41 µm) met een lage soortendensiteit (180 – 812 ind m^{-2}) en diversiteit (6 – 15 soorten per staal). Samen met een dominantie van de polychaet *Nephtys cirrosa* en de amphipode *Urothoe brevicornis*, werd de macrobenthische gemeenschap geclassificeerd tot de overgangsgemeenschap van de *N. cirrosa* en *Ophelia borealis – Glycera lapidum* gemeenschap.
Gedurende 32 jaar vertoonde het macrobenthos vooral een sterke temporele variatie, gerelateerd aan de variabele weersomstandigheden van dit gebied (bv. koude winters en zware stormen). De gemeenschappen op de Thorntonbank en Goote Bank vertoonden steeds een gelijkaardige fluctuatie. Deze trend werd onderbroken bij de installatie van de zes gravitaire funderingen in 2008.
Een significant verschil in macrobenthische gemeenschapssamenstelling werd vastgesteld tussen de Thorntonbank en de Goote Bank, met een herstel vanaf 2009. Het effect van de installatie werken (baggeren) was merkbaar door korte termijn veranderingen in het macrobenthos. Niettemin werd daarna een snel herstel waargenomen, wat bevestigt dat de macrobenthische gemeenschappen in deze gebieden een hoge veerkracht hebben ontwikkeld tegen zowel omgevingsstress als antropogene stress. De benthische ecosysteem kwaliteitsindex BEQI werd onderzocht als indicator en vertoonde gelijkaardige resultaten.

Binnen de concessie gebieden geldt een algemeen scheepvaartverbod (inclusief visserij) met een 500 m veiligheidszone rondom de operationele windmolenparken. Boomkorvisserij heeft een direct effect op de zeebodem door het afschrapen van de bovenste 3 – 6 cm, de resuspensie van sedimenten en het beschadigen of verwijderen van niet doelgerichte benthos. Met de installatie van 55 monopile funderingen op de Bligh Bank in 2009 werd voor het eerst een groot gebied (21 km²) gesloten voor vissersvaartuigen in het BDNZ. Om een potentieel herstel van kwetsbare soorten te onderzoeken, werd in **Hoofdstuk 3** de macrobenthische gemeenschap in het gebied gesloten voor visserij, vergeleken met een omliggend controle gebied, twee tot drie jaar na de installatie van het windmolenpark. Visserijactiviteit rond de windmolenparken werd geregistreerd via Vessel Monitoring Systeem (VMS) data door het Instituut voor Landbouw en Visserij Onderzoek (ILVO). Drie jaar na de installatie van het windmolenpark op de Bligh bank werden subtiele veranderingen in de macrobenthische gemeenschap waargenomen in het gebied gesloten voor visserij. De benthische aasgarnaal *Gastrosaccus spinifer* (30 ± 15 ind m^{-2}), koker bouwende polcyhaet Terebellidae sp. (196 ± 151 ind m^{-2}) en de echinoderm *Echinocyamus pusillus* (73 ± 71 ind m^{-2}), kwetsbaar voor boomkorvisserij, vertoonden een verhoogde densiteit in dit gebied. Regelmatig werden illegale vissersvaartuigen binnen het windmolenpark geregistreerd via VMS data en visuele observaties wat het herstel van de kwetsbare soorten kan vertragen. Niettemin bestaat er de mogelijkheid dat het gebied naar een ecologisch belangrijke habitat zal evolueren.

Op een kleinere schaal, rondom één gravitaire fundering op de Thorntonbank, werden drastische veranderingen van de zeebodemkenmerken en macrobenthische gemeenschapssamenstelling waargenomen en gedocumenteerd in **Hoofdstuk 4**. Langs vier gradiënten en op vijf afstanden werd het macrobenthos en de belangrijkste kenmerken van de permeabele, zanderige sedimenten onderzocht. Drie tot vier jaar binnen de operationele fase werden veranderingen aangetroffen tot 50 m verwijderd van de erosiebescherming rondom de fundering. De korrelgrootte van het sediment verminderde merkelijk van 427 µm op 200 m tot 312 ± 3 µm op 15 m van de fundering langs de zuidwestelijke en noordwestelijke gradiënten. Het organisch materiaalgehalte verhoogde van 0.4 ± 0.01 % op 100 m tot 2.5 ± 0.9 % op 15 m van de fundering. De waargenomen veranderingen in sedimentkenmerken veroorzaakten een verhoging in macrobenthische densiteit van 1390 ± 129 ind m^{-2} op 200 m tot 18583 ± 6713 ind m^{-2} op 15 m. De gemeenschap werd gedomineerd door de juveniele zeester (*Asterias rubens*) samen met de koker bouwende polychaeten *Lanice conchilega* en *Spiophanes bombyx.* Hieruit konden we besluiten dat de gemeenschap weg evolueert van de natuurlijk voorkomende overgangsgemeenschap van *N. cirrosa* en *O. borealis* – *G. lapidum*. Het optreden van dichte *L. conchilega* aggregaties en een verhoging van de macrobenthische densiteit kan de voedselbeschikbaarheid voor demersale vissoorten verhogen binnen het windmolenpark. Samen met de extra beschutting tegen sterke stromingen rondom de fundering, zouden de zones dichtbij de wind turbines kunnen uitgroeien tot belangrijke refuge gebieden voor juveniele vissoorten in de toekomst.

De verfijning van de sedimenten, geobserveerd rondom de gravitaire fundering, zal de permeabiliteit van het sediment verlagen. In grove, permeabele sedimenten, ontstaan er door druk gedreven advectieve poriewaterstromingen aan het sediment-water oppervlak. Deze poriewaterstromingen zullen de mineralisatie van organisch materiaal en het recycleren van nutriënten versnellen. Een verkennend proefonderzoek, om te bepalen hoe de gereduceerde permeabiliteit de werking van het benthische ecosysteem zou veranderen rond offshore windmolenparken, werd uitgevoerd in **Hoofdstuk 5**. Twee experimenten werden opgesteld in benthische kamers die drie sedimenttypes met een dalende permeabiliteit bevatten (grof, matig en fijn). De filtratiecapaciteit van de drie sedimenttypes werd bepaald door de water penetratiediepte te meten, na toevoeging van een kleurstof en het aantal gereduceerde diatomeeëncellen (*Skeletonema costatum*) in de waterkolom te tellen tijdens een tweede experiment. De water penetratiediepte verminderde van 6.5 ± 0.2 cm in het grof sediment tot 0.8 ± 0 cm in het fijn sediment. Een gelijkaardige trend werd vastgesteld na toevoeging van de ketenvormende *S. costatum* cellen. Bovendien werden de sediment gemeenschap zuurstofconsumptie snelheden en fluxen van nutriënten gemeten, vóór en na het toevoegen van de diatomeeëncellen (organisch materiaal). De advectieve poriewaterstromingen in het grof sediment vergemakkelijkten de mineralisatie van de toegevoegde organische materie (diatomeeën), wat tot uiting kwam in de hoge zuurstofconsumptie snelheden (23.7 ± 1.9 mmol O_2 m^{-2} d^{-1}). Lage zuurstofconsumptie snelheden werden gemeten in het fijn sediment (4.8 ± 0.1 mmol O_2 m^{-2} d^{-1}) waar de uitwisseling van opgeloste stoffen vooral afhangt van moleculaire diffusie. Lage mineralisatie van organisch materiaal werd ook geobserveerd in de verfijnde sedimenten met een reductie in NH_4^+ (ammonium) fluxen (van 834 ± 100 µmol m^{-2} d^{-1} tot 228 ± 782 µmol m^{-2} d^{-1}) en NO_x (nitraat + nitriet) fluxen (van 239 ± 156 µmol m^{-2} d^{-1} tot 96 ± 31 µmol m^{-2} d^{-1}) in de waterkolom. Deze resultaten suggereren dat de werking van het benthisch ecosysteem en de mineralisatie van organisch materiaal kan gewijzigd worden door een verfijning van het sediment rondom de funderingen van offshore windmolenparken.

Tot slot, heeft dit doctoraatsonderzoek vastgesteld dat de installatie en aanwezigheid van nieuwe harde substraten in een zanderig habitat, potentieel heeft om omgevingsveranderingen in het marien ecosysteem te veroorzaken wat mogelijk een nieuw habitat kan creëren in het BDNZ. Met de uitbreiding van het concessiegebied voor offshore windmolenparken naar 238 km² op lange termijn, zal een groot deel van het BDNZ onderworpen worden aan de aanwezigheid van veel harde substraten met gerelateerde stromingsveranderingen en aanrijking van organisch materiaal. Samen met het verbod voor boomkorvisserij zal het volledige concessiegebied kunnen evolueren naar een ecologisch rijk en complex habitat. Doch, kan een verhoging van antropogene activiteiten in het BDNZ ook leiden tot een biodiversiteitsverlies wat een homogeniteit van het gebied kan veroorzaken. Een aantal studies hebben de ecologische evolutie van veranderingen in de macrobenthische gemeenschap op kleine schaal gemodelleerd naar abrupte, benthische regime shiften op grotere schaal. Het opnemen van benthische indicatoren (bv. BEQI) in monitoringsstudies kan nuttig zijn om de gemodelleerde regime verschuivingen in de toekomst te detecteren en te onderscheiden van natuurlijke fluctuaties.

Bovendien is het wenselijk dat toekomstige monitoringsprogramma's een grensoverschrijdende samenwerking aanmoedigen tussen Europese landen met een wetenschappelijk gestuurde en adaptieve benadering, zodat we de ecosysteemprocessen rond offshore windmolenparken kunnen begrijpen in verschillende ruimtelijke schalen. Deze benadering zal ertoe bijdragen dat beheerders, beleidsmakers en windmolenparkontwikkelaars passende mitigerende maatregelen treffen indien nodig, en een aangepaste ruimtelijke planning scheppen op een regionaal gebied zoals de Noordzee.

CHAPTER 1

General Introduction

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1. Context setting of offshore renewable energy

Since ancient times, mankind used water and wind as their main energy resource which accelerated and enhanced human civilisation. With the construction of waterwheels, wind mills and sailing ships many cities and transportation networks were created. The introduction of steam engines in the eighteenth century enhanced industrialisation. Soon enough, steam engines ran on wood and coal, evolving into large power plants fuelled by coal, oil or natural gas in the late nineteenth century. The production of electricity in the twentieth century mainly relied on these fossil fuels together with the generation of heat from nuclear power plants (Asif and Muneer, 2007). However, the creation of radioactive nuclear waste (Nieto-Perez, 2013), the exhaustion of fossil fuels (Hubbert, 1956; Shafiee and Topal, 2009) and the increased emissions of greenhouse gases (e.g. CO_2) into the atmosphere combined with rising concerns of global warming (Dickinson and Cicerone, 1986; Zecca and Chiari, 2010) led to the pursuit of renewable energy resources. With the implementation of European Directive 2001/77/EC, Europe enforced renewable energy target figures to every member state by 2010 in order to reduce greenhouse gas emissions. Consequently, Europe has become one of the world leaders in realising renewable energy production over the past ten years. The recent Renewable Energy Directive 2009/28/EC obliges Europe to produce 20 % renewable energy of the total energy consumption and 10 % renewable energy for transport by 2020, again setting mandatory member state targets (EuropeanCommission, 2013). In 2012, Europe reached a renewable energy production of 13 % and a decrease in greenhouse gas emissions with 18 % compared to emissions in 1990 due to additional mitigating measures (EuropeanCommission, 2014).

Renewable energy sources range from solar and wind generated energy to biofuels produced from energy crops, agricultural waste or biomass (Asif and Muneer, 2007; Brennand, 2004). As a result of these high European targets and the “not in my backyard” opinion of most citizens, countries soon turned to their coastal waters to produce renewable energy out at sea. Offshore wind power (Corbetta et al., 2013), wave energy (Lopez et al., 2013), tidal energy (Ben Elghali et al., 2007) and power, generated from ocean currents (Tucker, 2007) or ocean thermal energy conversion (Pelc and Fujita, 2002; Rajagopalan and Nihous, 2013) have the potential to generate electricity and can assist to meet the renewable energy targets. Wave and tidal energy prototype devices were first established in the United Kingdom off the Orkney Islands (Scotland) and have since then increased throughout the UK and Europe (Melo et al., 2013). The first operational wave and tidal energy farms are scheduled for 2016/17 in the UK (RenewableUK, 2013). By far, Offshore Wind Farms (OWFs) are the prominent offshore renewable energy production systems in Europe. Across 11 countries, a total of 2080 wind turbines in 69 OWFs were fully operational at the end of 2013 (Corbetta et al., 2014). A total capacity of 6562 MW or 0.7 % of Europe’s electricity consumption was generated and is expected to reach 3 GW by 2015. With 56 % of the installed capacity, the UK is the leading country followed by Denmark (19 %) and Belgium (9 %). By 2017, the offshore wind farm industry is expected to expand further towards deeper waters (Arapogianni and Genachte, 2013), creating floating structures in water depths exceeding 120 m (compared to an average depth of 20 m at present).

2. Offshore wind farms in the Belgian part of the North Sea

The Renewable Energy Directive 2009/28/EC requires Belgium to reach a 13 % renewable energy contribution of the final energy consumption by 2020. With an estimated contribution of 4.1 % in 2011, Belgium successfully met the first interim target (Eurostat, 2013). The implementation of offshore wind energy will vastly assist in achieving the renewable energy targets by 2020. The Royal Decree of 20 December 2000 (amended by the Royal Decree of 17 May 2004) designated the offshore wind farm concession area (238 km²) in the Belgian part of the North Sea (BPNS) (Fig. 1). The spatial policy for OWFs in the BPNS has been implemented into the draft of the Belgian Marine Spatial Plan (KoninkrijkBelgië, 2013).

Table 1. Timeline of the Offshore wind farm installation in the Belgian part of the North Sea

2008	2009 - 2010	2011 - 2012	2013 - 2014	
6 gravity based foundations	55 monopile foundations	48 jacket foundations	72 monopile foundations	1 monopile Demo turbine
C-Power (Thorntonbank)	Belwind (Bligh Bank)	C-Power (Thorntonbank)	Northwind (Lodewijkbank)	Belwind-Alstom (Bligh Bank)

An overall prohibition on shipping (including fishing vessels) has been implemented with a 500 m safety zone around every operational offshore wind farm. In 2014, Belgium will have three operational OWFs in the BPNS (C-Power, Belwind and Northwind) together with four additional granted domain concessions (Norther, Rentel, Seastar and Mermaid). At present, a total wind capacity of 680 MW has been installed. Table 1 provides a timeline overview of the construction phases of the Belgian OWFs from 2008 – 2014.

The domain concessions are granted for 20 years, dividing the OWF installation into three phases: the construction, operational and decommissioning phase.

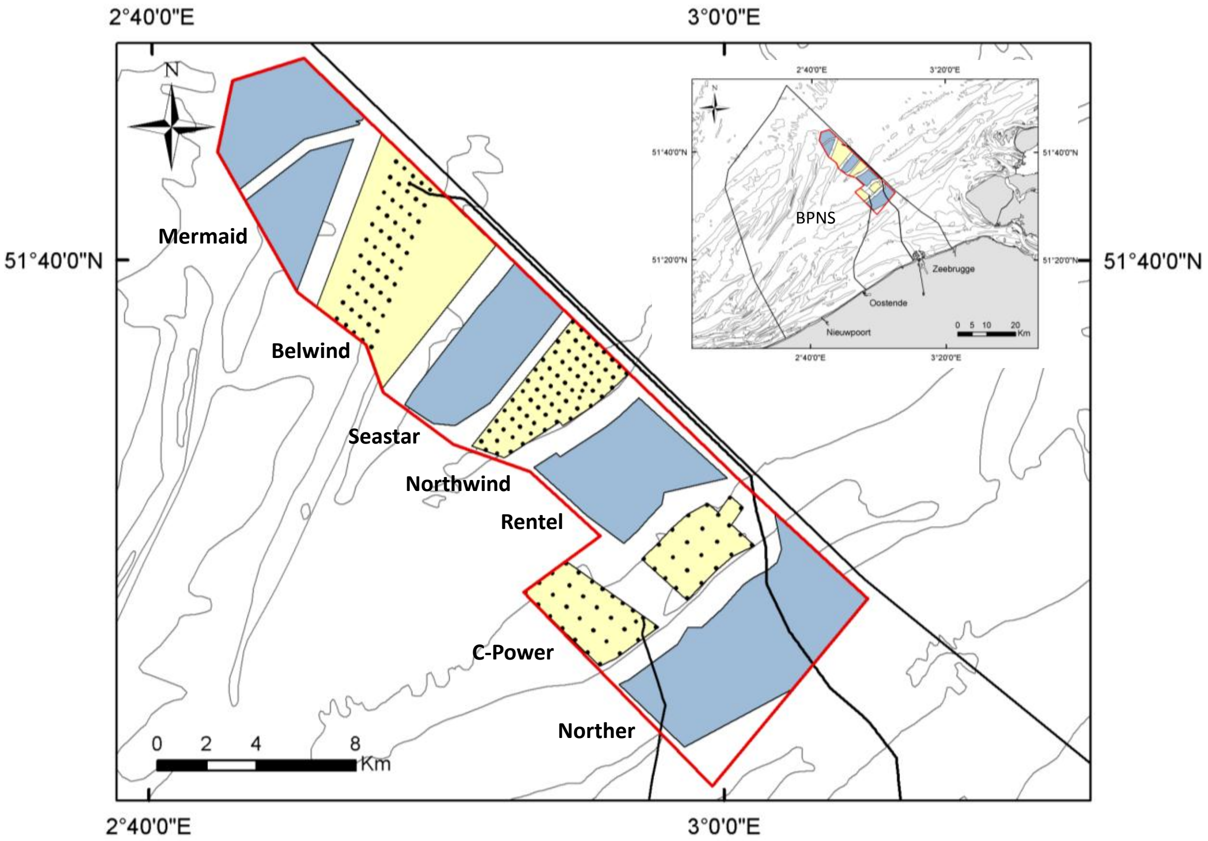

Fig. 1 Wind farm concession area (red area) in the Belgian part of the North Sea. Three offshore wind farms have been constructed (Yellow areas) on the Thorntonbank (C-Power), Bligh Bank (Belwind) and Lodewijkbank (Northwind). Two power cables from C-Power and Belwind to the shoreline. Four additional domain concessions have been granted to Norther, Rentel, Seastar and Mermaid (Blue areas). Wind turbines marked as black dots.

The OWF developers have used three different foundation types in the BPNS: gravity based, jacket and monopile foundations (Fig. 2) each with different (pre-) construction related activities such as dredging or pile driving:

- Gravity based foundations (GBFs) are large, hollow, concrete foundations, constructed onshore, transported to offshore locations and placed on the seabed (Peire et al., 2009). The foundations are then filled with sand for ballast and stabilisation. A scour (erosion) protection system is placed around the foundation consisting of a filter layer (crushed gravel) and an upper armour layer (quarried rock). Before construction, the seabed is thoroughly prepared and disturbed as dredging activities level off the seabed to create foundation pits (Brabant and Jacques, 2010).

- Jacket foundations are steel structures which consist of four legs. Slightly thinner pin-piles are driven into the seabed, which are then connected to the four steel legs of the foundation. No scour protection system has been applied around the jacket foundations in the BPNS. Seabed preparation activities are carried out to remove loose sand dunes and level off the surface (Brabant et al., 2011).

- Monopile foundations are the most widespread in European waters covering 76 % of the installed foundations (Corbetta et al., 2014). Monopiles consist of steel piles which are driven into the seabed and connected to the turbine through a transition piece (Belwind, 2011). Erosion pits are backfilled with a scour protection system. Limited seabed preparation activities are applied for this construction design.

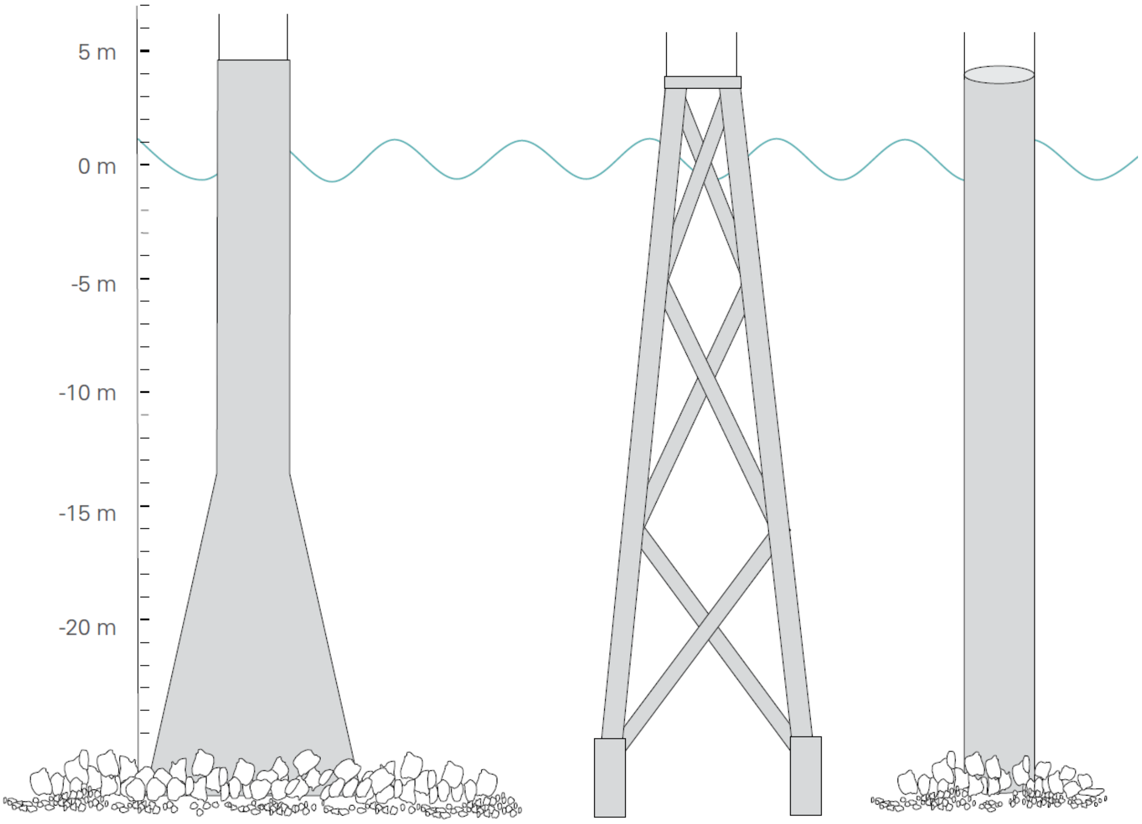

Fig. 2 Three foundation types present in the Belgian part of the North Sea, from left to right: gravity based, jacket and monopile foundation (Rumes et al., 2013).

3. OWF Environmental monitoring programme in Belgium

The construction and long-term presence of OWFs raise concerns for the marine environment (Petersen and Malm, 2006) as many hard substrate constructions are being installed in naturally sandy habitats. Therefore, a compulsory Belgian environmental monitoring programme was established in 2005 (Degraer et al., 2013a), coordinated by the Marine Ecology and Management

Section of the Operational Directorate Natural Environment (OD Nature, formerly Management Unit of the North Sea Mathematical Models, MUMM) from the Royal Belgian Institute of Natural Sciences (RBINS). The programme provides Belgium with a framework to assess possible environmental impacts of OWFs and support policy and developers to mitigate observed effects (in case of severe damage to the marine environment) and incorporate findings into the development of future OWFs (Brabant and Jacques, 2010). The monitoring programme included the different ecosystem components to cover the entire marine environment as much as possible: the seabed bathymetry, water turbidity, underwater noise (pile driving and operational noise), marine mammals (e.g. harbour porpoises and seals), hard-substrate epifaunal organisms (biofouling), seabirds, soft-substrate epibenthos and fish, hard-substrate related fish (e.g. cod and pouting) and the soft-substrate macrobenthos were monitored by OD Nature, the research Institute for Nature and Forest (INBO), the Institute for Agricultural and Fisheries Research (ILVO) and the Marine Biology Research Group of Ghent University (Degraer et al., 2013a). In most cases, a Before After Control Impact (BACI) design was applied where the baseline of the environment was assessed prior to construction of the OWFs and compared to the state of the environment after the impact (Smith et al., 1993). The design also includes a comparison between the impacted and control sites which contain similar environmental conditions.

This PhD research was performed within the framework of the monitoring programme concerning the effects on the soft-substrate macrobenthos, carried out by the Marine Biology Research Group of Ghent University (Marbiol-UGent). Up to date, a baseline and large-scale monitoring programme has been set up on the three sandbanks (Thorntonbank, Bligh Bank and Lodewijkbank) where OWFs have been installed (Table 2).

Table 2. Overview of the soft-substrate macrobenthos monitoring carried out by Marbiol-UGent

Thorntonbank (C-Power)			Bligh Bank (Belwind)		Lodewijkbank (Northwind)
2005	2008 - 2013	2010 - 2012	2008	2011 - 2013	2010 - 2012
Baseline study	Large-scale monitoring	Small-scale monitoring	Baseline study	Large-scale monitoring	Baseline study

A targeted, smaller scale monitoring around one gravity based foundation was carried out on the Thorntonbank to unravel ecological process-related effects induced by the construction and presence of the hard substrate. A control site was located on a nearby sandbank (Goote Bank) which contained similar environmental conditions to the Thorntonbank (De Maersschalck et al., 2005). The large-scale monitoring is still on-going but will experience reconsiderations during 2014 to optimise the future monitoring strategy in the BPNS (Degraer et al., 2013b). Data collected up until 2012 were analysed within the framework of this PhD research (Table 2, red box); the monitoring data of the Lodewijkbank are not discussed in this thesis.

4. The macrobenthos in the Belgian part of the North Sea

Macrobenthos are organisms living buried in the seabed, retained on a 1mm sieve, and mainly consisting of polychaetes (bristle worms), small crustaceans (e.g. amphipods and isopods), echinoderms (e.g. starfish) and bivalves (Degraer et al., 2006). The macrobenthos plays an important role in the marine trophic food web acting as a food source for demersal fish species (Braber and De Groot, 1973), crustaceans (Choy, 1986) and birds (Degraer et al., 1999). By consuming organic matter in the seabed (deposit feeders and predators) or filtering plankton from the water column (suspension feeders) the macrobenthos assists in the degradation of organic matter (Heip et al., 1995). Together with the active particle reworking of sediments (bioturbation) and the enhanced solute transport (bio-irrigation), the macrobenthos facilitates the coupling of nutrients between the seabed and the water column (benthic-pelagic coupling) and mineralisation processes (Braeckman et al., 2010; Kristensen et al., 2012; Meysman et al., 2006).

The distribution of macrobenthic communities is mainly related to the environmental conditions of the area such as sediment type, hydrodynamics, depth, food availability and temperature (Creutzberg et al., 1984; Degraer et al., 2008; Snelgrove and Butman, 1994; Van Hoey et al., 2004; Vanaverbeke et al., 2011). Consequently, the macrobenthos have been widely used in monitoring studies as a good indicator for sediment disturbance (Van Hoey et al., 2010; Ysebaert and Herman, 2003) and are relatively easy to sample. However, a good understanding of the natural variability of the macrobenthos in a temporal and spatial context is essential (Kröncke, 2011; Reiss et al., 2010; Van Hoey et al., 2005, 2007).

In the BPNS, four main macrobenthic communities have been distinguished, connected by transitional species assemblages (Van Hoey et al., 2004). The species rich and highly abundant *Abra alba – Kurtiella bidentata* community dominates the coastal area (Fig. 3 B) which is characterised by fine muddy sands enriched with organic matter (Van Hoey et al., 2005). A co-domination with the *Macoma balthica* community (Fig. 3 A) has been documented in the eastern coastal area of the BPNS (Degraer et al., 2008). Further offshore, the *Nephtys cirrosa* and *Ophelia borealis – Glycera lapidum* communities (Fig. 3 C +D) inhabit the areas with medium to coarse sands and are characterised by a lower abundance and species diversity (Van Hoey et al., 2004).

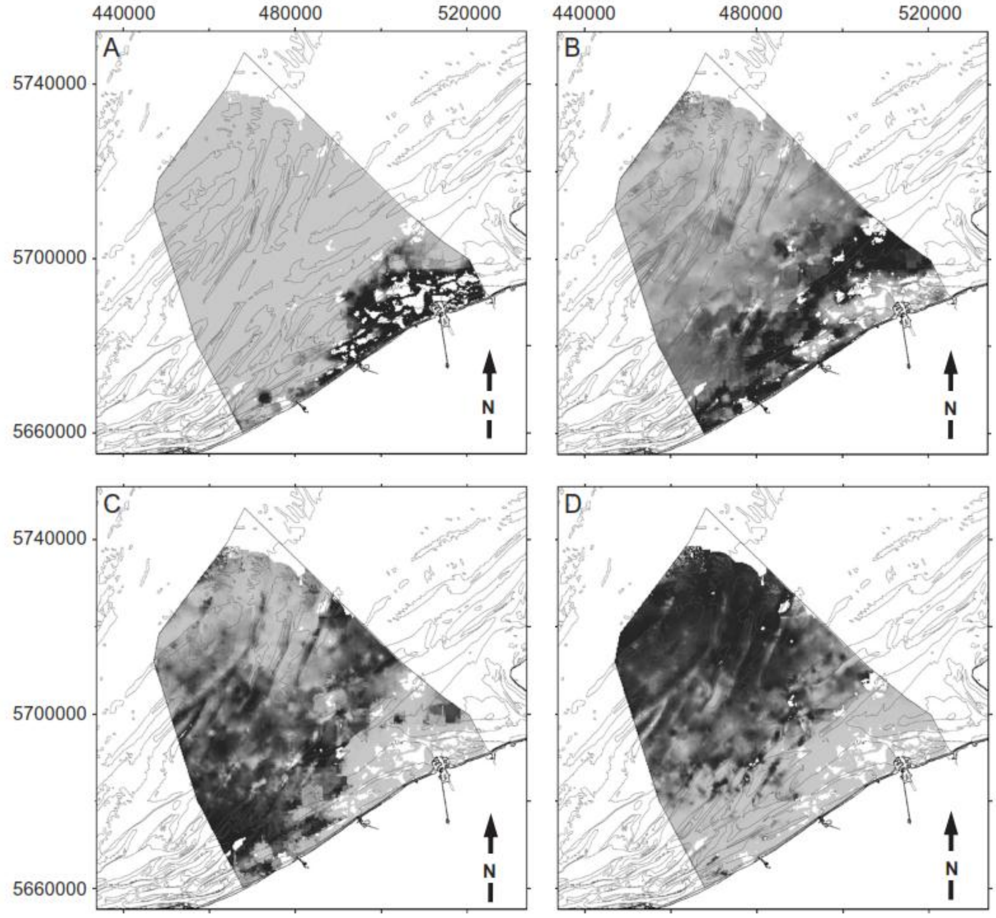

Fig. 3 Predicted habitat suitability maps for the *Macoma balthica* (A), *Abra alba* (B), *Nephtys cirrosa* (C) and *Ophelia limacina* (*borealis*) community (D) in the Belgian part of the North Sea. Black: maximum modelled habitat suitability, Light grey: 0% habitat suitability and White: no environmental data or prediction beyond the range of the model development data set (Degraer et al., 2008)

Situated in the eastern, offshore part of the BPNS, the OWF concession area is dominated by a species poor, *Nephtys cirrosa* and *Ophelia borealis – Glycera lapidum* transitional community (De Maersschalck et al., 2006). With a median grain size between 250 – 500 µm (De Maersschalck et al., 2006; Reubens et al., 2009) the seabed is highly permeable (Vanaverbeke et al., 2011; Wilson et al., 2008). Permeable sediments are characterised by strong advective pore-water flows (Huettel et al., 1998) which oxygenate the sediments and enhance nutrient mineralisation (Ehrenhauss et al., 2004a; Ehrenhauss et al., 2004b; Huettel and Rusch, 2000; Rusch et al., 2006). When the permeability of sediments decrease due to sediment refinement, the efficiency of the mineralisation processes can reduce considerably (Ehrenhauss and Huettel, 2004; Ehrenhauss et al., 2004b). Covering over 70 % of the coastal waters and continental shelves worldwide (Emery, 1968), sandy sediments constitute an important part of the marine ecosystem.

5. Anthropogenic pressures to macrobenthic communities

Global pressures to and impact on the marine environment (Halpern et al., 2008) have led to the implementation of the Water Framework Directive (WFD) 2000/60/EC and the European Marine Strategy Framework Directive (MSFD) 2008/56/EC. The latter was implemented to create and maintain a Good Environmental Status (GES) of the marine ecosystems in Europe by 2020 (EuropeanCommission, 2011). Initially, member states have to describe the current status of their marine environment including physical, chemical and biological characteristics together with anthropogenic activities (BelgischeStaat, 2012). In the BPNS, numerous human activities have been present for years, with specific pressures affecting the physical characteristics of the seabed, altering the macrobenthic habitat.

Since 1976, aggregate (sand and gravel) extraction activities were permitted in four large areas (321 km²) in the BPNS (BelgischStaatsblad, 1969). The creation of deep gullies (10 – 50 cm), increased turbidity and changing sediment grain size are documented effects related to extraction activities (De Backer et al., submitted; Degrendele et al., 2010). The macrobenthos are directly affected by the removal of sediments, decreasing their abundance, diversity and biomass. With a fast re-colonisation potential in high stress areas such as the BPNS (strong currents and frequent storms), the macrobenthic community is adapted to frequent disturbances and capable of recovering after a period of 12 months to 4 years (Boyd et al., 2005; Desprez, 2000; Newell et al., 1998; van Dalfsen et al., 2000).

Maintenance dredging works are regularly carried out around harbours, the seaport of Zeebrugge and the Western Scheldt estuary (Van den Eynde et al., 2013). Dredged material is dumped at specific assigned sites in the BPNS, drastically changing the morphology and sediment composition (increased mud content) of the seabed around these sites (Du Four and Van Lancker, 2008). Main effects to the macrobenthos are related to the burial or smothering of organisms after dumping and the long-term impact of the modified physical habitat (Lauwaert et al., 2011).

Furthermore, beam trawl fisheries have affected the seabed of the BPNS for centuries. The heavy trawls plough the seabed to a depth of 3 – 6 cm, causing sediment re-suspension and removal or damage to non-targeted, fragile benthic organisms (Bergman and Hup, 1992; Dayton et al., 1995; Jones, 1992; Rabaut et al., 2008).

With the installation of OWFs, an additional human impact has been introduced to the BPNS since 2008. The construction, operational and decommissioning phases of the OWFs each create specific pressures to the benthic environment (Gill, 2005; Hiscock et al., 2002; Petersen and Malm, 2006).

The ICES (International Council for the Exploration of the Sea) Working Group on Marine Benthal and Renewable Energy Developments (WGMBRED) have summarised these different pressures for the construction and operational phases and determined possible cause-effect relationships in the benthic ecosystem: 1) a biogeochemical reactor, 2) a source of biodiversity and 3) a source of food resources for higher trophic levels (ICES, 2012, 2013). Figure 4 illustrates the abiotic and biotic processes related to the construction and operational phase of OWFs and the cause-effect relationships affecting the biogeochemical reactions in the benthic ecosystem.

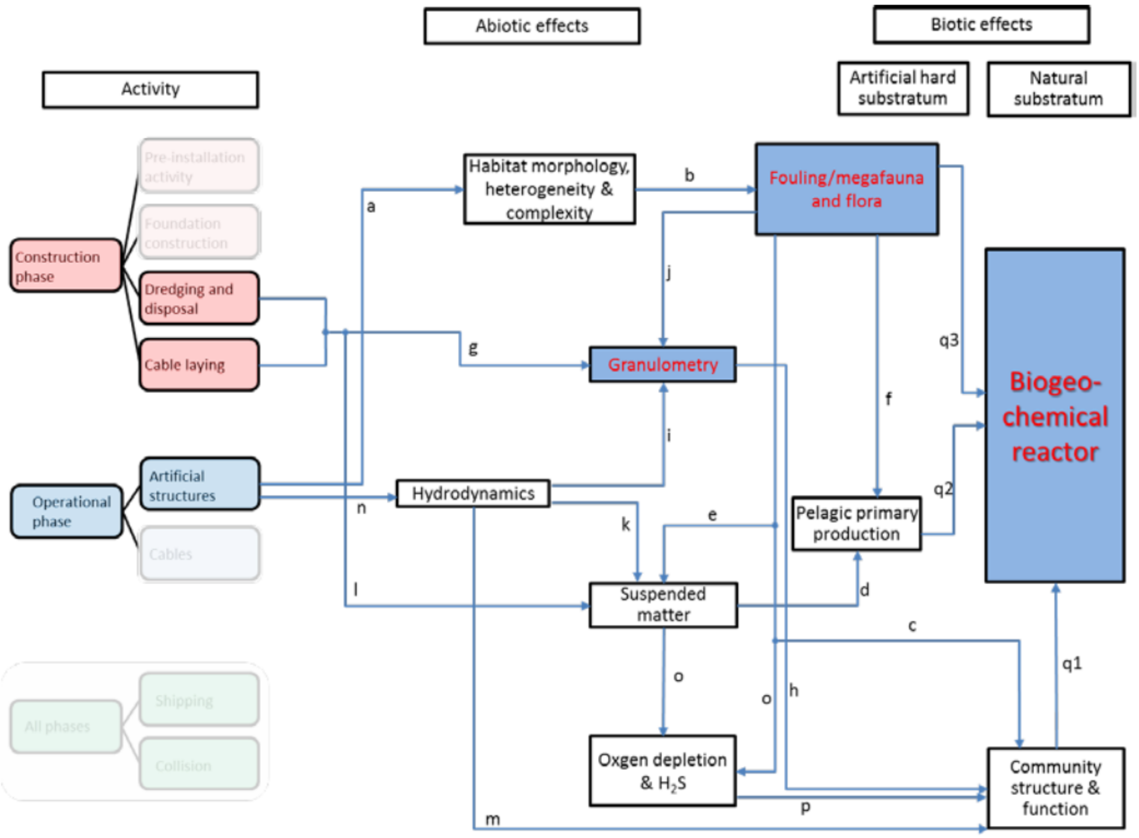

Fig. 4 One of three conceptual figures representing the abiotic and biotic processes during the construction and operational phase of OWFs which are linked to the biogeochemical reactions in the benthic ecosystem. Letters indicate the different research questions and can be retrieved in the ICES WGMBRED Report, 2013

Effects related to the (pre-) construction phase of OWFs (including cable installations) largely depend on the installed foundation type and the intensity of seabed preparation and dredging/extraction activities (Van den Eynde et al., 2010). As described above, the extraction of sediments will directly remove inhabitant macrobenthos and alter the sediment characteristics (granulometry) of the benthic habitat, possibly changing the macrobenthic community structure.

Ecological effects during the much longer (> 20 years) operational phase are mainly related to the physical presence of the hard substrate foundations in a naturally sandy environment (Hiscock et al., 2002). Firstly, the colonisation of the hard substrates by epifaunal species such as mussels and cnidarians (De Mesel et al., 2013) can increase the food availability to the surrounding soft-substrate macrobenthos due to a depositional flow of faecal pellets and detritus (Maar et al., 2009; Ysebaert et al., 2009). Secondly, modified local hydrodynamic conditions (current flows) can occur around these large foundations (Airoldi et al., 2005; Leonhard and Pedersen, 2005). With the creation of sheltered areas or even erosion pits around certain parts of the foundation, the settlement success of macrobenthic larvae could be facilitated. Furthermore, changing hydrodynamics can alter sedimentological characteristics (Hiscock et al., 2002; Leonhard and Pedersen, 2005; Schröder et al., 2006), creating an adapted macrobenthic community to these changed sediments. The extent of hydrodynamic change will also widely depend on the installed foundation type. Gravity based foundations will, for example, seriously decrease current flows compared to open jacket foundations.

Throughout the 20 year operational phase, a prohibition to beam trawl fisheries has been implemented within the Belgian OWFs. This will create a large and unique no-take fishery area for the BPNS allowing to determine to what extent the macrofaunal communities could recover from years of trawling impacts. Species highly vulnerable to beam trawling activities, such as slow growing and fragile echinoderms, bivalves and tube building polychaetes (Bergman and Hup, 1992; Kaiser and Spencer, 1996; Kröncke, 2011) will be allowed sufficient time and space to increase in abundance and slowly create dense, rich patches (Defew et al., 2012; Kröncke, 2011).

Furthermore, the potential changes in sedimentological characteristics and macrobenthic community structure and complexity could affect the benthic ecosystem functioning by altering biogeochemical processes (e.g. benthic-pelagic coupling) and trophic interactions in the marine food web (Dannheim et al., 2014).

Effects related to the decommissioning phase of the OWFs have not been discussed as activities related to this phase have been determined but could still be changed closer to the time. The observed effects of OWFs throughout the operational phase will most likely determine the extent of removal as subsurface structures could be left in place. Nevertheless, environmental effects of the decommissioning phase will probably be comparable to the effects documented during the construction phase.

6. Wider changes to the marine benthic landscape

So far, environmental monitoring projects were only able to focus on short term and small-scale research around one foundation or wind farm (Coates et al., 2014; Degraer et al., 2013b; Lindeboom et al., 2011; Reubens et al., 2011; Schröder et al., 2006; Wilhelmsson and Malm, 2008; Wilhelmsson et al., 2006). There is a great lack of knowledge on how the development of OWFs will affect the marine ecosystem in the longer term and on larger regional scales (Fig. 5) such as within the complete wind farm concession zone in the BPNS or at the scale of the entire North Sea (Burkhard and Gee, 2012; Wilhelmsson et al., 2010).

Apart from the seven approved domain concessions in the BPNS with three operational OWFs (Brabant et al., 2013), the UK and the Netherlands will simultaneously increase the amount of OWFs installed in the Southern Bight of the North Sea. In UK waters, seven OWFs are operational or under construction (TheCrownEstate, 2013) and a further large concession zone (East Anglia), with several additional OWFs, is planned in the Southern Bight. Two OWFs are operational in Dutch waters (Prinses Amalia and Egmond aan Zee) (Lindeboom et al., 2011) together with an additional twelve granted domain concessions (Rijksoverheid Nederland). With a further expansion of OWFs in most European countries, the North Sea will soon become a mosaic of numerous OWFs (Fig. 5). Creating complex habitats, the construction of OWFs can lead to unknown cumulative effects in the marine ecosystem (Gill, 2005).

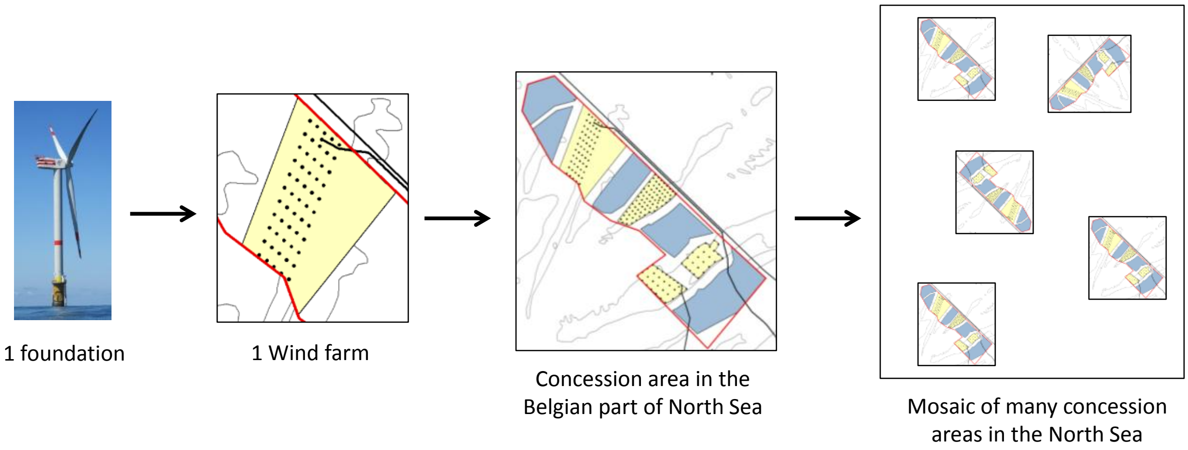

Fig. 5 Conceptual figure of the multiple scales related to the construction of OWFs with the smallest scale around one foundation to the larger scale with a mosaic of different concession areas in the North Sea.

There is a limited understanding on how marine organisms respond to changes in the spatial configuration of their habitat (Bostrom et al., 2011). Therefore, the multi-scale ecology of species and communities was addressed in the marine environment by applying the well-studied terrestrial landscape ecology (Bostrom et al., 2011; Pittman et al., 2011; Turner, 2005). Derived from the theory of island biogeography (MacArthur and Wilson, 1967), a landscape is defined as “an area of land containing a mosaic of habitat patches, often within which a particular ‘focal’ or ‘target’ habitat patch is embedded” (Dunning et al., 1992). When applying this definition to the marine environment, a seascape is generally defined as a spatially heterogeneous area in the coastal environment characterised by a mosaic of patches (vegetation), which are viewed as islands embedded in a matrix (sediment) (Bostrom et al., 2011).

In terms of OWFs, the enclosed, complex habitats within the concession areas will represent the focal patches, surrounded by the homogenous soft, sandy sediment (matrix) (Fig. 4). Various ecological processes, related to the spatial configuration of landscapes, have been explored, mainly concerning the fragmentation of sea grass meadows in the marine environment. The patch size, boundary shape, complexity and isolation (Bostrom et al., 2011) will affect ecological processes such as predator-prey dynamics (food web interactions) and movements of organisms (Hovel and Lipcius, 2001; Irlandi et al., 1995; Micheli and Peterson, 1999). Landscape corridors between patches can facilitate larval dispersal (Tewksbury et al., 2002), creating a stepping stone effect between OWF areas.

Geographic Information Systems (GIS), remote sensing and computer models have started to address the large-scale ecological patterns to assist in the management of marine ecosystems (Burkhard and Gee, 2012; Burkhard et al., 2011; Degraer et al., 2008; Vanhellemont and Ruddick, 2014, in press; Verfaillie et al., 2006). The production of benthic habitat maps and remote sensing imagery of coastal systems will largely facilitate the application of landscape ecology in the marine environment (Pittman et al., 2011). Vanhellemont and Ruddick (2014, in press) illustrated large current plumes behind every foundation of the London Array and Thanet OWFs in the UK by applying remote sensing imagery. With a spatial extent of the plume up to 10 km, this is a first step in understanding how suspended matter in the water column could be altered on a large-scale in OWFs. In combination with in-field research, these techniques can enhance our understanding of the changing ecological processes around OWFs at different spatial scales.

A few model studies have illustrated different ecological scenarios for OWFs with the possible evolution from heterogeneous, complex ecosystems at a small-scale (foundation or OWF level), to a larger scale ecological regime shift (abrupt change) with increased ecosystem functions (Burkhard and Gee, 2012). The final outcome will highly depend on the habitat connectivity between OWFs and the landscape configuration. The identification of seascape configuration thresholds will be important for the future management of OWFs, in order to avoid the modelled regime shifts of the marine ecosystem (Pittman et al., 2011) and preserve a Good Environmental Status within the MSFD framework. Application of benthic indicators in monitoring studies can further assist in identifying these threshold levels for the benthic system (Van Hoey et al., 2010).

7. The PhD research

- **Objectives**

This PhD research contributes to the environmental monitoring programme concerning the effects of Offshore Wind Farms (OWFs) on the marine environment (Degraer et al., 2013a) in the Belgian Part of the North Sea (BPNS). The OWF concession area situated in the eastern part of the BPNS is mainly characterised by permeable, sandy sediments with a relatively species poor macrobenthic community. The construction of numerous hard substrates in a sandy, relatively homogenous, environment such as the North Sea can cause changes to the macrobenthic community. With data collected from 2005 – 2012, this PhD covers the first phase of the macrobenthic survey around OWFs in the BPNS. Both the large-scale monitoring and the process-related targeted monitoring (see before) has provided us with the opportunity to investigate the environmental changes within the soft-substrate macrobenthic communities, related to the construction and operation of OWFs at different spatial scales.

The objectives of this PhD research are related to either the construction or operational phase of the OWFs in the BPNS and the response of the macrobenthos and seabed characteristics to these activities. The three main objectives of this PhD thesis are:

a) To determine the natural, temporal variability of macrobenthic communities in sandy, permeable sediments. How resilient are these communities in recovering from the construction activities of OWFs?

b) To identify if the exclusion of fisheries within OWFs affects the macrobenthos during the first years of the operational phase of an OWF on a large-scale.

c) To unravel changes to the macrobenthic community and sedimentological characteristics on a small spatial scale (in the direct environment of one foundation) and determine how the observed small-scale changes could affect the benthic ecosystem functioning.

With an expansion from three to seven OWFs in the BPNS, the conducted research performed for this PhD thesis will contribute to improve future monitoring strategies of offshore renewable energy developments in Belgium. The baseline knowledge attained on the effects OWFs enforce on the soft-sediment macrobenthos during the first years after construction can be further applied to optimise monitoring programmes in other European countries and regions with sandy coastal shelf seas. With a focus on both small- and large-scale effects, the results documented in this thesis will form a reference point in analysing cumulative effects of different OWFs in the North Sea after a longer term.

- **Outline of the PhD thesis**

This PhD thesis is a compilation of different research articles, apart from the general introduction and discussion, which have been published, submitted or in preparation for submission to a scientific peer-reviewed journal. Each chapter can be read as an independent unit which has led to an overlap between certain sections (introduction and discussion) of different chapters. The cited literature of all chapters can be retrieved in the reference list at the end of this thesis. All chapters have the PhD candidate as first author.

In **chapters 2 – 4** the data from field samples of the soft-sediment macrobenthos in and around the OWFs on a small- and large-scale area analysed. In this thesis, the term 'large-scale' was applied for one OWF (approximately 21 km²) and 'small-scale' for research performed around one foundation. **Chapter 5** was achieved from a laboratory based benthic chamber experiment.

Chapter 2 contains a long-term data analysis based on both historical data and the data related to the construction of OWFs. The historical data was acquired from the MACRODAT database of the Marine Biology Research group (Ghent University) and also data provided by the ILVO-Bioenvironmental Research group. Large-scale effects related to the construction activities of six gravity based foundations (GBFs) on the Thorntonbank were explored. Firstly, a long-term analysis (1980 – 2012) was applied on data collected on the Thorntonbank and the reference sandbank (Goote Bank) to distinguish the natural, temporal variability of the macrobenthic community from any anthropogenic related fluctuations. With the typical variable weather conditions of the BPNS, strong temporal variations in the macrobenthos were observed. Significant differences in the macrobenthic community composition between the impacted and reference sandbank were only detected in the construction year after severe sand extraction and seabed preparation activities had taken place. The Benthic Ecosystem Quality Index (BEQI) was assessed as an indicator and reflected similar results. With a rapid recovery from 2009 onwards, the macrobenthos illustrated a high resilience to the physical seabed disturbance related to the construction of GBFs. *This paper will be submitted to a special issue of Hydrobiologia as: Coates D.A., Van Hoey G., Colson L., Vincx M. and*

Vanaverbeke J. Rapid macrobenthic recovery after construction activities of an offshore wind farm in the Belgian part of the North Sea on a large-scale.

Chapter 3 discusses the large-scale exclusion of beam trawl fisheries within an operational OWF with 55 monopile foundations on the Bligh Bank. The macrobenthic community within the fishery enclosed area of 21 km² was compared with a surrounding control area two to three years after the construction of the OWF. Regular fishing activities around the OWF were registered through Vessel Monitoring System (VMS) data by ILVO. Three years after the exclusion of fisheries, subtle changes were observed within the macrobenthic community in the No Fishery area. Species sensitive to trawling activities, such as the tube-building polychaetes Terebellidae sp. and the echinoderm *Echinocyamus pusillus*, showed increased abundances within the OWF. However, regular illegal fishing activities within the OWF were registered through VMS data and visual observations, possibly slowing down the recovery rates of these fragile organisms. Nevertheless, this short-term study suggests the possible evolution of the area into an ecological important habitat and will facilitate future research in exploring how the area will evolve after a longer term closure to fisheries (> 20 years). *This paper has been submitted to Fisheries Research as Coates D.A., Kapasakali D.-A., Vincx M. and Vanaverbeke J. Short-term effects of fishery exclusion in offshore wind farms on macrofaunal communities in the Belgian part of the North Sea.*

In **Chapter 4**, a smaller scale was explored by sampling around one GBF on the Thorntonbank three and four years after construction. Along four gradients and five distances around the GBF, the macrobenthos and the main characteristics of the permeable, sandy sediments were investigated. Environmental characteristics illustrated a change in close vicinity to the foundation with a reduction in sediment grain size and an increase in total organic matter content up to a distance of 50 m. These observed changes inevitably triggered a significant increase in the macrobenthic abundance and diversity together with a shift in species dominance from *Nephtys cirrosa* to the ecosystem-engineer *Lanice conchilega* closer to the foundation. The results from this chapter suggest a viable prediction to an overall change to the seabed characteristics and macrobenthic community on a larger scale in the future, creating an ecological important refugium area for species from higher trophic levels. *The results of this paper have been published in Marine Environmental Research as Coates D.A., Deschutter Y., Vincx M. and Vanaverbeke J. (2014) Enrichment and shifts in macrobenthic assemblages in an offshore wind farm area in the Belgian part of the North Sea. Marine Environmental Research. 95 : 1 – 12.*

To increase our knowledge on how the altered sediment characteristics observed in **Chapter 4** could affect the benthic ecosystem functioning, an experimental pilot study was carried out and documented in **Chapter 5**. Two experiments were set up in benthic chambers containing three sediment types with a decreasing permeability. The filtration capacity of the three sediment types was assessed by measuring the water penetration depth after addition of a dye and by counting the reduction of diatom cells (*Skeletonema costatum*) in the water column during a second experiment. Additionally, sediment community oxygen consumption (SCOC) rates and nutrient fluxes were also measured before and after the addition of the diatom cells (organic matter). Mineralisation rates of the added organic matter were facilitated by the advective-pore water flows in the coarse, permeable sediments. The results suggest that the reduction in sediment grain size observed in **Chapter 4** could alter the ecosystem functioning and mineralisation processes of organic matter.

In **Chapter 6**, the results from this PhD thesis are discussed in a wider ecological setting, addressing the importance of macrobenthic research in order to assist in understanding the changing marine ecosystem processes around OWFs. From the obtained knowledge, recommendations are formulated for future monitoring in Belgium and for the collaboration between European countries in order to address the wider spatial landscape issues.

CHAPTER 2

Rapid macrobenthic recovery after construction activities of an offshore wind farm in the Belgian part of the North Sea on a large-scale

Paper to be submitted to a special issue of Hydrobiologia as

Coates D.A., Van Hoey G., Colson L., Vincx M. and Vanaverbeke J. Rapid macrobenthic recovery after construction activities of an offshore wind farm in the Belgian part of the North Sea on a large-scale

Abstract

The development of offshore wind farms (OWFs) in the North Sea has increased considerably during the past years to contribute to alternatives for fossil fuel energy. Activities related to the construction of OWFs are mainly associated to dredging, causing direct effects to the macrofauna in the surrounding seabed. The sediment characteristics and macrofaunal assemblages before and after the construction (2005 – 2012) of an OWF with six gravity based foundations on a subtidal sandbank (Thorntonbank) in the Belgian part of the North Sea were studied on a large-scale. We distinguished natural from anthropogenic related fluctuations in macrofaunal communities by analysing a long term dataset (1980 - 2012) of both the impacted (Thorntonbank) and reference sandbank (Goote Bank). Both sandbanks are characterised by coarse, sandy substrates (331 ± 20 µm - 410 ± 41 µm) and a macrobenthic community with a low species abundance (180 – 812 ind m^{-2}) and diversity (6 – 15 species per sample). Strong temporal variations were observed, related to the variable weather conditions (cold winters and severe storms) in the area. Significant differences in macrobenthic community composition were observed due to the installation of the OWF in the construction year (2008) followed by a rapid recovery from 2009 onwards. These patterns were also reflected by the benthic ecosystem quality index BEQI. Even though the construction of OWFs creates a large physical disturbance to the seabed, the macrobenthic community of these dynamic sediments have illustrated a fast recovery potential.

Keywords

Wind power - Macrobenthic communities - Temporal variability - North Sea - Benthic indicators

1. Introduction

Renewable energy resources such as offshore wind, tidal and current energy are widely being developed in the marine environment as an alternative for fossil fuels. In North-west Europe, the development of offshore wind farms (OWFs) has escalated during the past decade with 69 commissioned OWFs across 11 countries (Corbetta et al., 2014). The ecological impacts on the natural processes and surrounding macrobenthic biodiversity on a large-scale and long term are hard to determine at this point in time as many studies have only been able to focus on short term or small-scale effects (Bergström et al., 2012; Coates et al., 2014; Leonhard and Pedersen, 2005; Lindeboom et al., 2011).

During the (pre-) construction phase of an OWF, direct impacts are enforced on the seabed according to the chosen foundation type (Hiscock et al., 2002). Drilling or pile-driving are main activities during the installation of monopile foundations with limited seabed preparation works, while intense dredging activities are necessary for gravity based foundations (GBFs). Due to the nature of these construction activities, effects on the macrofaunal communities within the surrounding soft substrate can be related to the physical disturbance associated to aggregate extraction activities. A direct effect of marine aggregate extraction on macrobenthic assemblages is the removal of sediments leading to a decreased abundance, diversity and biomass of benthic organisms in dredged areas (Boyd et al., 2003; Desprez, 2000; Newell et al., 1998). Dredging activities also create a sediment plume through the re-suspension of finer particles, potentially clogging the gills of suspension-feeding organisms. Many studies documented a fast re-colonisation after cessation of dredging, with an increase of short lived opportunistic species such as mobile bivalves (e.g. *Spisula* sp. and *Tellina* sp.) and polychaetes (e.g. *Capitella* sp., *Spio* sp. and *Spiophanes bombyx),* causing shifts in the macrofaunal community composition (Desprez, 2000; Newell et al., 1998; van Dalfsen et al., 2000). However, full recovery, in terms of abundance and diversity, of the initial long-lived macrobenthic community can take from 12 months up to 4 years (Desprez, 2000; Newell et al., 1998). The recovery rate is however closely related to the duration and intensity of the extraction activity (Boyd et al., 2005; Boyd et al., 2003). Additionally, macrofauna living in high stress areas with strong currents and storm disturbances such as the North Sea are already naturally adapted to changes, leading to a more efficient and rapid recovery of the community after anthropogenic disturbance (Bonne, 2010; Desprez, 2000; Foden et al., 2009; Kenny and Rees, 1996; Vanosmael and Heip, 1984). Long-term datasets are therefore prerequisites in environmental monitoring studies to gain knowledge on the natural evolution of macrobenthic communities and distinguish them from fluctuations caused by anthropogenic activities such as dredging (Frojan et al., 2008) but also climate variability (Kröncke et al., 1998) or eutrophication (Pearson and Rosenberg, 1978). Thorough analysis of monitoring data is necessary to unravel the detailed patterns and cause-effect relationships. However, tools that provide a fast and easy to calculate signal of the occurring changes have become essential in environmental impact assessment (EIA) processes and management guidance. Therefore, indicator tools (e.g. BEQI, Benthic Ecosystem Quality Index), accompanied by thresholds levels, were developed to assess the degree of anthropogenic impacts and the environmental status of the soft sediment macrobenthos (Van Hoey et al., 2013).

Six GBFs were constructed on a subtidal sandbank (Thorntonbank) in the Belgian part of the North Sea (BPNS) in the first half of 2008. Pre-construction activities included seabed preparation with short-term but intense dredging in areas in and around the concession zone (Merken, 2008; Peire et al., 2009). The OWF was expanded during a second construction phase in 2011 with the installation of 48 jacket foundations where activities were mainly limited to drilling and pile-driving. In this study, we analysed the macrobenthic community on the Thorntonbank and a nearby sandbank (Goote Bank) from 1980 – 2012. The long-term dataset helps distinguish natural fluctuations from any construction effects on the macrobenthic community on a large-scale. Community descriptors such as abundance, diversity and species composition are analysed over 32 years to characterise the macrofauna and its natural evolution on both sandbanks. Following the BEQI index algorithm, the degree of impact (positive or negative), caused by the construction or operation of the OWFs, was evaluated based on a comparison of the benthic characteristics between the impact and control areas of the Thorntonbank and Goote Bank. The main hypothesis of this study can be divided into two sections: 1) Any construction effects enforced upon the macrobenthos will rapidly be suppressed due to the fast recovery potential of the macrobenthic community and 2) Operational effects of the six gravity based turbines at a large-scale will be minimal after short-term (4 years) monitoring.

2. Material and methods

2.1. Study area

The Thorntonbank and Goote Bank are situated in the eastern part of the Belgian Part of the North Sea (BPNS) and belong to the Zeelandbanks which are situated parallel to the coastline (Northeast orientation) at a distance of 15 – 30 km offshore (Fig. 1). Crests are below 10 m MLLWS (Mean Lowest Low Water Spring Level) (Dewicke et al., 2003) and the sediments consist of medium sands with a grain size between 250 and 500 µm (De Maersschalck et al., 2006). The Thorntonbank is situated in the Belgian wind farm concession zone where the first six GBFs were constructed in the first half of 2008 and 48 jacket structures in 2011 - 2012.

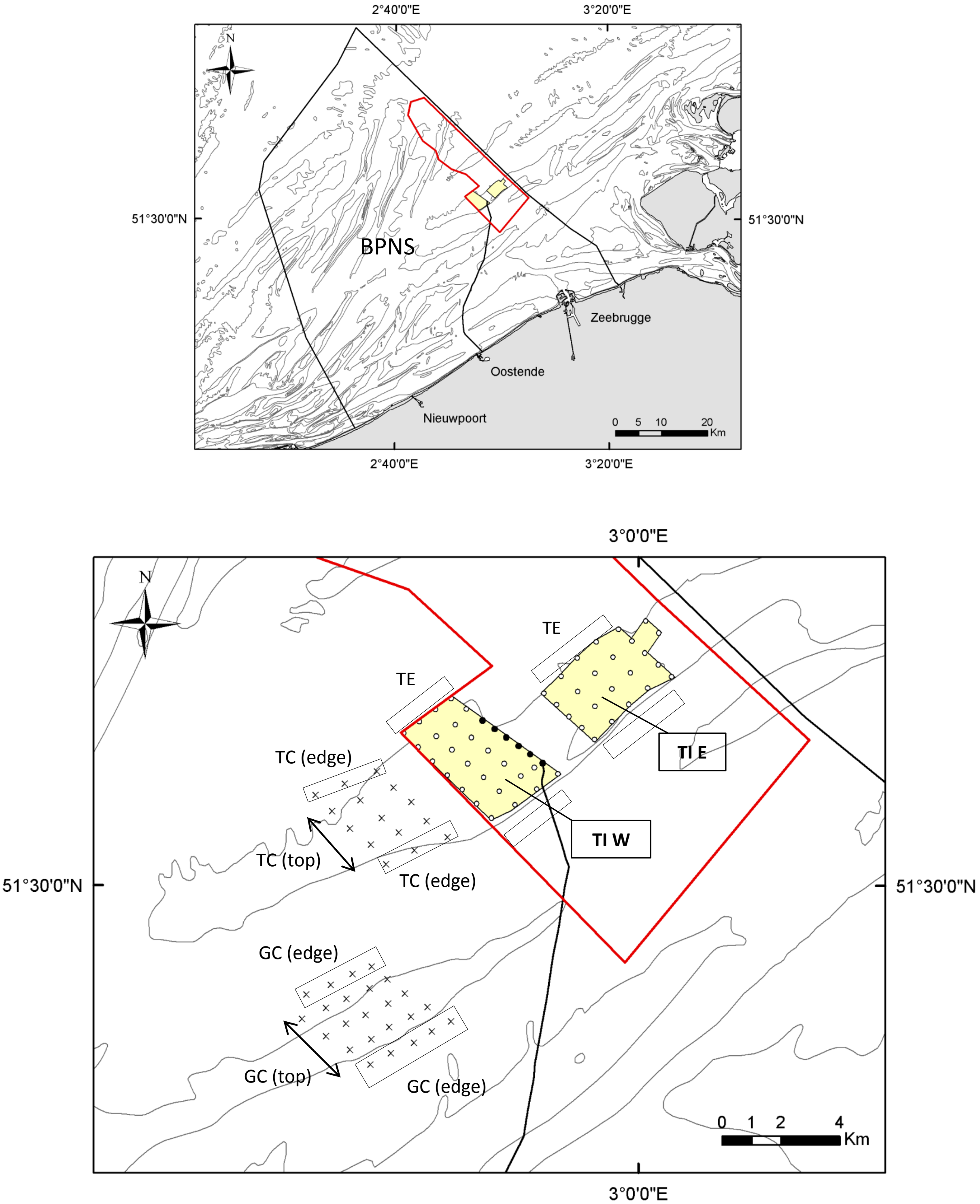

Fig. 1 Upper map: Map of the Belgian part of the North Sea (BPNS) with the Belgian offshore wind farm concession zone (Red box). Map below: Six gravity based foundations (black dots) were constructed in 2008 in the western impact area (TI W) of the Thorntonbank, 48 jacket foundations (white circles) were constructed in 2011 in the eastern (TI E) and western impact area. Two reference areas on the Goote Bank (GC) and the Thorntonbank (TC) were simultaneously sampled together with the edges around the impact areas (TE)

2.2. Historical data

Historical macrobenthic abundance data was available from the database MACRODAT containing all collected macrobenthic samples by Ghent University from 1976 to present in combination with data provided by the ILVO-Bioenvironmental Research group. All data collected from August to November on the Thorntonbank and Goote Bank were selected, resulting in a dataset for both sandbanks in the years 1980, 1985, 1986 and 1998. The amount of stations sampled per sandbank and per year was limited (Table 1).

Table 1. Number of sampled stations on the Thorntonbank and Goote Bank from 1980 - 2012

# Stations	**1980**	**1985**	**1986**	**1998**	**2005**	**2008**	**2009**	**2010**	**2012**
Thorntonbank	2	2	1	19	60	51	57	20	14
Goote Bank	2	5	2	1	16	25	22	25	4

2.3. Short-term data

Samples on the Thorntonbank and Goote Bank were collected within the framework of the wind farm monitoring programme between 2005 and 2012. The Thorntonbank was divided into a control (TC), edge (TE), eastern (TI E) and western (TI W) impact area. In 2008, the first six GBFs were constructed at TI W and the 48 jacket foundations, constructed in 2012, were installed at both the TI E and TI W impact areas. The edge area (TE) was defined to determine any edge effects around the concession area of the OWF. Alongside the control area on the Thorntonbank (TC), the Goote Bank was sampled as a second reference area as it contained similar physical characteristics of the seabed (De Maersschalck et al., 2005). Samples were obtained from the *RV* Belgica during autumn in 2005 (reference year) and yearly from 2008 to 2012 by means of a 0.1026 m² Van Veen grab. Due to the construction of jacket foundations, samples could not be obtained in 2011 and from 2012 no stations within the concession zones could be sampled.

Samples were sieved alive over a 1mm mesh sized sieve, fixed in a 4 % formaldehyde-seawater solution and coloured with rose Bengal. After rinsing and sorting, organisms were identified to species level and conserved in an 8 % neutralised formaldehyde solution. Biomass or ash free dry weight (AFDW) was determined for every species per sample, either through a conversion factor of the wet weight (Brey, 2001) or by regression factors of the length or width. When neither conversion factors nor regressions existed for a certain species, AFDW was determined by cremation. Samples were dried for 48 h at 60 °C and burned in a muffle furnace for 2 h at 550 °C, the AFDW was then calculated as the difference between the dry and ash weight.

Throughout the monitoring programme, environmental data such as grain size distribution and total organic matter content were sampled parallel with the biotic data. After drying at 60 °C the grain size distribution was measured on a subsample of the Van Veen grab with a Malvern Mastersizer 2000G, hydro version 5.40 (laser diffraction method) (Malvern, 1999). Grain size fractions are given as volume percentages with a range from fine clay (< 4 µm) to coarse gravel/shell material (> 1600 µm). The total organic matter (TOM) content was also determined by applying following equation: TOM % = [(DW – AW) / (DW - CrW)] x 100. The dry weight (DW) was determined after 48 h at 60 °C and the ash weight (AW) after 2 h 20 min at 550 °C. Every used crucible was weighed (CrW) in order to determine TOM %.

2.4 Benthic indicator

The benthic indicator BEQI (Benthic Ecosystem Quality Index, www.beqi.eu), evaluates the difference in benthic characteristics (density, biomass, number of species and species composition) between two datasets (e.g. control versus impact). The difference between the two datasets (expressed as Ecological Quality Ratio, EQR) is scaled between 0 and 1 and divided into five classes: bad [< 0.2], poor [0.2 - 0.4], moderate [0.4 - 0.6], good [0.6 - 0.8] and high [0.8 - 1]. When the EQR reaches a value below 0.6, the difference between the two datasets (control – impact) is unwanted and a detailed analysis of the outcome is advised. In order to perform a proper indicator assessment of a possible impact, the influence of the natural variability in benthic characteristics on the indicator outcome has to be minimised. Therefore, the compared datasets in the assessment design should have the same habitat characteristics (such as sediment type, depth region, etc.), the same time period (season, year) and contain enough samples to obtain a confident assessment (Van Hoey et al., 2010). The assessment confidence (statistical power) of each parameter is separately scored within the BEQI tool, based on the probability of creating a Type II error. This depends on the variance in the data, the effect size and the choice of the significance level, which is set to 0.05 in the BEQI - tool (Van Hoey et al., 2010). The ecological status of the different areas defined in paragraph 2.3 were determined (TI W, TI E, TC and GC), together with a separate consideration of the samples in the edge zones in each of those areas (TI WE, TI EE, TCE and GCE), to incorporate possible differences between the top and gully (edge) of the sandbanks**.** To increase the assessment confidence, the designs were based on all data (three replicates per station) available within each area/period. The data from the Thorntonbank and Goote Bank control zones were analysed together.

The accuracy of the BEQI assessment is based on the variability within the data in three classes (good, moderate and poor). Only the indicator outcomes that scored moderate or good were included in the results. Subsequently, an appropriate selection of the control dataset is advised, as different control data samples will have an influence on the final indicator judgment (Van Hoey et al., 2013). Therefore, different control datasets were applied in this study, based on samples from the control areas of the same year (within year assessment), samples from the area before 2005 ('historic data') and samples from the baseline sampling (2005).

2.5 Data analysis

With an unequal set of replicates obtained throughout the years, it was opted to select all first replicates of every station to create a stratified random sampling design (Degraer et al., 2008). The historical database (1980 - 2012) was subjected to a data quality control where taxa that were not sampled quantitatively (hyperbenthos, meiobenthos and fish) and extremely rare taxa (defined as species with a single occurrence and a maximum of three individuals) were excluded from the analysis. Certain species were lumped to genera or family level due to inconsistent species identification throughout different studies (e.g. Anthozoa, *Aricidea* sp., *Callianassa* sp., *Cirratulidae* sp., *Eteone* sp., *Magelona* sp., *Pseudocuma* sp. and *Spio* sp.). After the data control, a total set of 328 macrobenthic samples and 133 taxa was further used for data analysis from 1980 up to 2012.

From 2005 – 2012 the abundance data from the five different monitoring areas was also analysed separately, as biomass and environmental data were only available in this period. 294 samples and 131 species were finally included in this analysis. The number of individuals per m² (abundance) was determined together with the species richness (N_0) and Shannon Wiener diversity index (H'). Dominant species were calculated for all samples as species with a mean contribution of more than 15 % to the mean total density. Results are expressed as mean ± standard error (SE).

The Plymouth Routines in Multivariate Ecological Research (PRIMER) programme (version 6.1.6) was used for statistical analyses with the PERMANOVA add-on software (Anderson et al., 2008; Clarke and Gorley, 2006). Multivariate and univariate PERMANOVA's (Permutational ANOVAs) were carried out in this study with a two-factor design containing sandbank and year as factors in the historical data analysis (1980 – 2012) and zones and year as factors in the short-term monitoring (2005 – 2012). Type III sums of squares was applied as the design was unbalanced, with the number of permutations set to 9999 and the permutation of residuals under a reduced model. When the main PERMANOVA test showed significant effects for factors, a pair-wise comparison was applied between levels of factors. When the main test showed a significant interaction between factors a pair-wise test was performed within the interaction factor. A significance level of $p < 0.05$ was applied for all tests. Monte-Carlo p-values P(MC) were used if the amount of possible permutations to obtain meaningful results was low (< 100 permutations) (Anderson et al., 2008). Homogeneity of dispersions was tested with PERMDISP using distances among centroids. Data was log (x+1) transformed if PERMDISP was significant, if thereafter PERMDISP was still significant the transformation was preserved and results were carefully analysed. After square-root transformation, the resemblance measure Bray-Curtis similarity was used for the multivariate community analysis of the biotic data (abundance and density). The Euclidean distance similarity matrix was separately applied for the univariate analysis of total density, diversity and biomass and for environmental data (grain size and TOM %) after normalisation. Furthermore, a SIMPER analysis of the multivariate abundance data was carried out to determine the species with the highest contribution to the similarity within communities. The total abundance of these species was analysed with PERMANOVA.

3. Results

3.1 Long-term changes (1980 – 2012)

Throughout 32 years, the total mean macrobenthic density reached a maximum in 1986 on the Thorntonbank (682 in m^{-2}) and in 2008 on the Goote Bank (812 ± 141 ind m^{-2}). Minimum values were measured on both sandbanks in 1998, with 180 ± 21 ind m^{-2} on the Thorntonbank and 244 ind m^{-2} on the Goote Bank (Fig. 2).

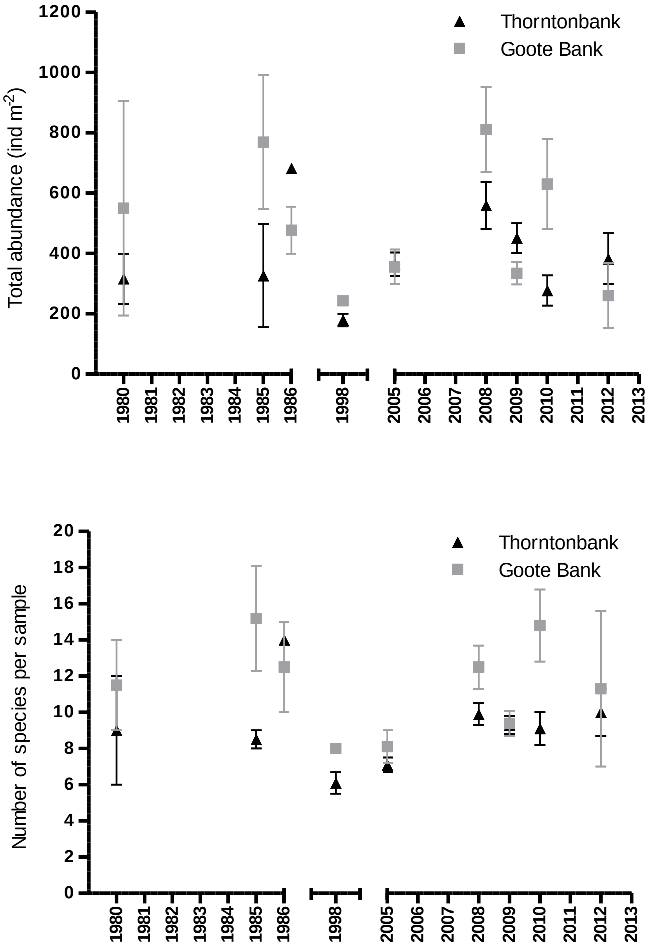

Fig. 2 Total abundance (ind m^{-2}) and total number of species (sample^{-1}) on the Thorntonbank and Goote Bank from 1980 – 2012

The main PERMANOVA test only showed significant changes for the factor Year (*p* = 0.0319). Significantly lower abundances were measured in 1998 compared to 1985 and 1986 (Pair-wise PERMANOVA test). Mean abundances in the construction year (2008) were significantly higher than 2005, 2009, 2010 and 2012 (Table 2).

The total mean species number ranged between 6 ± 0.6 and 15 ± 2.9 species per sample with significant differences between years (PERMANOVA, *p* = 0.002). With 7 ± 0.4 and 8 ± 0.9 species per sample in 2005 on the Thorntonbank and Goote Bank respectively, the total mean number of species was significantly lower compared to 1985, 1986 and 2008 - 2010 (Table 2). Similarly, the diversity index Shannon H’ showed significant differences between years (*p* = 0.0001) with a significantly lower value in 2005. In 2010, the diversity showed a significant increase compared to 2005, 2008 and 2009. The Goote Bank also showed a higher diversity (*p* = 0.0423) over the years in comparison to the Thorntonbank.

Table 2. Univariate, pair-wise PERMANOVA test for term year to determine significant differences (significant results p < 0.05)

Average abundance	**t**	**p-value (perm)**	**Unique perms**
1985 - 1998	2.2324	0.0499	9779
1986 - 1998	5.0508	0.0002	7465
2005 - 2008	3.66	0.0006	9836
2008 - 2009	3.4306	0.001	9847
2008 - 2010	2.0028	0.0451	9814
2008 - 2012	2.076	0.0466	9822
Species number	**t**	**p-value (perm)**	**Unique perms**
1985 - 2005	2.1051	0.0332	9835
1985 - 2009	1.3102	0.2	9839
1986 - 2005	2.0037	0.0438	9836
2005 - 2008	4.1386	0.0002	9840
2005 - 2009	2.9131	0.0041	9826
2005 - 2010	3.3749	0.0016	9827
Shannon H' diversity	**t**	**p-value (perm)**	**Unique perms**
1980 - 2005	2.735	0.0079	9826
1985 - 2005	2.6304	0.0112	9825
1986 - 2005	2.46	0.0202	9841
2005 - 2008	4.0754	0.0002	9848
2005 - 2009	3.6243	0.0004	9832
2005 - 2010	5.5775	0.0001	9840
2005 - 2012	3.2248	0.0016	9841
2008 - 2010	2.4886	0.0135	9848
2009 - 2010	2.9481	0.0039	9858

The macrobenthic community composition based on total densities showed a significant interaction for the term sandbank x year (p = 0.0008) in the main PERMANOVA test. Pair-wise comparisons for the term sandbank x year for pairs of levels of the factor sandbank illustrated a significant difference in community composition between the Thorntonbank and Goote Bank from 2008 until 2010 (Table 3). It must be noted that results for PERMDISP showed a significance for the term sandbank x year (p = 0.0087). On the Thorntonbank, communities were mainly dominated by *Nephtys cirrosa* (20 %), *Gastrosaccus spinifer* (19 - 36 %) or *Bathyporeia elegans* (27 %) in the period between 1980 and 1986. From 1998 onwards, *N. cirrosa* (15 – 49 %) and *Urothoe brevicornis* (18 – 28 %) dominated the communities with the exception of 2008 where the opportunistic *Spiophanes bombyx* co-dominated (18 %) with *N. cirrosa* (28 %). *Spiophanes bombyx* was only dominant on the Goote Bank in 2008 (38 %). Dominant species between the 1980-1986 and 1998-2012 also varied with a dominance of *B. elegans* (22 %) and *Ophelia borealis* (21 %) from 1980 – 1986 and a main dominance of *N. cirrosa* (13 – 28 %) from 1998 – 2012.

Table 3. Multivariate abundance analysis. with the pair-wise test for term sandbank x year for pairs of levels of factor sandbank (Thorntonbank – Goote Bank). significant results p < 0.05 with p-values based on the amount of permutations: p-value (perm) and Monte-Carlo p-values: P (MC)

Within level Year	t	p-value (perm)	Unique perms	p-value (MC)
1980	11.367	0.6694	3	0.3671
1985	14.408	0.0512	21	0.1108
1986	0.63189	1	3	0.6956
1998	12.726	0.0992	20	0.1254
2005	12.703	0.1224	9931	0.1299
2008	17.494	0.0012	9912	0.0029
2009	15.828	0.0062	9929	0.0086
2010	16.619	0.0037	9916	0.007
2012	13.284	0.0643	2940	0.1036

3.2 Short-term changes (2005 – 2012)

3.2.1 Sediments

The sediments in all monitoring areas on the Thorntonbank and Goote Bank were characterised by coarse, sandy substrates with an average median grain size measured between 332 ± 20 µm and 410 ± 41 µm for the period between 2005 and 2012 (Table 4). No significant differences in year or monitoring areas were observed (PERMANOVA). Low TOM contents were measured in the sediments ranging between 0.5 ± 0.12 % and 1.2 ± 0.11 %. A significant difference between zones (p = 0.0153) was observed in the main PERMANOVA test with significantly higher TOM contents on the Goote

Bank in comparison to all monitoring areas on the Thorntonbank. Furthermore, the eastern and western impact areas had a significantly lower TOM content than the Thorntonbank edge area (Table 5). PERMDISP also had a significant result (p = 0.0081) for factor zone.

Table 4. Number of sampled stations from 2005 – 2012. Mean total abundance (ind m^{-2}), total number of species per sample, Shannon H’ diversity index per sample, biomass (mg m^{-2}), median grain size (µm) and total organic matter content (TOM %) in the five monitoring areas: Goote Bank (GC), Thorntonbank control (TC), Thorntonbank Edge (TE), Thorntonbank eastern impact (TI E) and western impact area (TI W)

2005	**GC**	**TC**	**TE**	**TI E**	**TI W**
# stations	16	15	15	19	11
Abundance	356 ± 57	472 ± 74	428 ± 105	231 ± 49	361 ± 67
Species number	8 ± 0.9	13 ± 0.8	8 ± 1.1	6 ± 0.6	8 ± 1.1
Shannon H'	1.5 ± 0.1	1.4 ± 0.1	1.3 ± 0.1	1.3 ± 0.1	1.5 ± 0.1
Biomass	690 ± 395	253 ± 100	205 ± 76	96 ± 15	164 ± 45
MGS	345 ± 9	337 ± 7	368 ± 17	361 ± 7	354 ± 13
TOM %	0.85 ± 0.08	0.58 ± 0.02	0.74 ± 0.07	0.53 ± 0.02	0.53 ± 0.03

2008	**GC**	**TC**	**TE**	**TI E**	**TI W**
# stations	25	15	12	18	6
Abundance	812 ± 141	449 ± 88	602 ± 158	661 ± 171	447 ± 207
Species number	13 ± 0.7	10 ± 1.1	11 ± 1.2	10 ± 1.0	7 ± 1.7
Shannon H'	1.7 ± 0.1	1.8 ± 0.1	2 ± 0.1	1.6 ± 0.1	1.3 ± 0.1
Biomass	2789 ± 677	1170 ± 340	3376 ± 1403	1789 ± 667	1578 ± 380
MGS	340 ± 15	340 ± 24	366 ± 18	363 ± 8	353 ± 9
TOM %	0.88 ± 0.09	0.90 ± 0.33	0.75 ± 0.05	0.57 ± 0.03	0.58 ± 0.03

2009	**GC**	**TC**	**TE**	**TI E**	**TI W**
# stations	22	15	13	19	10
Abundance	334 ± 37	555 ± 127	568 ± 126	323 ± 41	389 ± 77
Species number	9 ± 0.7	10 ± 0.9	10 ± 1.1	9 ± 0.8	9 ± 0.8
Shannon H'	1.7 ± 0.1	1.7 ± 0.1	1.7 ± 0.1	1.6 ± 0.1	1.7 ± 0.1
Biomass	958 ± 160	1471 ± 344	1002 ± 144	1515 ± 317	694 ± 104
MGS	364 ± 12	340 ± 11	380 ± 22	372 ± 12	370 ± 3
TOM %	0.82 ± 0.05	0.60 ± 0.02	0.64 ± 0.05	0.52 ± 0.05	0.57 ± 0.06

2010	**GC**	**TC**	**TE**	**TI E**	**TI W**
# stations	25	4	12	2	2
Abundance	630 ± 149	246 ± 98	279 ± 77	249 ± 112	356 ± 83
Species number	15 ± 2.0	9 ± 3.0	9 ± 1.1	9 ± 3.0	12 ± 1.5
Shannon H'	2.1 ± 0.1	1.7 ± 0.4	1.7 ± 0.1	1.9 ± 0.3	2.1 ± 0.1
Biomass	3294 ± 725	951 ± 265	1118 ± 265	886 ± 21	2495 ± 1676
MGS	353 ± 11	344 ± 4	376 ± 16	365 ± 47	331 ± 2
TOM %	1.15 ± 0.11	0.64 ± 0.11	0.82 ± 0.13	0.49 ± 0.12	0.62 ± 0.03

2012	**GC**	**TC**	**TE**	**TI E**	**TI W**
# stations	4	4	10	/	/
Abundance	261 ± 108	314 ± 128	410 ± 110	/	/
Species number	11 ± 4.3	10 ± 1.7	10 ± 1.8	/	/
Shannon H'	2 ± 0.3	1.9 ± 0.2	1.7 ± 0.2	/	/
Biomass	816 ± 152	1002 ± 203	758 ± 162	/	/
MGS	410 ± 41	360 ± 34	359 ± 18	/	/
TOM %	0.94 ± 0.22	0.75 ± 0.17	0.69 ± 0.10	/	/

Table 5. Pair-wise PERMANOVA test of the Total Organic Matter (TOM) content for term zones (significant results p < 0.05). Goote Bank (GC), Thorntonbank control (TC), Thorntonbank Edge (TE), Thorntonbank eastern impact (TI E) and western impact area (TI W)

TOM contents	**t**	**p-value (perm)**	**Unique perms**
GC - TC	2.049	0.0482	9831
GC - TE	28.498	0.0049	9821
GC - TI E	50.113	0.0002	9828
GC - TI W	36.667	0.0009	9834
TE - TI E	36.722	0.0021	9823
TE - TI W	23.893	0.0199	9838

3.2.2 Macrofaunal communities

With a significant effect for the factor Year (*p* = 0.0314) (main test PERMANOVA), the mean total abundance was significantly higher in 2008 in comparison to 2005 (*p* = 0.0101) and 2012 (*p* = 0.0187). Between 2005 and 2012 the mean total number of species ranged between 6 ± 0.6 and 15 ± 2.0 species per sample. The main PERMANOVA test showed significant differences between years (*p* = 0.0008) with significantly higher mean total number of species in 2008, 2009 and 2010 compared to the reference year 2005 (Table 6). Even after log transformation, PERMDISP for both mean total abundance and mean total species number remained significant with respectively *p* = 0.0272 and *p* = 0.0266.

Table 6. Univariate, pair-wise PERMANOVA test for term year (significant results p < 0.05)

Species number	**t**	**p-value (perm)**	**Unique perms**
2005 - 2008	3.6201	0.0006	9816
2005 - 2009	3.699	0.0004	9853
2005 - 2010	2.3731	0.0188	9818
Shannon H' diversity	**t**	**p-value (perm)**	**Unique perms**
2005 - 2008	4.3003	0.0001	9840
2005 - 2009	4.2403	0.0001	9850
2005 - 2010	4.3908	0.0001	9825
2005 - 2012	3.5167	0.001	9835
2008 - 2010	2.0725	0.0432	9839
2009 - 2010	2.1306	0.0383	9816
Biomass	**t**	**p-value (perm)**	**Unique perms**
2005 - 2008	12.694	0.0001	9838
2005 - 2009	12.93	0.0001	9824
2005 - 2010	7.3065	0.0001	9817
2005 - 2012	5.3859	0.0001	9848
2008 - 2009	2.4688	0.014	9843

Both diversity (H') (p = 0.0001) and total biomass (p = 0.0001) were significantly affected by the factor year (Table 6). Significantly lower biomass values were observed in 2005. Another significant decrease was measured for biomass in 2009 in comparison to 2008 (p = 0.014), most likely due to the occurrence of *Ophiura albida* (3 - 13 ind m^{-2}) in 2008 and its absence from 2009.

The macrobenthic community composition based on total abundance and biomass showed a significant interaction for the term zones x year (p = 0.0446 for abundance and p = 0.0084 for biomass). Pair-wise comparisons within the interaction term for pairs of levels of the factor zones showed a significant difference in macrofaunal community composition (based on abundance) in the construction year (2008). The communities at TI W, TI E and TE were significantly different from those in the control zones on both the Thorntonbank and Goote Bank (see Addendum I, Table 1). TI E showed significant interactions with the control zones from 2005 until 2009. From 2010 onwards, no significant difference in community composition was detected between the two impact and control zones. Similar patterns were measured for the macrobenthic community composition based on biomass with mainly significant differences in community between the impact zones (TI E and TI W) compared to the two control zones (TC and GC) in 2008. PERMDISP analysis remained significant for community composition based on abundance (p = 0.0014) and biomass (p = 0.0004).

In the reference year (2005) the macrobenthic community in all areas was dominated by *Nephtys cirrosa* (25 – 41 %) and *Urothoe brevicornis* (20 – 37 %). A shift in dominant species occurred in 2008 with a high dominance of *Spiophanes bombyx* (16 – 38 %) with the highest mean contribution to the average total density on the Goote Bank (38 %), which disappeared again from 2009 onwards. The mean abundance (ind m^{-2}) of the most characteristic species (based on SIMPER) for each area is given in Table 7.

Table 7: Mean abundance (ind m^{-2}) of the main characteristic species (based on SIMPER) in every zone: Goote Bank (GC), Thorntonbank control (TC), Thorntonbank Edge (TE), Thorntonbank eastern impact (TI E) and western impact area (TI W).

		2005	**2008**	**2009**	**2010**	**2012**
GC	*Nephtys cirrosa*	67	66	76	38	12
	Spiophanes bombyx	17	367	19	47	0
	Spio sp.	7	19	19	34	19
	Urothoe brevicornis	131	61	36	28	5
	Ophelia borealis	7	6	20	23	17
	Eteone sp.	4	11	7	6	0
	Nephtys caeca	0	23	3	2	5
	Bathyporeia elegans	0	29	11	6	2
	Glycera lapidum	7	26	8	10	2

		2005	2008	2009	2010	2012
TC	*Nephtys cirrosa*	94	99	136	78	44
	Spiophanes bombyx	32	88	134	7	15
	Urothoe brevicornis	231	40	97	15	88
	Spio sp.	12	18	16	15	7
	Ophelia borealis	3	5	13	17	2
	Bathyporeia elegans	0	26	45	10	37
	Bathyporeia guilliamsoniana	34	12	4	22	5
	Eteone sp.	5	8	10	5	0
	Thia scutellata	13	2	5	7	0
		2005	**2008**	**2009**	**2010**	**2012**
TE	*Nephtys cirrosa*	63	82	112	58	31
	Urothoe brevicornis	198	46	111	71	66
	Spiophanes bombyx	24	150	108	5	19
	Spio sp.	10	28	11	11	14
	Bathyporeia guilliamsoniana	19	17	14	11	14
	Bathyporeia elegans	0	40	12	8	35
	Echinocardium cordatum	6	8	58	14	7
	Ophelia borealis	1	5	4	11	2
	Thia scutellata	9	8	10	5	1
	Nephtys juv.	0	0	1	15	19
		2005	**2008**	**2009**	**2010**	**2012**
TI E	*Nephtys cirrosa*	65	126	131	68	/
	Spiophanes bombyx	14	242	28	34	/
	Spio sp.	13	21	19	24	/
	Urothoe brevicornis	85	38	33	5	/
	Eteone sp.	3	37	9	5	/
	Bathyporeia elegans	0	77	14	34	/
	Ophelia borealis	3	3	14	15	/
		2005	**2008**	**2009**	**2010**	**2012**
TI W	*Nephtys cirrosa*	89	115	140	97	/
	Spiophanes bombyx	32	185	56	39	/
	Urothoe brevicornis	132	6	39	58	/
	Bathyporeia guilliamsoniana	50	18	4	24	/
	Spio sp.	13	15	7	15	/
	Eteone sp.	4	29	13	5	/
	Bathyporeia elegans	0	41	41	15	/

3.2.3 Benthic indicator

The average BEQI scores and the accompanying status (Table 8) classified the difference between control and impact area in most periods and assessments as acceptable (good). Only in 2008 and for the TI WE zone in some years (2010, 2012), certain assessments indicated an unacceptable situation (moderate status). Comparing the average BEQI scores over the years, the lowest values were measured in 2008. The power class of the confidence of the assessments was mainly classified as good to moderate. Except in the years 2010 and 2012 in the TI E area, where no confidence assessment could be made with the available data.

Table 8. Average BEQI results for each design/period. Values in bold indicates a good power for all BEQI parameters, values normal indicates a moderate power for one of the BEQI parameters and asterix means no confident assessment possible with the available data. Status codes: Black: high status, Dark grey: good status, Light grey: moderate status

Control	Area	Zone	2008	2009	2010	2012
within year	TI (TI E + TI W)	Top+Edge	**0.628**	**0.802**	**0.695**	*
<2005	TI (TI E + TI W)	Top+Edge	**0.553**	**0.743**	**0.676**	**0.677**
2005	TI (TI E + TI W)	Top+Edge	**0.611**	**0.708**	**0.637**	**0.664**
within year	TI E	Top	**0.628**	**0.731**	**0.764**	
<2005	TI E	Top	**0.588**	**0.687**	0.607	
2005	TI E	Top	**0.542**	**0.764**	0.691	
within year	TI W	Top	0.674	**0.789**	**0.653**	
<2005	TI W	Top	0.618	0.716	0.728	
2005	TI W	Top	0.676	**0.697**	0.769	
within year	TI E	Edge	0.666	0.701	**0.641**	*
<2005	TI E	Edge	0.57	0.608	*	*
2005	TI E	Edge	**0.598**	**0.636**	*	*
within year	TI W	Edge	0.45	0.671	0.44	0.414
<2005	TI W	Edge	0.616	0.609	**0.538**	0.511
2005	TI W	Edge	**0.609**	0.625	**0.576**	0.524

4. Discussion

4.1 Macrobenthic assemblages on two offshore sandbanks

The long-term analysis of the macrofauna shows a community with a relatively low species abundance (180 – 812 ind m^{-2}) and diversity (6 – 15 species per sample) on the Thorntonbank and Goote Bank in the eastern part of the BPNS. The community on both sandbanks is dominated by a few species with the polychaete *Nephtys cirrosa* and the amphipod *Urothoe brevicornis* as the main re-occurring dominant species throughout the years. Different macrobenthic communities were characterised in the BPNS by Van Hoey et al. (2004) based on species composition and habitat preferences (sedimentology and bathymetry). Four main macrobenthic communities were distinguished, linked through six transitional species assemblages (Van Hoey et al., 2004). The biological characteristics on both offshore sandbanks in this study classify the macrofauna into a transitional community between the *Nephtys cirrosa* and the *Ophelia borealis – Glycera lapidum* community. Physical parameters of the area support this with a depth range of 10 - 20 m and coarse sandy sediments with a median grain size between 331 ± 20 µm and 410 ± 41 µm.

Throughout 32 years, a strong temporal variability was detected on the Thorntonbank and Goote Bank typical for the macrofauna of the coastal region (Frid, 2011; Kröncke, 2011; Kröncke et al., 1998; Van Hoey et al., 2007) with variable weather conditions (Fromentin and Ibanez, 1994), food availability (Kröncke, 2011) but also recruitment intensity (Desroy and Retiere, 2001). Various studies have detected changes at a decadal scale (Wieking and Kroncke, 2001) in the macrofaunal community structure in the North Sea and related them to fluctuations in the winter North Atlantic Oscillation (NAO) Index (Kröncke et al., 1998; Kröncke et al., 2013; Reiss et al., 2006). Negative and positive NAOs in Europe are caused by changing atmospheric pressures between Iceland and the Azores (Osborn, 2011). The NAO influences the hydroclimatic state of the North Sea through changes in precipitation rate, sea surface temperature and current flows (Wieking and Kroncke, 2001). Extremely cold winters are reflected in a negative NAO and have been shown to strongly affect the macrobenthic community parameters (e.g. abundance, species number and biomass) by decreasing the sea surface temperature (Kröncke et al., 2013; Neumann et al., 2009; Weijerman et al., 2005). After the strong winter of 1995/1996, Reiss et al. (2006) and Van Hoey et al. (2007) detected a decrease in abundance and diversity and a change in macrobenthic community structure up to 1998. Even though the long-term dataset was incomplete in this study, a significant decrease in macrobenthic abundance was similarly detected in 1998. The significantly lower diversity in 2005 could not immediately be linked to clear changes in NAO or sea surface temperature; however a slight decrease in sea surface temperature was measured in 2003 and an abrupt biological regime shift in the macrobenthic community was observed by Kröncke et al. (2013) after a negative NAO in 2000/2001. Dissimilarities between macrofaunal communities of the control areas and TI E were already detected before construction (2005), suggesting that the natural variability between these areas cannot be neglected.

4.2 Macrobenthic response to construction activities

While temporal trends for both sandbanks were obvious during the investigated 32 years, differences in macrofaunal communities between the Thorntonbank and Goote Bank were only detected in the autumn of 2008 until 2010, equalising again in 2012. The TI W (where the six foundations were installed in 2008) and TE areas showed a shift in macrobenthic community composition in comparison to the control areas in the construction year due to small changes in species dominance. In addition, BEQI revealed a moderate ecological status in 2008, in contrast to the good status acquired in other years, providing a signal of change within this period. We suggest that the observed changes are related to the (pre-) construction dredging activities of the GBFs on the Thorntonbank which commenced in the first half of 2008 (approximately 6 - 8 months before sampling). The construction activities included dredging for cable laying and for the creation of foundation pits (7 m below the seabed), together with the removal of sediments to disposal areas 300 m away from the foundation pits (Peire et al., 2009). Van Dalfsen et al. (2000) observed community changes one year after short-term (< 1 year) dredging activities in the North Sea and the Mediterranean Sea with an increased abundance of opportunistic polychaete species. A recovery of the initial long-lived macrofaunal species was observed within two years. The macrofaunal community in the present study progressed towards a recolonisation from 2009 onwards, approximately 1.5 years after construction activities ceased. Kenny and Rees (1996) similarly illustrated a fast recolonisation after cessation of dredging with a complete recovery of dominant species only 8 months later. Dredging has also been related to the direct removal of species and a decrease in species abundance, diversity and biomass (Newell et al., 1998; Vanaverbeke et al., 2007). Slightly lower species diversity was indeed observed in 2008 at the TI W area in comparison to the control areas (but not significantly). However, a higher total abundance was observed in the construction year which can be related to an overall increase of the opportunistic r-selected species *Spiophanes bombyx* in all areas except the control area on the Thorntonbank where the mean abundance of *S. bombyx* increased in 2009.

With a strong relation to the sedimentological characteristics of the surrounding seabed (Van Hoey et al., 2004), macrofauna are highly susceptible to the removal of sediments through dredging activities. Dredging can affect the seabed topography and sediment composition by direct removal of the substrates (Desprez, 2000; Newell et al., 1998). In active areas such as the BPNS, dredged sites will rapidly be infilled by surrounding fine and mobile sands (Desprez, 2000). In this study, limited to no changes in the sediment grain size was observed 6 - 8 months after construction, most likely due to a fast recovery of the sedimentological characteristics (Desprez, 2000; van Dalfsen et al., 2000), which will simultaneously facilitate the macrofaunal recovery (Bonne, 2010). Previous studies have illustrated that short-term dredging activities can have minor effects on the sediments while macrobenthic communities can be seriously altered for a certain amount of time (Desprez, 2000; Kenny and Rees, 1996; van Dalfsen et al., 2000).

The OWF was completed with the construction of 48 jacket foundations in 2011 (Brabant et al., 2012). No direct changes in species composition were observed on the Thorntonbank after this period. Impact samples could not be obtained in 2012 but effects to the macrofaunal community were also lacking between the edge and control areas, while a significant difference in macrofaunal community was observed between these areas after construction of the GBFs in 2008. Either the pre-construction dredging activities related to GBFs were more intense or the disturbance to the community was missed by the lack of samples in 2011 and a rapid recovery by 2012. Bonne et al. (2010) observed comparable results on an actively dredged subtidal sandbank in the BPNS (Kwintebank), where a recovery of the macrobenthic community was suggested one year after cessation of dredging. With a rapid recovery, the macrobenthic community on the Thorntonbank appears to be very resilient and well adapted to physical disturbances. Strong currents, cold winters and severe storms frequently occur in the BPNS causing natural disturbances to the seabed and naturally creating dynamic macrofaunal communities that are well adapted to change (Kenny and Rees, 1996). However, many other anthropogenic activities have also dominated in these areas such as beam trawl fisheries and sand extraction (Bonne, 2010; Eastwood et al., 2007; Foden et al., 2009; Vanaverbeke et al., 2007; Vanosmael and Heip, 1984). Bottom fishing gears such as beam trawling impact the macrobenthic communities by disturbing the seabed up to a depth of 3 – 6 cm, subsequently affecting species abundance and diversity by causing mortality and injury (de Groot, 1984; Rabaut et al., 2008). Furthermore, both the Thorntonbank and Goote Bank have a history as sand extraction sites (Vanosmael and Heip, 1984), suggesting an additional adaptation of the macrobenthic community on these sandbanks to intense anthropogenic stress. The impact related to the construction of six GBFs could therefore be comparably to the impact associated to previous sand extraction activities in the area.

At present, no substantial short-term changes due to the operational phase of the six GBFs were observed at a large spatial scale. The observed changes mostly fell within the acceptable range measured by BEQI (good status). As the entire OWF was only completed in 2012, it was too early to detect effects related to the exclusion of beam trawl fisheries, changing hydrodynamics or increased organic enrichment at a large-scale. As these effects have been observed in the area but at a smaller scale (Coates et al., 2014), the macrobenthic community within the OWF should be closely followed up during the coming decade.

4.3 Indicators: a quick tool to pick up signals of change?

The applicability of BEQI in signalling changes of the soft-substrate macrobenthic ecosystem within and around the OWF on the Thorntonbank was tested using abundance, species number and species composition. The detected changes could be the result of certain anthropogenic activities, related to the construction of the OWF, as explained above. The application of benthic indices such as BEQI, provide a fast tool to determine if changes affect sediments classified as a ‘good’ status. In most periods and assessments, the BEQI status could be classified as good. No unacceptable differences were observed between the benthic characteristics in the control and impact area. Changes ranged within the expected natural variability of that area and year. Therefore, indicator tools reinforce the observed patterns of the classical analyses in this study suggesting a suitable use to provide management advice. The different assessments (control dataset, or pooling of data) present small changes in results but with certain consistent patterns (Table 8). Two aspects are essential in this type of assessment. Firstly, the choice of the control data, where we used three sets in this study: (1) comparison of control and impact data within the same year, (2) data from the reference year 2005 or (3) 'historic' data from the location (from before 2005). In general, the assessment with control data within the year reveals slightly higher EQR values in the impact site, compared to the use of temporal control datasets. This can be related to the fact that benthic characteristics show a year-to-year variability affecting the assessment results. In this study, different levels of confidence (statistical power) were obtained with the number of samples available (Table 8). Therefore, it is important to have a balanced design within monitoring programmes of anthropogenic activities.

5. Conclusions

The macrofaunal community on the Thorntonbank has been shown to be highly dynamic and adapted to short-term stress. The community on the impacted sandbank showed main differences with the control areas during the construction of gravity based foundations, followed by a rapid recovery. For most areas and periods the BEQI indictor had a good status score for the benthic characteristics between the impact and control areas, indicating no severe impacts on the benthic soft sediment community at this point in time. However, cumulative effects related to the construction and operation of several OWFs on the sediment composition, topography and macrofaunal community remains unknown. Abundance levels of the macrofaunal species might recover after construction but the risk exists that the pre-impacted communities will shift to a different state, possibly affecting species at a higher trophic level (Kenny and Rees, 1996). This could especially be of importance in areas where the macrofauna is less resilient to anthropogenic disturbance. Furthermore, this study illustrated the effectiveness of indicator tools, such as BEQI, as fast methods to provide management advice in impact studies.

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CHAPTER 3

Short-term effects of fishery exclusion in offshore wind farms on macrofaunal communities in the Belgian part of the North Sea

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Abstract

With the wide scale construction of offshore wind farms (OWFs) throughout the entire North Sea, large areas are permanently being closed to fishery activities. Bottom trawling has affected macrobenthic assemblages for centuries, especially the fragile and long-lived species. Due to the closure of fisheries in many OWFs, opportunities are provided to investigate the potential recovery of vulnerable species. The soft-substrate macrobenthic community was investigated from 2008 – 2012, before and after the construction of an OWF in the Belgian part of the North Sea, situated on the Bligh Bank which is part of the Hinder Bank sandbanks. The fishery enclosed area (21 km²) within the OWF (No Fishery area) was compared with a surrounding control area where regular fishing activities were registered through vessel monitoring system (VMS) data throughout the period 2010 - 2011. Three years after the exclusion of fisheries, subtle changes within the macrobenthic community were observed in the No Fishery area. The benthic mysid shrimp *Gastrosaccus spinifer* (30 ± 15 ind m^{-2}), tube-building polychaetes Terebellidae sp. (196 ± 151 ind m^{-2}) and the echinoderm *Echinocyamus pusillus* (73 ± 71 ind m^{-2}), sensitive to trawling activities, showed an increased abundance in the No Fishery area. With an expansion of the wind farm concession area to 238 km² in the future, the likely increase of dense Terebellidae patches (e.g. *Lanice conchilega* reefs) within the No Fishery area could create an ecological important large-scale refugium. This study creates a baseline for the evaluation of long-term changes due to the fishing impacts and effects related to the presence of OWFs.

Keywords

Fishery exclusion - Wind power - North Sea - Short-term - Macrobenthos

1. Introduction

The development of Offshore Wind farms (OWFs) across the North Sea is increasing rapidly to create renewable energy sources as an alternative for fossil fuels (Corbetta et al., 2013). The seascape of the North Sea is being altered in a large area, causing unknown and probably cumulative effects to the marine environment at different scales. Environmental effects of OWFs to the soft-substrate macrofauna can be divided into relatively short-term effects (several months) during the (pre)-construction phase (Coates et al., submitted) and effects related to the much longer (> 20 years) operational phase. The operational phase of an OWF could alter the soft substrate macrofaunal community due to the prohibition of beam trawl fisheries in most OWFs. The North Sea has been heavily trawled for centuries with an increasing trend in the 1960s and 1970s (de Groot, 1984; Jones, 1992; Kaiser et al., 2002). Beam trawling has a direct physical impact by scraping and ploughing the seabed at least up to a depth of 3 – 6 cm, re-suspending sediments and removing or damaging non-targeted benthos (Bergman and Hup, 1992; Dayton et al., 1995; Jones, 1992; Rabaut et al., 2008). Over the past decades, macrofaunal assemblages in heavily trawled areas have shifted towards an alternative stable state with a dominance of opportunistic, short-lived and fast-growing species with high reproduction rates (Collie et al., 2000; Frid et al., 2000; Jennings et al., 2001; Kaiser et al., 2002; Kaiser and Spencer, 1996). Furthermore, additional food supplies due to bycatch discards (Enever et al., 2007) attract and increase abundances of scavenging and predatory species (e.g. *Asterias rubens* and *Pagurus* sp.) to these fishing areas (Dannheim et al., 2014; Rumohr and Kujawski, 2000).

Long-term changes and potential recovery of the macrofauna are not only dependent on the frequency and scale of trawling but also on the nature of the sediments and the existing resilience to natural disturbances (Collie et al., 2000; Kaiser et al., 2002). Macrofaunal assemblages in sandy substrates are adapted to the high natural stress of the area due to strong tidal currents and frequent storms (Rijnsdorp et al., 1998), leading to a faster recovery after the physical disturbance of anthropogenic activities (Bonne, 2010; Coates et al., submitted). However, these substrates are mostly located in areas which have been subjected to frequent trawling for centuries (Kaiser et al., 2002; Rijnsdorp et al., 1998). Slow-growing or fragile macrofaunal species, living in the upper layer of sandy sediments such as certain bivalves (e.g. *Spisula* sp.), echinoderms (e.g. *Echinocardium cordatum*) and tube forming polychaetes (Terebellidae sp.), are known to be highly vulnerable to frequent trawling activities and have declined in abundance throughout the past century (Bergman and Hup, 1992; de Groot, 1984; Jennings et al., 2001; Kaiser and Spencer, 1996; Kröncke, 2011; Tuck et al., 1998). A long-term study on the Dogger Bank, a fine sandy bank in the Southern North Sea, attributed the disappearance of dense *Spisula* and *Mactra* patches throughout the 20th century to the increased fishing pressure (Kröncke, 2011). Kröncke (2011) observed the random occurrence of new patches which would disappear after weeks or months most likely due to fishing pressure. These results suggest the viable ability of such bivalves to re-establish dense patches if trawling would be prohibited over longer periods. In the Irish Sea, Kaiser and Spencer (1996) observed a higher abundance of tube building polychaetes such as *Lagis koreni* and Terebellidae sp. in unfished areas in comparison to fished areas. Likewise, video analysis in a closed sandy Bay off the coast of Scotland revealed a higher number of *Lanice conchilega* (Terebellidae sp.) beds in comparison to a Bay open to frequent trawling (Defew et al., 2012). The higher occurrence of these habitat-structuring beds would lead to the development of a more diverse community, as closely associated species of the *L. conchilega* beds (e.g. *Eumida sanguinea*) are known to be highly sensitive to fishery impacts (Rabaut et al., 2008). The long-term prohibition of beam trawling within OWFs could provide fragile and long-lived species with the space and time to re-establish and recover, potentially increasing the habitat complexity of the area (Defew et al., 2012).

At present, three OWFs have been constructed in the Belgian part of the North Sea (BPNS). The OWF constructed on the Bligh Bank was the first to cover a large area with 55 monopile foundations. Within the Belgian OWFs all vessels, including beam trawl fisheries, are prohibited. Data from the satellite based Vessel Monitoring System (VMS) is used to estimate fishing effort over time (Foden et al., 2010). VMS data provides information on the position of fishing vessels larger than 15 meters (Mills et al., 2007). Vandendriessche et al. (2013b) investigated the presence of Belgian, Dutch and British fishing vessels around the Belgian OWFs based on VMS data and visual observations of smaller vessels (< 15 m). In the BPNS, a maximum of 401 - 800 VMS registrations have been detected per grid cell (3 km²) per year, with a decreasing trend (51 – 200 VMS registrations) in more offshore areas. Inside the Bligh Bank OWF, 1 – 50 VMS registrations per grid cell were observed in 2010 and 2011. In 2011, a slight increase in fishing pressure was observed around the wind farm with 51 – 100 registrations per grid cell.

Until now, the macrofauna has not been investigated within a large area closed to fishery activity in the BPNS. The OWF creates an ideal situation to record and closely follow-up any macrobenthic recovery processes related to the prohibition of beam trawl fisheries during a long period. The macrofaunal species inhabiting the sandy substrates of the Bligh Bank form a typical community for the BPNS, which has adapted to the natural stress of the area and anthropogenic impacts (e.g. beam trawl fisheries) (Reubens et al., 2009). The recovery potential of macrofaunal communities in sandy sediments have only been derived from short-term experimental trawling studies. Collie et al. (2000) suggested a recovery potential of 100 days after one trawling event for a macrobenthic community dominated by short-lived species. With a clear history of frequent trawling in the southern North Sea and an average rate of 2 – 3 disturbances per year (Collie et al., 2000), a much longer recovery time is expected (Rijnsdorp et al., 1998). Recovery rates of 12 months to 4 years, after sediment extraction in terms of abundance and diversity (Newell et al., 1998), could suggest a possible timeframe for the increased recruitment of long-lived and fragile organisms inside fishery excluded areas (Hiddink et al., 2006).

The soft sandy sediments and accompanying macrofauna were sampled around (control area) and inside (No Fishery area) the Bligh Bank OWF before construction (2008), during (2009) and 2 - 3 years after construction and implementation of fishery exclusion (2011 – 2012). After three years of fishery exclusion, we hypothesise that the macrofaunal community inhabiting the sandy substrates of the Bligh Bank, will demonstrate first signs of recovery, with an increase in abundance of known fragile and long-lived macrobenthic species to the area.

2. Material and methods

2.1 Study area

The Belgian OWF concession zone is situated at the eastern side of the BPNS (Fig. 1). The Bligh Bank belongs to the most eastern part of the Hinder Banks, approximately 40 - 50 km offshore. The construction of the first phase of the Bligh Bank OWF commenced in 2009 and was completed in early 2010. The OWF consists of 55 monopile foundations with a total capacity of 165 MW. The foundations are located 500 – 650 m apart at a water depth ranging between 15 – 40 m. All vessels including beam trawl fisheries have been excluded from this area since 2009 with a 500 m safety radius around the OWF (Fig. 1 Blue area), creating a large area of approximately 21 km² closed to fishing activities.

2.2 Sampling design and treatment

A Before, After Control Impact (BACI) design was applied with the baseline study (Before) carried out in 2008. Since then, samples were collected in autumn (September – October) in 2009 (construction year), 2011 and 2012 (Table 1). Samples could not be obtained in 2010 as the research vessel was prohibited from entering the wind farm. The collected samples on the Bligh Bank were divided into two areas: the No Fishery area inside the 500 m exclusion zone and a control area outside the exclusion zone. All stations were positioned to cover the entire area, including the gullies and tops of the sandbanks. Samples were collected inside the No Fishery area from a small survey vessel (Geosurveyor IV) in 2011 and the R.V. Simon Stevin in 2012. All other samples were obtained from the *RV* Belgica.

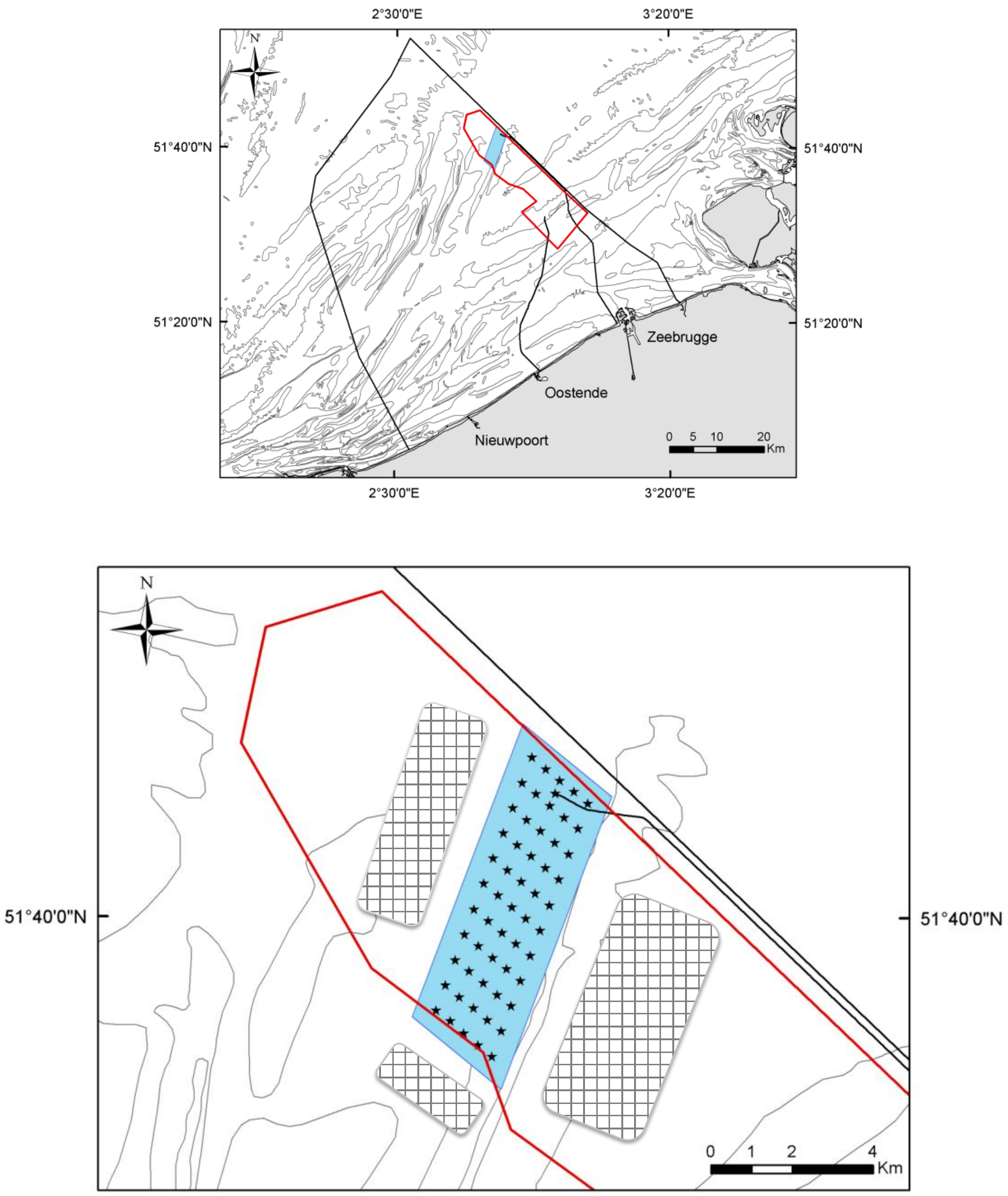

Fig. 1. Upper map: The Belgian part of the North Sea containing the offshore wind farm concession area (red area). Map below: The Bligh Bank offshore wind farm consisting of 55 monopile foundations (stars) with samples taken inside the No Fishery area (blue area) and control samples outside the No Fishery area (hatched areas).

2.2.1 Biotic data

Samples were collected by means of a 0.1026 m² Van Veen grab, sieved on-board over a 1 mm sieve table and subsequently fixed in an 8 % formaldehyde-seawater solution. In the laboratory, samples were stained with 1 % Rose Bengal and rinsed over a 1 mm sieve. After sorting, organisms were identified to species level, whenever possible, and subsequently stored in a 4 % neutralised formaldehyde solution. Biomass (mg) or ash free dry weight (AFDW) was determined for every species per sample by a conversion factor of the wet weight (Brey, 2001) or by regression factors of the length or width with the AFDW. When neither conversion factors nor regressions existed for a certain species, AFDW was determined by cremation. Samples were dried for 48 h at 60 °C and burned in a muffle furnace for 2 h at 550 °C. The AFDW was calculated as the difference between dry weight (DW) and ash weight (AW).

2.2.2 Environmental data

Sediment samples for grain size analysis were taken as a subsample (50 ml) from the Van Veen grab. Median grain size d (0.5) was determined on dried samples (60 °C) with a Malvern Mastersizer 2000G, hydro version 5.40 (laser diffraction method) (Malvern, 1999). Grain size fractions were determined as volume percentages with a range from fine clay (< 4 µm) to coarse gravel/shell material (> 1600 µm). The total organic matter content (TOM %) was measured per sample by applying the following equation: TOM % = [(DW – AW) / (DW - CrW)] x 100. DW was determined after 48 h at 60 °C and the AW after 2 h 20 min at 550 °C. Every used crucible was weighed (CrW) in order to determine TOM %.

2.3 Data analysis

Species that were not sampled quantitatively with the Van Veen grab (hyperbenthos, meiobenthos and fish) were excluded from the dataset together with extremely rare taxa, defined as species with a unique occurrence and a maximum of three individuals per sample. Certain species were lumped to a higher taxonomic level to take inconsistent species identification into consideration (e.g. *Aricidea* sp., Capitellidae, Cirratulidae, *Diastylis* sp., *Glycera* sp., *Magelona* sp., *Spio* sp., Terebellidae sp.). After the data quality control, a total set of 117 macrobenthic samples and 105 taxa was further used for analysis. The number of individuals per m² (abundance), species richness (N_0), Shannon Wiener diversity index (H’) and biomass (mg m^{-2}) were calculated. Results are expressed as mean ± standard error (SE).

The Plymouth Routines in Multivariate Ecological Research (PRIMER) programme (version 6.1.6) with the PERMANOVA add-on software was used for statistical analyses (Anderson et al., 2008; Clarke and Gorley, 2006). Multivariate and univariate PERMANOVAs (Permutational ANOVAs) were carried out in this study with a two-factor design containing areas (No Fishery area and control) and years (2008 - 2012) as factors. Type III sum of squares was applied as the design was unbalanced, with the number of permutations set to 9999 and the permutation of residuals under a reduced model. When the main PERMANOVA test showed significant effects for factors, a pair-wise comparison was applied between levels of factors. When the main test showed a significant interaction between factors, a pair-wise test was performed within the interaction factor. A significance level of $p < 0.05$ was applied for all tests. The resemblance measure Euclidean distance was applied for the univariate analysis of total density, diversity, biomass and the environmental data (grain size and TOM %). The resemblance measure Bray-Curtis similarity was applied for the multivariate analysis of abundance and biomass data after square -root transformation. Homogeneity of dispersions was tested with PERMDISP using distances among centroids. Data was log (x+1) transformed if PERMDISP was significant. Principal coordinates analysis (PCO) was performed to visualise the multivariate data after calculation of distances among centroids for levels of the factor area or year. Furthermore, a SIMPER analysis was conducted to determine the species with an important contribution to the similarity within communities in the No Fishery and control area. Subsequently, the total abundance of these species was analysed with PERMANOVA.

3 Results

3.1 Environmental characteristics

The average median grain size ranged between 389 ± 9 µm in the No Fishery area in 2012 and 440 ± 16 µm in the control area in 2009 (Table 1). The main PERMANOVA test showed a significance for the factor area ($p = 0.01$) with a higher grain size in the control area. Total organic matter content in the sediments ranged between 0.52 ± 0.05 % in the control area in 2011 and 0.85 ± 0.23 % in the No Fishery area in 2012 (Table 1). The main PERMANOVA test showed a significance for the factor year ($p = 0.0443$). Pair-wise tests showed significantly higher TOM contents in 2012 in comparison to 2008 ($p = 0.0194$) and 2011 ($p = 0.0364$).

3.2 Macrobenthic density, diversity and biomass

The mean total abundance in the control area ranged between 256 ± 30 ind m^{-2} in 2011 and 458 ± 72 ind m^{-2} in 2012. A peak was measured in the No fishery area in 2012 with an average abundance reaching 1027 ± 701 ind m^{-2}, due to a high average abundance of Terebellidae sp. (196 ind m^{-2}), *Eumida sanguinea* (112 ind m^{-2}), *Echinocyamus pusillus* (73 ind m^{-2}) and *Heteromastus filiformis* (64 ind m^{-2}) at one or two stations (see Addendum II, Table 1). A similar peak in average biomass was measured in the No Fishery area in 2012 with a value of 5733 ± 4974 mg m^{-2}. The biomass of *Echinocardium cordatum* was excluded from all samples due to the presence of a few large individuals. The main range in average biomass ranged between 514 ± 111 mg m^{-2} in the No Fishery area in 2011 and 1864 ± 511 mg m^{-2} in the control area in 2012.

The mean number of species (N_0) showed a minimum of 9.8 ± 1.1 species per sample in the control area in 2009 and a maximum number of species per sample of 14.7 ± 1.7 in the control area in 2012.

The Shannon Wiener diversity index (H') showed a similar pattern with a range between 1.6 ± 0.2 in the control area in 2008 and 2.2 ± 0.1 in the control area in 2012. No significant differences in year, area or the interaction factor year x area were observed for all univariate biotic parameters (PERMANOVA).

Table 1. Number of sampled stations from 2008 – 2012 in the No Fishery area and Control area on the Bligh Bank in the Belgian part of the North Sea. Mean total abundance (ind m^{-2}), species number per sample, Shannon H' diversity index per sample, biomass (mg m^{-2}), median grain size (µm) and total organic matter content (TOM %)

	2008	**2009**	**2011**	**2012**
No Fishery area				
# stations	6	16	9	9
Abundance (ind m^{-2})	361 ± 72	434 ± 69	369 ± 87	1027 ± 701
Species number	10.3 ± 1.7	11 ± 0.9	10.4 ± 1.4	12.3 ± 3.4
Shannon H'	1.7 ± 0.2	1.9 ± 0.1	1.9 ± 0.2	1.7 ± 0.3
Biomass (mg m^{-2})	802 ± 253	1338 ± 370	514 ± 111	5733 ± 4974
MGS (µm)	395 ± 8	411 ± 13	397 ± 8	389 ± 9
TOM %	0.59 ± 0.06	0.74 0.09	0.63 ± 0.10	0.85 ± 0.23
Control area				
# stations	15	25	18	18
Abundance (ind m^{-2})	436 ± 67	345 ± 50	256 ± 30	458 ± 72
Species number	10.7 ± 1.3	9.8 ± 1.1	10.3 ± 1.2	14.7 ± 1.7
Shannon H'	1.6 ± 0.2	1.7 ± 0.1	1.9 ± 0.1	2.2 ± 0.1
Biomass (mg m^{-2})	1656 ± 320	1818 ± 360	1392 ± 479	1864 ± 511
MGS (µm)	421 ± 10	440 ± 16	418 ± 12	409 ± 7
TOM %	0.59 ± 0.05	0.63 ± 0.06	0.52 ± 0.05	0.75 ± 0.07

3.3 Macrofaunal community analysis

105 species were identified in both areas from 2008 to 2012 with 38 malacostraca species, 36 polychaete species, 17 mollusc species and 6 echinoderm species as the four main taxonomic groups. The main PERMANOVA test showed a significant difference in macrobenthic community structure between the two areas (p = 0.0061) and years (p = 0.0001) based on total abundances. Pair-wise tests based on years showed a significant difference in community structure between all years except 2008 – 2009 (Table 2). Based on total biomass, the macrobenthic community structure also showed a significant difference between both areas (p = 0.0003) and years (p = 0.0001) in the main PERMANOVA test. Pair-wise tests based on years also illustrated a significant difference between all

years (Table 2). PCO plots based on distances between centroids with the factor year, clearly illustrate the changes in macrobenthic community structure both for abundance (Fig. 2 Left) and biomass (Fig. 2 Right) from 2008 – 2012.

Table 2. Multivariate pair-wise PERMANOVA analysis for factor year to determine significant differences (p < 0.05) in community structure based on abundance and biomass

	Average abundance			**Average biomass**		
Groups	**t**	**p-value (perm)**	**Unique perms**	**t**	**p-value (perm)**	**Unique perms**
2008 - 2009	1.2892	0.0584	9922	1.397	0.0232	9916
2008 - 2011	1.8533	0.0002	9917	1.8129	0.0002	9908
2008 - 2012	2.2784	0.0001	9924	1.8455	0.0001	9907
2009 - 2011	1.9981	0.0001	9914	1.9873	0.0001	9917
2009 - 2012	2.5152	0.0001	9918	2.1395	0.0001	9925
2011 - 2012	1.7953	0.0003	9934	1.5056	0.0061	9907

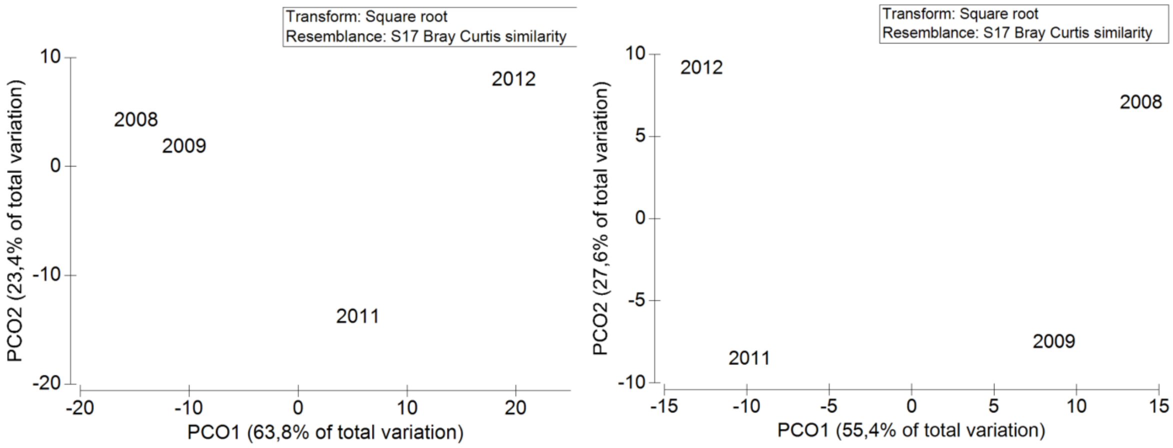

Fig. 2. PCO plot based on distances among centroids with factor year as grouping factor for abundances (left) and biomass (right)

3.4 Characteristic species and dominance

The main characteristic species for every area were determined based on a SIMPER analysis and represented in Table 3. The main PERMANOVA test for the total abundance of *Nephtys cirrosa* and *Spiophanes bombyx* showed a significant effect for the factor year (*p* = 0.0128 and *p* = 0.0025) with a main decrease in abundance in 2012 (Table 4). Even after transformation, PERMDISP was significant (*p* = 0.001) for the factor year for abundances of *S. bombyx*.

Table 3. Mean abundance (ind m^{-2}) and biomass (mg m^{-2}) of the main characteristic species (based on SIMPER) in the No Fishery and control area

		Average Abundance				**Average biomass**			
		2008	**2009**	**2011**	**2012**	**2008**	**2009**	**2011**	**2012**
No Fishery	*Nephtys cirrosa*	106 ± 26	98 ± 16	42 ± 7	36 ± 9	355 ± 103	392 ± 59	131 ± 22	245 ± 77
	Spio sp.	11 ± 5	13 ± 4	32 ± 12	11 ± 3	3 ± 1	4 ± 1	7 ± 2	3 ± 1
	Bathyporeia guilliamsoniana	16 ± 6	67 ± 23	11 ± 6	28 ± 10	32 ± 19	153 ± 65	17 ± 13	22 ± 10
	Bathyporeia elegans	8 ± 8	34 ± 9	19 ± 11	55 ± 36	6 ± 6	28 ± 8	14 ± 10	16 ± 10
	Glycera sp.	44 ± 30	11 ± 5	12 ± 4	44 ± 37	26 ± 15	28 ± 15	17 ± 14	59 ± 36
	Ophelia borealis	0 ± 0	7 ± 3	10 ± 4	18 ± 5	0 ± 0	72 ± 30	21 ± 9	89 ± 50
	Spiophanes bombyx	62 ± 50	46 ± 19	3 ± 2	2 ± 1	114 ± 98	22 ± 13	1 ± 0	1 ± 1
	Terebellidae sp.	5 ± 3	42 ± 28	19 ± 16	196 ± 151	1 ± 0	1 ± 1	13 ± 7	4570 ± 4545
	Aonides paucibranchiata	19 ± 13	6 ± 3	11 ± 7	53 ± 52	4 ± 3	1 ± 0	1 ± 1	17 ± 17
	Gastrosaccus spinifer	5 ± 3	2 ± 2	42 ± 29	30 ± 15	22 ± 16	18 ± 18	104 ± 83	93 ± 49
	Angulus pygmaeus	0 ± 0	5 ± 2	4 ± 3	8 ± 4	0 ± 0	10 ± 5	3 ± 2	31 ± 26
Control	*Nephtys cirrosa*	139 ± 22	73 ± 10	70 ± 8	37 ± 6	757 ± 164	429 ± 59	284 ± 41	444 ± 63
	Bathyporeia guilliamsoniana	36 ± 9	53 ± 12	19 ± 6	42 ± 10	57 ± 20	110 ± 24	23 ± 7	37 ± 10
	Glycera sp.	16 ± 6	25 ± 10	12 ± 2	11 ± 4	11 ± 8	30 ± 10	6 ± 2	3 ± 1
	Spio sp.	8 ± 3	16 ± 7	18 ± 5	3 ± 1	2 ± 1	3 ± 1	5 ± 2	1 ± 1
	Bathyporeia elegans	9 ± 5	8 ± 3	12 ± 3	20 ± 8	3 ± 2	9 ± 4	13 ± 5	5 ± 2
	Spiophanes bombyx	52 ± 18	17 ± 5	2 ± 1	7 ± 2	95 ± 33	4 ± 2	1 ± 1	4 ± 1
	Aonides paucibranchiata	11 ± 6	9 ± 5	14 ± 5	17 ± 5	4 ± 3	1 ± 0	2 ± 1	3 ± 1
	Angulus pygmaeus	5 ± 2	8 ± 3	17 ± 4	9 ± 4	3 ± 2	8 ± 4	22 ± 10	9 ± 4
	Terebellidae sp.	8 ± 3	13 ± 6	12 ± 6	62 ± 28	35 ± 16	0 ± 0	6 ± 3	49 ± 26
	Nephtys juv.	0 ± 0	0 ± 0	11 ± 3	30 ± 7	0 ± 0	0 ± 0	7 ± 3	17 ± 5
	Echinocyamus pusillus	15 ± 9	7 ± 2	9 ± 4	5 ± 2	7 ± 3	8 ± 3	0 ± 0	5 ± 2
	Thia scutellata	8 ± 3	6 ± 2	3 ± 1	3 ± 1	96 ± 92	68 ± 34	32 ± 21	61 ± 41
	Processa modica	4 ± 3	4 ± 1	3 ± 1	9 ± 3	15 ± 14	50 ± 19	46 ± 19	89 ± 34

Table 4. Univariate pair-wise PERMANOVA analysis for factor year to determine significant differences ($p < 0.05$) in total abundance

	Nephtys cirrosa			***Spiophanes bombyx***		
Groups	**t**	**p-value (perm)**	**Unique perms**	**t**	**p-value (perm)**	**Unique perms**
2008 - 2009	19.91	0.0499	9846	13.51	0.1790	9859
2008 - 2011	3.72	0.0004	9818	30.25	0.0035	9856
2008 - 2012	4.91	0.0001	9820	28.90	0.0048	9839
2009 - 2011	24.67	0.0141	9831	28.14	0.0043	9878
2009 - 2012	41.81	0.0001	9837	25.86	0.0066	9847
2011 - 2012	24.66	0.0173	9852	10.35	0.3198	9738

The abundance of Terebellidae sp. showed a significant difference for the factor year (*p* = 0.0224) (PERMANOVA). Pair-wise tests showed an overall increase in the abundance of Terebellidae sp. in 2012 in comparison to 2009 (*p* = 0.0233) and 2011 (*p* = 0.0239). However, PERMDISP was significant (*p* = 0.021) for the factor year even after transformation. A significant interaction for the term year x area (*p* = 0.0436) was measured for densities of *Gastrosaccus spinifer*. Pair-wise comparisons for the term year x area for pairs of levels of the factor year, showed a significant increase in density between 2009 and 2012 (*p* = 0.0181) in the No Fishery area. Pair-wise comparisons for the term year x area for pairs of levels of the factor area, showed no significant differences in abundance of *G. spinifer*. Only borderline differences in terms of significance were observed between the two areas in 2011 (*p* = 0.0525) and 2012 (*p* = 0.0527). PERMDISP was significant for the factor year (p = 0.006) and area (*p* = 0.004), even after transformation. Furthermore, typical fragile species expected to show vulnerability to fishery impacts e.g. *Echinocardium cordatum*, *Spisula* sp. *Echinocyamus pusillus* did not show any significant differences. In terms of biomass only *S. bombyx* and *G. spinifer* illustrated significant differences. The total biomass of *S. bombyx* showed a significant difference between years (*p* = 0.0006; PERMDISP *p* = 0.001) with a significantly higher biomass in 2008 (Table 5).

Table 5. Univariate pair-wise analysis (PERMANOVA) for factor year to determine significant differences ($p < 0.05$) in total biomass for *Spiophanes bombyx*

Groups	**t**	**p-value (perm)**	**Unique perms**
2008 - 2009	3.2083	0.0037	9845
2008 - 2011	3.0173	0.0027	9852
2008 - 2012	2.9686	0.0034	9865
2009 - 2011	1.9	0.0413	9910
2009 - 2012	1.6417	0.08	9903
2011 - 2012	1.4794	0.1496	9848

The total biomass of *G. spinifer* was significantly different between the two areas ($p = 0.0044$; PERMDISP $p = 0.001$). Dominant species are represented in Table 6 with a relatively stable distribution over time with *N. cirrosa* as the main dominant species in both areas (11 – 35 %). However, in 2012 *Bathyporeia guilliamsoniana* dominated in the No Fishery area (20 %).

Table 6. Dominant macrobenthic species from 2008 – 2012 in the No Fishery and control area. Mean contribution (%) to the average abundance

	No Fishery area		**Control**	
	Species	**%**	**Species**	**%**
2008	*Nephtys cirrosa*	35	*Nephtys cirrosa*	35
2009	*Nephtys cirrosa*	27	*Nephtys cirrosa*	28
2011	*Nephtys cirrosa*	16	*Nephtys cirrosa*	32
2012	*Bathyporeia guilliamsoniana*	20	*Nephtys cirrosa*	11

4 Discussion

With the BACI design applied in this study, we were able to relate changes to the macrofauna in time with changes observed between the unfished and control area. The control area was situated on the same sandbank as the unfished area to ensure the comparability of habitats and a similar fishing effort in the past (Grizzle et al., 2009). Three years after the prohibition of fisheries was implemented inside the Bligh Bank OWF, no significant differences could be observed between the unfished and control area (no significant interaction effects between area and year). Main changes illustrated the natural variability in species composition in both areas. A three year fishery exclusion was possibly too short to determine significant changes to the complete macrofaunal community. However, subtle changes within the macrofaunal community were observed in the unfished area in 2012 with an unusually high abundance of certain species. The abundance of the sand burrowing mysid shrimp *G. spinifer* showed higher (marginally significant) differences between the unfished and control area in 2011 and 2012. An increase in maximum mean abundance of the tube building Terebellidae sp. was observed from 2008 to 2012 in both areas. The abundance of Terebellidae sp. increased by a factor 40 in the unfished area (5 ± 3 ind m^{-2} to 196 ± 151 ind m^{-2}) in comparison to a factor 8 in the control area (8 ± 3 ind m^{-2} to 62 ± 28 ind m^{-2}). Therefore, we can suggest that the recruitment of Terebellidae sp. was more successful in the no fishery area.

In the German Bight, higher energy flows through *Owenia fusiformis*, a terebellid tube-building polychaete, were observed in untrawled areas when compared to trawled areas (Dannheim et al., 2014). Most terebellid individuals in our study were too small to be identified to species level. High densities of *Lanice*-bed associated species (i.e *Eumida sanguinea* (112 ind m^{-2})) suggest that terebellid species at the Bligh Bank are *L. conchilega*. *Lanice conchilega* patches have occasionally been observed on a large-scale before, but mostly on a nearby sandbank (Thorntonbank) and never with such high abundances of the associated species (Coates and Vincx, 2010; De Maersschalck et al., 2006; Reubens et al., 2009). Increased densities around a gravity-based foundation on the Thorntonbank are explained by changing hydrodynamics and the depositional flow of organic matter in the wake of the foundation (Coates et al., 2014). Increased densities were restricted to the immediate vicinity (< 50 m) of the foundation. The increased terebellid densities at the Belwind OWF were observed further away from the wind turbines, where no differences in hydrodynamics (Van den Eynde et al 2013) or significant increased levels of sedimentary organic matter (this study) were observed. Hence, we suggest that the detected terebellid patches with their associated species are most likely related to the prohibition of fishing within the OWFs. However, an expansion of the observed small-scale enrichment by Coates et al. (2014) is not ruled out for the future and could be facilitated by the prohibition of beam trawl fishing. The quantity and density of terebellid patches will then increase inside the unfished areas, simultaneously increasing the habitat complexity of the area (Defew et al., 2012; Petersen and Malm, 2006). Limited studies have dealt with the effects of long-term and large-scale prohibition of trawling on the macrofauna in comparable habitats. In the western Gulf of Maine an early stage of infaunal recovery was illustrated, with a higher abundance and diversity in the sandy sediments of a large area (30 x 110 km²) four to six years after fishery closure (Grizzle et al., 2009). Similarly, a study carried out in the north-west Mediterranean Sea observed significantly higher densities of vulnerable filter feeding organisms in muddy sediments (2.7 km²) after a 20 year fishing prohibition (de Juan et al., 2007). In terms of biomass, recovery rates are estimated to be longer (Hiddink et al., 2006), in accordance with our results as only the total biomass of *S. bombyx* and *G. spinifer* illustrated changes. Therefore, it is expected that the patterns observed in this study will only become stronger after longer-term monitoring throughout the 20 year concession period.

Various studies have illustrated the vulnerability of *E. cordatum* to beam trawling (Bergman and Hup, 1992; de Groot, 1984; Jennings et al., 2001; MacDonald et al., 1996). With a sexual maturity age of three years (Fish and Fish, 1996), the time frame applied in this study was possibly too short and it was therefore too early to detect an increase of *E. cordatum*. However, the smaller Pea urchin *E. pusillus* did show slight changes with an increased average abundance in 2012 (73 ind m^{-2}). With a reproductive maturity age of one year (Fish and Fish, 1996), this species could be illustrating the first signs of recovery for fragile echinoderms. Furthermore, VMS data and visual observations revealed the entrance of illegal trawling vessels into the OWF throughout the sampling period (Vandendriessche et al., 2011). Depending on the frequency of illegal trawling, the recovery rates of fragile and long-lived species will take even longer (Rijnsdorp et al., 1998) providing a viable explanation for why the macrofaunal community has not shown a significant recovery at this point in time.

A changing macrobenthic community and increased amounts of dense terebellid patches in the unfished area will enhance the attraction of higher trophic levels such as demersal fish species within the OWF (Bergström et al., 2012; Kaiser et al., 2002; Petersen and Malm, 2006). Although it appears too early to detect significant large-scale changes in the macrobenthic community, a study carried out on the diet of the common dab in 2010 did illustrate a higher stomach fullness index inside another Belgian OWF in comparison to a reference area (Derweduwen et al., 2012; Vandendriessche et al., 2013c). With amphipods, decapods, mysids and polychaetes as the main prey species of dab (Vandendriessche et al., 2013c), the increased abundance of *G. spinifer* and Terebellidae sp. observed in this study, will enhance the food availability for demersal fish species within the OWF. An increase of the predatory pressure inside the OWF could also clarify why so far, no significant differences have been observed between the unfished and control area. At a large-scale (> 180 m distance from the turbines), no significant attraction of demersal fish species within the Belgian OWFs have thus far been recorded (Vandendriessche et al., 2013a). However, larger individuals of plaice and turbot were observed within the Bligh Bank OWF, suggesting a refugium effect due to the prohibition of fishing activities (de Juan et al., 2007; Vandendriessche et al., 2013c). Together with a possible increase of terebellid patches, the enclosed area within the OWF could evolve into an ecological important area during the coming 20 years.

5 Conclusions

With signs of a changing macrobenthic community, the enclosed OWF may be acting as a de-facto marine protected area (MPA) from fishery impacts. At this point in time, the enclosed habitat is not necessarily the most valuable area in the BPNS in terms of conservation (Inger et al., 2009), as large-scale effects of the physical presence of the OWF (e.g. changing hydrodynamics and increased food availability) are still unknown. Our results suggest that the area could evolve into a rich and sheltered habitat providing a refugium for foraging and breeding organisms from higher trophic levels (de Juan et al., 2007; Rabaut et al., 2010). However, in accordance with other studies there is still a major knowledge gap as to how the area will evolve in the long term (> 20 years), underlining the need for further long term research at different spatial scales and trophic levels. With a total area of 238 km² which will prohibit beam trawl fishery activities after construction of all Belgian OWFs, an unknown large-scale situation for the Southern North Sea will be created.

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CHAPTER 4

Enrichment and shifts in macrobenthic assemblages in an offshore wind farm area in the Belgian part of the North Sea

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Abstract

The growing development of offshore wind energy installations across the North Sea is producing new hard anthropogenic structures in the natural soft sediments, causing changes to the surrounding macrobenthos. The extent of modification in permeable sediments around a gravity based wind turbine in the Belgian part of the North Sea was investigated in the period 2011 - 2012, along four gradients (south-west, north-east, south-east, north-west). Sediment grain size significantly reduced from 427 µm at 200 m to 312 ± 3 µm at 15 m from the foundation along the south-west and north-west gradients. The organic matter content increased from 0.4 ± 0.01 % at 100 m to 2.5 ± 0.9 % at 15 m from the foundation. The observed changes in environmental characteristics triggered an increase in the macrobenthic density from 1390 ± 129 ind m^{-2} at 200 m to 18583 ± 6713 ind m^{-2} at 15 m together with an enhanced diversity from 10 ± 2 at 200 m to 30 ± 5 species per sample at 15 m. Shifts in species dominance were also detected with a greater dominance of the ecosystem-engineer *Lanice conchilega* (16 - 25 %) close to the foundation. This study suggests a viable prediction of the effects offshore wind farms could create to the naturally occurring macrobenthos on a large-scale.

Keywords

Benthic ecology - Community composition - Permeable sediments - Environmental impact - Small scale - Wind power - North Sea

1. Introduction

Anthropogenic structures have become a widespread phenomenon in the marine environment with the presence of many shipwrecks, coastal defence structures, oil and gas platforms and renewable energy devices such as offshore wind farms (OWFs) (Moffat et al., 2010; OSPARcommission, 2009; Zintzen et al., 2008). These hard substrates create a new habitat within the natural soft sediments influencing the surrounding benthic biodiversity (Ambrose and Anderson, 1990; Davis et al., 1982; Langhamer, 2010; Petersen and Malm, 2006). Worldwide, over 70 % of the continental shelves and coastal waters are covered by sandy, permeable sediments (Boudreau et al., 2001; Janssen et al., 2005; Meysman et al., 2007). Highly permeable sediments (permeability > 10^{-12} m²) are characterised by a low organic matter content due to high mineralisation rates caused by advective pore-water flows (Rusch et al., 2006). Typical macrobenthic assemblages of permeable sediments are characterised by a low density and diversity, dominated by a few species such as mobile polychaetes (e.g. *Nephtys cirrosa*) and crustaceans (*Bathyporeia* sp. and *Urothoe brevicornis*) (De Maersschalck et al., 2006; Van Hoey et al., 2004). The macrobenthos plays an essential role in the marine ecosystem functioning by degrading organic matter and transferring energy to higher levels in the marine food web (Hiscock et al., 2006; Newell et al., 1998), acting as a food source for demersal fish species and crustaceans (Schuckel et al., 2011; Sell and Kroncke, 2013). Changes to the abundance of one or more macrobenthic species can cause existing macrofaunal communities to shift to another state (Hiscock et al., 2002), consequently affecting the entire food web by altering trophic interactions.

Macrobenthic assemblages within the soft substrate are highly dependent on a wide variety of environmental conditions such as sediment grain size, hydrodynamics, food availability and trophic interactions (Martin et al., 2005; Snelgrove and Butman, 1994; Van Hoey et al., 2004; Vanaverbeke et al., 2011). With an increasing demand for offshore renewable energy production in Europe (Corbetta et al., 2013), the installation of OWFs in the natural, permeable soft sediments will most likely induce changes to the surrounding soft-bottom environment (Gill, 2005; Petersen and Malm, 2006). The morphology of the natural seabed will inevitably be altered due to pre-construction activities of OWFs depending on the selected foundation type. Seabed preparation activities are performed prior to the installation of gravity foundations and include dredging and drilling, which causes re-suspension of sediments and physical removal of the natural soft substrate and its associated macrobenthic assemblage (Hiscock et al., 2002; Peire et al., 2009; Petersen and Malm, 2006). Various studies have observed changes to the sedimentological characteristics due to modified local current flows and their creation of sheltered areas directly around artificial foundations (Airoldi et al., 2005; Leonhard and Pedersen, 2005; Schröder et al., 2006). Consequently, the native macrobenthos could shift to species which are more adapted to the changed sediments. The local biodiversity around

OWFs increases due to the colonisation of the foundations by epifaunal hard-substrate related species such as cnidarians and mussels (Kerckhof et al., 2009; Krone et al., 2013; Wilhelmsson and Malm, 2008). The hard-substrate epifauna enrich the surrounding sediments through the depositional flow of faecal pellets and detritus, increasing food availability in the area (Maar et al., 2009; McKindsey et al., 2011; Ysebaert et al., 2009). This process can lead to an enrichment of the macrobenthic biomass and abundance in the soft-substrate around foundations (Coates et al., 2011; Maar et al., 2009). Changes to the natural sediment morphology around OWFs, due to pre-construction works or changing hydrodynamics, could decrease the permeability of sediments. This process will facilitate the retention of deposited organic matter (Janssen et al., 2005), increasing food availability to the macrobenthos even further. Other mechanisms influencing sediment permeability include the transport of organic particles (e.g. detritus and faecal pellets) in the sediments (Volkenborn et al., 2007a) by advective pore-water flows (Huettel and Rusch, 2000), resulting in the physical bioclogging of interstitial spaces and further reduction of the sediment permeability (Zetsche et al., 2011).

Previous studies concerning the effects on macrofauna around artificial substrates have detected a wide range of community changes where the intensity mainly depends on the extent of physical modifications and biological factors such as predation (Maar et al., 2009; Martin et al., 2005; OSPARcommission, 2009; Schröder et al., 2006). However, temporal and spatial scale effects are equally important in the assessment of community changes (OSPARcommission, 2009), but largely unknown for OWFs. On a small-scale, Martin et al. (2005) observed an overall increase in the local species diversity around different coastal defence structures in Europe, with a patchy distribution of environmental and biological factors. Small-scale effects (1 m – 1 km) have mainly been detected shortly after the construction of an artificial structure. One year after the installation of a research platform in the German Bight, changes were observed up to a distance of 15 m with higher abundances of mobile predators and a decline of typical soft substrate species (Schröder et al., 2006). Larger-scale changes to the macrobenthic assemblages are closely linked to the regional environmental processes (sediment characteristics and currents) of the area (Martin et al., 2005). Coastal defence structures, which were closely located to each other, showed differences in species composition and trophic structure both in Spain and the UK (Martin et al., 2005). On a regional scale, artificial structures can also act as stepping stones for the dispersal of the larval stages of non-indigenous species (ICES, 2012; OSPARcommission, 2009). The community could be affected by changes in competition or predation between species, consequently modifying trophic links in the food web. This could be of concern in the long-term as many OWFs are being constructed in the entire North Sea (ICES, 2012).

In the Belgian part of the North Sea (BPNS) the first OWF was constructed in 2008 with six gravity based wind turbine foundations (GBF) (Brabant et al., 2012). A rapid colonisation of the hard substrate by epifaunal organisms was observed with the creation of an intertidal mussel-barnacle belt only two years after construction (Kerckhof et al., 2009). A multi-species community established on the subtidal part of the foundation with two tube-building amphipods as some of the most abundant species (Kerckhof et al., 2010). Four years after construction, a total of 84 epifaunal species had already been observed (De Mesel et al., 2013). Around the GBF, an attraction of pouting (*Trisopterus luscus*) and Atlantic cod (*Gadus morhua*) was also observed (Reubens et al., 2013). Increased densities of foraging fish and crustaceans can increase predatory pressure to the surrounding macrofaunal assemblages (Ambrose and Anderson, 1990; Schröder et al., 2006). Maar et al. (2009) observed a decreased macrofaunal biomass at a local scale around a turbine foundation in Denmark and related this to the enhanced abundance and therefore predatory pressure of shore crabs. The soft-substrate macrofauna was widely studied on a large-scale before and after the GBFs were constructed with baseline samples collected in 2005 and from 2008 till present (Coates and Vincx, 2010; De Maersschalck et al., 2006; Reubens et al., 2009). Samples were not only collected on the impacted sandbank but also on a neighbouring sandbank as a reference. The community showed resilience to disturbances linked to (pre-) construction activities with a fast recovery potential and no operational effects on a large-scale four years after the construction of six GBFs (Coates et al., submitted).

Up to now, information concerning the small-scale effects (< 200 m) of the presence of GBFs to the surrounding permeable sediments and macrofauna was lacking. With an increasing demand for OWFs in the North Sea it is important to determine if and how the macrobenthic assemblage is changing at different spatial scales. In this study, the macrofaunal community was sampled in close vicinity to one GBF, three and four years after construction. The main hypothesis can be divided in three sections: 1) the sedimentary characteristics of permeable sediments change in close vicinity to a GBF 2) the organic matter content in the surrounding seabed increases and 3) the macrobenthic community composition of the surrounding seabed changes. Along four transects, the soft substrates were sampled with a Van Veen grab from a distance of 15 m to 200 m away from one GBF. To determine any effects on the macrobenthos, density, diversity and biomass were measured around the foundation during a period of two years.

2. Material and methods

2.1. Study site

The first offshore wind farm in the BPNS was installed on the Thorntonbank in 2008. As part of the Zeeland Ridges, the Thorntonbank is located in the eastern part of the BPNS, almost parallel to the coastline and 27 km offshore (Fig. 1). The main tidal flow is oriented on a south-west – north-east axis with a tidal range of 4 - 5 m during spring tide (Van den Eynde et al., 2010). The natural soft sediment in the area consists of medium sands with a grain size between 250 and 500 µm (De Maersschalck et al., 2006). Sampling was carried out around one of six gravity based wind turbine foundations (D5, coordinates WGS 84: 51° 32.88'N – 2°55.77'E). The foundations are located 500 m apart from each other at a depth of 22.5 m at mean low water during spring tide (MLWS). The foundations have a base diameter of 23.5 m on the seabed with a surrounding scour protection system (boulders) consisting of a filter layer (crushed gravel) with a diameter of 55.5 m and an upper armour layer (quarried rock) with a diameter of 51 m (Peire et al., 2009) (Fig. 2). All foundations are connected by power cables, arriving and leaving the foundations at the north-east side.

2.2. Sample collection and treatment

2.2.1. Biological data

Over a period of two years (2011 - 2012) macrobenthic samples were collected during late spring (30th May 2011 and 11th June 2012), along four transects (south-west, north-east, south-east and north-west) around the gravity based turbine (Fig. 1). Along every transect, the macrobenthos was collected with a Van Veen grab (surface area 0.025 m²) from a small survey vessel (GEO.xyz bvba) at 15, 25, 50, 100 and 200 m distance, starting from the edge of the scour protection boulders. Three replicates were obtained at every location (Table 1). Real-time positioning was ensured by an on-board surveyor. Replication was limited on the south-east gradient in 2012 due to deteriorating weather conditions during sampling. The presence of high power cables on the seabed restricted sampling on the north-east gradient to 100 and 200 m in both years.

All samples were sieved on-board over a 1 mm sieve table and fixed in an 8 % formaldehyde-seawater solution. In the laboratory, samples were stained with 1 % Rose Bengal and rinsed over a 1 mm sieve. After sorting, organisms were identified to species level, whenever possible, and subsequently stored in a 4 % neutralised formaldehyde solution.

Biomass or ash free dry weight (AFDW) was determined for every species per sample by a conversion factor of the wet weight (Brey, 2001) or by regression factors of the length or width with the AFDW. When neither conversion factors nor regressions existed for a certain species, AFDW was determined by cremation. Samples were dried for 48 h at 60 °C and burned in a muffle furnace for 2 h at 550 °C. The AFDW was calculated as the difference between the dry weight (DW) and ash weight (AW).

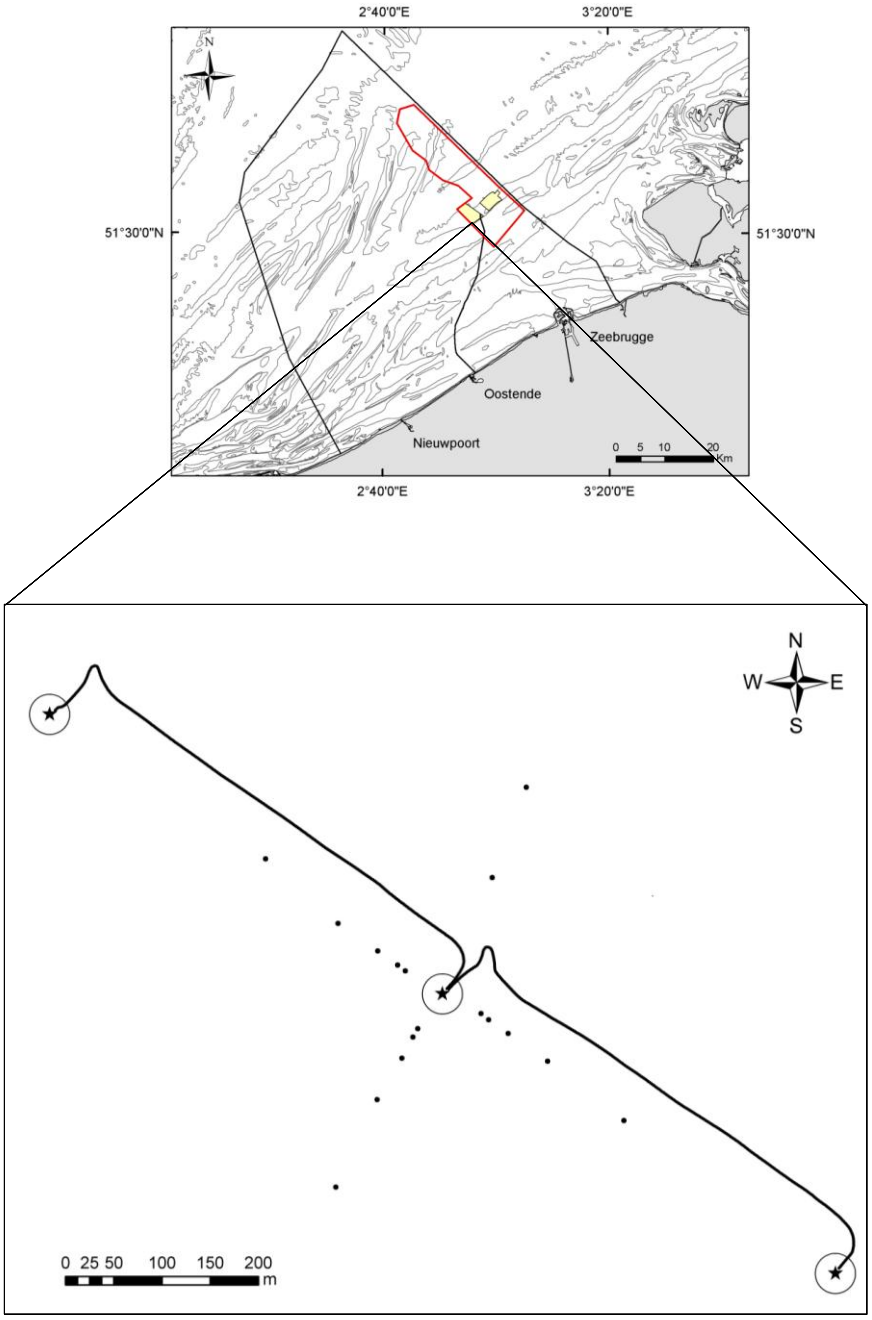

Fig. 1 Upper map: Map of the Belgian part of the North Sea with the designated wind farm concession area (red area). Map below: The gravity based foundations (D4 – D6) are represented by stars and the surrounding scour protection boulders by circles. The black dots represent the sampling positions around the D5 gravity based foundation in four directions (south-west, north-west, south-east and north-east) and five distances (15 m – 200 m). Power cables arrive and leave at the north-east side, represented by the black line. The foundations are located on the Thorntonbank, 27 km offshore

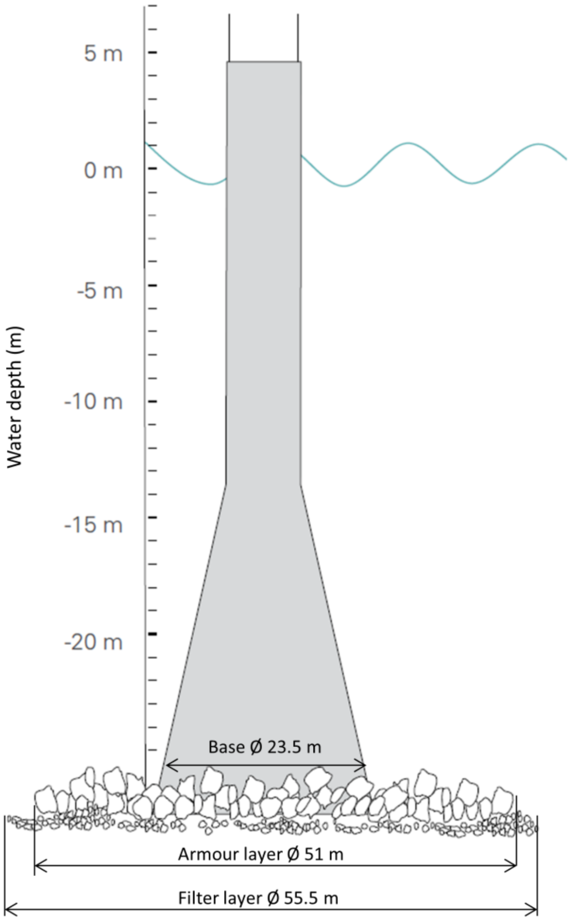

Fig. 2 Schematic representation of the gravity based foundation constructed on the Thorntonbank with a base diameter of 23.5m, armour layer of 51 m and filter layer of 55.5 m. Adapted from (Rumes et al., 2013)

2.2.2.Environmental data

Data for water temperature (°C), significant wave height (m) and tides (TAW, Tweede Algemene Waterpassing) were recorded from the Weshinder measuring pylon (MP7) and a measurement buoy at Ostend (Monitoring Network Flemish Banks). Sediment samples for grain size analysis were taken as a subsample (50 ml) from the Van Veen grab. Median grain size d (0.5) was determined on dried samples (60 °C) with a Malvern Mastersizer 2000G, hydro version 5.40 (laser diffraction method) (Malvern, 1999). Grain size fractions were determined as volume percentages with a range from fine clay (< 4 µm) to coarse gravel/shell material (> 1600 µm). The total organic matter content (TOM %) was measured per sample by applying following equation: TOM % = [(DW – AW) / (DW - CrW)] x 100. DW was determined after 48 h at 60 °C and the AW after 2 h 20 min at 550 °C. Every used crucible was weighed (CrW) in order to determine TOM %.

Table 1. Number of replicated macrobenthic samples around the gravity based foundation on the Thorntonbank in the spring of 2011 and 2012. Four gradients sampled: south-west (SW), north-west (NW), south-east (SE) and north-east (NE) along five distances (15 m – 200 m) from the foundation (NA = Not Applicable)

Distance	**30/05/2011**				**11/06/2012**			
	SW	**NW**	**SE**	**NE**	**SW**	**NW**	**SE**	**NE**
15 m	3	3	3	NA	3	3	1	NA
25 m	3	3	3	NA	3	3	1	NA
50 m	3	3	3	NA	3	3	1	NA
100 m	3	3	3	3	3	2	1	3
200 m	3	3	3	3	3	3	1	3

2.3. Data analysis

Before analysis, the number of individuals per m² (abundance) was determined and a data quality control was carried out; e.g. species that were not sampled quantitatively (hyperbenthos, meiobenthos and fish) were excluded from the dataset together with extremely rare taxa, here defined as species with a unique occurrence and a maximum of two individuals per sample. The dataset was checked for inconsistent species identifications; lumping certain species to genera or family level (e.g. Anthozoa, *Aricidea* sp., *Harmothoe* sp., *Phyllodoce* sp., *Spio* sp. and Cirratulidae). Species richness (N_0) and dominant species were calculated for all samples. A total set of 91 samples and 94 species was obtained.

The Plymouth Routines In Multivariate Ecological Research (PRIMER) programme, version 6.1.6 with PERMANOVA add-on software, was applied for statistical analyses (Anderson et al., 2008; Clarke and Gorley, 2006). A significance level of $p < 0.05$ was used in all tests. Results were expressed as mean ± standard error (SE). Multivariate and univariate permutational ANOVAs (PERMANOVA) were carried out with a 3-factor design including year (Ye), gradient (Gr) and distance (Di) analysed in this order. With an unequal number of replicates, the design was unbalanced. Therefore, a Type III sum of squares was applied with the number of permutations set to 9999 and the permutation of residuals under a reduced model. When the main test showed significant effects of factors, a pair-wise comparison was applied between levels of factors. When the main test showed significant interactions between factors, a pair-wise comparison was performed within the interaction factor. The resemblance measure Bray-Curtis similarity was used for multivariate analysis of the biotic data (density and biomass) after square-root transformation, visualised by a principal coordinates (PCO) analysis. The homogeneity of dispersions was tested with PERMDISP using distances among centroids. If PERMDISP was significant ($p < 0.05$), data was log (x+1) transformed.

Euclidean distance similarity matrices were applied for environmental data (grain size and TOM %) and the univariate analysis of total density, biomass and diversity. Before analysis of the environmental variables, data was normalised. To determine the correlation between the multivariate resemblance matrix of abundance data (square root transformation, Bray-Curtis similarity) and the resemblance matrix of environmental variables grain size and TOM % (log (x+1) transformation, normalised, Euclidean distance) a step-wise DistLM was applied (with the AIC selection criterion) to establish how much the environmental data determined the variation in multivariate abundance.

3. Results

3.1. Environmental variables

On the 30th May 2011 the average water temperature (measured at Westhinder MP7) ranged between 13.1 – 13.3 °C, the significant wave height fluctuated between 40 and 50 cm. On the 11th June 2012 the average water temperature was measured between 13.5 – 13.6 °C with a significant wave height increasing from 60 cm to 100 cm. High water (4.08 m TAW) was at 10.44h in Ostend on 30th May 2011 and at 17.15h on 11th June 2012 (4.17m TAW).

Table 2: Pair-wise PERMANOVA test for term Gr x Di for pairs of levels of factor distance, within level south-west and north-west of the factor gradient to determine significant differences (p < 0.05) in median grain size

	south-west gradient			**north-west gradient**		
	t	**p-value (perm)**	**Unique perms**	**t**	**p-value (perm)**	**Unique perms**
15 m - 100 m	4.041	0.0061	8691	39.368	0.0065	8828
25 m - 100 m	3.010	0.0221	8815	10.259	0.3203	8813
50 m - 100 m	6.328	0.0026	8739	0.474	0.6522	6847
15 m - 200 m	1.484	0.1742	8720	66.227	0.0018	8769
25 m - 200 m	0.096	0.9167	8716	34.025	0.0146	8848
50 m - 200 m	2.306	0.0524	8805	17.112	0.1381	8914
15 m - 25 m	1.649	0.1682	8791	14.886	0.1538	8761
15 m - 50 m	0.032	0.9726	8783	32.553	0.0141	8882
25 m - 50 m	2.780	0.0162	8860	13.131	0.2304	8869
100 m - 200 m	2.889	0.0246	8784	2.621	0.0346	8835

The mean grain size around the foundation ranged between 312 ± 3 µm at 15 m on the north-west gradient and 427 µm at 15 m on the south-east gradient in 2012 (Fig. 3 A + B). The main PERMANOVA test (3-factor design) showed significant interactions for the terms Ye x Gr (*p* = 0.0109) and Gr x Di (*p* = 0.0001). Pair-wise comparisons for the term Ye x Gr for pairs of levels of the factor year, showed significantly higher mean grain sizes in 2012 in comparison to 2011 ($p = 0.0103$) on the south-east gradient. Pair-wise comparisons for the term Gr x Di for pairs of levels of the factor distance always showed significantly lower mean grain sizes closer to the foundation, in comparison to the furthest stations at 100 m or 200 m on the south-west and north-west gradients (Fig. 3 A). Stations at 15 to 50 m on the south-west gradient all had significantly lower mean grain sizes in comparison to 100 m ranging from 315 ± 21 µm at 15 m to a maximum of 403 ± 17 µm at 100 m in 2011 (Table 2). On the north-west gradient, the mean grain size at 15 m was significantly lower to 100 m and 200 m together with a significant difference between 25 m and 200 m (Fig. 3 A). The mean grain size on the north-west gradient ranged between 311 ± 3 µm at 15 m in 2012 and 412 ± 15 µm at 200 m in 2011. On the south-east gradient, a significantly higher mean grain size was observed at 15 m ($p = 0.004$) in comparison to 50 m with a peak of 427 µm at 15 m in 2012.

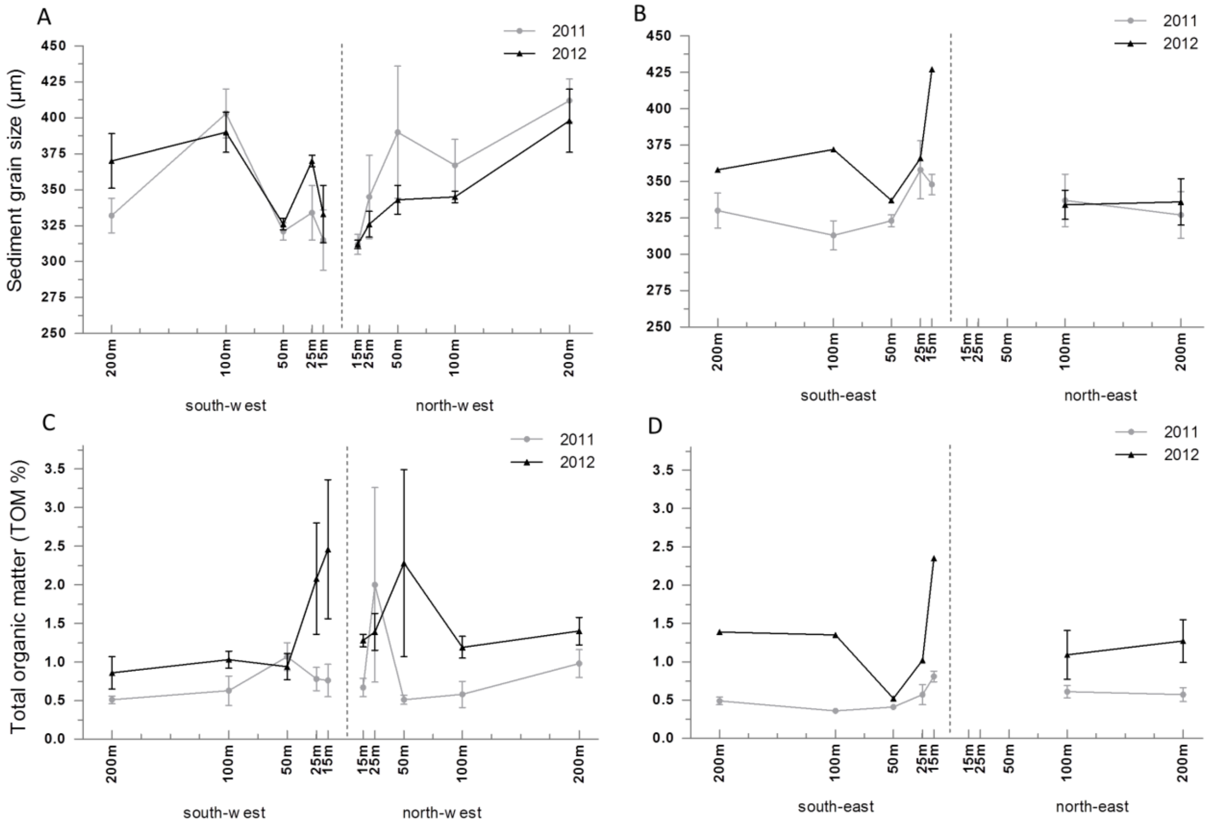

Fig. 3 A-B: Mean sediment grain size (µm) ± standard error and C-D: mean organic matter content (TOM %) ± standard error along four gradients (south-west, north-west, south-east and north-east) and five distances (15 m – 200 m) around a gravity based foundation.

In 2011, the mean total organic matter content (mass %) showed a minimum of 0.36 ± 0.01 % at 100 m on the south-east gradient and a maximum of 2.0 ± 1.3 % at 25 m on the north-west gradient (Fig. 3 C + D). The total organic matter content ranged from 0.52 % at 50 m on the south-east gradient to 2.46 ± 0.9 % at 15 m on the south-west gradient in 2012. The main PERMANOVA test (3-factor design) only showed a significance for the factor year ($p = 0.0008$) with significantly higher organic matter content in 2012 compared to 2011.

3.2. Macrobenthic data

3.2.1. Macrobenthic density, diversity and biomass

The highest mean macrobenthic density (62227 ± 54445 ind m^{-2}) was measured in 2011 at 25 m from the foundation on the south-west gradient due to the dominance of *Asterias rubens* juv. (19 - 52 %). Disregarding *A. rubens* juv., the highest mean macrobenthic densities in 2011 and 2012 were both measured on the south-west gradient at 15 m distance from the foundation with respectively 9339 ± 3073 ind m^{-2} and 18583 ± 6713 ind m^{-2} (Fig. 4 A). The main PERMANOVA test demonstrated significant interactions for the terms Ye x Gr ($p = 0.043$) and Gr x Di ($p = 0.001$) for densities. Pair-wise comparisons for the interaction term Ye x Gr for pairs of levels of the factor year, only showed significantly lower mean densities in 2012 compared to 2011 at the south-east gradient ($p = 0.0202$) (Fig. 4 B). PERMDISP showed a significant effect ($p = 0.0004$) for the term Ye x Gr, suggesting a high dispersion between samples. As for pair-wise comparisons of the interaction term Gr x Di for pairs of levels of the factor distance, significant results were measured on the north-west, south-west and south-east gradients. On the north-west gradient, significantly higher mean macrobenthic densities were measured at 15 m in comparison to 50 m ($p = 0.0075$) and 200 m ($p = 0.032$) (Fig. 4 A). On the south-west gradient, significantly higher mean densities were measured in samples taken closer to the foundation (15, 25 and 50 m) in comparison to both 100 m and 200 m (Table 3). On the south-east gradient, significantly lower abundances were measured at 200 m in comparison to 50 m ($p = 0.0095$) and 100 m ($p = 0.0018$) distance (Fig. 4 B).

The mean number of species (N_0) in samples from 15 m to 200 m ranged between 9.7 ± 2.3 species per sample at 200 m on the north-west gradient in 2011 and 30.3 ± 4.9 species per sample at 15 m on the south-west gradient in 2012 (Fig. 4 C + D). Significant interactions were observed for the term Gr x Di ($p = 0.0007$) from the main PERMANOVA test. Pair-wise comparisons for pairs of levels of the factor distance, revealed significant differences in distance on the north-west and south-west gradients. On the north-west gradient, a significantly higher number of species per sample was measured at 15 m in comparison to 50 m ($p = 0.0218$). However, a significantly lower species richness was measured at 50 m compared to 100 m ($p = 0.0352$). On the south-west gradient, a significantly higher number of species was measured at 15, 25 and 50 m in comparison to 100 and 200 m (Table 3). No significant differences were observed on the north-east and south-east gradients.

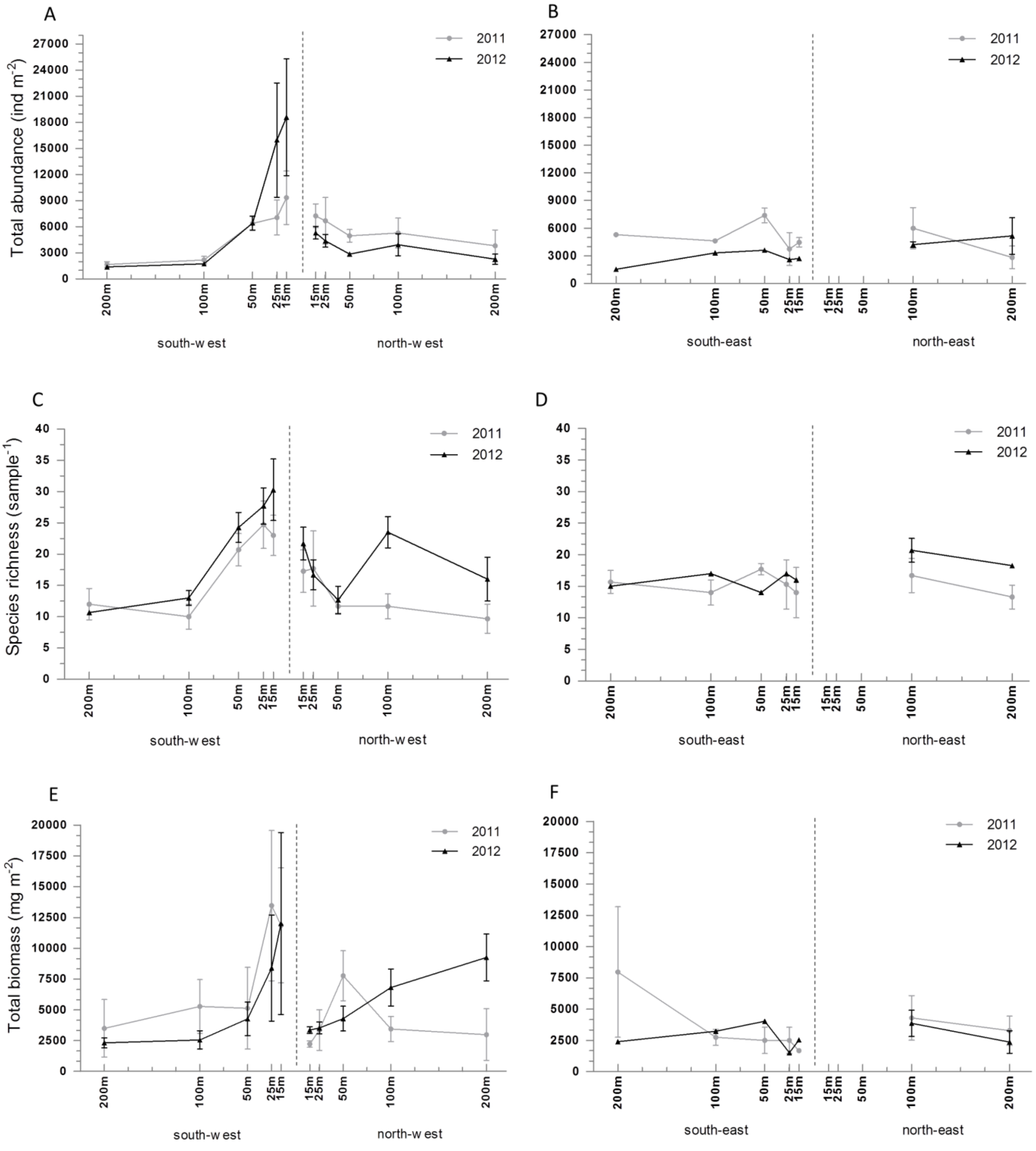

Fig. 4 A - B: Mean total abundance (ind m^{-2}) ± standard error, C - D: mean number of species (sample $^{-1}$) ± standard error and E - F: mean biomass (mg m^{-2}) ± standard error along four gradients (south-west, north-west, south-east and north-east) and five distances (15 m – 200 m) around a gravity based foundation.

Table 3. Pair-wise PERMANOVA test for term Gr x Di for pairs of levels of factor distance, within level south-west of factor gradient to determine significant differences (p < 0.05) in average abundance and number of species

	Average abundance			**Species richness**		
	t	**p-value (perm)**	**Unique perms**	**t**	**p-value (perm)**	**Unique perms**
15 m - 100 m	6.7366	0.0027	8820	4.8095	0.0023	5782
25 m - 100 m	4.8598	0.0033	8773	5.4893	0.0025	3204
50 m - 100 m	8.2993	0.0025	8667	5.2015	0.0016	4287
15 m - 200 m	7.7533	0.0021	8713	4.7958	0.003	7002
25 m - 200 m	5.6638	0.0022	8745	5.4467	0.0031	7479
50 m - 200 m	10.655	0.0027	8617	5.1236	0.0025	7463
15 m - 25 m	0.48572	0.6344	8863	0.13169	0.8998	8121
15 m - 50 m	2.0238	0.077	7926	1.2155	0.2647	8334
25 m - 50 m	1.4672	0.1813	7951	1.226	0.2494	6581
100 m - 200 m	1.4658	0.1866	5382	9.71E-02	0.9284	3852

The species *Echinocardium cordatum* was excluded from the analysis due to extreme outliers caused by a few large specimens. Without this species, the average total biomass between 15 and 200 m ranged from 1504 mg m^{-2} at 25 m on the south-east gradient in 2012 to 13461 ± 6101 mg m^{-2} at 25 m on the south-west gradient in 2011 (Fig. 4 E + F). In both years, the average total biomass on the south-west gradient was again highest in close vicinity to the turbine and decreased with increasing distance from the scour protection system. The average total biomass on the south-west gradient in 2011 ranged from 3450 ± 2343 mg m^{-2} at 200 m to 13461 ± 6101 mg m^{-2} at 25 m and from 2316 ± 407 mg m^{-2} at 200m to 12009 ± 7385 mg m^{-2} at 15 m in 2012. However, no significant interactions in biomass were measured for factors Ye, Gr and Di or any of the interaction terms (PERMANOVA). The main species, contributing to the high biomass values at 25 m on the south-west gradient in 2011, were *A. rubens juv.*, *A. rubens*, *Ophiura ophiura* and *Nephtys cirrosa*. In 2012, *A. rubens*, *Lanice conchilega* and *Spiophanes bombyx* contributed to the high biomass values at 15 m distance on the south-west gradient.

3.2.2 Community analysis and dominant species

A total of 94 species were identified with 4 major taxonomic groups: 39 species of Malacostraca, 31 polychaete species, 12 mollusc species and 7 echinoderm species. Analysing macrobenthic community structure based on total densities, significant interactions were found for the term Gr x Di (*p* = 0.0001) in the main PERMANOVA test. Pair-wise tests for the term Gr x Di for pairs of levels of the factor distance, showed a significant difference in community structure on the south-east gradient between 50 m and 200 m (*p* = 0.0114). On the north-west and south-west gradients, significant differences were mainly detected between samples taken closer to the foundation (15 m - 50 m) and samples collected further away at 100 m and 200 m (Table 4).

Table 4. Multivariate abundance analysis, with the pair-wise PERMANOVA test for term Gr x Di for pairs of levels of factor distance (15 m – 200 m) for the south-west and north-west gradients

	south-west gradient			**north-west gradient**		
	t	**p-value (perm)**	**Unique perms**	**t**	**p-value (perm)**	**Unique perms**
15 m - 100 m	3.0622	0.0025	8897	2.1201	0.0025	8929
25 m - 100 m	2.8367	0.0027	8929	1.3979	0.0824	8911
50 m - 100 m	2.5593	0.0022	8909	1.1416	0.2769	8903
15 m - 200 m	2.8698	0.0022	8917	2.5183	0.0028	8893
25 m - 200 m	2.6384	0.0028	8924	1.8396	0.017	8888
50 m - 200 m	2.273	0.0049	8921	1.2938	0.1209	8876
15 m - 25 m	0.9325	0.4893	8917	1.035	0.3766	8858
15 m - 50 m	1.6755	0.0337	8916	1.6325	0.0269	8942
25 m - 50 m	1.3803	0.1118	8857	0.96857	0.4647	8859
100 m - 200 m	1.2469	0.176	8892	1.3099	0.1469	8926

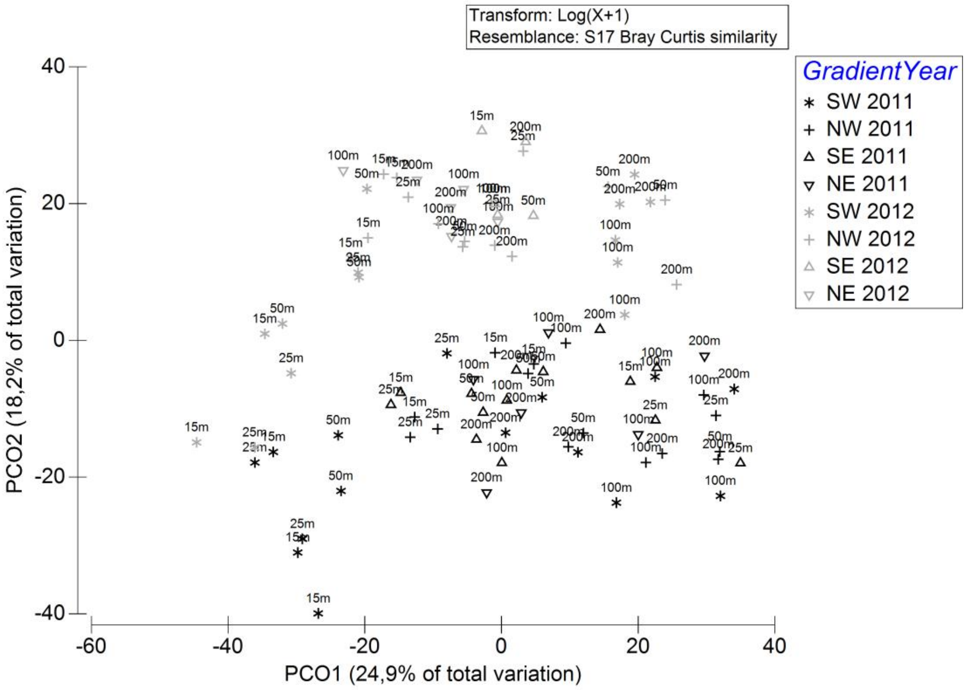

Fig. 5 PCO plot (Principal Coordinates analysis) based on macrobenthic densities around a gravity based foundation on four gradients: south-west (SW), north-west (NW), south-east (SE) and north-east (NE) along five different distances 15 m, 25 m, 50 m, 100 m and 200 m

No significant difference in community structure with distance was observed on the north-east gradient. A PCO plot illustrates the difference in community structure with samples taken from 15 m to 50 m on the south-west gradient, clustered to the left of the plot in both years (Fig. 5). The community structure based on biomass again showed a significant interaction for the term Gr x Di (*p* = 0.0001). Pair-wise test for pairs of levels of the factor distance showed similar results to the abundance analysis with a significant difference in community structure at stations sampled close to the turbine (15 m - 50 m) with samples taken at 100 m and 200 m on the north-west and south-west gradients (Table 5**).** No significant difference in community structure based on biomass was observed on the north-east and south-east gradients. PERMDISP showed a slight yet significant effect for the term Gr x Di (*p* = 0.0493).

Table 5. Multivariate biomass analysis, with the pair-wise PERMANOVA test for term Gr x Di for pairs of levels of factor distance (15 m – 200 m) for the south-west and north-west gradients

	south-west gradient			**north-west gradient**		
	t	**p-value (perm)**	**Unique perms**	**t**	**p-value (perm)**	**Unique perms**
15 m - 100 m	2.9839	0.0022	8861	1.6422	0.0151	8907
25 m - 100 m	2.5183	0.0029	8872	1.2391	0.1597	8938
50 m - 100 m	2.1991	0.0069	8844	0.97485	0.522	8899
15 m - 200 m	3.115	0.0019	8863	2.0417	0.0032	8882
25 m - 200 m	2.577	0.0035	8916	1.6562	0.0159	8923
50 m - 200 m	2.1883	0.0058	8867	1.1995	0.184	8867
15 m - 25 m	0.8348	0.6616	8904	0.94249	0.5531	8905
15 m - 50 m	1.8419	0.0168	8860	1.6099	0.0221	8948
25 m - 50 m	1.2452	0.1819	8866	1.2685	0.1428	8920
100 m - 200 m	1.048	0.3654	8916	1.1599	0.2511	8973

Results of the step-wise DistLM showed that the environmental predictor variables grain size and total organic matter content explained 15.6 % of the variation in community structure together and 10.8 % (*p* = 0.0001) and 4.7 % (*p* = 0.0002) for grain size and total organic matter content separately. The main dominant species in 2011 on all gradients was *A. rubens* juv. (19 – 52 %) up to a distance of 50 m, with a maximum mean density of 55169 ± 53387 ind m^{-2} at 25 m on the south-west gradient (Table 6). From 100 m onwards, *A. rubens* juv. was replaced by *Spio* sp. as dominant species (21 – 67 %) over all gradients. On the north-west and south-east gradients, *Spio* sp. also dominated from 15 m to 50 m (18 – 63 %). A significant interaction for the term Ye x Gr (*p* = 0.0014) was measured for densities of *Spio* sp. (PERMANOVA). Pair-wise tests for pairs of levels of the factor gradient showed significantly higher densities of *Spio* sp. on the north-west (*p* = 0.0002) and south-east (*p* = 0.0001) gradients in 2011, compared to the south-west.

Table 6. Dominant macrobenthic species from 15 m – 200 m distance in 2011 and 2012 on all four gradients south-west (SW), north-west (NW), south-east (SE) and north-east (NE). Mean contribution (%) to the average abundance per station.

2011	**15 m**		**25 m**		**50 m**		**100 m**		**200 m**	
SW	*Asterias rubens* juv.	43.1	*Asterias rubens* juv.	48.9	*Asterias rubens* juv.	52.3	*Spio* sp.	53.5	*Spio* sp.	40.1
	Nemertea sp.	19.6								
NW	*Spio* sp.	34.8	*Spio* sp.	35.4	*Spio* sp.	63.0	*Spio* sp.	67.4	*Spio* sp.	52.2
	Spiophanes bombyx	20.0	*Asterias rubens* juv.	32.3					Nemertea sp.	14.6
SE	*Asterias rubens* juv.	29.3	*Spio* sp.	39.4	*Spio* sp.	44.2	*Spio* sp.	56.7	*Spio* sp.	48.9
	Spio sp.	17.6	*Asterias rubens* juv.	19.0	*Spiophanes bombyx*	23.7	*Spiophanes bombyx*	15.1	*Spiophanes bombyx*	20.8
NE							*Spio* sp.	54.4	*Spio* sp.	20.7
									Spiophanes bombyx	19.6
									Nephtys cirrosa	16.0
2012										
SW	*Lanice conchilega*	23.5	*Lanice conchilega*	25.0	*Spiophanes bombyx*	19.6	*Spio* sp.	40.3	*Spio* sp.	21.0
	Spiophanes bombyx	21.3	*Spiophanes bombyx*	20.6	*Urothoe brevicornis*	14.7			*Nephtys cirrosa*	18.0
NW	*Spiophanes bombyx*	24.4	*Bathyporeia elegans*	23.2	*Urothoe brevicornis*	38.3	*Spio* sp.	25.5	*Gastrosaccus spinifer*	26.7
	Lanice conchilega	15.8	*Urothoe brevicornis*	21.4						
SE	*Urothoe brevicornis*	43.3	*Ophiura* juv.	15.4	*Spio* sp.	39.3	*Spio* sp.	22.0	*Nephtys* juv.	21.1
			Bathyporeia elegans	13.8			*Spiophanes bombyx*	19.5	*Spiophanes bombyx*	18.4
NE							*Spio* sp.	15.8	*Magelona* juv.	28.3

A complete shift in dominant species was observed in 2012, with significantly lower densities of *Spio* sp. on the north-west (p = 0.0001) and south-east (p = 0.0003) gradients. Main dominant species on the south-west gradient shifted to *Lanice conchilega* (24 – 25 %) and *Spiophanes bombyx* (20 – 21 %) up to 50 m. Both species dominated at 15 m on the north-west gradient with 24 % for *S. bombyx* and 16 % for *L. conchilega*. As in 2011, *Spio* sp. was the main dominant species on all gradients at 100 m.

4. Discussion

Our investigations in the immediate vicinity of an offshore gravity based wind turbine show that changes to the sedimentary characteristics (grain size distributions and organic matter) occur directly around the turbine, inducing an important impact on the associated soft-sediment macrofauna.

4.1 Sediment characteristics close to the foundation

The sediments directly around the turbine are classified as medium sands (250 - 500 µm). A finer grain size was observed close to the turbine (15 - 50 m) in comparison to stations positioned further away (100 - 200 m) on the north-west and south-west gradients, with an inter-annual variation in grain size distribution on the south-east gradient. The high spatial and temporal variability observed does not coincide with results found at a large-scale on the Thorntonbank before and after construction (Coates et al., submitted), indicating a direct effect of the presence of the foundation on the seabed characteristics. Two causes are suggested as the main drivers for these observed patterns. First of all, pre-construction activities in this area of the sandbank included dredging of natural sands (Van den Eynde et al., 2010) and replacement by a foundation layer, containing crushed gravel. After construction, a scour protection system, consisting of a filter and upper armour layer, was installed to prevent erosion (Peire et al., 2009). These activities will have affected the sedimentological characteristics of the seabed around the foundations considerably. This can partially explain the gradient of increasing grain size with increasing distance from the foundation. A second reason for the spatial and yearly variability and refinement process is the changing hydrodynamics around the foundation. The main tidal currents on the Thorntonbank are directed on a north-east - south-west axis, creating certain areas of low current speed in the wake of the foundation (Hiscock et al., 2002; Van den Eynde, 2005; Zettler and Pollehne, 2006). As samples close to the foundation were absent on the north-east gradient these statements could only be based on the results observed on the south-west gradient. However, similar results were found in the Danish Horns Rev OWF with a tendency to a lower grain size at 5 m and 25 m, compared to 100 m distance around several monopile foundations, due to a reduction in current speed close to the foundation (Leonhard and Pedersen, 2005). A study carried out around FINO 1, a German offshore research platform, found highly heterogeneous sediments up to 5 m away from the platform. The re-suspension of fine, mobile sands away from the foundation was suggested with the settlement of coarse, dead shells close by due to changes in local current speeds (Hiscock et al., 2002; Schröder et al., 2006). FINO 1 is an open jacket structure with four piles, allowing the main current flow to pass through the construction. However, in our study we investigated a large GBF with a diameter of 23.5 m where decreased current flows will prevent the re-suspension of finer sands along certain gradients close to the turbine.

In our study, a trend to higher organic matter content was observed at 15 m and 25 m away from the foundation on the south-west gradient (Fig. 3 C), possibly due to the flow of the main tidal currents in this direction. In this case, hydrodynamics cannot be the only driving force, as a peak of organic matter was also measured at 15 m on the south-east gradient in 2012. It is generally accepted that the hard-substrate epifauna growing on foundations contribute to the organic matter input on the seabed by sedimentation of faeces and detritus (Barros et al., 2001; Maar et al., 2009; Zettler and Pollehne, 2006), thus modifying the sedimentary habitat and the biodiversity structure of the soft-substrate macrofauna (Coates et al., 2011; Ysebaert et al., 2009). An extensive colonisation of the GBF (discussed in this study) by 84 epifaunal species have been recorded since construction (De Mesel et al., 2013). Together with our results, we suggest that the hard-substrate epifauna is contributing to a higher organic matter input onto the seabed, resulting in changing sedimentological conditions directly around the foundation. Sediments on the Thorntonbank are highly permeable because of their coarse grain size (Vanaverbeke et al., 2011), resulting in a rapid mineralisation of deposited organic matter (Rusch et al., 2006) due to an increased pore-water flow within the sediments (pore-water advection) (Janssen et al., 2005). With a grain size refinement close to the foundation, permeability will diminish, increasing the availability of organic matter within the sediments (Janssen et al., 2005). With a lack of replication on the south-east gradient in 2012 and absent samples close to the foundation on the north-east gradient, significant differences between gradients could not be observed. However, the average percentage of organic matter content was much higher in our study, compared to samples taken at a larger scale on the Thorntonbank before and after construction (De Maersschalck et al., 2006; Reubens et al., 2009). An average maximum of 1.15 % was recorded on a neighbouring sandbank in autumn (Coates et al., submitted) in comparison to a maximum of 2.5 ± 0.9 % at 15 m from the foundation in 2012, which is more than double the amount. It must be taken into consideration that the organic matter input into the sediment varies with the time and magnitude of spring phytoplankton bloom (Franco et al., 2007; Vanaverbeke et al., 2004). However, no great difference in organic matter content was measured between spring (0.52 ± 0.04 %) and autumn (0.60 ± 0.02 %) in samples taken before construction on the Thorntonbank (De Maersschalck et al., 2006). These results illustrate a foundation effect on the total organic matter close to the foundation, regardless of the sampled gradients. This could suggest a higher importance of the increased flow of organic matter to the seabed, possibly overriding the local hydrodynamic effects.

4.2 Macrobenthic response

An enrichment of the soft-sediment macrobenthic assemblage around the foundation was mainly measured on the south-west gradient with a significant increase in abundance and species richness in close vicinity to the foundation. The differences in macrobenthic assemblage, with distance but also gradient, could be explained by changing hydrodynamic processes around the foundation. In the wake of the foundation, tidal flow will decrease (Hiscock et al., 2002), producing sheltered habitats and enhancing larval settlement (Qian, 1999) and recruitment of macrobenthic species mainly along the south-west and north-east gradients. The presence of sand pits (4 – 4.5 m depth) on the south-west gradient, due to dredging works prior to construction (Van den Eynde et al., 2010), could also enhance the entrapment of larvae on this gradient. Extra refuge areas are produced where the current velocities reduce and the concentration of larvae on the seabed can increase. The reduced current flow increases the depositional flux of organic matter onto the seabed from the fouling epifaunal organisms, contributing to the changing surrounding macrobenthic assemblage by increasing food availability (McKindsey et al., 2011; Ysebaert et al., 2009). A significantly enriched abundance and biomass of the benthic macrofauna was observed on the north-east side of a GBF in the Danish Nysted OWF (Maar et al., 2009). The enhanced sedimentation of epifaunal faecal pellets and detritus in this direction, due to the main currents, was seen as the key driving force. However, changing local hydrodynamics was not excluded as an influencing factor. The macrofaunal community around an offshore wave energy converter off the Swedish coast was also richer than at a reference site due to hydrodynamic changes, organic enrichment but also decreased grain sizes (Langhamer, 2010).

In our study, the enrichment of macrofaunal abundance and biomass, in close vicinity to the foundation, was mainly related to the occurrence of the juvenile phase of the hard substrate related species *Asterias rubens* juv. and the tube-building polychaetes *Lanice conchilega* and *Spiophanes bombyx*. Kerckhof et al. (2012) found a strong increase in the relative abundance of *A. rubens* on this foundation in the spring of 2011. However, the dominance of *A. rubens* juv. on the seabed disappeared in 2012. The abundance of juvenile starfish has a high yearly variability due to large annual variations in recruitment intensity which depend on food availability and temperature (Guillou et al., 2012). The exact time of spawning during springtime can vary between April-May with settlement occurring between June-August (Nichols and Barker, 1984). It is possible that sampling was carried out before the settlement of *A. rubens* larvae took place in 2012, or the recruitment could have been lower due to more unfavourable environmental conditions. The latter is most plausible as low densities of *A. rubens* juv. were measured in 2012. This decreased dominance possibly provided *L. conchilega* and *S. bombyx* with the opportunity to expand their dominance both on the south-west (up to 25 m) and the north-west gradients (up to 15 m) in comparison to 2011. The tube-dwelling terebellid polychaet *L. conchilega* often co-dominates with *S. bombyx*, a selective deposit feeder, which is known to be positively associated with *L. conchilega* (Rabaut et al., 2007). *Lanice conchilega* occurs in sediments ranging from mud to coarse sands (Degraer et al., 2006; Hartmann-Schröder, 1996) with a preference for muddy and fine sediments (Van Hoey et al., 2008). Studies have observed an increase in the finer sediment fractions of the seabed around adult *L. conchilega* as it actively builds tubes from coarser particles (Rabaut et al., 2007) and influences local hydrodynamics by decreasing bottom current flows near dense assemblages (Eckman, 1983). As a well-known ecosystem-engineer (Rabaut et al., 2007; Van Hoey et al., 2008), the high abundance of *L. conchilega* close to the foundation has the potential to enhance the already observed changes in hydrodynamic flow, grain size distribution and food availability in our study. Therefore, the habitat complexity and heterogeneity around the foundation will increase even more. This would explain the significantly higher macrobenthic abundances and species richness observed close to the foundation mainly on the south-west and north-west gradients where *L. conchilega* dominated. Various other studies also found a significant and positive correlation between the macrobenthic abundance, diversity and biomass with increasing densities of *L. conchilega* both for intertidal (Callaway, 2006; De Smet et al., 2013; Zuhlke, 2001) and subtidal areas (Rabaut et al., 2007; Van Hoey et al., 2008).

The enrichment of the macrofaunal community around the foundation can serve as an additional food source for higher trophic levels (Schuckel et al., 2011), possibly attracting more demersal fish species inside the OWF. In 2009 and 2010 the diet of dab within the wind farm did strongly differ from the reference area with a higher stomach fullness index, suggesting a higher food availability (Derweduwen et al., 2012). However, observations at a small-scale (< 500 m) were limited to pelagic fish (Reubens et al., 2013) with a main predation restricted to hard-substrate epifaunal species (Reubens et al., 2011). Furthermore, aggregations of *L. conchilega* have been shown to serve as nurseries for plaice (Rabaut et al., 2010). The potential expansion of *L. conchilega* in the long-term and the creation of sheltered areas around the foundations could create an ideal (refuge) habitat for juvenile flatfish species on a small-scale. Together with the exclusion of fisheries inside OWF, these refuge habitats could evolve into important ecological systems on a larger scale in the future.

The macrobenthic communities of the BPNS were arranged into 10 assemblages, reflecting differences in sedimentary characteristics (Van Hoey et al., 2004). Before construction, the macrobenthic community on the Thorntonbank was characterised by a low average species richness (6 ± 0.6 – 13 ± 0.8 species per sample) and abundance (231 ± 49 ind m^{-2} – 472 ± 74 ind m^{-2}) (Coates et al., submitted), inhabiting medium sands containing a low organic matter content (De Maersschalck et al., 2006). The native macrobenthos was classified into the transitional species poor *Nephtys cirrosa* and *Ophelia limacina* – *Glycera lapidum* community (De Maersschalck et al., 2006; Van Hoey et al., 2004). In the present study, a high average species richness (10 ± 2 - 30 ± 5 species per sample) and abundance (1390 ± 129 ind m^{-2} – 18583 ± 6713 ind m^{-2}) was observed, coinciding with a shift in dominant species. Therefore, it can be concluded that the community has evolved away from the transitional *N. cirrosa* and *O. limacina* – *G. lapidum* community. We suggest that the macrobenthic community is shifting towards a variation of the rich *Abra alba* – *Mysella bidentata* community, normally found in shallow and muddy sands (Van Hoey et al., 2004).

4.3 Advice for justifying effects in the future

Despite the spatial and temporal deviations from an ideal sampling design, our results reveal altered macrofaunal communities in close vicinity to a gravity based foundation. To our knowledge, this is the first small-scale study that has observed a significant enrichment of the soft-substrate macrofaunal assemblages in the vicinity of a GBF and this during two consecutive years. The local effects of OWFs on macrobenthic assemblages should be included in future monitoring programmes around OWFs. Alternative sampling methods such as Scuba Diving techniques could be applied to sample close to foundations and in challenging areas (e.g. north-east gradient in this study). However, it is advisable to focus on a single sampling technique within one study design.

With the construction of an additional 48 turbines within this wind farm and the development of several other OWFs not only in this area but the entire North Sea, the observed results can suggest a viable prediction to an overall change in the macrobenthic communities at a larger scale with unknown ecological responses to existing food web interactions. However, if the observed enrichment of the macrofauna spatially increases, demersal fish species could be attracted to the OWFs creating ecological important refuge areas. Food web analysis is a logical next step in future research, providing a more rigorous and comprehensive understanding of the ecological processes (Fry, 2006) occurring around OWFs. Isotope analysis combined with other techniques such as stomach content analysis, the use of lipid biomarkers and isotope mixing models can be used to acquire knowledge on the structure and the stability of the food webs around OWFs (Rooney et al., 2006; Rooney and McCann, 2012; Thompson et al., 2012). These techniques can provide insight into the temporal and spatial variation of food source availability and into the feeding habits of certain fish species around artificial reefs (Braeckman et al., 2012; Mablouké et al., 2013); all of which is indispensable knowledge for future management of these areas. Furthermore, the research should also be expanded to the three foundation types (GBF, monopile and jacket foundation) constructed in the BPNS as different physical and biotic interactions can occur.

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CHAPTER 5

Indirect effects of offshore wind farms on benthic ecosystem functioning of permeable sediments: an experimental pilot study

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Abstract

The seabed of the Belgian offshore wind farm concession zone is characterised by coarse, permeable sediments. Sediments with a high permeability are well oxygenated which accelerates the mineralisation of deposited organic matter after phytoplankton blooms. Research around an offshore wind turbine in the Belgian part of the North Sea has revealed an enrichment of the macrobenthic community, together with sediment refinement up to a 50 m distance from the foundation. With an increasing development of offshore wind farms in the North Sea, there is a need to understand how the altered sediment characteristics could affect the functioning of these sediments. In this pilot study, two experimental chamber experiments were set-up to assess the filtration capacity and fluxes across the sediment-water interface of three sediment types with decreasing permeability (coarse, intermediate and fine) before and after the addition of diatom (*Skeletonema costatum*) cells. A stirring disk was installed to create advective pore-water flows in the permeable sediments. After sediment refinement, water penetration depth (coloured with Rhodamine WT dye) decreased from 6.5 ± 0.2 cm in the coarse sediment to 0.8 ± 0 cm in the fine sediment. A similar trend was observed after the addition of the chain-forming *S. costatum* cells in the second experiment. After an incubation of 23 h, the uptake of diatom cells into the sediment reduced with decreasing permeability. The advective pore-water flow in the coarse sediments facilitated the mineralisation of added organic matter (diatom cells) as reflected by the high Sediment Community Oxygen Consumption (SCOC) rates (23.7 ± 1.9 mmol O_2 m^{-2} d^{-1}). Lowest SCOC rates were measured in the fine sediments 4.8 ± 0.1 mmol O_2 m^{-2} d^{-1} where solute exchange mainly relies on molecular diffusion. Low mineralisation of organic matter was also observed in the refined sediments, with a reduction in the NH_4^+ (ammonium) flux (from 834 ± 100 µmol m^{-2} d^{-1} to 228 ± 782 µmol m^{-2} d^{-1}) and NO_x (nitrate + nitrite) flux (from 239 ± 156 µmol m^{-2} d^{-1} to 96 ± 31 µmol m^{-2} d^{-1}) into the water column. We could therefore conclude that the refinement of permeable sediments around offshore wind farms affects the ecosystem functioning and mineralisation process of organic matter.

Keywords

Sediment refinement - Permeability - Ecosystem functioning - Mineralisation - SCOC

1. Introduction

Annual phytoplankton blooms, which mainly consist of diatoms (Reid et al., 1990; Rousseau et al., 2002) and the haptophyte *Phaeocystis* (Lancelot et al., 2005), are a major source of organic matter in North Sea sediments during spring. Approximately 20 % of the primary production (organic matter) has been observed to reach the seabed after spring phytoplankton bloom (Lancelot et al., 2005; Provoost et al., 2013). The settled particulate organic matter (phytodetritus) is buried into the sediments where it is mineralised to inorganic nutrients by the microbial community, fuelling the phytoplankton with the necessary nutrients (Blackburn, 1988; Middelburg et al., 2004). Oxygen plays a crucial role in the benthic degradation process of organic matter (Glud, 2008). For example, ammonium (NH_4^+) oxidises into nitrate (NO_3^-) using oxygen as electron acceptor (Billen and Lancelot, 1988). Sediment characteristics highly affect oxygen concentrations together with the efficiency of sediment-water exchange rates of nutrients. In fine grained (muddy), cohesive sediments, oxygen concentrations sharply decrease with depth (Vanaverbeke et al., 2011), limiting the mineralisation of organic matter to the sediment surface (< 0.5 cm) unless sediments are actively mixed by organisms (bioturbation), or where bio-irrigation provides the deeper sediment layers with oxygen (Braeckman et al., 2010). Molecular diffusion between the sediments and overlying water is the key particle transport mechanism in fine sediments (Huettel and Gust, 1992). In coarse, permeable (> 10^{-12} m²) sediments, pressure driven advective pore-water flows at the sediment-water interface transport organic matter into deeper layers of the sediments (Ehrenhauss and Huettel, 2004; Huettel and Rusch, 2000; Precht and Huettel, 2003). The pore-water transport is initiated when bottom currents interact with the seabed topography (Huettel and Rusch, 2000; Precht and Huettel, 2003). Furthermore, these permeable sediments are well oxygenated (Ziebis et al., 1996), accelerating the mineralisation of organic matter and the recycling of nutrients (Ehrenhauss et al., 2004a; Ehrenhauss et al., 2004b; Huettel and Rusch, 2000).

With an increasing development of offshore constructions, there is a need to understand how the biogeochemical properties of the natural soft sediments surrounding these constructions could change. The Belgian offshore wind farm concession zone is characterised by coarse sediments which cover 70 % of the continental shelves worldwide (Emery, 1968). A small-scale study around one gravity based foundation in the Belgian part of the North Sea (BPNS) investigated the macrofaunal communities along four gradients (south-west, north-east, south-east and north-west) (Coates et al., 2014). A reduction in sediment grain size up to a 50 m distance from the foundation was observed along the south-west and north-west gradients. The permeability of coarse sediments (> 200 µm) in the BPNS have been derived from grain size analysis (Wilson et al., 2008) and exceeded 10^{-10} m² (Vanaverbeke et al., 2011). In close vicinity to the foundation, the lowest permeability calculated was

9.2 10^{-12} m² at the south-west gradient, suggesting a decreasing permeability around gravity based wind turbines in the BPNS (See Addendum III, Fig. 2). Furthermore, an increased chlorophyll *a* concentration was also measured along the south-west gradient in samples taken at one and seven meters from the foundation in 2010 (Addendum III, Fig. 1).

Sediment refinement can greatly influence the sediment-water exchange rates (benthic-pelagic coupling), which in its turn affects the mineralisation rates of organic matter in the sediments (Ehrenhauss et al., 2004b). The smaller interstitial space in finer sediments obstruct the advective pore-water flows, thereby decreasing fluid exchange rates between the sediment and the overlying water column (Ehrenhauss and Huettel, 2004; Huettel and Rusch, 2000; Precht and Huettel, 2003). Enhrenhauss and Huettel (2004) measured a decrease in pore-water exchange rates from 20 l m^{-2} d^{-1} in coarse sediments (500 – 1000 µm) to 0 l m^{-2} d^{-1} in fine sediments (63 – 125 µm) simultaneously decreasing the transport depth of diatoms from 5.5 cm to 0.3 cm. Decreasing oxygen penetration depths creates slower mineralisation rates in refined sediments (Ehrenhauss and Huettel, 2004; Ehrenhauss et al., 2004b).

The aim of this study was to investigate whether a decreasing permeability of subtidal sediments of the BPNS, due to the construction of offshore wind turbines, would affect the benthic ecosystem functioning. Natural, permeable sand from an offshore sandbank was mixed with commercial and intertidal sediment to create three artificial sediment types with a decreasing permeability. We hypothesise that 1) the oxygen and nutrient fluxes across the sediment-water interface, 2) the ability to filter diatom cells from the water column and 3) the benthic ecosystem functioning will be affected in refined sediments from the BPNS.

2. Material and methods

2.1. Field sampling and sediment refinement

The experimental set-up of this study required coarse (permeable) and fine sands. Permeable sediments and natural seawater were collected from the offshore station 330 (51°26.0’ N ; 02°48.5’ E, 20 m depth) on the Goote Bank in the Belgian part of the North Sea (BPNS). Sampling was performed with a Van Veen grab (0.1026 m²) from the *RV* Simon Stevin in March 2013.

Prior to the experiment, all sediments were dried at 60 °C for 72 h to remove all metazoan fauna. Sediment grain sizes were measured with the Malvern Mastersizer 2000G, hydro version 5.40 (laser diffraction method) (Malvern, 1999). Station 330 consists of medium sands with a median grain size (d_{50}) of 388 µm and 239 µm for the first decile of the grain size distribution (d_{10}). The sediment permeability for station 330 is estimated at 1.8 x 10^{-11} m², calculated and corrected from the empirical relation according to Hazen $k_H = 1.1019 * 10^3\ m^{-2}\ s * d^2_{10} * \nu$ (k_H = permeability and ν = kinematic viscosity) (Rusch et al., 2001).

The fine sands were collected from an intertidal mudflat (Paulina) located along the Western Scheldt estuary in the Netherlands (51°21.4' N ; 3° 42.85' W) in February 2013. The sediments collected at Paulina had a median grain size of 139 µm, d_{10} of 45 µm and an estimated permeability of 6.5 10^{-13} m². The commercial available sand had a median grain size of 191 µm, d_{10} of 128 µm and an estimated permeability of 5.2 10^{-12} m².

To remove all organic matter, the Paulina sand was burnt for 4 h in a muffle furnace at 550 °C, thoroughly washed with distilled water and subsequently dried at 60 °C (Ehrenhauss and Huettel, 2004). The sand from station 330 was not burnt to ensure bacterial activity throughout the experiments. The permeable sands were artificially mixed with Paulina sediment and commercially available sand to obtain three sediment types from fine to coarse grained sands with different permeability (Table 1).

Table 1. Sediment characteristics of the three sediment treatments. d_{10} = first decile of the grain size distribution, d_{50} = median grain size, k_H = permeability

Treatment	**Mixture**	**d_{10} (µm)**	**d_{50} (µm)**	**k_H (m²)**
Coarse	100 % 330	238.9	387.7	1.8×10^{-11}
Intermediate	50 % commercial sand - 50 % 330	143.5	276.9	6.5×10^{-12}
Fine	80 % Paulina - 20 % 330	57.0	199.4	1.0×10^{-12}

2.2. Experimental set-up

2.2.1.Culturing algae

A strain of the diatom *Skeletonema costatum* was obtained from the Laboratoire d'Ecologie des Systèmes Aquatiques from the Université Libre de Bruxelles (ULB). *Skeletonema costatum* is a chain-forming diatom naturally occurring in the BPNS (Rousseau et al., 2002; Takabayashi et al., 2006). The diatoms were cultivated in sterile natural sea water (31 PSU), enriched with a f/2 culture medium (Guillard, 1975) at 15 °C during 30 – 40 days in a 12 h : 12 h light and dark regime. Diatom counts were applied under an inverted Carl Zeiss microscope (400 x magnification).

2.2.2. Benthic stirring chambers

The experiments were carried out in a climate controlled room set to 15 °C. Three cylindrical cores (19 cm diameter, 28 cm high) were used for each sediment type in both experiments (Fig. 1) with 10 cm of sediment and 5.1 l of natural sea water (31 PSU). In each chamber, the water was stirred by horizontally rotating disks of 17 cm diameter rotating 5 cm above the sediment at 60 rpm. The disk rotation creates an artificial pressure gradient in the water column, generating lowest pressure at the centre and highest at the edge (Ehrenhauss and Huettel, 2004; Huettel and Rusch, 2000).

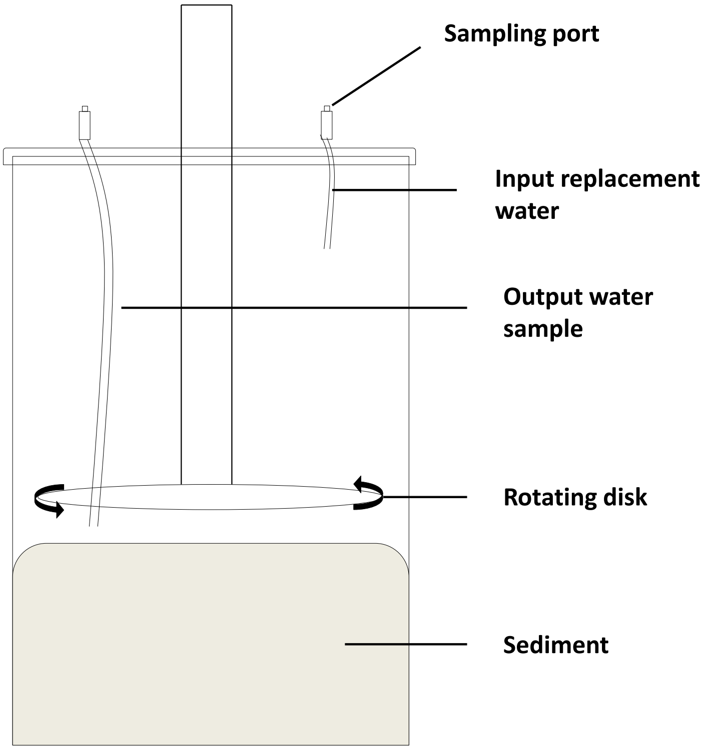

Fig. 1 Experimental set-up, benthic chamber with rotating disk to create advective pore-water flows within the sediment

2.2.3. Filtration capacity measurements

To estimate the water penetration depth in the sediments, a tracer dye Rhodamine-WT (1 %) was added to three benthic stirring chambers for every sediment type. After the addition of 20 ml Rhodamine to the water column, the experiment was run in total darkness for 48 h. The penetration depth of the red dye into the sediment was measured at two positions along the outside of the core after 6, 24 and 48 h. The decrease in Rhodamine dye concentration in the water column was measured by taking a 10 ml water sample at T_0, 2 h (T_1), 4 h (T_2), 6 h (T_3) and 48 h (T_4). Absorbance levels were measured with a spectrophotometer (UV-1601) at 540 nm.

2.2.4. Measurements of sediment-water fluxes

During a second experiment, the oxygen and nutrient fluxes ($NH_4{}^+$ and NO_3^- + NO_2^-) across the sediment-water interface were investigated for the three sediment types. The experiment was first incubated without algae to determine oxygen and nutrient fluxes before organic matter addition (Ehrenhauss et al., 2004b). Each core was hermetically sealed with an air-tight lid, to insure accurate respiration measurements, and incubated in the dark for approximately 23 h. The Sediment Community Oxygen Consumption (SCOC) in the water column was continuously measured in mmol l^{-1} through oxygen sensor spots which were attached to the inside of the core and connected to a fibre-optic oxygen meter (*FireSting* O_2) through a spot adapter and fibre-optic cable (Pyro-Science). The slope of the linear regression of the decreased oxygen concentration in time was calculated in mmol m^{-2} d^{-1}. Water samples (20 ml) for nutrient measurements were taken through the output sampling port at regular time intervals (at the start and every 3 h) throughout the experiment. The sampled water volume was simultaneously replaced with seawater through the input sampling port. Water samples were filtered through 25 mm GF/F filters (Whatman), frozen immediately (-20 °C) and later analysed using automated colorimetric techniques after defrosting. Nutrient fluxes (µmol m^{-2} d^{-1}) were calculated from the difference between the final and initial sample concentration over time. No compensation was made for the dilution due to replacement water as the addition (2 %) was expected to be negligible.

Identical measurements were carried out during a second run of the experiment, after the addition of 20 ml live algae culture. The cell numbers of *S. costatum* added per treatment were 7.6 ± 1.6 cells ml^{-1} (coarse sediment), 14.9 ± 2 cells ml^{-1} (intermediate sediment) and 25.5 ± 2.3 cells ml^{-1} (fine sediment). Apart from the 20 ml water sample for nutrient measurements, an additional 5 ml was obtained for diatom cell counts during the same time intervals. The samples were fixed with 200 µl ethanol and stored for later analysis. Three diatom cell counts were performed on 1 ml of every sample in suspension culture plates (21 cm^2 wells) with an inverted Carl Zeiss microscope (400 x magnification). Cell counts of the remaining diatoms in the water column provided additional information on the filtration capacity of the three sediment types.

2.3. Statistical analysis

R 3.0.2 software (www.r-project.org) was applied for statistical analyses. Homogeneity of variances were tested with the Levene's test. For the filtration capacity experiment, one-way ANOVAs were applied to detect any differences among treatments (sediment types). Tukey HSD-post hoc tests were carried out to determine pair-wise tests between treatments. For the flux experiment, two-way ANOVAs were carried out to determine any interaction effects (st x a) between factors sediment type and the addition of algae. When a significant interaction was measured between factors, a pair-wise Tukey HSD post hoc test was performed within the interaction factor. All results are expressed as average ± SE of the three replicates.

3. Results

3.1. Filtration capacity of the sediments

After a 48 h incubation period, the penetration depth of the Rhodamine dye was significantly different between all three sediment types (ANOVA $F_{2,15} = 266.5$ $p < 0.05$; Tukey HSD test all $p < 0.001$) (Fig. 3). The highest penetration depth was measured in the coarse sediment with an average depth of 6.5 ± 0.2 cm. Depth decreased with grain size with 4.2 ± 0.4 cm in the intermediate and 0.8 ± 0 cm in the finer sediments.

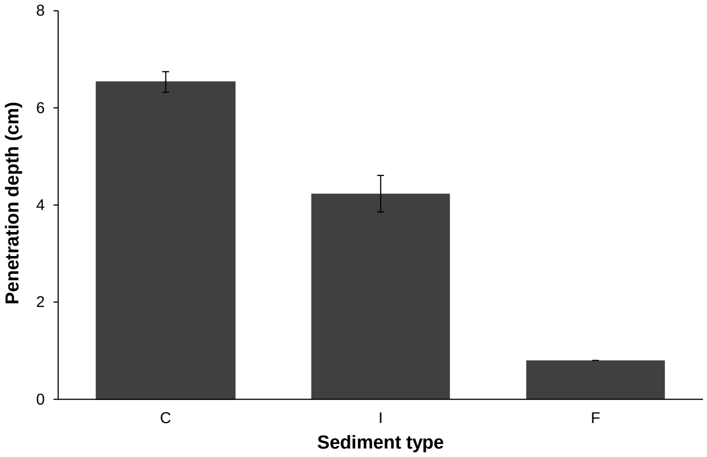

Fig. 3 Penetration depth in coarse (C), intermediate (I) and fine (F) sediments after 48 h incubation period with Rhodamine dye

The concentration of Rhodamine dye decreased over time in the water column in every set-up (Fig. 4). The rate of decreasing dye concentration increased with the sediment permeability. The permeable, coarse sediment illustrated a high filtration capacity with a decrease of the dye concentration in the water column by -0.432 ± 0 mmol m^{-2} d^{-1}. The filtration capacity of the intermediate and fine sediments was lower with respectively -0.216 mmol m^{-2} d^{-1} and -0.288 ± 0.038 mmol m^{-2} d^{-1}. Only one replicate was applied for the intermediate sediment type as unreliable measurements, suggesting the dye concentration increased in the water column, were excluded. No significant difference was measured between the coarse and fine sediment (ANOVA $F_{2,3} = 5.4$ $p = 0.1023$).

After the addition of live diatom cells in the second experiment, a reduction of cells present in the water column was observed throughout the 24 h incubation period (Fig. 5). A reduction from 7.6 ±

1.6 cells ml^{-1} to 0.4 ± 0.4 cells ml^{-1} was measured in the overlying water of the coarse sediment, while the fine sediment incubation hardly showed a decrease (25.5 ± 2.7 cells ml^{-1} to 22.6 ± 1.4 cells ml^{-1}).

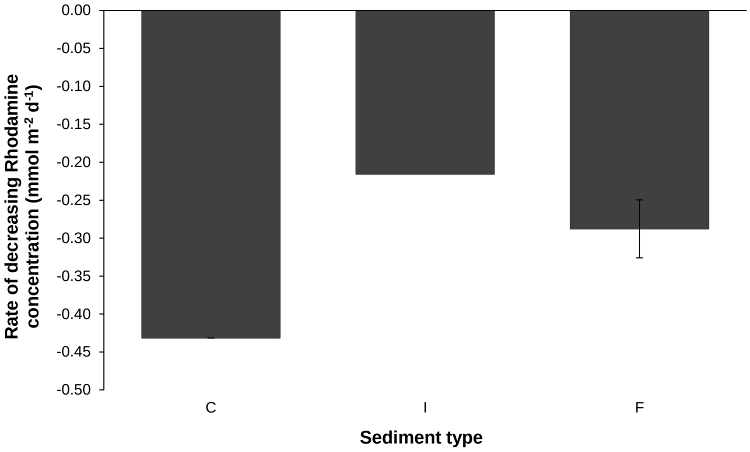

Fig. 4 Rate of decreasing Rhodamine concentration (mmol m^{-2} d^{-1}) in coarse (C), intermediate (I) and fine (F) sediments after 48 h incubation period

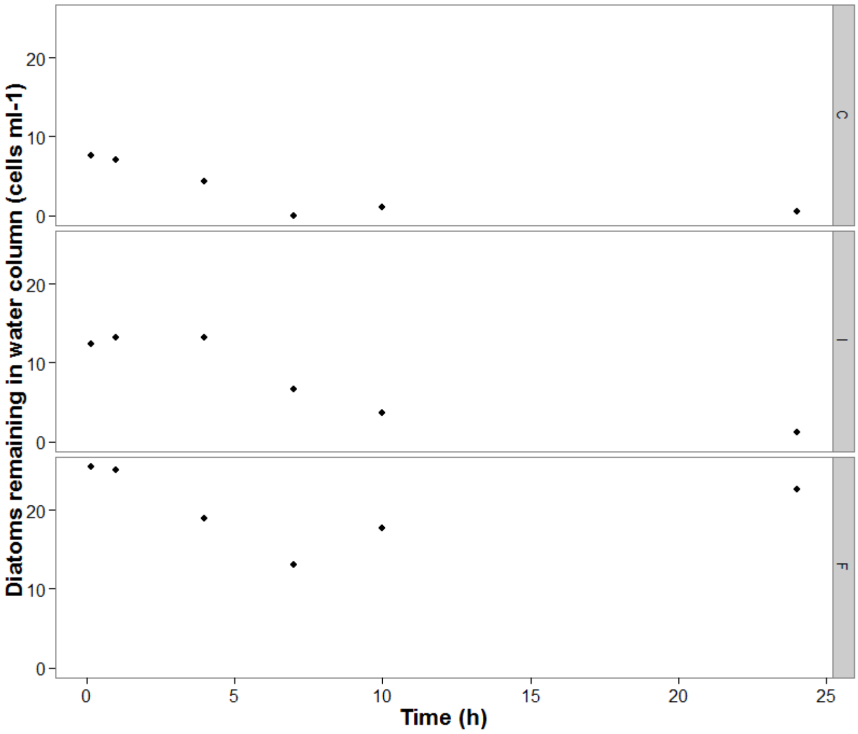

Fig. 5 Diatom cell counts (ml^{-1}) remaining in the water column during 23 h incubation period

3.2. Sediment Community Oxygen Consumption (SCOC) rates

To determine the difference in SCOC rates between the three sediment types, the experiment was run with and without the addition of algae (*S. costatum*) (Fig. 6). Two-way ANOVA illustrated a significant interaction effect between the factors sediment type and addition of algae ($F_{2,11}$ = 18.187, $p < 0.001$). Without algae, the SCOC rate of the intermediate sediment was significantly lower than the coarse (Tukey HSD, $p = 0.002$) and fine (Tukey HSD, $p < 0.001$) sediments. After the addition of algae, the SCOC rate of the coarse sediment was significantly higher in comparison to the fine sediment (Tukey HSD, $p = 0.008$). The fine sediment also illustrated a significant decrease in SCOC rate (Tukey HSD, $p < 0.001$) after the addition of algae. No significant difference in SCOC rates were measured before and after the addition of algae in the coarse and intermediate sediments (Tukey HSD, $p > 0.005$).

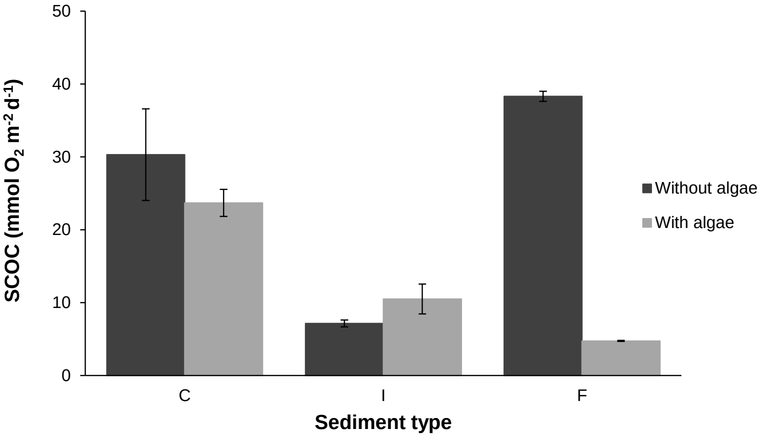

Fig. 6 The sediment community oxygen consumption (SCOC) rates (mmol O_2 m^{-2} d^{-1}) in the coarse (C), intermediate (I) and fine (F) sediments with and without the addition of algae (*S. costatum*)

3.3. Dissolved inorganic nitrogen (DIN) fluxes

Before the addition of algae, the sediments acted as an NH_4^+ sink with the strongest NH_4^+ influx measured in the fine sediments (-895 ± 222 µmol m-2 d-1) (Fig. 7A). After the addition of algae, the NH_4^+ flux changed and sediments released NH_4^+ into the water column (Fig. 8A). A decreasing NH_4^+ efflux was observed with decreasing grain size. The efflux of NH_4^+ from the coarse sediments was 5.5 and 2.6 times the efflux measured from the intermediate and fine sediments. Two-way ANOVA measured a significant difference in NH_4^+ flux between the experiment carried out with and without the addition of algae ($F_{1,12}$ = 13.834, $p = 0.003$).

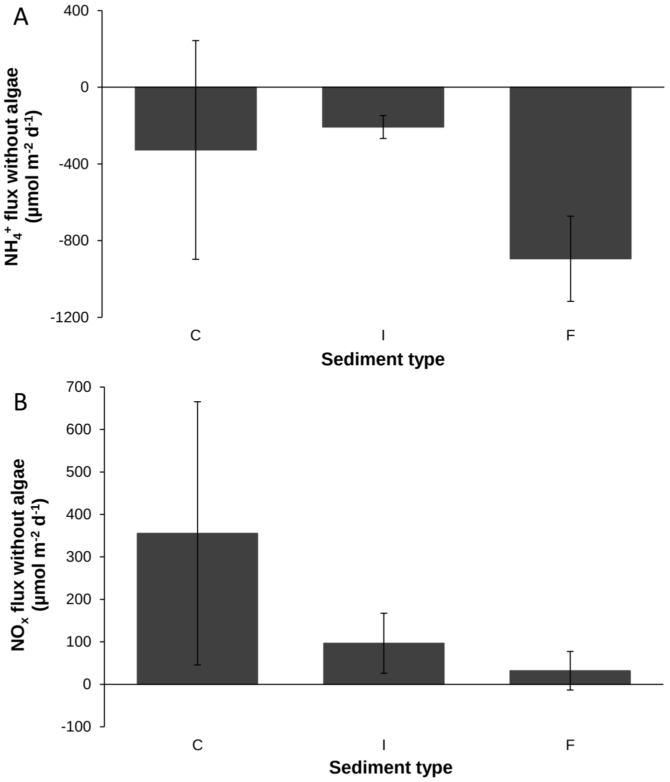

Fig. 7 NH_4^+ and NOx (NO_3^- + NO_2^-) flux (µmol m^{-2} d^{-1}) in the coarse (C), intermediate (I) and fine (F) sediments before the addition of algae

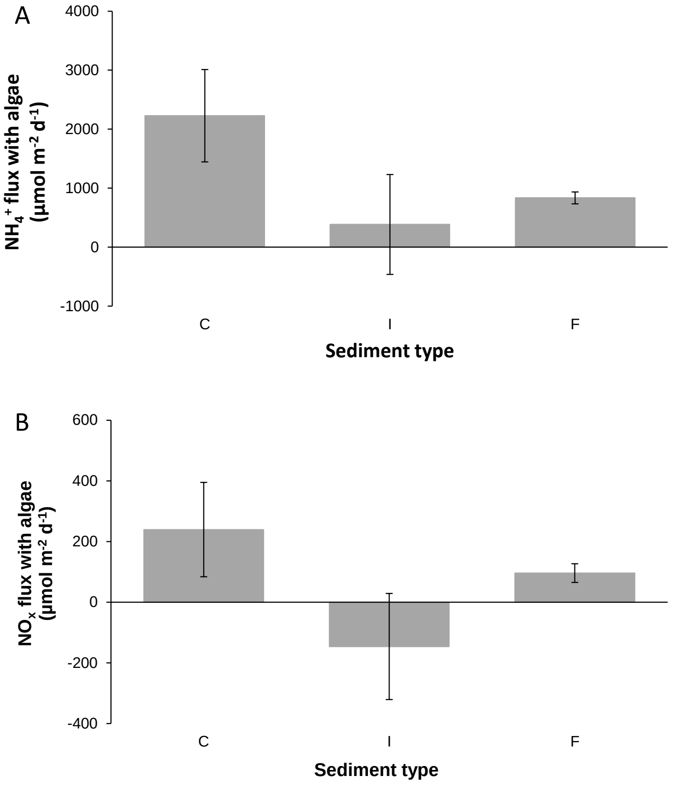

Fig. 8 NH_4^+ and NOx (NO_3^- + NO_2^-) flux (µmol m^{-2} d^{-1}) in the coarse (C), intermediate (I) and fine (F) sediments with the addition of algae (*S. costatum*)

Before the addition of algae, a decreasing trend in NO_x (NO_3^- + NO_2^-) effluxes was observed with decreasing grain size (Fig. 7B). The efflux of NOx in the coarse sediments was 3.7 and 11 times higher in comparison to the NOx efflux measured in the intermediate and fine sediments. Fluxes decreased slightly after the addition of algae (Fig. 8B). In the intermediate sediment the NO_x flux was directed into the sediment. Two-way ANOVA did not detect any significant differences in NO_x fluxes between experiments or sediment types.

4. Discussion

Within this study, certain limitations were observed during the preparation phase of the three sediment types. By mixing sediment types collected from two completely different sites, initial situations of organic matter content could still have been different even after pre-treatment procedures. The finest sediment collected from the Paulina mudflat was burnt and washed to remove all organic matter, following the procedures of Ehrenhauss and Huettel (2004). For this specific sediment, the pre-treatment procedures need to be further tested and adjusted for future experimental set-ups. The commercially available sand, used to obtain the intermediate sediment, could have had variances in sand grain structure compared to the naturally formed sand grains which are constantly reworked by bottom currents. Furthermore, a complete homogenisation of the mixed sediments was often difficult to achieve. However, the main results of this pilot study are comparable with earlier findings from similar experiments (Ehrenhauss and Huettel, 2004; Ehrenhauss et al., 2004b; Huettel and Gust, 1992; Huettel and Rusch, 2000) and show clear differences in water penetration depth, filtration capacity and fluxes of DIN across the sediment-water interface between sediment types. Therefore, we believe that the obtained results from this study represent a simulation of the effects of the refinement of sediments on the biogeochemical properties of sediments in the BPNS.

4.1. Filtration capacity

The artificial refining of permeable sediments from the BPNS showed a decreased water penetration depth in the sediment and a decreased trapping of *Skeletonema costatum* cells, reflecting a less efficient benthic-pelagic coupling. With a high uptake of the Rhodamine dye in the coarse sediment, a strong advective pore-water flow was illustrated. The lower permeability of the intermediate and fine sediments clearly obstructed the advective flow of the coloured water into deeper layers of the sediments. Comparably, a decreased uptake of the chain-forming diatom cells was observed in the fine sediment resulting in a lower efficiency of the uptake of organic matter after sediment refinement. These results are comparable with previous studies where Huettel and Rusch (2000) observed a doubled degradation of algae cells in permeable sediments compared to impermeable bottoms. The uptake of algae is not only determined by sediment permeability and advective pore-water flows, the chain length of diatom cells are of equal importance. With an average size of *S. costatum* cells between 5 and 10 µm, Ehrenhauss and Huettel (2004) observed that *S. costatum* chains could not penetrate into sediments with a permeability below 7 x 10^{-12} m² as pore sizes were too small (30 – 53 µm diameter). Together with decreased advective pore-water flows, the transport of diatom cells was restricted to the upper 0.3 - 0.7 cm in fine sands compared to 5.5 cm in coarser sands (Ehrenhauss and Huettel, 2004).

The decreased filtering capability in refined sediments can lead to an accumulation of organic matter in the upper centimetres of the seabed, even forming accumulated aggregates at the sediment surface (Ehrenhauss and Huettel, 2004; Ehrenhauss et al., 2004a). Consequently, molecular diffusion and bioturbation by organisms become important mechanisms to assist in solute transport and mineralisation processes (Braeckman et al., 2010). For years, permeable sands were viewed as large empty deserts poor in organic matter with a low contribution to nutrient cycling (Boudreau et al., 2001). However, the results from this and previous stirring chamber experiments propose permeable sediments, in energetic coastal areas such as the BPNS, to be important and fast filters for organic matter and phytoplankton cells, vastly contributing to the recycling process of nutrients (Ehrenhauss and Huettel, 2004; Ehrenhauss et al., 2004a; Ehrenhauss et al., 2004b; Huettel and Gust, 1992; Huettel and Rusch, 2000; Janssen et al., 2005). Therefore, a reduction in permeability can greatly affect the benthic-pelagic coupling by decreasing the uptake and recycling of organic matter after phytoplankton blooms.

4.2. Ecosystem functioning

Oxygen is an important solute regulating mineralisation processes of organic matter in the seabed (Glud, 2008; Middelburg et al., 2004; Ziebis et al., 1996). The settled particulate organic matter is buried into the sediments where the degradation into carbon dioxide and ammonium (NH_4^+) takes place. Subsequently, oxygen oxidises ammonium into nitrite (NO_2^-) and nitrate (NO_3^-). Nitrate can further be denitrified into nitrogen gas. All components of the nitrogen cycle slowly escape back into the water column (Billen and Lancelot, 1988). Measuring the oxygen and nutrient fluxes across the sediment-water interface provides information on how ecosystem functioning or mineralisation processes change when permeability decreases.

Before algal addition in the second chamber experiment, SCOC rates significantly decreased in the intermediate sediments in comparison to the coarse sediments (Fig. 6). SCOC rates of the coarse, subtidal sediments were expected to be low due to the low organic matter content during this time of the year (March) (Franco et al., 2010). A SCOC value between 5 - 10 mmol O_2 m^{-2} d^{-1} was measured at station 330 in April by Franco et al (2010). They considered these rates to be underestimated due to the lack of advective pore-water transport during measurements (Janssen et al., 2005), which is confirmed by our experiment with a SCOC value of 30.3 mmol O_2 m^{-2} d^{-1} in the coarse sediment.

Previous studies have illustrated the stimulation of oxygen consumption after the addition of a diatom pulse (Ehrenhauss and Huettel, 2004), as oxygen is directly used to mineralise the added organic matter (Ehrenhauss et al., 2004b). However, SCOC rates were lower after the addition of algae in our study, but not significantly for the coarse and intermediate sediments. In the fine sediment, the significant decrease in SCOC rates could be a combination of the insufficient removal of organic matter in the sediment mix during pre-treatment procedures (see material and methods) and an experimental artefact due to the frequent intrusion of light throughout the experiment. In the fine sediment, living diatoms were hardly trapped into the sediment, accumulating the cells in the water column and enhancing oxygen production which decreased SCOC rates after the addition of algae. A complete exclusion of light into the chambers has to be taken into consideration for future experiments. Additionally, SCOC rates declined significantly with decreasing sediment permeability after the addition of diatom cells (Fig. 6). Similarly, Ehrenhauss and Huettel (2004) measured highest oxygen consumption rates in the medium and coarse sands after algae addition. The advective oxygen transport in permeable sediments enhances the availability of oxygen within the sediments, creating a faster mineralisation of nutrients. After the addition of algae, DIN fluxes did increase with increasing permeability after an incubation period of 23 h. A high efflux of NH_4^+ was measured in the coarse, permeable sediment, illustrating a greater uptake of the particulate organic matter (diatom cells) which is degraded into NH_4^+ in the sediments. The lower uptake of organic matter into the finer sediments consequently produces a lower NH_4^+ efflux. Ehrenhauss et al (2004b) also observed the fastest mineralisation process in coarse sediments. However, they measured higher NOx (NO_3^- +NO_2^-) concentrations than NH_4^+ after a 132 h incubation. The incubation period in our study (23 h) may have been too short to complete mineralisation processes, suggested by high NH_4^+ but lower NO_x effluxes (Fig. 8). Before the addition of algae (Fig. 7), nitrification in the sediment (witnessed from the NO_x efflux) used NH_4^+ from the water column as a source (witnessed from the NH_4^+ influx). After the addition of algae, NH_4^+ from the degradation of the added organic matter was the source of nitrification.

4.3. The indirect effect around offshore wind turbines

Our experimental results suggest that the refinement of sediments around wind turbines in offshore wind farms (Coates et al., 2014) could lead to a reduction in the advective transport through the sediments, which brings along a decrease in organic matter cycling (Ehrenhauss et al., 2004b). With the extensive colonisation of the foundation by hard-substrate epifaunal species (De Mesel et al., 2013), the deposition of organic matter on the seabed will be enhanced in these areas due to the higher depositional flow of faecal pellets and detritus (Coates et al., 2014). Around the offshore wind turbine, highest chlorophyll *a* concentrations (0 - 2 cm) were measured at the south-west gradient (see Addendum III), suggesting a greater accumulation of the deposited detritus on this side of the foundation due to dominating current regimes in this direction and the wake effect of the foundation (Coates et al., 2014). Together with a decreased permeability and a subsequent reduction in solute transport to deeper layers, the deposited organic matter could considerably accumulate in the upper centimetres of the seabed around offshore wind turbines (Ehrenhauss and Huettel, 2004). Further detailed research is required to investigate the effects of sediment refinement to the ecosystem functioning around offshore wind farms.

Additionally, a shift and enrichment of the macrobenthic community was also observed in close vicinity to the offshore foundation with a dominance of the tube-building ecosystem engineer *Lanice conchilega* (Coates et al., 2014). Intertidal studies have observed the effect of bioturbating ecosystem engineers to maintain permeable sediments and counter the expansion of mud flats (Volkenborn et al., 2007a; Volkenborn et al., 2007b). Incorporating food web dynamics and energy flows of key organisms into future experiments will provide additional information on the ecosystem functioning of subtidal permeable sediments around offshore wind farms in the BPNS.

Acknowledgements

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CHAPTER 6

General Discussion, Recommendations and Conclusions

1. Overall aim of this PhD

With the installation of Offshore wind farms (OWFs) throughout the entire North Sea, numerous hard substrates are being introduced into naturally, soft sandy sediments. Up till recently, sand extraction, dredging and beam trawl fisheries were the three main activities affecting the soft sediment macrobenthos in the Belgian part of the North Sea (BPNS). The introduction of an additional anthropogenic impact affecting the physical characteristics of the seabed raises concerns on how the local macrobenthos inhabiting this sediment will be affected. Therefore, the overall aim of this PhD thesis was related to unravelling the response of the macrobenthos to the construction and operational phases of OWFs. A multi-scale approach was applied in this thesis to identify 1) large-scale effects related to the construction or operational phases of OWFs and 2) small-scale changes in the operational phase of one foundation. In the following chapter, the main results of the conducted research are discussed and set into a wider ecological context. From our obtained knowledge, recommendations for future research and environmental monitoring around OWFs are suggested.

2. Changing macrobenthic communities around offshore wind farms

Before the construction of OWFs in the BPNS, the baseline status of macrobenthic communities on the Thorntonbank, Bligh Bank and reference bank (Goote Bank) were determined. All sandbanks had similar macrobenthic communities, characterised by a low species abundance and diversity. The polychaete *Nephtys cirrosa* was observed as the most dominant species on the Bligh Bank, while *N. cirrosa* co-dominated with the amphipod *Urothoe brevicornis* on the Thorntonbank and Goote Bank. Inhabiting coarse, sandy sediments, the macrobenthic community was classified into the transitional community of the *N. cirrosa* and *Ophelia borealis – Glycera lapidum* communities (Van Hoey et al., 2004). Being part of two different subtidal sandbank systems (Hinderbanks and Zeelandbanks), the Bligh Bank and Goote Bank did illustrate slight differences in biotic and sedimentological characteristics due to a higher coastal influence at the Goote Bank (Van Hoey et al., 2005; Vanaverbeke et al., 2000). Our results correspond to previous habitat mapping carried out on the BPNS where macrobenthic communities have been documented to illustrate an east-west and onshore-offshore gradient (Degraer et al., 2008).

A good knowledge of the temporal variability of the macrobenthic communities in the concession areas was needed to distinguish natural from anthropogenic triggered fluctuations related to the installation of OWFs. The macrobenthic communities illustrate a strong natural, temporal variability, corresponding with similar studies in nearby coastal regions (Frid, 2011; Kröncke et al., 2011; Van Hoey et al., 2007). Main fluctuations in the abundance and diversity could be linked to extreme weather conditions such as cold winter temperatures (Reiss et al., 2006), suggesting a macrobenthic community well-adapted to long-term natural disturbances. Throughout the long-term data analysis, the communities on the Thorntonbank and Goote Bank always demonstrated a similar fluctuation. The construction (dredging) activities related to the gravity based foundations (GBFs) in 2008 were capable of generating short-term changes to the macrobenthic community on the Thorntonbank. A significant difference in macrobenthic community composition between the Thorntonbank and reference bank (Goote Bank) was observed, recovering from 2009 onwards. With a rapid recovery thereafter, the high resilience of the macrobenthic assemblages in these areas to environmental and anthropogenic stress was confirmed. Such short-term, large-scale changes due to the construction activities of OWFs have not been detected before in other OWF monitoring studies. Lindeboom et al. (2011) did not detect any statistical differences in macrobenthic communities between the impacted and reference areas at a large-scale a few months after the completion of the OWF Egmond aan Zee (OWEZ) in the Dutch coastal zone. The type of installed foundation and related construction activities could impose different impacts to the macrobenthos. Monopile foundations were installed at OWEZ which involve much less seabed preparation (dredging) activities in comparison to the GBFs constructed at the Thorntonbank. Our results did illustrate comparable outcomes to various studies concerning the impacts of dredging and sand extraction activities (Boyd et al., 2005; Desprez, 2000).

With six GBFs constructed on the Thorntonbank, large-scale effects related to the presence of the hard substrate foundations were largely absent in the first years of the operational phase, corresponding to research carried out in similar ecosystems (Leonhard and Pedersen, 2005; Lindeboom et al., 2011). After downsizing to a smaller scale, drastic changes to the sedimentological characteristics and macrobenthic community composition were rapidly observed directly around one GBF. Three to four years into the operational phase, changes were detected up to a 50 m distance away from the scour protection system around the GBF characterised by a significant decrease in median grain size. With a local increase in biodiversity and a community dominated by the juvenile starfish (*Asterias rubens*) together with the tube-building polychaetes *Lanice conchilega* and *Spiophanes bombyx,* the macrobenthos around the foundation is shifting from the species and density poor *N. cirrosa* and *O. limacina – G. lapidum* community to the rich *Abra alba* community. Furthermore, changes to the local current flow mainly lead to substantial effects along the south-west side of the foundation. With a decreasing grain size, changing hydrodyanmics (Hiscock et al., 2002; Schröder et al., 2006) and a possible increased food supply due to the down flux of faecel pellets and detritus (Barros et al., 2001; Maar et al., 2009) from the hard-substrate epifouling organisms (De Mesel et al., 2013), environmental characteristics are altering the local macrobenthic community on a small-scale, directly due to the physical presence of the GBF.

The prohibition of beam trawl fisheries inside OWFs, creates a unique opportunity for researchers to determine how the cessation of centuries of beam trawling will alter the macrobenthos. It is expected that fragile and long-lived species could re-establish and increase in abundance with time (Defew et al., 2012; Kröncke, 2011). Three years after the construction of the OWF on the Bligh Bank, no significant differences in macrobenthic community composition were observed between the no fishery area inside the OWF and the fished area outside. However, subtle changes at the species level were detected in the no fishery area with a slight increase in dense tube-building terebellid patches and associated species such as *Eumida sanguinea*. The observation of regular illegal beam trawl fisheries within the OWF could have slowed down the recovery potential of the fragile, long-lived macrobenthic organisms. Similarly, five years after the construction of the OWEZ wind farm in the Dutch coastal zone, only subtle changes were observed to the macrobenthos with larger shell lengths or widths of the bivalves *Tellina fabula* and *Ensis directus* (Bergman et al., 2012). Both our study and the study at OWEZ suggest a longer term period is needed to detect the possible recovery potential of the soft substrate macrobenthic communities after centuries of beam trawling.

3. The importance of macrofauna in environmental monitoring studies

Belgium was one of the first European countries to implement an environmental monitoring programme to assess the potential impacts of OWFs on various aspects of the marine environment. The coordination of the programme through one initiative (The Marine Ecology and Management Section of the Operational Directorate Natural Environment, OD Nature) has led to successful overarching research (Degraer et al., 2013b). Monitoring programmes concerning the macrobenthos have also been implemented in the Netherlands, Germany, Denmark and Sweden. However, most programmes mainly focused on either small-scale (Leonhard and Pedersen, 2005; Maar et al., 2009; Schröder et al., 2006; Wilhelmsson and Malm, 2008) or large-scale (Bergman et al., 2012; Lindeboom et al., 2011) aspects. The programme in Belgium provided researchers with a unique framework to study every aspect of the marine environment in more detail. The multi-scale approach and yearly monitoring strategy applied in this PhD thesis has provided us with an important time series of data for the macrobenthos in the BPNS. With possible impacts ranging from the construction phase up to four years into the operational phase of OWFs on different spatial scales, this research offers an important baseline of knowledge for other countries on how OWFs can affect the natural soft substrate macrobenthos.

With the introduction of numerous hard substrates in the BPNS, the macrobenthos is an important component of the ecological monitoring programme as it provides us with direct information on how the natural soft, sandy sediments and their inhabitants are changing. As a food source for many demersal fish species and crustaceans, the macrobenthos has a crucial trophic position in the marine food web. A change to the macrobenthic species composition has the potential to alter food web energy flows (Dannheim et al., 2014) and interactions between organisms. The occurrence of dense *L. conchilega* patches and increased macrobenthic abundances on a small-scale could attract more demersal fish species inside the OWF (Rees et al., 2005; Ryer et al., 2013). So far, a higher abundance of demersal fish species has not been observed (Vandendriessche et al., 2013a). The presence of a higher food availability inside the OWF was suggested due to a higher measured stomach fullness index of the flatfish species dab (Derweduwen et al., 2012). As known ideal nurseries for plaice (Rabaut et al., 2010), the dense *L. conchilega* patches could offer protection to juvenile flatfish from visual predators (Auster et al., 1997). Together with the additional shelter from strong currents around the foundations, the areas close to the turbines could expand further into ecologically important habitats (Fig. 1).

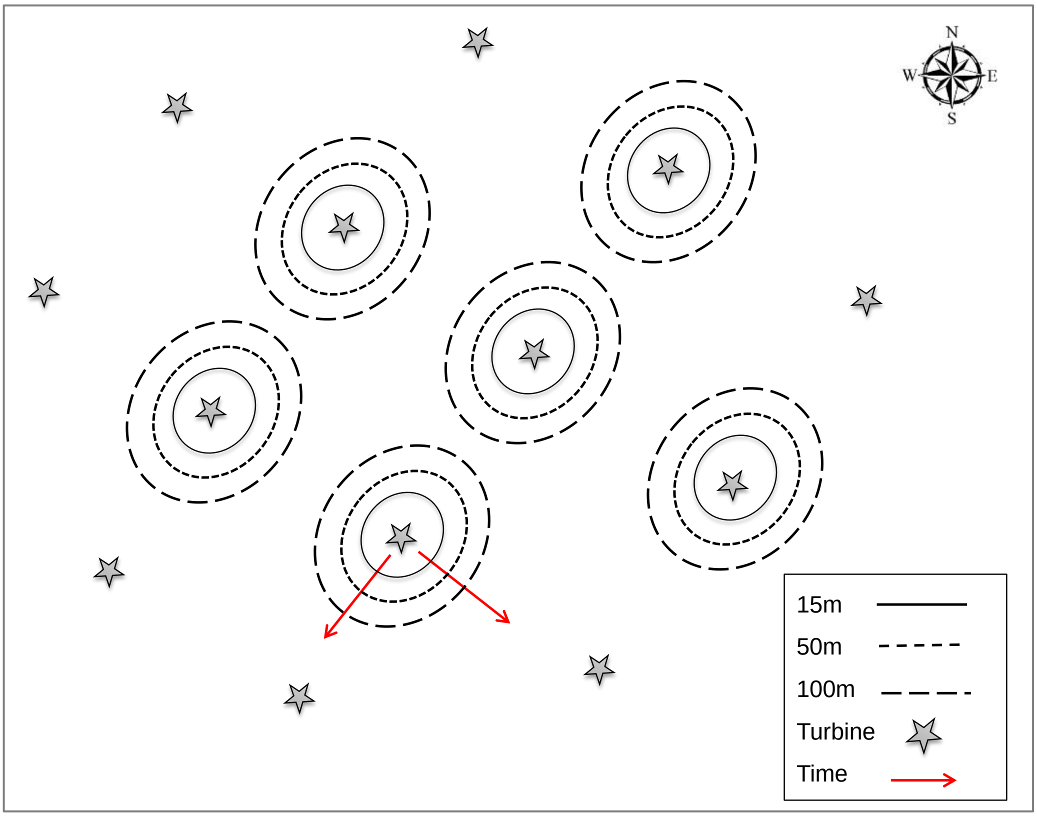

Fig.1 Illustration of the hypothetical expansion of the rich macrobenthic community around the D5 gravity based foundation on the Thorntonbank.

Simultaneously, the increased macrobenthic abundance within the patches (Callaway, 2006; Rabaut et al., 2007) increase the food availability (prey) for juvenile flatfish species. In the longer term, an expansion of the observed small-scale enrichment around the GBFs (Fig. 1) could be facilitated due to the prohibition of beam trawling (Kröncke, 2011). This hypothesis seems to be confirmed by the first signs of a slight increase in terebellid patches on a larger scale (21 km²) within the Bligh Bank OWF. The results suggest that an increased habitat complexity could emerge within the OWFs which could lead to ecological important refuge habitats for higher trophic levels and even commercially important species (Wilhelmsson et al., 2006).

However, with the evolution away from the transitional *N. cirrosa* and *O. limacina – G. lapidum* community at a small-scale, the habitat heterogeneity of the BPNS could also be negatively affected in the longer term. De Backer et al. (submitted) demonstrated similar diversity-disturbance responses of macrobenthic assemblages in the BPNS to three different human activities with a physical impact to the seabed (sand extraction, dredging and OWFs). Through the increasing pressure of anthropogenic activities, the loss of the *N. cirrosa* and *O. limacina – G. lapidum* habitat could decrease the diversity at a larger scale, inducing a homogenisation of the macrobenthic assemblages and sediments in the BPNS. This could result in a reduction of the habitat heterogeneity, further affecting the biodiversity and ecosystem processes of the area (Hewitt et al., 2008; Thrush et al., 2006). The experimental pilot study in Chapter 5 demonstrated that a decreasing grain size around offshore foundations could seriously affect the benthic ecosystem functioning by decreasing the uptake of oxygen and solutes into deeper layers of the sediment, decreasing mineralisation processes. The benthic-pelagic coupling and nutrient cycling could be severely altered around OWFs after a longer period. Furthermore, an increased deposition of organic matter to the seafloor due to the down flux of faecal pellets and detritus from organisms fouling the hard-substrate is highly possible. Both factors could lead to an accumulation of organic matter in the upper centimetres of the seabed around these offshore foundations. However, as consumers of organic matter, either in the seabed or from the water column, macrobenthic organisms are crucial for the mineralisation of organic matter (Heip et al., 1995). The increased occurrence of dense *L. conchilega* patches around the foundation could therefore facilitate the mineralisation processes. As a bioturbating ecosystem engineer, *L. conchilega* has the potential to maintain the permeability of sediments (Volkenborn et al., 2007a; Volkenborn et al., 2007b) and further enhance the mineralisation of the deposited organic matter (Braeckman et al., 2010).

An ecosystem based approach in marine impact assessments aims at improving the knowledge and management of impacts related to anthropogenic pressures by including different ecological interactions (Lange et al., 2010). The benthos is the only component that provides a direct link between ecological processes occurring in the seabed and the pelagic system in the water column. The importance of benthos in OWF monitoring studies has been illustrated through the identification of numerous cause-effect relationships between the installation of OWFs and the marine benthos (ICES, 2013), affecting the local or regional marine biodiversity. Integrated into four (biodiversity, food webs, seafloor integrity and invasive species) of eleven descriptors of the Marine Strategy Framework Directive (MSFD), the benthos is essential for the protection of the marine environment including its species and habitats (BelgischeStaat, 2012). With most part of the BPNS covered by sandy sediments, observing the changing macrobenthos and sedimentological characteristics around OWFs will be a crucial component in order to achieve a good environmental status (MSFD) of the sandy habitats in the BPNS.

4. Cumulative effects and up-scaling

With an increased biodiversity around the foundations and a prohibition to beam trawl fisheries, the OWFs could act as de-facto Marine Protected Areas (MPAs). MPAs have been implemented to manage human pressures (e.g. fishing pressure) in order to protect and conserve marine biodiversity (Jennings, 2009; Pedersen et al., 2009). Therefore, offshore wind farm sites cannot simply be labelled as MPAs as the marine habitats are being subjected to a human pressure (artificial hard-substrates) and the occurring assemblages will not necessarily represent the naturally occurring habitats of the BPNS. Furthermore, cumulative effects of different anthropogenic activities within the OWF concession area could further alter the naturally occurring macrobenthic communities. In 2013, reef balls were installed within the OWFs on the Thorntonbank and Bligh Bank, consisting of concrete modules and the deposition of a scour protection system (Vande Lanotte et al., 2012). The installation of these reef balls is an experimental phase of a seal action plan to determine if these additional artificial reefs would increase the biodiversity of the area. Additionally, the OWF concession area is also a designated mariculture area (Delbaere et al., 2013) but has not been implemented at this point in time. Mariculture activities would have the potential to increase the amount of deposited organic matter to the seabed even more (Maar et al., 2009).

In this PhD study, only short-term effects to the soft-substrate macrobenthos on foundation or OWF level (21 km²) were documented. With the expansion of OWFs to a concession area of 238 km² in the longer term, a large area of the BPNS will be subjected to the presence of many hard substrate foundations with related hydrodynamic changes and organic matter enrichment. Together with the prohibition of beam trawl fisheries, an evolution to a large ecologically rich and complex habitat is realistic. Furthermore, the development of a Belgian Offshore Grid (BOG) will efficiently transport energy produced at the OWFs to inshore locations (Elia, 2013) by creating an offshore high-voltage substation. The first permit has been requested for the creation of the “Alfa Island” which will assemble the power from at least five OWFs and transport it through one cable to the shoreline. The Alfa Island will be situated 45 km offshore, parallel to the OWF concession area and will contain a small harbour. This island will expand the already existing concession area, potentially inducing larger changes to the hydrodynamic conditions and widening the area closed to beam trawl fisheries. Additionally, concession zones for artificial energy islands (atolls) have been planned in the BPNS in order to efficiently store any excess energy produced from the OWFs (Vanbavinckhove and Pirlet, 2013).

Few studies have modelled the ecological evolution from changes to the macrobenthic community at a small-scale to abrupt, benthic regime shifts on a larger scale (Burkhard and Gee, 2012). The incorporation of benthic indicators (e.g. BEQI) in monitoring studies can be helpful to quickly detect these modelled regimes shifts in the future and discriminate them from natural fluctuations. Benthic indicators provide monitoring programmes with accurate threshold values of the habitat status which can be implemented to deliver management advice to the wind farm industry (Van Hoey et al., 2013). However, we must take into consideration that the current reference of these threshold values are based on a heavily disturbed baseline (e.g. from beam trawling) and should be taken into consideration through regular re-assessments.

Further up-scaling the observed results from this PhD thesis to the entire North Sea is not straightforward as seabed characteristics and hydrodynamic conditions can differ between regions. However, our results can be globally applied to sandy coastal shelf seas with similar abiotic conditions. Our in-field research can assist modelling studies to fill in knowledge gaps and determine if the observed local changes will induce longer term large-scale effects across wider regions such as the North Sea (Burkhard and Gee, 2012; Burkhard et al., 2011). The combination with remote sensing imagery will be inevitable as changing currents can be easily visualised by satellite imagery registering suspended matter content in the water column (Vanhellemont and Ruddick, 2014, in press).

The creation of large mosaic patterns of OWF concession areas in the North Sea raises concerns with respect to the creation of landscape corridors between artificial habitats. This stepping stone effect between OWF concession areas could further be facilitated by the creation of energy islands. Larval dispersal between areas can be assisted by the created landscape corridors (Tewksbury et al., 2002), potentially introducing non-indigenous species to certain regions. This could increase the competition pressure between organisms (Gill, 2005; Petersen and Malm, 2006), facilitating the possibility of abrupt large-scale changes occurring to the natural macrobenthic assemblages.

5. Recommendations for future monitoring and research in the BPNS

Throughout this PhD study, many difficulties were encountered concerning the sampling design in both the small and larger scale studies. The following paragraph will propose suggestions to help avoid these shortcomings in the future. Furthermore, new research questions are formulated and suggestions are made on how we can follow-up the evolution of the natural macrobenthic community at a larger scale around OWFs.

The environmental monitoring programme in Belgium has been successful and has set the example for many other countries. However, we were only able to focus on short-term effects of maximum four years into the operational phase of one or two OWFs. A longer term, adaptive monitoring programme will definitely still be needed during the coming decade. A wider scale, ecosystem based approach will be inevitable in future monitoring (Gill, 2005) as the current trend of offshore activities are leaning towards an overcrowded marine system of combined anthropogenic pressures.

- Acquiring knowledge on the frequency rate of illegal fishing activities will assist monitoring studies in identifying the recovery rates of macrobenthic communities inside OWFs. Therefore, future monitoring of the soft-sediment macrobenthos should be actively combined with Vessel Monitoring System (VMS) data from both Belgian and foreign vessels (Vandendriessche et al., 2013b). Furthermore, VMS data can contribute to the detection of edge effects around OWFs.

- Around the GBF on the Thorntonbank, an enrichment of the sediments was observed up to a 50 m distance away from the scour protection system. The main question arising is if this observed enrichment will spatially expand during the coming years (Fig. 1). To avoid replication problems, the sampling design should be simplified by focusing on the most interesting gradients together with a replication around different foundations instead of one.

- We must take into consideration that the small-scale impact study was only investigated around one GBF on the Thorntonbank. Small-scale effects to the surrounding sediments around monopile and jacket foundations could create similar or completely different impacts to the macrobenthos. Hydrodynamic conditions around the other foundations could be different due to variations in pile thickness or current flows through for example the jacket foundations. Furthermore, no scour protection system was installed around these foundations, decreasing the hard-substrate area for bio-fouling organisms and the food input onto the seabed. Assessing these differences between foundations will be essential in assisting modelling studies to up-scale the small-scale observations to a regional or global scale.

- In order to achieve a reliable Before After Control Impact design (BACI) in monitoring studies, suitable reference areas should be selected with similar environmental conditions to the impacted sites. An additional reference area should be assigned for the most offshore concession areas which are part of the Hinderbanks. As part of the Zeelandbanks, characteristics of the current reference bank (Goote Bank) are different due to the higher coastal influence.

- As a descriptor of the MSFD, unravelling changes to the structure and stability of the marine food web around OWFs will be required in order to achieve and manage a Good Environmental Status of the BPNS. The assessment of trophic interactions and food source availability will provide an enhanced understanding of the ecological processes around offshore foundations (Rooney and McCann, 2012). Introducing techniques such as isotope and stomach content analysis into monitoring studies will provide additional information on the changing food web dynamics. The incorporation of the pelagic system into the future monitoring programme will further assist food web studies. The deposited faecal pellets and detritus around the foundations could be quantified through sediment traps. Identifying the composition of the deposited matter will further provide information on the driving forces behind the observed changes in this PhD thesis to the seabed characteristics and macrofauna.

- The pilot study documented in Chapter 5 has illustrated the ability of OWFs to alter the benthic ecosystem functioning by decreasing the permeability of the surrounding sediments. Incorporating key macrobenthic species (e.g. *Lanice conchilega*) into future experiments will increase our knowledge on how the macrofauna can assist in the benthic ecosystem functioning around offshore foundations in permeable sediments.

- International collaboration and cross-border initiatives will be needed in Europe to tackle the large-scale, regional questions concerning the North Sea. Collaboration between scientists and exchange of data will assist modelling studies in determining possible long-term changes at a regional scale. Furthermore, a faster detection of possible landscape corridors for invasive species will be possible. The ICES Working Group on Marine Benthal and Renewable Energy Developments (WGMBRED) has initiated international collaboration and will be essential for continuation in the future.

- Marine spatial patterns are becoming increasingly more complex, illustrating the need for an adaptive spatial planning of OWFs according to the seascape ecology. Incorporating the seascape ecology into the management of the marine system will assist researchers and policy makers in achieving a good environmental status of the North Sea.

6. Conclusions

The multi-scale approach applied in this study, greatly assisted in understanding how ecological processes in the seabed are changing around offshore foundations in the BPNS. The construction activities of six GBFs did produce short-term changes to the macrobenthos. However, a rapid recovery was observed suggesting that 1) the macrobenthic community is highly resilient to disturbance due to the natural variability in this area and 2) short-term construction activities of OWFs will not cause major impacts to the macrobenthic community. However, we must be clear that these observations were based on the construction of six foundations. The cumulative construction activities of several OWFs should still be monitored to detect abrupt shifts in macrobenthic communities due to changing seabed characteristics. Combining traditional monitoring with benthic indicators to provide a fast detection of changes has been shown to be beneficial and should be incorporated into the longer term research of OWFs. The application of suitable benthic indicators will facilitate monitoring programmes in identifying ecological impacts and providing valuable management advice.

With the prohibition of beam trawl fisheries, an additional change will occur during the operational phase of OWFs. Fragile and long-lived species will be provided with the time and space to recover from centuries of fishing pressure. Our results have suggested the evolution to increased dense terebellid patches but due to illegal fisheries in the area and the short period of the study, no significant changes could be determined at this point in time.

Changing hydrodynamics, increased organic matter input and a decreasing permeability of the sediments in close vicinity to an offshore foundation can cause shifts in the macrobenthic community. An increased biodiversity of the macrobenthos was observed up to a distance of 50 m away from the foundation creating complex, rich patches of the ecosystem engineer *Lanice conchilega,* which could evolve into ecologically important refuge areas for young fish species at a larger scale in the future. However, the decreasing permeability of these sediments can change the benthic ecosystem functioning, stressing the importance to 1) determine how the macrobenthos will assist in the ecosystem functioning of permeable sediments and 2) how the structure and functioning of the marine food web is changing around OWFs.

Offshore renewable energy has become a wide scale hope to reduce greenhouse gas emissions. However, this research has demonstrated that the construction and presence of new hard substrates in a sandy environment has the potential to cause environmental changes to the marine ecosystem, possibly creating a new habitat within the Belgian part of the North Sea. The future monitoring programmes should have a research driven approach (e.g. experimental and in-field research) in order to fully understand the ecosystem processes around OWFs at different scales. This approach will further assist managers, policy makers and wind farm developers to take appropriate mitigating measures if needed and generate an adaptive spatial planning on a regional scale such as the North Sea.

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ADDENDA

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ADDENDUM I

Appendix to Chapter 2

Table 1. Multivariate abundance and biomass analysis with the pair-wise test for term year and zones for pairs of levels of factor Zones: Goote Bank (GC), Thorntonbank control (TC), Thorntonbank Edge (TE), Thorntonbank eastern impact (TI E) and western impact area (TI W). Significant results p < 0.05

	ABUNDANCE				**BIOMASS**			
2005	t	P(perm)	Unique perms	P(MC)	t	P(perm)	Unique perms	P(MC)
GC - TC	1.0306	0.3881	9933	0.3866	1.0378	0.3685	9952	0.3466
GC - TE	1.0254	0.3817	9925	0.3823	1.2614	0.124	9931	0.142
GC - TI E	1.6144	0.0094	9930	0.0182	1.7514	0.003	9945	0.0071
GC - TI W	1.0036	0.4292	9908	0.4003	1.2562	0.1248	9930	0.1573
TC - TE	1.0855	0.3007	9939	0.2974	0.96934	0.4394	9920	0.4313
TC - TI E	1.5876	0.0389	9944	0.0379	1.5345	0.0242	9939	0.0322
TC - TI W	0.7725	0.7556	9935	0.7132	0.66329	0.8971	9936	0.8615
TE - TI E	0.91666	0.5061	9954	0.5069	1.1023	0.2551	9927	0.2716
TE - TI W	1.2576	0.1478	9941	0.1571	1.0724	0.2906	9935	0.3078
TI E - TI W	1.5923	0.0331	9950	0.0381	1.6677	0.0075	9938	0.0124
2008	t	P(perm)	Unique perms	P(MC)	t	P(perm)	Unique perms	P(MC)
GC - TC	1.0862	0.2926	9943	0.2911	0.90149	0.6103	9932	0.5665
GC - TE	1.3865	0.0484	9918	0.063	1.3005	0.0659	9919	0.0929
GC - TI E	1.5619	0.0162	9924	0.0219	1.4828	0.0203	9919	0.0275
GC - TI W	1.6486	0.0091	9854	0.0133	2.0029	0.0005	9851	0.0009
TC - TE	1.4498	0.0204	9933	0.0395	1.24	0.1166	9931	0.1483
TC - TI E	1.3195	0.0692	9922	0.0957	1.1479	0.2229	9947	0.2273
TC - TI W	1.4471	0.0201	9087	0.0499	1.7787	0.0009	9127	0.0079
TE - TI E	1.4285	0.0235	9930	0.045	1.5153	0.0108	9939	0.0306
TE - TI W	1.5834	0.0094	7705	0.0282	1.9873	0.0011	7676	0.0036
TI E - TI W	1.0984	0.2653	9547	0.2804	1.4059	0.0463	9591	0.0696
2009	t	P(perm)	Unique perms	P(MC)	t	P(perm)	Unique perms	P(MC)
GC - TC	1.1061	0.261	9935	0.2683	1.3183	0.0801	9939	0.101
GC - TE	1.3298	0.0642	9937	0.0895	1.4725	0.0253	9938	0.0362
GC - TI E	1.3638	0.049	9929	0.0658	1.4491	0.0322	9920	0.0439
GC - TI W	1.1068	0.2691	9946	0.282	1.1704	0.2011	9923	0.2197
TC - TE	1.1042	0.2499	9932	0.2726	1.2336	0.1296	9921	0.1536
TC - TI E	0.98708	0.4293	9936	0.4352	0.99806	0.4303	9908	0.4231

TC - TI W	0.75053	0.8213	9903	0.77	0.87194	0.6578	9927	0.604
TE - TI E	1.3732	0.0402	9926	0.0625	1.3014	0.0646	9919	0.0904
TE - TI W	0.92667	0.5644	9901	0.5228	0.88479	0.678	9880	0.626
TI E - TI W	1.0285	0.3806	9927	0.3756	0.88377	0.6835	9930	0.6262

2010	t	P(perm)	Unique perms	P(MC)	t	P(perm)	Unique perms	P(MC)
GC - TC	0.85073	0.7098	8096	0.6555	0.72773	0.9027	7871	0.8564
GC - TE	1.6222	0.0084	9907	0.0101	1.571	0.0087	9902	0.0137
GC - TI E	0.82819	0.7819	351	0.6779	0.94169	0.5384	325	0.5044
GC - TI W	1.1859	0.1355	351	0.1907	1.1727	0.153	325	0.1999
TC - TE	0.81973	0.7522	1810	0.6747	0.79172	0.7744	1813	0.6998
TC - TI E	0.58554	0.9355	15	0.8015	0.61071	0.9384	15	0.7869
TC - TI W	0.80981	0.8048	15	0.5963	0.74027	0.8026	15	0.6703
TE - TI E	0.96335	0.4331	91	0.4651	1.0207	0.3931	91	0.4003
TE - TI W	1.1067	0.2726	91	0.2929	1.1749	0.2213	91	0.2325
TI E - TI W	1.284	0.3338	3	0.2797	1.1639	0.3341	3	0.3451

2012	t	P(perm)	Unique perms	P(MC)	t	P(perm)	Unique perms	P(MC)
GC - TC	1.4199	0.0588	35	0.1173	1.6711	0.0296	35	0.0532
GC - TE	1.2417	0.1409	1001	0.1655	1.4646	0.0345	998	0.068
TC - TE	1.112	0.2642	1001	0.2796	1.2937	0.0939	1000	0.1416

ADDENDUM II

Appendix to Chapter 3

Table 1. Species list of mean densities (ind m^{-2}) for every year (2008 – 2012) in the Control and No Fishery area on the Bligh Bank

	2008	2008	2009	2009	2011	2011	2012	2012
AVERAGE	**Control**	**No Fishery**	**Control**	**No Fishery**	**Control**	**No Fishery**	**Control**	**No Fishery**
Abludomelita obtusata	0	0	0	0	0	0	1	2
Abra alba	0	0	4	1	2	2	0	0
Aequipecten opercularis	1	0	0	0	1	0	0	0
Amphipode juv.	0	0	1	0	1	2	1	3
Angulus fabula	0	0	0	0	0	0	9	1
Angulus juv.	0	0	0	0	0	0	6	0
Angulus pygmaeus	5	0	8	5	17	4	9	8
Angulus tenuis	0	0	2	1	0	0	0	0
Anthozoa sp.	1	2	4	3	0	0	0	0
Aonides paucibranchiata	11	19	9	6	14	11	17	53
Arca sp.	1	0	0	0	0	0	0	0
Aricidea sp.	0	0	0	4	0	0	0	1
Atylus sp.	0	2	0	0	0	0	0	1
Bathyporeia elegans	9	8	8	34	12	19	20	55
Bathyporeia guilliamsoniana	36	16	53	67	19	11	42	28
Bathyporeia juv.	0	0	0	2	0	0	2	0
Bathyporeia pelagica	23	2	1	1	0	0	0	0
Bathyporeia pilosa	0	0	2	0	0	0	1	1
Bathyporeia sarsi	0	0	0	0	0	0	1	0
Bathyporeia sp.	0	0	2	6	1	2	3	1

Bivalvia juv.	0	0	0	0	0	0	1	0
Brachyura juv.	8	0	1	1	2	8	3	3
Branchiostoma lanceolatum	1	0	2	4	0	3	2	35
Capitellidae sp.	0	0	1	0	1	0	3	28
Cirratulidae sp.	0	0	1	0	0	0	0	2
Corophium sp.	1	0	0	0	1	0	2	1
Crangon sp.	1	0	0	0	0	1	0	0
Decapoda juv.	0	0	1	2	1	10	0	0
Diplodonta rotundata	0	0	0	0	1	0	0	0
Echinocardium cordatum	3	2	1	8	0	6	1	0
Echinocyamus pusillus	15	8	7	9	9	0	5	73
Echinoidea juv.	0	0	0	0	4	4	10	9
Edwardsia sp.	0	0	0	0	3	0	27	2
Eteone longa	1	3	7	14	1	2	0	0
Eumida sanguinea	0	0	0	0	0	0	1	112
Eunereis longissima	0	0	1	0	0	0	2	3
Eunice sp.	0	0	0	0	0	0	1	0
Eurydice spinigera	0	0	1	1	1	1	0	1
Eurytemora sp.	1	0	2	0	0	0	0	0
Euspira sp.	1	0	1	0	1	0	5	2
Exogone hebes	0	0	0	0	1	0	4	0
Gastrosaccus spinifer	2	5	0	2	2	42	7	30
Glycera sp.	16	44	25	11	12	12	11	44
Harmothoe sp.	0	0	1	1	0	0	0	9
Hesionura elongata	0	2	0	1	0	0	4	0
Heteromastus filiformis	1	0	3	0	0	1	0	64
Jassa sp.	38	3	0	0	0	0	0	0
Kurtiella bidentata	0	0	0	0	1	0	0	8
Liocarcinus sp.	3	3	0	0	1	0	0	0
Lumbrineris sp.	1	0	0	1	0	0	1	0

Macoma balthica juv.	0	0	0	0	0	0	3	0
Maerella tenuimana	0	0	0	1	1	0	1	0
Maldanidae sp.	0	0	0	0	0	0	0	5
Megaluropus agilis	0	2	0	1	0	1	2	0
Monocorophium acherusicum	0	0	0	0	0	0	2	1
Mya sp.	0	0	0	0	0	0	1	2
Myrianida prolifera	0	0	0	0	0	1	0	0
Nephtys cirrosa	139	106	73	98	70	42	37	36
Nephtys juv.	0	0	0	0	11	2	30	23
Nephtys longosetosa	0	0	0	1	1	1	1	0
Notomastus latericeus	1	3	12	5	2	1	8	48
Nototropis falcatus	0	0	0	1	1	0	0	0
Nototropis swammerdamei	2	6	0	0	1	18	3	36
Oligochaeta sp.	0	0	0	0	0	1	1	0
Ophelia borealis	3	0	1	7	9	10	10	18
Ophiura albida	6	5	4	1	0	0	1	2
Ophiura juv.	0	0	0	0	1	5	12	28
Ophiura ophiura	0	0	4	2	1	1	0	2
Paguridae sp.	1	0	0	0	0	1	0	0
Parougia eliasoni	1	13	0	2	0	0	0	0
Perioculodes longimanus	0	0	0	1	1	0	1	0
Pestarella tyrrhena	0	0	1	0	0	0	4	0
Philocheras trispinosus	0	0	0	1	1	1	0	0
Pholoe minuta	0	0	0	0	0	0	1	13
Phyllodoce rosea	3	5	1	6	0	0	0	1
Pisidia longicornis	0	0	0	0	1	0	0	2
Pisione remota	3	0	0	0	0	1	1	0
Poecilochaetus serpens	3	2	0	0	0	0	1	2
Polychaeta sp.	0	0	0	0	0	0	3	0
Polynoinae sp.	0	0	0	0	0	0	0	1

Pontocrates arenarius	0	3	0	1	1	1	0	0
Processa modica	4	0	4	2	3	0	9	1
Pseudocuma gilsoni	1	0	0	0	1	0	1	1
Pseudocuma longicorne	0	10	0	2	1	1	0	0
Pseudocuma simile	1	0	0	0	1	0	0	0
Schistomysis juv.	0	0	0	0	0	44	0	0
Scolelepis bonnieri	0	0	1	4	0	0	0	0
Scolelepis squamata	0	0	0	0	0	2	0	1
Scoloplos armiger	0	0	0	2	0	1	0	0
Sipuncula sp.	0	0	2	0	0	0	0	8
Spio sp.	8	11	16	13	18	32	3	11
Spiophanes bombyx	52	62	17	46	2	3	7	2
Spisula elliptica	0	2	1	2	1	0	0	0
Spisula juv.	0	0	0	0	0	0	2	4
Spisula subtruncata	3	2	4	0	5	2	2	0
Streptosyllis websteri	0	0	0	0	0	0	1	0
Striarca lactea	0	0	0	0	0	0	3	0
Syllis gracilis	1	2	0	1	0	0	0	1
Tanaissus lilljeborgi	0	0	0	0	1	0	1	0
Terebellidae sp.	8	5	13	42	12	19	62	196
Thelepus cincinnatus	1	0	2	0	0	0	0	0
Thia scutellata	8	3	6	7	3	0	3	0
Upogebia sp.	0	0	2	0	0	0	1	0
Urothoe brevicornis	11	2	31	4	5	31	41	0
Urothoe sp.	0	0	0	0	0	0	3	0

ADDENDUM III

Appendix to Chapter 4

A pilot study was carried out from 2010 – 2012 with samples collected at one and seven meters from the scour protection system at the D5 gravity based foundation on the Thorntonbank.

Scuba divers sampled the soft sediments with an airlift suction device (0.1026 m²). Species were identified to species level and the sediment characteristics (total organic matter content, median grain size and chlorophyll *a* concentrations) were analysed.

Table 1. Number of replicates from 2010 – 2011. Mean total abundance (ind m^{-2}), total number of species (N0), Shannon H' diversity index per sample, biomass AFDW (mg m^{-2}), median grain size d(0.5) (µm) and total organic matter content (TOM %) and abundance of the most dominant species on the south-west (SW), north-west (NW), south-east (SE) and north-east (NE) gradients at 1 and 7 m from the scour protection system around the D5 offshore wind turbine on the Thorntonbank

1m	**2010**				**2011**	**2012**			
	SW	**NW**	**SE**	**NE**	**SW**	**SW**	**NW**	**SE**	**NE**
Replicates	1	1	1	1	1	2	1	1	2
Abundance (ind.m^{-2})	9162	3177	4893	2154	2778	1418	887	3899	1155
Species number (N_0)	28	32	27	20	20	24	27	31	20
Species diversity (H')	1.66	2.42	1.69	2.01	1.97	2.40	2.95	1.96	2.36
AFDW (mg.m^{-2})	9537	1702	3515	878	424	5864	1448	2765	1754
d(0.5) µm	285	339	372	294	275	319	322	329	294
TOM %	0.60	0.27	0.48	0.51	0.76	0.92	0.53	0.61	0.64
Dominant species									
Asteriidae juv.	4834	107	653	146	877	283	10	1832	10
Lanice conchilega	1550	58	0	88	117	312	78	302	151
Spiophanes bombyx	897	78	88	975	0	15	19	19	68
Monocorophium acherusicum	127	1277	2778	175	936	0	0	39	5
Jassa herdmani	341	400	263	312	0	112	97	809	44
Echinoidea juv.	0	0	0	0	0	0	0	0	0
Urothoe brevicornis	0	39	39	0		19	97	10	0

7m	**2010**				**2011**	**2012**			
	SW	**NW**	**SE**	**NE**	**SW**	**SW**	**NW**	**SE**	**NE**
Replicates	1	1	1	1		2	1	1	2
Abundance (ind.m^{-2})	11501	3246	750	8723		585	536	2203	361
Species number (N_0)	32	29	18	29		16	20	18	15
Species diversity (H')	2.01	2.44	2.32	1.91		2.29	2.83	2.29	2.37
AFDW (mg.m^{-2})	5378	3973	498	4230		265	3313	464	109
d(0.5) µm	334	311	430	285	285	406	320	334	426
TOM %	0.59	0.39	0.69	0.84	9.79	0.52	0.53	0.48	0.73
Dominant species									
Asteriidae juv.	4961	0	214	1384		63	10	429	5
Lanice conchilega	1832	58	0	1949		73	0	39	5
Spiophanes bombyx	1062	195	0	1082		10	29	595	0
Monocorophium acherusicum	185	253	39	68			0	0	0
Jassa herdmani	731	409	78	29		24	19	224	15
Echinoidea juv.	39	1209	0	2943		0	0	0	0
Urothoe brevicornis	0	49	136	0		44	29	58	15

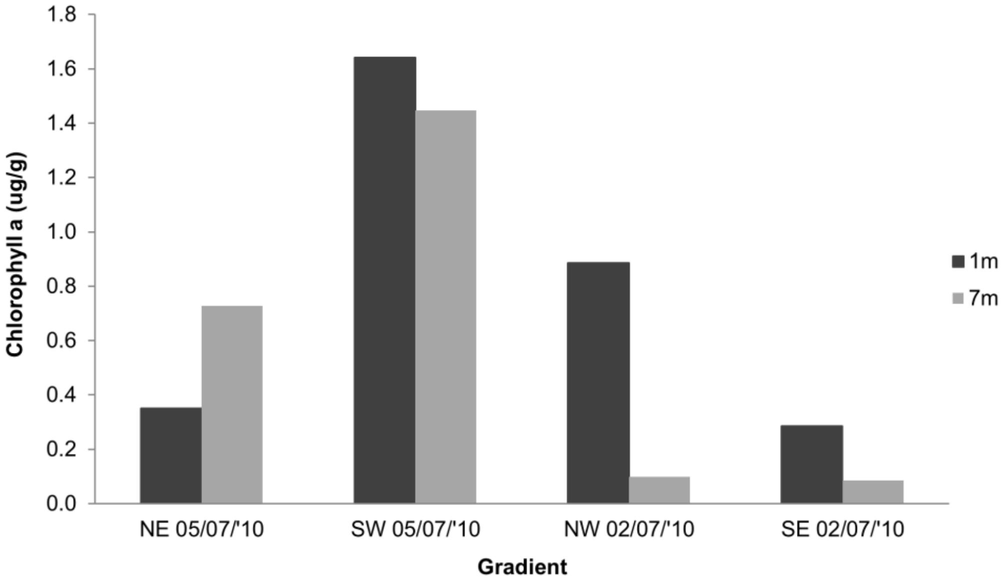

Fig. 1 Chlorophyll *a* (µg g^{-1}) concentrations on the north-east (NE), south-west (SW), north-west (NW) and south-east (SE) gradients at 1 and 7 m from the scour protection system around the D5 offshore wind turbine on the Thorntonbank

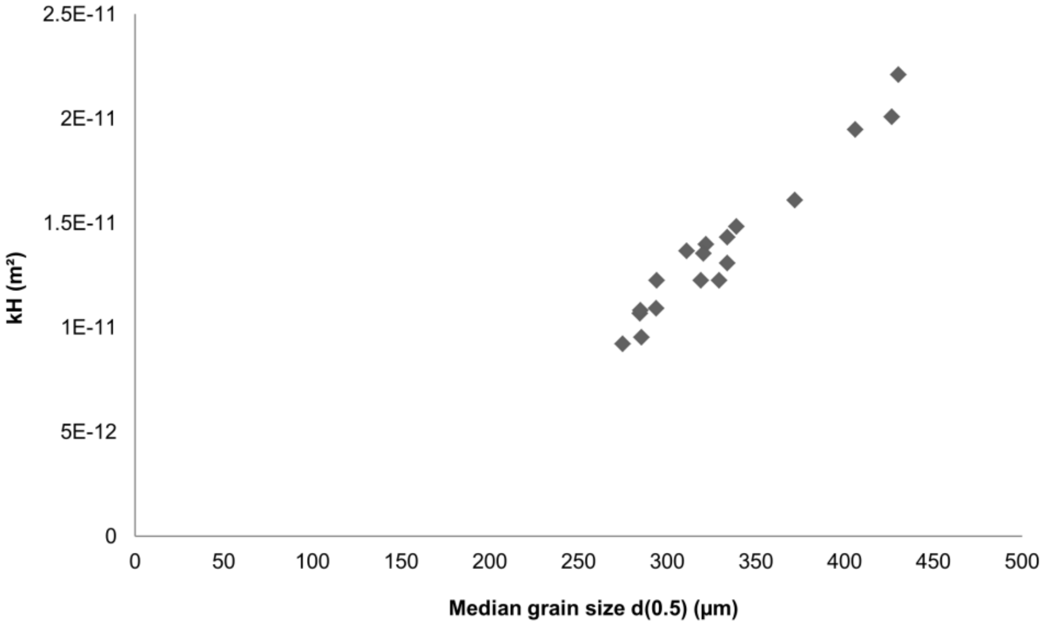

Fig. 2 Sediment permeability k_H (m²) and median grain size d(0.5) (µm) of samples taken at 1 and 7 m from the scour protection system around the D5 offshore wind turbine on the Thorntonbank

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Publication List

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Publication List

A1 – Peer reviewed Articles

1. Coates D.A., Deschutter Y., Vincx M., Vanaverbeke J. (2014) Enrichment and shifts in macrobenthic assemblages in an offshore wind farm area in the Belgian part of the North Sea. Marine Environmental Research. 95 : 1-12

2. Coates D.A., Kapasakali Danae-Athena, Vincx M., Vanaverbeke J. (submitted) Short-term effects of fisheries exclusion in offshore wind farms on macrofaunal communities in the Belgian part of the North Sea. Fisheries Research.

3. De Backer A., Van Hoey G., Coates D., Vanaverbeke J., Hostens K., (submitted) Similar diversity-disturbance responses to different physical impacts: three cases of small-scale biodiversity increase in the Belgian part of the North Sea. Marine Pollution Bulletin.

4. Coates D.A., Van Hoey G., Colson L., Vincx M., Vanaverbeke J. (to be submitted) Macrobenthic recovery after construction of an offshore wind farm in the Belgian part of the North Sea on a large-scale. Special issue Hydrobiologia.

Poster presentations

1. Rappé K., Claessens M., Coates D., Janssen C.R., Vincx M. (2009) Biomarker responses in *Mytilus edulis* following in situ exposure in Belgian coastal harbours. 9th VLIZ Young Scientists' Day, Bruges, Belgium. 06/03/2009.

2. Claessens M., Rappé K., Coates D., Vincx M., Janssen C.R. (2009) Use of multibiomarker experiments with caged mussels for risk assessment of marine pollutants. 10th VLIZ Young Scientists' Day, Ostend, Belgium. 27/11/2009.

3. Degraer S., Brabant R., Braeckman U., Coates D., Courtens W., Derweduwen J., Di Marcantonio M., Haelters J., Hostens K., Jacques T., Kerckhof F., Norro A., Reubens J., Stienen E., Vanaverbeke J., Van Colen C., Vandendriessche S., Van den Eynde D., Van de walle M., Vanermen N., Van

Hoey G., Vigin L., Vincx M., Wittoeck J. (2009) Monitoring the impact of offshore wind farms on the marine environment: an obligate multidisciplinary and integrated programme. 10th VLIZ Young Scientists' Day, Ostend, Belgium. 27/11/2009.

4. Coates D., Reubens J., Vanden Eede S., Vincx M. (2009) Monitoring the impact of offshore wind farms on the soft-sediment macrobenthos. 10th VLIZ Young Scientists' Day, Ostend, Belgium. 27/11/2009.

5. Coates D., Vanaverbeke J., Vincx M. (2010) The impact of Offshore wind farms on the soft-sediment macrobenthos in the Belgian part of the North Sea. 45th European Marine Biology Symposium, Heriot-Watt University, Edinburgh, Scotland. 23-27/08/2010.

6. Reubens J., Coates D. (2010) Windmolens: voor mens en dier. Nacht van de onderzoekers, Brussels, Belgium. 24/09/2010.

7. Coates D., Vanaverbeke J., Vincx M. (2011) A close up on the soft-sediment macrobenthos around offshore wind turbines. 11th VLIZ Young Scientists' Day, Bruges, Belgium. 25/02/2011.

8. Coates D., Vanaverbeke J., Vincx M. (2011) Temporal analysis of the macrobenthic data along the wind farm concession areas in the Belgian part of the North Sea. ICES/NAFO symposium on the variability of the North Atlantic and its marine ecosystems during 2000-2009, Santander, Spain. 10-12/05/2011.

9. Coates D., Vanaverbeke J., Vincx M. (2012) A historical view of the macrobenthic communities on the Thorntonbank and Goote Bank. 12th VLIZ Young Scientists' Day, Bruges, Belgium. 24/02/2012.

10. Coates D., Vanaverbeke J., Vincx M. (2012) Soft-sediment macrobenthos around offshore wind farms in the Belgian part of the North Sea. ICES Workshop on the effects of Offshore wind farms on Marine Benthos (WKEOMB)– Facilitating a closer international collaboration throughout the North Atlantic Region, Bremerhaven, Germany. 27-30/03/2012.

11. Coates D., Vanaverbeke J., Vincx M. (2013) 31 years of macrobenthic data on the Thorntonbank and Goote Bank: a parallel evolution or not? 13th VLIZ Young Scientists' Day, Bruges, Belgium.

12. Coates D., Kapasakali D. Vincx M. , Vanaverbeke J. (2013) Large and small scale changes in macrobenthic communities in Belgian Offshore wind farms. WINMON.BE 2013 Conference. Monitoring environmental impacts of offshore wind farms in the Belgian part of the North Sea – Learning from the past to optimize the future monitoring. Brussels, Belgium, 26-28/11/2013.

Oral presentations

1. Coates D., Vanaverbeke J., Vincx M. (2010) Future plans to monitor the impact of offshore wind farms on the soft-sediment macrobenthos. OcteMBSS, Ghent, Belgium, 26/02/2010.

2. Coates D., Vincx M., Vanaverbeke J. (2011) A close up on the soft-sediment macrobenthos around offshore wind turbines. World Conference on Marine Biodiversity, Aberdeen, Scotland. 26-30/09/2011.

3. Coates D. (2012) Monitoring offshore wind farms in the Belgian part of the North Sea. Mares Science Café, Ghent, Belgium. 06/02/2012.

4. Coates D., Vanaverbeke J., Vincx M. (2012) The added value of small-scale monitoring of the soft-sediment endobenthos around offshore wind farms in the BPNS. ICES annual science conference, Bergen, Norway. 19-21/09/2012.

5. Coates D., Vanaverbeke J., Vincx M. (2013) Macrofaunal enrichment near wind turbines on the Thorntonbank: succession or steady-state? UndeceMBSS, Ghent, Belgium, 08/03/2013.

6. Coates D., Reubens J. (2013) Attraction & Production? WINMON.BE 2013 Conference. Monitoring environmental impacts of offshore wind farms in the Belgian part of the North Sea – Learning from the past to optimize the future monitoring. Brussels, Belgium, 26-28/11/2013. Joint presentation.

B2 - Reports

1. Coates D. and Vincx M. (2010) Monitoring the effects of offshore wind farms on the soft substratum macrobenthos: Year-1 Bligh Bank and Year-2 Thorntonbank. In: *Offshore wind farms in the Belgian part of the North Sea: Early environmental impact assessment and spatio-temporal variability.*, Degraer S, Brabant R, Rumes B (eds), Chapter 7, pp 83-104. Royal Belgian Institute for Natural Sciences, Management Unit of the North Sea Mathematical models.

2. Coates D., Vanaverbeke J., Rabaut M., Vincx M. (2011) Soft-sediment macrobenthos around offshore wind turbines in the Belgian Part of the North Sea reveals a clear shift in species composition. In: *Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring.*, Degraer S, Brabant R, Rumes B (eds), Chapter 6, pp 47-63. Royal Belgian Institute for Natural Sciences, Management Unit of the North Sea Mathematical models.

3. Van Hoey G., Coates D., Hostens K., Vincx M. (2011) The use of the Benthic Ecosystem Quality Index (BEQI) for the evaluation of the impact of the Thorntonbank wind farm on the soft-bottom macrobenthos. In: *Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring.*, Degraer S, Brabant R, Rumes B (eds), Chapter 11, pp 147-157. Royal Belgian Institute for Natural Sciences, Management Unit of the North Sea Mathematical models.

4. Coates D., Vanaverbeke J., Vincx M. (2012) Enrichment of the soft sediment macrobenthos around a gravity based foundation on the Thorntonbank. In: *Offshore wind farms in the Belgian part of the North Sea: Heading for an understanding of environmental impacts*. Degraer S., Brabant R, Rumes B (eds), Chapter 4, pp 41-54. Royal Belgian Institute for Natural Sciences, Management Unit of the North Sea Mathematical models.

5. Coates D., Van Hoey G., Reubens J., Vanden Eede S., De Maersschalck V., Vincx M., Vanaverbeke J. (2013) The macrobenthic community around an offshore wind farm. In: *Environmental impacts of offshore wind farms in the Belgian part of the North Sea: Learning from the past to optimize future monitoring programmes*., Degraer S. , Brabant R., Rumes B. (eds), Chapter 9, pp 87-97. Royal Belgian Institute for Natural Sciences, OD natural environments, Marine ecosystem management unit.

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7. Rumes B., Coates D., De Mesel I., Derweduwen J., Kerckhof F., Reubens S., Vandendriessche S. (2013) Does it really matter? Changes in species richness and biomass at different spatial scales. In: *Environmental impacts of offshore wind farms in the Belgian part of the North Sea: Learning from the past to optimize future monitoring programmes.*, Degraer S. , Brabant R., Rumes B. (eds), Chapter 18, pp 182- 189. Royal Belgian Institute for Natural Sciences, OD natural environments, Marine ecosystem management unit.

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