Offshore wind farms in the Belgian part of the North Sea

Heading for an understanding of environmental impacts

Edited by
Steven Degraer
Robin Brabant
Bob Rumes

2012

Royal Belgian Institute of Natural Sciences
Management Unit of the North Sea Mathematical Models
Marine Ecosystem Management Section

in collaboration with

Commissioned and produced in 2012 by:
Royal Belgian Institute of Natural Sciences (RBINS)
Management Unit of the North Sea Mathematical Models (MUMM)
www.mumm.ac.be

Edited by:
Steven Degraer (steven.degraer@mumm.ac.be)
Robin Brabant (robin.brabant@mumm.ac.be)
Bob Rumes (bob.rumes@mumm.ac.be)

Cover photo:
The first phase of the C-Power wind farm on the Thorntonbank (photo J. Haelters / MUMM / RBINS)

Status ☐ draft
☒ final version
☐ revised version of document
☐ confidential

Available in ☒ English
☐ Dutch
☐ French

This report should be cited as:
Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2012). Offshore wind farms in the Belgian part of the North Sea: Heading for an understanding of environmental impacts. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. 155 pp. + annexes.

If a separate chapter is cited, the authors and the title of that chapter need to be mentioned.

If you have any questions or wish to receive a digital version of this document, please send an e-mail to info@mumm.ac.be, quoting the reference, or write to:

MUMM
100 Gulledelle
B–1200 Brussels
Belgium
Phone: +32 2 773 2111
Fax: +32 2 770 6972
http://www.mumm.ac.be/

Acknowledgements
This research is financed by C-Power nv, Belwind nv and Northwind nv, in fulfillment of the environmental monitoring programme of their environmental permits. The authors want to thank C-Power, Belwind and Northwind for their willing cooperation. This monitoring exercise benefited from the use of the research vessel Belgica (operated by the Belgian Navy under charter of the RBINS), the research vessel Zeeleeuw (operated by the Flanders Institute of the Sea) and the observation aircraft of RBINS for collecting the necessary data at sea.

Table of contents

Annexes

Chapter 1. Executive summary

S. Degraer[1*], R. Brabant[1], D. Coates[2], W. Courtens[3], J. Derweduwen[4], J. Haelters[1], K. Hostens[4], J.-S. Houziaux[1], F. Kerckhof[1], A. Norro[1], T. Onkelinx[3], B. Rumes[1], E.W.M. Stienen[3], J. Vanaverbeke[2], S. Vandendriessche[4], M. Van de walle[3], N. Vanermen[3], W. Van Roy[1], P. Verschelde[3], H. Verstraete[3], L. Vigin[1] , M. Vincx[2] & T. Willems[4]

[1]Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models (MUMM), Gulledelle 100, 1200 Brussels and 3e en 23e Linieregimentsplein, 8400 Ostend
[2]Ghent University, Biology Department, Marine Biology Research Group, Krijgslaan 281 (S8), 9000 Ghent
[3]Research Institute for Nature and Forest, Ministry of the Flemish Government, Kliniekstraat 25, 1070 Brussels
[4]Institute for Agricultural and Fisheries Research (ILVO-Fisheries), Bio-Environmental Research Group, Ankerstraat 1, 8400 Ostend

**Corresponding author: S.Degraer@mumm.ac.be*

Hard substrate epifauna on the C-Power D5 turbine Photo A. Norro / RBINS / MUMM

1.1. Introduction

With the launching of its new Climate Plan in 2008, the European Union committed itself to produce 20% of its electricity from renewable energy sources by 2020. Targets vary between member states and for Belgium the goal was set at 13%, to achieve by 2020. Offshore wind energy production is expected to be a major contributor in reaching this goal. The Royal Decree of 17 May 2004 delineated a zone for the production of electricity from water, current and wind in the Belgian part of the North Sea (BPNS). Since 2004, four companies, C-Power (Thorntonbank: 54 turbines, 325 MW), Belwind (Bligh Bank: 110 turbines, 330 MW), Northwind (formerly Eldepasco, Lodewijkbank: 72 turbines, 216 MW) and Norther (south of the Thorntonbank: 47-86 turbines, 258-470 MW), were granted a domain concession and an environmental permit to build and exploit an offshore wind farm. By July 2012 three other companies, Rentel, Seastar and Mermaid, obtained a concession, but still have to obtain an environmental permit. Both C-Power and Belwind already started the installation of an offshore wind farm. C-Power has built six gravity based foundation (GBF) wind turbines on the Thorntonbank in 2008, which were the first wind turbines in Belgian waters. Another 49 jacket foundations were installed in 2011 and the first half of 2012, which are now being equipped with the turbines. At the Belwind concession area, construction activities started in autumn 2009: in a first phase, 56 monopiles were driven into the seabed after which the foundations were equipped with turbines. The 55 wind turbines and one offshore high voltage station are operational since early 2011. Construction activities at Northwind will commence in early 2013. Belwind's second phase is foreseen to start in 2014.

To allow for a proper evaluation and auditing of the environmental impacts of offshore wind farms, the environmental permit includes a monitoring program to ensure (1) the ability to mitigate or even halt the activities in case of extreme damage to the marine ecosystem and (2) an understanding of the environmental impact of offshore wind farms to support policy, management and design of future offshore wind farms. The former objective is basically tackled through the baseline monitoring, focusing on the *a posteriori*, resultant impact quantification, while the latter monitoring objective is covered by the targeted or process monitoring, focusing on the cause-effect relationships of *a priori* selected impacts[1]. As such, the baseline monitoring deals with observing rather than understanding impacts and hence leads to area-specific results, which might form a basis for halting activities. Targeted monitoring on the other hand deals with the understanding of the processes behind the impacts and hence leads to more generic results, which might form a sound basis for impact mitigation. For more details on baseline and targeted monitoring we refer to Degraer & Brabant (2009).

The monitoring program targets physical (i.e. hydro-geomorphology and underwater noise), biological (i.e. hard substrate epifouling and fish communities, soft substrate macrobenthos, epibenthos and fish, seabirds and marine mammals), as well as socio-economical (i.e. seascape perception and offshore renewables appreciation) aspects of the marine environment. The Management Unit of the North Sea Mathematical Models (MUMM) of the Royal Belgian Institute of Natural Sciences coordinates the monitoring and specifically covers hydro-geomorphology, underwater noise, hard substrate epifauna, radar detection of seabirds, marine mammals and socio-economic aspects. In 2011, MUMM further collaborated with different institutes to complete the necessary expertise in the following domains: noise (INTEC, Ghent University), seabirds (Research Institute for Nature and Forest, INBO), soft substrate epibenthos and fish (Institute for Agricultural and Fisheries Research, ILVO-Fisheries), and soft substrate macrobenthos (Marine Biology Section, Ghent University). For details on the specific research strategies followed and methodologies used, one is referred to the individual chapters.

---

[1] While the first two integrated reports (Degraer & Brabant, 2009; Degraer *et al.*, 2010) mainly dealt with the results of the baseline monitoring aspect, last year's report (Degraer *et al.*, 2011) as well as this report rather focus on selected findings from the targeted or process monitoring in search for a proper understanding of the environmental impacts.

1.2. This report's focus

The first phase of the monitoring program (2005-2012) started the year before the (anticipated) construction of the first wind turbines at the Thorntonbank (i.e. 2005). At the end of this first phase, an overview and discussion of the monitoring activities and outcomes between MUMM, its monitoring partners and the wind farm industry is planned. This workshop will be the first thorough impact evaluation of marine wind farms in Belgian waters. It will also allow for a first valuation of these impacts: although some already observed effects (e.g. local enrichment of soft substrate benthos and attraction of some bird species) might seem positive, an appropriate ecological valuation is needed here. Concepts such as ecological pit falls, i.e. species being attracted to suboptimal habitat, are yet to be evaluated. Being a major focus of the first phase reporting, the application of these concepts will further help identifying the most critical issues to be tackled in the next phase of the environmental monitoring.

Although an exhaustive and thorough evaluation of possible impacts of marine wind farms in the BPNS will hence only be possible after the first six years of monitoring, important monitoring results become available along the monitoring trajectory. These results are published in yearly scientific reports, each focusing on a selection of scientific targets. A first set of scientific reports presented data on the baseline situation at future impact and reference sites (De Maersschalck *et al.*, 2006; Henriet *et al.*, 2006; Vanermen *et al.*, 2006). The first integrated report focused on the appropriateness of the general settings of the monitoring program, e.g. selection of reference sites and conditions, as well as strategic and technical recommendations for future monitoring (Degraer & Brabant, 2009). A second integrated report targeted the first scientific results on the evaluation of the early and or localized environmental impacts of the GBF wind turbines (C-Power) and monopiles (Belwind), as well as on the natural spatio-temporal variability (i.e. dynamic equilibrium) of various ecosystem components (Degraer *et al.*, 2010). The third integrated report focused on a selection of targeted monitoring results and attempted to construct a hypothesized impact scenario, including presumed cause-effect relationships between the various ecosystem components and activities related to the offshore wind farm projects (Degraer *et al.*, 2011). Finally, this year's integrated report continues building on a common understanding of the environmental impacts of offshore wind farms: (1) the cause-effect relationships with the benthos component of the marine ecosystem, as introduced in last year's report, were further explored and substantiated, while the 2011 monitoring program also allowed for (2) strengthening of the visual detection of impacts on seabirds and getting prepared for going offshore with the bird radar, and (3) an in depth analysis and quantification of harbour porpoise *Phocoena phocoena* disturbance by piling activities.

The above mentioned focuses of this year's report by no means preclude the fact that more data have been collected within both the C-Power and Belwind concession areas. These data will however be addressed in one of the upcoming yearly scientific reports, each having a selected focus.

1.2.1. Effects on the benthos and its food chain

As already hinted in the previous integrated reports (Degraer *et al.*, 2009, 2010, 2011), major effects onto the benthos component of the marine ecosystem become more pronounced as the wind farms 'grow older and bigger'. In this context, the effects can be linked (mainly) via the food chain from hard substrate epifouling organisms to the natural soft bottom macrobenthic and epibenthic communities to demersal and benthopelagic fish. Indirect effects of the exclusion of bottom trawling within the wind farm or altered hydrodynamic and sedimentological conditions can however not be ruled out. This year's report provided further evidence of these ever extending effects, from which a prognosis for further effect development can be deducted.

1.2.2. Hard substrate epifouling communities

The ecological succession within the epifouling communities started of swiftly with a no less than 49 species present only a couple of months after installation of the wind turbine foundation (Kerckhof *et al.*, 2009, 2010), including a wide range of non-indigenous species especially in the intertidal zone (Kerckhof *et al.*, 2011). Whether or not the patterns detected on a concrete gravity based foundation (GBF, C-Power) can be generalized to other wind farms yet remained a question. The epifouling community of other foundation types (*in casu* steel monopile foundations, Belwind) installed in other water masses (*in casu* clear English Channel water) in different years however showed a relatively high resemblance. No less than 50% of the total species pool (41 taxa out of 78 taxa) was shared between both wind farms and both wind farms were dominated by the amphipod *Jassa herdmani* with up to 90000 ind./m² (i.e. about 70% of the enumerable fauna). Other dominant species in both wind farms comprised the starfish *Asterias rubens* and the amphipod *Stenothoe valida*. The first year of succession also showed a highly similar trajectory, with particularly mobile species, e.g. *J. herdmani* and another amphipod *Corophium acherusicum* displaying a typical summer – winter oscillation pattern in both wind farms. This similarity was however less obvious for the sessile species, e.g. the polychaete *Pomatoceros triqueter* and the hydroid *Clytia hemisphaerica*, and seemed to strongly diverge after the first year. Concrete GBFs at the C-Power site also hosted more species than the steel monopiles (70 taxa versus 49 species). The presence of the coastal, soft sediment species such as the bivalves *Abra alba* and *Mysella* (*Kurtiella*) *bidentata*, only in the C-Power farm and attributed to the rather turbid waters in this area, further contributed to the dissimilarity between both wind farms.

1.2.3. Soft sediment macrobenthic communities

Changes in (1) currents, altering the settlement patterns of macrobenthos larvae, (2) sediment composition, altering the habitat suitability for benthic organisms and (3) local productivity, increasing the local food supply to the benthos, do impact the natural local soft sediment benthic communities. A first glimpse of this effect, which can be characterized as a general enrichment of the benthos close to the wind turbines, *in casu* gravity based foundations (C-Power, first phase), was already provided by Coates *et al.* (2011), who demonstrated the effect to extend to about 15 m. This year's findings confirmed the effects and demonstrated its further spatial extension to a distance of at least 50 m from the erosion protection layer. With densities up to about 55000 ind./m², particularly juvenile starfish *A. rubens* overwhelmingly dominated the macrobenthos nearby the wind turbine foundation. A gradient of strong effects close to the erosion protection layer to low/no impact at 100 m distance was observed. The southwest oriented transect for example, exhibited (1) a decrease in median grain size towards the wind turbine (about 400 µm at 100 m distance to about 250 µm at 1 m), (2) an increase in macrobenthic density (juvenile starfish excluded) from about 2000 ind./m² (100 m) to some 9000 ind./m² (15 m), (3) an increase in species richness from 10 spp./0.1 m² (100 m) to 23 spp./0.1m² (25 m) and consequently, (4) a significant change in community composition from the natural *Nephtys cirrosa* community to a community closely related to the rich, nearshore *Abra alba* community and characterized by the polychaete *Spiophanes bombyx*. These patterns are particularly visible in the southwest – northeast directions, as may be explained by the wake effect of the foundations onto the tidal currents.

1.2.4. Soft sediment epibenthos and fish communities

1.2.4.1. Current findings

Effects were also observed in the more mobile epifauna and demersal and benthopelagic fish. Although some differences between impact and reference sites are visible, e.g. lower and higher densities of demersal fish (about 100 versus 200 ind./1000m²), especially correlated to the decrease in lesser weaver *Echiichthys vipera* densities, and epibenthos (about 20 versus 5 ind./1000m²), respectively, on the impacted sandbank tops in autumn 2011, the unequivocal detection of consistent long term trends in diversity, species' densities and biomass and community composition remains difficult. However, the starfish *A. rubens* (about 10 versus <3 ind./1000m²) and the sea urchin *Psammechinus miliaris* (about 10 versus <1 ind./1000m²) tended to consistently occur in elevated densities within the wind farm from 2011 onwards. Especially for the latter species, this increased density may be caused by their dislodgement from the artificial hard substrates and their consequent prolonged survival in soft sediments in absence of bottom trawling. Yet, significant effects became particularly clear when comparing the size spectra inside and outside the wind farm. In our last year's report (Vandendriessche *et al.*, 2011), especially the occurrence of larger and absence of smaller swimming crabs *Liocarcinus holsatus* and the brown shrimps *Crangon crangon* within the wind farm (Belwind) attracted our attention. In this study also plaice *Pleuronectes platessa* could be added to the list of such species, while turbot *Psetta maxima* might be added to the list as it was caught with a few larger individuals only within the wind farm. The absence of fisheries (i.e. refugium effect) alongside a possible increased predation of smaller individuals (i.e. predation effect) and altered sediment composition and macrobenthic communities (i.e. food availability effect), are hypothesized to be at the basis of this finding. Food availability indeed differentiates the wind farms from the surrounding unimpacted sandbanks, as exemplified by dab *Limanda limanda* (this study) and pouting *Trisopterus luscus* (Reubens *et al.*, 2011). Dab predates on amphipods, decapods and polychaetes and inside the wind farm also on the hard substrate amphipod *Phtisica marina*. Furthermore, stomach analysis generally showed a higher fullness index within the wind farm, indicating the higher food availability nearby the wind turbines as well.

1.2.4.2. Future prospects

Although e.g. densities of the sea urchin *P. miliaris* and the starfish *A. rubens* are still far too high, the hard substrate epifouling community resembles the *Metridium senile* biotope (*sensu* Connor *et al.*, 2004). A further evolution towards this biotope with affinities to the surrounding natural gravel bed fauna, might hence be expected during the coming years or decade. Highly species-specific interactions (e.g. the sea slug *Onchidoris bilamellata* predating on barnacles) are known to play a major role within this succession. Whether or not this community will however ever reach the mature quality of the *M. senile* biotope is yet to be answered, as many epifouling communities on artificial hard substrates are known to never reach successional maturity.

An extension of the enrichment effect onto the soft sediment benthos was observed between 2010 and 2011 and now extended to at least 50 m from the erosion protection layer. A further spatial extension of the effect may hence be expected as the fining of the sediment and the increase of the food availability continues. As the distance between the erosion protection layers in the C-Power wind farm is only about 350 m, a full coverage spread of this enrichment of the soft sediment macrobenthic communities throughout wind farm should not be considered impossible.

The effect on densities, but more particularly the size spectra of some epibenthic and fish species is expected to get more pronounced as their populations continue developing in the unique environmental conditions of offshore wind farms. The environmental uniqueness will most likely further be strengthened as the increased food availability inside the wind farms extends spatially. Also the chance of its detection with the ongoing baseline monitoring will get significantly higher as the signal of more larger fish and less smaller fish gets stronger. We therefore expect to be able to statistically demonstrate this effect for even more epibenthic and fish species in the future.

1.2.5. Attraction and avoidance of seabirds

1.2.5.1. Current findings

Being highly mobile species exhibiting a high seasonal and year-to-year variability in densities and spatial distribution, the impact assessment onto seabirds is hampered by a substantial amount of uncertainty (Vanermen *et al.*, 2010, 2011). Quantifying this uncertainty through advanced statistical modeling (i.e. zero-inflated negative binomial modeling) and power analysis should hence be considered a major step forward when evaluating the true impacts of offshore wind farms on seabirds. So far only seabird attraction of e.g. little gull *Hydrocoloeus minutus*, common tern *Sterna hirundo*, and sandwich tern *Sterna sandvicensis*, could be demonstrated at the C-Power site. This attraction should however be expounded as until 2011 this wind farm was still one-dimensional (i.e. one line of six wind turbines), for which actual attraction to the wind farm or only to its edges cannot be distinguished. The two-dimensional Belwind site indeed showed both attraction (common gull *Larus canus* and herring gull *Larus argentatus*) and avoidance (common guillemot *Uria aalge* and northern gannet *Sula bassana*). Gulls are probably attracted from a sheer physical perspective, with the wind farm functioning as a stepping stone, a resting place or a reference feature in the wide open sea. Power analysis however revealed that changes in densities of 25 % will most likely never become detectable. In contrast, changes in densities of 50 % should be detectable after 10 years of monitoring.

1.2.5.2. Future prospects

While the ship-based visual counts allow quantifying the impact of offshore wind farms onto the local seabird populations, the investigation of seabirds will be complemented by radar observations (Brabant & Jacques, 2009). Radar research indeed provides detailed information on the flight patterns (incl. altitude) as well as on bird behaviour also when weather conditions do not allow for visual observations or during night time. Furthermore, its continuous data recording can be used to investigate subtle, though important changes in (flying) seabird spatial distribution as a function of environmental conditions, e.g. wind direction. A first test of the radar's applicability in the tern breeding colony in the port of Zeebrugge confirmed its capability in distinguishing birds from clutter and ships and in analyzing foraging flight patterns of terns in relation to wind direction. The radar data suggest that offshore feeding common terns *S. hirundo* depart from the colony with side- or tailwind, then forage with headwind while hovering, to take the side- or headwind back to the colony. As such, terns adapted their departure and arrival directions as a function of wind direction. The test phase has been particularly useful to learn how to properly work with the radar system, with lessons learnt at the level of (1) ground truthing and taxonomic resolution, (2) GIS processing of the vast amount of data collected and (3) strategic concepts for data analysis.

As the radar is installed onto the C-Power offshore high voltage station in March 2012, we expect to go offshore from the autumn migration period in 2012 onwards. With the tight integration of both visual and radar observations of seabirds we are entering a second stage in seabird monitoring.

1.2.6. Piling activities and harbour porpoises

Piling activities are known to generate excessive levels of underwater noise, with apparent source sound pressure levels (SPL_{p-p}) as high 270 dB re 1 µPa for monopiling (Norro *et al.*, 2010) or a zero to peak level (SPL_{z-p}) of 194 dB re 1 µPa, normalized to 750 m distance from the source. Although piling of the smaller pinpiles at the basis of jacket foundations was expected to be less noisy, no significant differences with monopiling could be demonstrated (pinpiling SPL_{z-p} normalized to 750 m: 172-189 dB re 1 µPa). Similarly, also the sound exposure level (SEL), varying between 145 and 168 dB re 1 μPa^2s normalized to 750 m, exerted no difference between mono- and pinpiling and near identical spectra were measured for both types of piling. However, although similar cumulative SEL values were observed for both foundation types, the jacket piling (four pinpiles per wind turbine) takes about 2.5 times more time than the monopiling and will as such have a prolonged impact onto e.g. marine mammals. Taking account of a discomfort SPL_{p-p} of 140 dB, harbour porpoises for example are expected to be disturbed to a distance of 19 km from the pinpiling location. Together with the prolonged duration, major effects of the pinpiling activities onto marine mammals might hence be expected.

Being the most abundant marine mammal in Belgian waters, wind farm piling activities are a major concern to the harbour porpoise *P. phocoena* (Haelters *et al.*, 2009, 2010, 2011). The species was particularly abundant at the end of March (about 2.5 ind./km², ~ 8500 porpoises), early April 2011 (1.3 ind./km²), when the C-Power pinpiling started. Dedicated aerial surveys during the C-Power pinpiling activities indeed showed a (modelled) disturbance up to about 22 km, as such affecting 2000 to 3800 porpoises (assuming a random distribution). Porpoise disturbance was confirmed by the nullification of acoustic detections near the piling location immediately after the start of the piling (5 dpm/h → 0 dpm/h). Twelve to 14 hours after cessation of the first piling activities the harbour porpoises temporarily reinvaded the impact area (7 dpm/h at about 5 km from the piling location). Such reinvasion was however much lower, if not non-existing from the second piling event onwards. Recovery of the spatial distribution could also be deducted from the aerial surveys, with a (modelled) impact radius of only about 13 km one day after cessation of piling. While the piling itself most likely is the main cause of the observed changes in spatial distribution, the contribution of other piling-related activities such as shipping cannot be fully ruled out. These activities might have caused the porpoises to leave the area prior to the start of the piling itself. The same goes for the acoustic harassment device (i.e. seal scarer), used to scare off marine mammals prior to piling to prevent major physiological damage due to the generation of excessive noise. These related activities however are all expected to disturb porpoises only up to a maximum of a few kilometers.

1.3. References

Brabant R. & Jacques T.G. (2009) Research strategy and equipment for studying flying birds in wind farms in the Belgian part of the North Sea. *In:* Degraer, S. & Brabant, R. (Eds.) (2009) Offshore wind farms in the Belgian part of the North Sea: State of the art after two years of environmental monitoring. Royal Belgian Institute for Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. pp. 223-235.

Connor, D.W., Allen J. H., Golding, N., Howell, K.L., Lieberknecht, L.M., Northen, K., Reker, J.B., (2004). The Marine Habitat Classification for Britain and Ireland Version 04.05. Joint Nature Conservation Committee, Peterborough.

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:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 6: 47-63.

Degraer, S. & Brabant, R., (Eds.) (2009). Offshore wind farms in the Belgian part of the North Sea: State of the art after two years of environmental monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. 287 pp. + annexes.

Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2010). Offshore wind farms in the Belgian part of the North Sea: Spatio-temporal variability and early impact assessment. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. 184 pp. + annexes.

Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. 157 pp. + annex.

De Maersschalk, V., Hostens, K., Wittoeck, J., Cooreman, K., Vincx, M. & Degraer, S. (2006) Monitoring van de effecten van het Thornton windmolenpark op de benthische macro-invertebraten en de visfauna van zachte substraten. 136 pp.

Haelters, J., (2009). Monitoring of marine mammals in the framework of the construction and exploitation of offshore wind farms in Belgian marine waters. *In:* Degraer, S. & Brabant, R. (Eds.). Offshore wind farms in the Belgian part of the North Sea: State of the art after two years of environmental monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. Chapter 10: 237-266.

Haelters, J., Kerckhof, F., Jacques, T.G. & Degraer, S., (2010). Spatio-temporal patterns of the harbour porpoise Phocoena phocoena in the Belgian part of the North Sea. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.). Offshore wind farms in the Belgian part of the North Sea: early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. Chapter 10: 153-163.

Haelters, J., Kerckhof, F., Vigin, L. & Degraer, S., (2011). Offshore wind farm impact assessment: monitoring of marine mammals during 2010. *In:* Degraer, S., Brabant, R. & Rumes, B. (Eds.). Offshore wind farms in the Belgian part of the North Sea: selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Brussels, 131-146.

Henriet, J-P., Versteeg, W., Staelens, P., Vercruysse, J. & Van Rooij, D. (2006) Monitoring van het onderwatergeluid op de Thorntonbank: Referentieonderzoek van het jaar nul. 53 pp.

Kerckhof, F., Norro, A., Jacques, T. & Degraer, S. (2009). Early colonisation of a concrete offshore windmill foundation by marine biofouling on the Thornton Bank (southern North Sea). *In:* Degraer, S. & Brabant, R. (Eds.) (2009). Offshore wind farms in the Belgian part of the North Sea: State of the art after two years of environmental monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 4: 39-51.

Kerckhof, F., Rumes, B., Norro, A., Jacques, T.G. & Degraer, S., (2010). Seasonal variation and vertical zonation of the marine biofouling on a concrete offshore windmill foundation on the Thornton Bank (southern North Sea). *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2010). Offshore wind farms in the Belgian part of the North Sea: early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. Chapter 5: 53-68.

Kerckhof, F., Degraer, S., Norro, A. & Rumes, B., (2011). Offshore intertidal hard substrata: a new habitat promoting non-indigenous species in the Southern North Sea: an exploratory study. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 4: 27-37.

Norro, A., Haelters, J., Rumes, B. & Degraer, S., (2010). Underwater noise produced by the piling activities during the construction of the Belwind offshore wind farm (Bligh Bank, Belgian marine waters). *In:* Degraer, S., Brabant, R., Rumes, B., (Eds.) (2010). Offshore wind farms in the Belgian part of the North Sea. Early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 4: 37-51.

Reubens, J., Degraer, S. & Vincx, M., (2011). Spatial and temporal movements of cod (Gadus morhua) in a wind farm in the Belgian part of the North Sea using acoustic telemetry, a VPS study. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 5: 39-46.

Vandendriessche S., Derweduwen J. & Hostens K., (2011). Monitoring the effects of offshore windmill parks on the epifauna and demersal fish fauna of soft-bottom sediments: baseline monitoring. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 7: 65-81.

Vanermen, N., Stienen, E.W.M., Courtens, W. & Van de Walle, M., (2006). Referentiestudie van de avifauna van de Thortonbank. 131 pp.

Vanermen, N., Stienen, E.W.M., Onkelinx, T., Courtens, W., Van de walle, M. & Verstraete, H. (2010). Monitoring seabird displacement: a modelling approach. *In:* Degraer, S., Brabant, R. & Rumes, B. (Eds.) (2010). Offshore wind farms in the Belgian part of the North Sea – Early environmental impact assessment and spatio-temporal variability: 133-152. Brussels: Management Unit of the North Sea Mathematical Models.

Vanermen, N., Stienen, E.W.M., Onkelinx, T., Courtens, W. & Van de walle, M., (2011). Seabirds & offshore wind farms: Power and impact analyses 2010. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 9: 93-129.

Chapter 2. Offshore wind energy development in the Belgian part of the North Sea & anticipated impacts: an update

R. Brabant[*], S. Degraer & B. Rumes

Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models (MUMM), Marine Ecosystem Management Section, Gulledelle 100, 1200 Brussels

[*] *Corresponding author:* *R.Brabant@mumm.ac.be*

Turbine transport and installation on the Thorntonbank Photo MUMM / RBINS

2.1. Context

The European Directive 2001/77/EC on the promotion of electricity produced from renewable energy sources in the internal electricity market, imposes upon each Member State a target figure of the contribution of the production of electricity from renewable energy sources that should have been achieved in 2010. For Belgium, this target figure was 6 % of the total energy consumption. In January 2008, the European Commission launched its new Climate Plan, and a new target for Belgium was set at 13 % to achieve by 2020. Offshore wind farms in the Belgian part of the North Sea (BPNS) are expected to make an important contribution to achieve that goal.

With the Royal Decree of 17 May 2004 a zone in the Belgian part of the North Sea (BPNS) was reserved for the production of electricity. It is located between two major shipping routes: the north and south traffic separation schemes (TSS). In 2011, the zone was adjusted on its Northern and Southern side because of the need for a safer shipping traffic in the vicinity of the wind farms. After this adjustment the total surface of the area is 238 km² (Figure 1). The initial surface was 264 km².

Prior to installing a wind farm, a developer must obtain (1) a domain concession in the zone reserved for wind energy development and (2) an environmental permit. Without an environmental permit, a project developer is not allowed to build and exploit a wind farm, even if a domain concession was granted.

When a project developer applies for an environmental permit an administrative procedure, mandatory by law, starts. That procedure has several steps, including a public hearing during which the public can express any objections. Later on during the permit procedure, the Management Unit of the North Sea Mathematical Models (MUMM) of the Royal Belgian Institute of Natural Sciences renders advice on the possible environmental impact of the future project to the Minister responsible for the marine environment. MUMM's advice includes an environmental impact assessment, based on an environmental impact study that is set up by the project developer. The Minister then grants or denies the environmental permit in a duly motivated decree.

The environmental permit includes a number of terms and conditions intended to minimize or mitigate the impact of the project on the marine ecosystem. Furthermore, as required by law, the permit imposes a monitoring programme to assess the effects of the project on the marine environment. The environmental monitoring is a legal obligation and is a competency of the federal government. The monitoring has two goals:

- to enable the authorities to mitigate or even halt the activities in case of extreme damage to the marine ecosystem;
- to understand and evaluate the impact of offshore wind farms on the different aspects of the marine environment and consequently support the future policy regarding offshore wind farms.

The monitoring is lead by MUMM, but MUMM collaborates with several other institutes that each have a specific expertise of the marine environment. The costs of the monitoring program are paid by the permit holders.

At present, four companies were granted a domain concession and an environmental permit to build and exploit an offshore wind farm: C-Power in 2004, Belwind in 2008, Northwind (formerly Eldepasco) in 2009 and Norther in early 2012. C-Power had its permit revised in 2006 and 2008, and the monitoring programme was adapted accordingly (Table 1).

C-Power and Belwind have already started their construction activities at the Thorntonbank and Bligh Bank, respectively, while Northwind's construction activities (72 turbines of 3MW) on the Lodewijkbank (formerly Bank zonder Naam) are expected to start in April 2013. The Norther project is located in the southernmost part of the wind energy zone and will presumably start its construction in 2014. More detailed information on projects can be found via www.c-power.be, www.belwind.be & www.northwindenergy.eu. Information on the Norther project can be found on the websites of www.airenergy.be and www.electrawinds.be.

Three other projects, Rentel, Seastar and Mermaid, were granted only a domain concession so far (Figure 1). Rentel, obtained a concession in between C-Power and Northwind (Figure 1). On July 2nd

2012, they have submitted their application to obtain an environmental permit. The concession of Seastar, in between Belwind and Northwind, was withdrawn in 2011, but, on June 1st 2012 the project obtained a new concession. Finally, on June 25th 2012, the project Mermaid was granted a domain concession in the northernmost part of the zone reserved for energy production. This is the seventh and, for now, last concession foreseen for the construction of wind farms in the Belgian part of the North Sea.

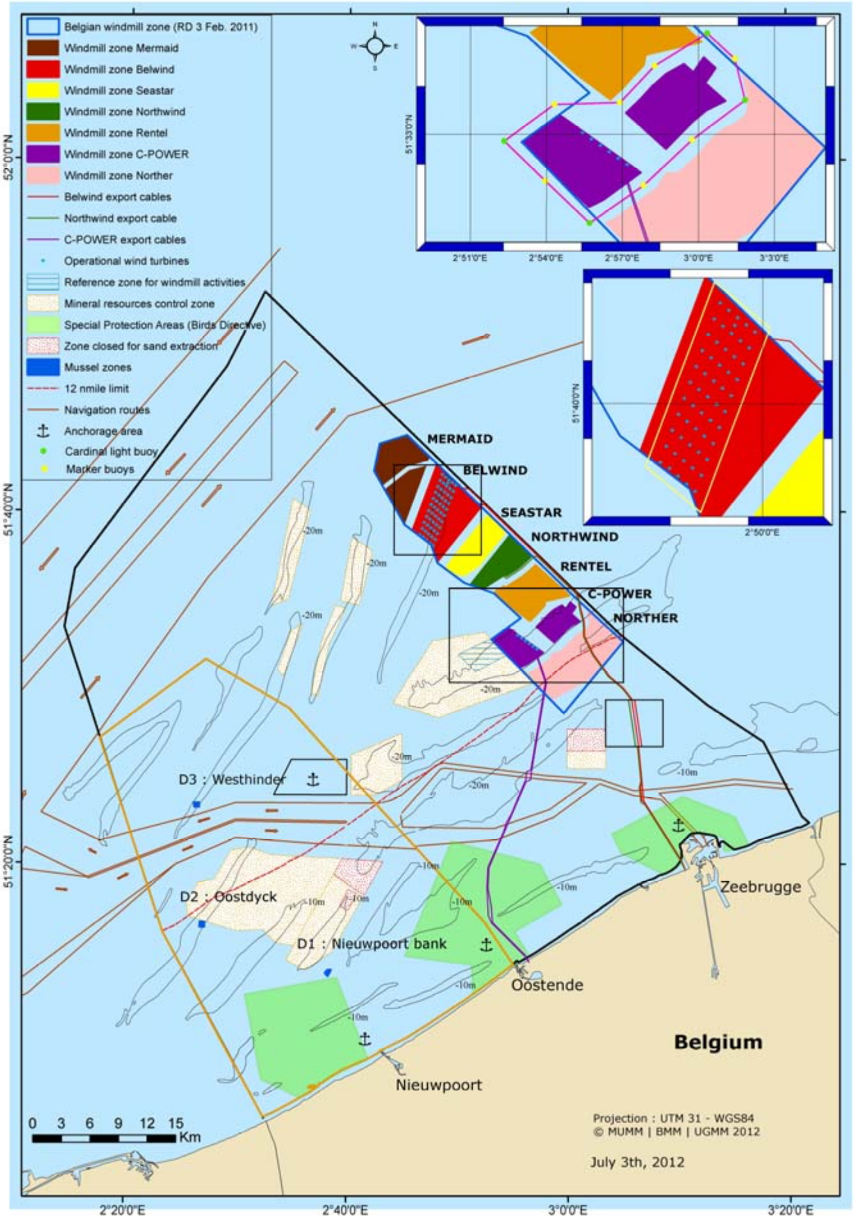


Figure 1. Zone reserved for the production of renewable energy by the Royal Decree of 17 May 2004 (http://www.mumm.ac.be/EN/Management/Atlas)

Table 1.
Overview of the dates when the projects were granted a domain concession and an environmental permit.

Project	Concession obtained	Permit application	Permit obtained
C-Power	27/06/03	17/6/2003 22/9/2005 -	14/04/2004 10/05/2006 25/04/2008
Belwind	5/6/2007	19/6/2007	20/2/2008
Northwind	15/5/2006	12/12/2008	19/11/2009
Norther	5/10/2009	10/5/2011	18/1/2012
Rentel	4/6/2009	2/7/2012	procedure ongoing
Seastar	1/6/2012	No application yet	
Mermaid	25/6/2012	No application yet	

2.2. Ongoing wind farm projects

2.2.1. C-Power

The C-Power project is located on the Thorntonbank (Figure 1). This is a sandbank located 27 km of the Belgian coast. Water depth in the concession area varies between 18 and 24 m.

The C-Power concession is divided in two sub-areas (A and B). Across the two sub-areas 54 turbines will be installed. Phase I (30,5 MW), a pilot phase, consists of six turbines that were installed on row D of sub-area A and the first 150 kV offshore cable (Figure 2). The six 5MW Repower turbines are operating since the 10th of May 2009. Phase II and phase III will each consist of 24 turbines of 6.15 MW. The installed capacity of the entire wind farm will be 325 MW.

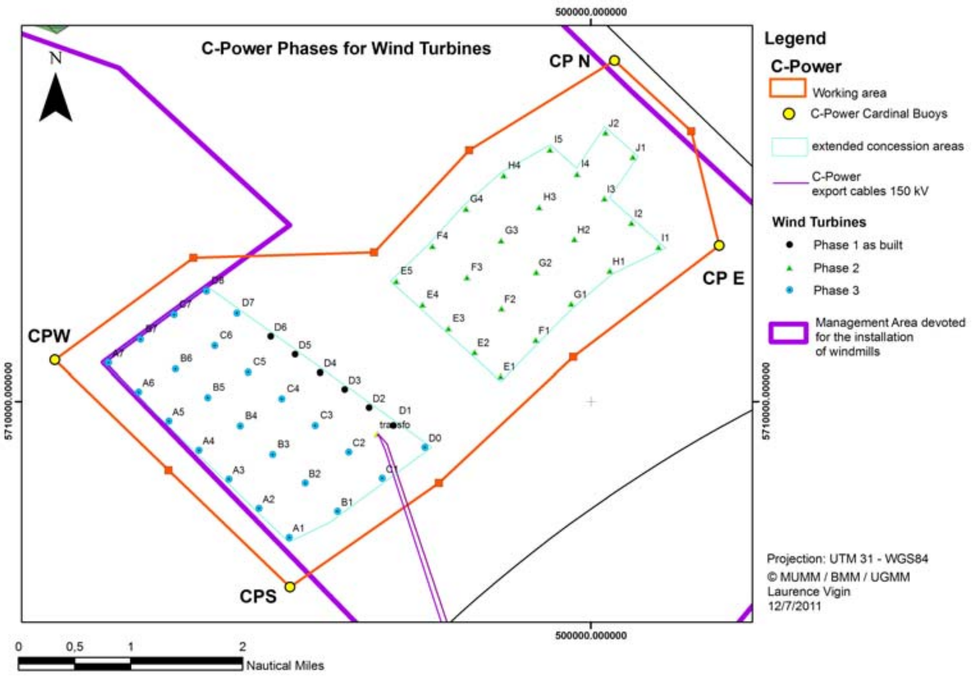


Figure 2. Layout of the C-Power project.

C-Power used gravity based foundations (GBF) for its phase I. These GBFs are hollow, concrete structures that are filled with sand, upon installation on the seabed. More detailed information can be obtained from Peire *et al.* (2009) and Brabant & Jacques (2010).

The foundation type for the phase II and III turbines is different from the pilot phase since jacket foundations, instead of the GBFs, were installed. These foundations consist of a steel jacket with four legs. The foundations were installed using the pre-piling concept: four pin-piles were driven into the seabed and the legs of the foundation were grouted on the pre-piles. The piles vary in length depending on the water depth at their location and are in the range of 21.0 to 49.5 m.

C-Power started on April 7th 2011 with the piling works. On August 21st 2011, all pin-piles for the phase II and phase III turbines were in place. Before the pre-piling of the pin-piles started, bottom surveys were conducted in 2010 and the seabed needed to be prepared. Details on this can be found in Brabant *et al.* (2011). All jacket foundations were installed in 2011 and the first half of 2012 (Figure 3). The installation of the 6 MW Repower turbines is now ongoing, 24 of the turbines were already installed by the end of June 2012. The offshore transformer station (OTS, figure 4) and the second export cable are already installed. Both sub sea power cables come ashore near Ostend.

Figure 3. Phase III jacket installation (Photo C-Power).

Figure 4. The OTS was installed on March 17th 2012. On the background the six phase I turbines (photo C-Power).

2.2.2. Belwind

The Belwind project is situated on the Bligh Bank at about 40 km off the Belgian coast (Figure 1 & 5). The water depth in the concession area varies between 15 and 40 m. Once finalized, the park will consist of 110 Vestas V90 turbines and an offshore high voltage station (OHVS), with a total installed capacity of 330 MW. The construction of the park is divided in two phases.

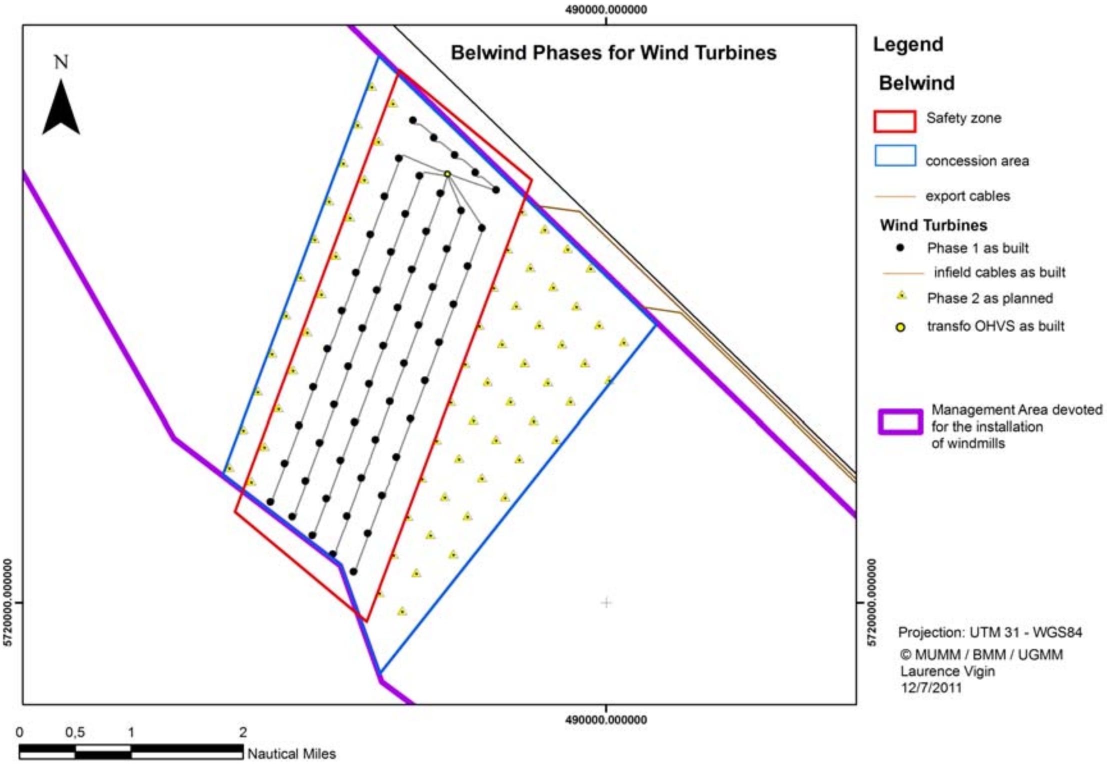


Figure 5. Lay out of the Belwind project.

In 2010, Belwind completed the first phase of their wind farm: 55 Vestas V90-3MW turbines, an OHVS, infield cables and an export cable. The 55 wind turbines are operational since January 13th, 2011 (Figure 6). Details on the construction of this first phase can be found in Brabant *et al.* (2011). No construction works were done in 2011. The start of the construction of phase II is foreseen in 2014.

Figure 6. Phase I wind turbines on the Bligh Bank (Photo MUMM / RBINS).

2.3. Anticipated environmental impacts

With the construction and exploitation of the above described projects a new offshore activity started in the BPNS. While offshore wind farms help achieving the goals set by 2001/77/EC on the promotion of electricity produced from renewable energy and help in the struggle against climate change, the construction and exploitation of offshore wind farms will also have certain impacts on the marine environment. These can be neutral, positive and/or negative for the marine ecosystem.

The environmental impact assessments (MUMM, 2004, 2007, 2009 & 2011) anticipated a variety of possible impacts. Some of those impacts are already being revealed during the first years of environmental monitoring (Degraer *et al.*, 2010 & 2011), e.g.:

- Increased erosion of the natural sandy sediments around wind turbine foundations because of accelerating currents next to the foundations;
- Increased turbidity during the construction of the wind farms;
- Increased underwater noise pressure, generated during the construction and exploitation phases and the associated impact on marine mammals and fish;
- Colonisation of the introduced hard substrata (i.e. foundations) by epifouling organisms and its consequent stepping-stone effect on invasive species;
- Attraction of fish by the introduced hard substrata;
- Changes within the soft-substratum macro- and epibenthos and fish as a result of e.g. fisheries displacement, altered sediment characteristics and organic enrichment of the sandy sediments by (local) deposition of organic matter produces by the hard substrate epifauna;
- Altered spatio-temporal distribution, densities and migration routes of seabirds and marine mammals;
- Altered public perception of offshore wind farms.

With the monitoring programme, MUMM and its partners (1) assess the extent of the anticipated impacts on the different aspects of the marine ecosystem and (2) aim at revealing the processes behind the impacts. The first objective is basically tackled through the baseline monitoring, focusing on the *a posteriori*, resultant impact quantification, while the second monitoring objective is covered by the targeted or process monitoring, focusing on the cause-effect relationships of *a priori* selected impacts. As such, the baseline monitoring deals with observing rather than understanding impacts and hence leads to area-specific results, which might form a basis for halting activities. Targeted monitoring on the other hand deals with the understanding of the processes behind the impacts of a selected set of hypothesized cause-effect relationships highly relevant to the wind energy sector. This step is not only a pre-requisite for effective regulatory application, but also permits (1) current and future impact mitigation, (2) better prediction of future impacts, as well as (3) moving away from site-specific observations to more generic knowledge. More details on this topic can be found in Degraer & Brabant (2009) and Degraer *et al.* (2010).

In 2009, we reported on the lessons learnt and recommendations from the first two years of environmental monitoring (Degraer & Brabant, 2009). The integrated Degraer *et al.* (2010) report focused on the natural spatio-temporal variability and the evaluation of the early and localized environmental impacts at the C-Power and Belwind sites. This report presents a selection of major findings from the baseline and targeted monitoring activities from 2011, and aims at a continued heading towards an understanding of environmental impacts of offshore wind farms.

2.4. References

Brabant, R., Degraer, S. & Rumes, B., (2011). Offshore wind energy development in the Belgian part of the North Sea & anticipated impacts: an update. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. pp. 9-16.

Brabant, R. & Jacques, T.G., (2010). Offshore wind energy development in the Belgian part of the North Sea and anticipated impacts. pp. 9-18. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2010). Offshore wind farms in the Belgian part of the North Sea: Spatio-temporal variability and early impact assessment. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. 184 pp. + annexes.

Degraer, S. & Brabant, R., (Eds.) (2009). Offshore wind farms in the Belgian part of the North Sea: State of the art after two years of environmental monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. 287 pp. + annexes.

Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2010). Offshore wind farms in the Belgian part of the North Sea: Spatio-temporal variability and early impact assessment. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. 184 pp. + annexes.

Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. 157 pp. + annex.

MUMM, (2004). Milieueffectenbeoordeling van het project ingediend door de n.v. C-Power. Rapport van het Koninklijk Belgisch Instituut voor Natuurwetenschappen, departement Beheerseenheid van het Mathematisch Model van de Noordzee (BMM). 155 pp.

MUMM, (2007). Milieueffectenbeoordeling van het BELWIND offshore windmolenpark op de Bligh Bank. Rapport van het Koninklijk Belgisch Instituut voor Natuurwetenschappen, departement Beheerseenheid van het Mathematisch Model van de Noordzee (BMM). 183 pp.

MUMM, (2009). Milieueffectenbeoordeling van het ELDEPASCO offshore windmolenpark op de Bank zonder Naam. Rapport van het Koninklijk Belgisch Instituut voor Natuurwetenschappen, departement Beheerseenheid van het Mathematisch Model van de Noordzee (BMM). 168 pp.

MUMM, (2011). Milieueffectenbeoordeling van het NORTHER offshore windmolenpark ten zuidoosten van de Thorntonbank. Rapport van het Koninklijk Belgisch Instituut voor Natuurwetenschappen, departement Beheerseenheid van het Mathematisch Model van de Noordzee (BMM). 189 pp.

Peire, K., Nonneman, H. & Bosschem, E., (2009). Gravity Based Foundations for the Thornton Bank Offshore Wind farm. Terra et Aqua, 115, 19 – 29.

Chapter 3. A comparison of the first stages of biofouling in two offshore wind farms in the Belgian part of the North Sea

F. Kerckhof*, B. Rumes, A. Norro, J.-S. Houziaux, & S. Degraer

Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models (MUMM), Marine Ecosystem Management Section, Gulledelle 100, 1200 Brussels and 3de en 23ste Linieregimentsplein, 8400 Oostende

**Corresponding author: F.Kerckhof@mumm.ac.be*

In remembrance of our colleague Jean-Sébastien Houziaux, who left us far too early.

Abstract

In this contribution, the species composition and ecological succession of the biofouling on the subtidal zone (circa littoral) of selected foundations of two offshore wind farms in the Belgian part of the North Sea, the C-Power farm on the Thorntonbank and the Belwind farm on the Bligh Bank, are studied through time. As observed in many other studies, the colonization of the bare substratum has been very fast in the two studied wind farms.

No less than 50% of the total species pool (41 taxa out of 78 taxa) was shared between both wind farms and both wind farms were dominated by the amphipod *Jassa herdmani* with up to 90000 ind./m² (i.e. about 70% of the enumerable fauna). Other dominant species in both wind farms comprised the starfish *Asterias rubens* and the amphipod *Stenothoe valida*. The first year of the succession also showed a highly similar trajectory, with particularly mobile species, e.g. *J. herdmani* and *Corophium acherusicum* displaying a typical summer – winter oscillation pattern in both wind farms. This similarity was however less obvious for the sessile species such as *Pomatoceros triqueter* and *Clytia hemisphaerica*, and seemed to strongly diverge after the first year. The concrete gravity based foundations (GBFs) at the Thorntonbank hosted more species than the steel monopiles at the Bligh Bank (70 taxa versus 49 species). The presence of some coastal, soft sediment species such as the bivalves *Abra alba* and *Mysella (Kurtiella) bidentata*, only in the C-Power farm, further contributed to the dissimilarity between both wind farms. The biofouling on both wind farms exhibits direct influence by the same pool of species originating from the surrounding artificial and natural hard substrata.

As for shipwrecks, specific features of the studied structures such as verticality, substratum composition or lower depths may explain specific patterns of taxonomic composition such as an impoverished sessile epifauna compared to the surrounding natural hard substrata. Predatory activities are an important factor controlling the succession of sessile species on the substratum. The high degree of patchiness suggests that the colonization process is still ongoing and the mature state, the *Metridium senile* biotope, characteristic for this type of substrata in the North Sea is not fully reached yet.

Apart from the slipper limpet *Crepidula fornicata*, no introduced species have been observed thus far in the permanently submerged part of the wind turbine foundations. This observation contrasts with the intertidal and splash zone, where many introduced species were present in the biofouling community.

Samenvatting

Deze bijdrage is gewijd aan het onderzoek van de soortensamenstelling en de ecologische successie van de aangroeigemeenschap op het subtidale (circalitorale) deel van de funderingen van 2 windmolenparken in het Belgische Deel van de Noordzee, namelijk het C-Power windmolenpark op de Thorntonbank en het Belwind windmolenpark op de Bligh Bank.

Zoals reeds in talrijke andere studies werd waargenomen, vond er een snelle kolonisatie plaats van het beschikbare substraat. Niet minder dan de helft van het totale aantal aanwezige soorten (41 van de 78 taxa) werd in beide windmolenparken aangetroffen. Op allebei de windmolenparken was de amphipode *Jassa herdmani* dominant aanwezig met dichtheden tot 90.000 ind./m² (dit is 70% van de telbare soorten). Andere dominante soorten in de beide windmolenparken waren de zeester *Asterias rubens* en het vlokreeftje *Stenothoe valida*. Tijdens het eerste jaar verliep de successie in beide windmolenparken zeer gelijkaardig met enkele mobiele soorten zoals *J. herdmani* en *Corophium acherusicum* die een typische zomer – winter oscillatie vertoonden. De sessiele soorten zoals *Pomatoceros triqueter* en *Clytia hemisphaerica* vertoonden een sterk verschillend kolonisatiepatroon waardoor de overeenkomsten na het eerste jaar minder duidelijk waren. Op de betonnen gravitaire funderingen van het C-Power windmolenpark op de Thorntonbank werden beduidend meer soorten aangetroffen dan op de stalen monopiles van Belwind op de Bligh Bank (respectievelijk 70 en 49 soorten). Het verschil tussen beide parken werd bovendien nog benadrukt door het voorkomen op de C-Power funderingen van enkele soorten die typisch zijn voor kustgebonden mobiele sedimenten zoals de tweekleppigen *Abra alba* en *Mysella (Kurtiella) bidentata*. De biologische aangroei op beide

parken vertoont sterke gelijkenissen met de aangroeigemeenschappen op de artificiële en natuurlijke harde substraten in de omgeving en de rekrutering gebeurt blijkbaar uit een gemeenschappelijke soortenpool.

In vergelijking met de omliggende natuurlijke harde substraten is de taxonomische samenstelling van de sessiele epifauna verarmd. Dit kan, net zoals bij scheepswrakken, verklaard worden door specifieke kenmerken van de bestudeerde structuren zoals verticaliteit, samenstelling van het substraat of een lagere waterdiepte. Daarnaast speelt de activiteit van predatoren een belangrijke bepalende rol bij de opeenvolging van sessiele soorten. Deze fenomenen zorgen voor de specifieke patronen. De hoge mate van ruimtelijke heterogeniteit (patchiness) suggereert dat het kolonisatieproces nog niet voltooid is. De climaxgemeenschap, de *Metridium senile*-gemeenschap, die kenmerkend is voor dit type van substraten in de Noordzee wordt bijgevolg nog niet maximaal bereikt.

Behalve het muiltje, *Crepidula fornicata*, werden geen andere geïntroduceerde soorten aangetroffen op de permanent ondergedompelde (subtidale) delen van de windmolenfunderingen. Deze waarneming contrasteert sterk met de intertidale en spatzone waar in de biologische aangroeigemeenschap overwegend geïntroduceerde soorten aangetroffen werden.

3.1. Introduction

With the construction of offshore wind turbines in the Belgian part of the North Sea (BPNS), a new habitat of artificial hard substratum is introduced in a region mostly characterized by sandy sediments. This largely enhances the habitat heterogeneity of the region and the effect of the introduction of these hard substrata is regarded as one of the most important changes of the marine environment caused by the construction of wind farms (Petersen & Malm, 2006).

It is well known that submerged artificial hard substrata are rapidly and intensively colonized by biofouling (e.g. Horn, 1974; Connell & Slatyer, 1977). This had been found to be the case with wind turbines in the North Sea (e.g. Schröder *et al.*, 2005; Kerckhof *et al.*, 2009; Kerckhof *et al.*, 2010). Fouling assemblages will develop successively, which may resemble epibioses on natural substrata (e.g. Connell, 2001). The wind turbines will also enable the establishment of species previously not present in an environment dominated by soft sediment habitats, as well as the further spread of non-indigenous species (stepping stone effect) (e.g. Kerckhof *et al.*, 2011). Certain warm water species may also take advantage of the increased presence of hard substrata to spread further into the North Sea due to climate change. Alternatively, the foundations and associated scour protection may allow for the re-establishment of biological communities previously present on nearby gravel beds.

Fish populations including commercial fish are also attracted (Reubens *et al.*, 2010). Furthermore, Belgian wind farms are closed to fisheries and may thereby act as protected areas. One of the major reasons causing fish to aggregate around such artificial structures and reefs emerging from the seafloor is the provision of food through the development of a species-rich hard substratum epifauna community (Reubens *et al.*, 2010). The systematic monitoring of this new biofouling community in the area is therefore an important factor to take into account when researching the biodiversity within wind farms.

The establishment of a biofouling community usually follows a clear successional development: the new structures will be gradually colonized by a number of species. These organisms will each influence the environment in a species-specific way, as such preventing other organisms to get established (i.e. inhibition) or creating the right circumstances for other species to join in (i.e. facilitation) (Connell & Slatyer, 1977). Consequently, the number of individuals of each species in the community will change and gradually new species will arrive that may progressively replace the first inhabitants. This long term process is known as ecological succession.

It is expected that differences in the nature of the substratum will create different (initial) conditions for the development of a biofouling layer, likely influencing the next colonization steps. On the other hand, the two wind farms are located at different distances from the shore (~25 km for C-Power, ~35 km for Belwind). Van den Eynde *et al.* (2010) showed that higher levels of turbidity are reached in winter at the Goote Bank, nearby the Thorntonbank (C-Power farm), compared to the Bligh Bank (Belwind farm). No data has yet been processed on the Thorntonbank itself (Michael Fettweis, pers. comm.). The C-Power site is thus expected to experience a more pronounced influence of coastal waters than the more offshore located Belwind farm though this may be weak. However, parameters such as the verticality of the substratum, its nature, or slight differences in environmental drivers are likely to drive site-specific differences, as was observed on some shipwrecks of the area (Zintzen, 2007). Furthermore, the installation of wind turbines was not carried out simultaneously at the two wind farms: the C-Power gravity-based foundations were installed by mid-spring 2008, while the steel foundations of Belwind were installed by early winter 2009-2010. Yearly variations in the benthic composition of the zooplankton (larvae and dispersing adults) are expected to strongly influence the initial species composition independently from other site-specific features.

In this contribution, the species composition of the subtidal zone (circa littoral) of selected foundations is studied through time in the two aforementioned wind farms. The analysis aims at tentatively detecting site-specific differences and disentangling the influence of the various environmental parameters driving the fouling process. We hypothesize that, even though the nature of the substratum, geographic position and installation time will drive site-specific differences, the faunistic patterns will be dominated by a common pool of species that also dominates on the surrounding natural and artificial hard substrata of the area. We further discuss the species composition and discuss the occurrence of introduced species.

3.2. Material and methods

3.2.1. Study area

In late spring 2008, the first six concrete foundations of the C-Power wind farm were installed on the Thornton bank, some 30 km off the Belgian coast. Between September 2009 and February 2010, 56 steel monopile foundations were installed at the Belwind wind farm on the Bligh Bank. Both banks belong to the Zeeland Banks system (Cattrijsse & Vincx, 2001). Local water depth within the wind farms ranges from 7 - 30 m and the surrounding soft sediment seabed is composed of medium sand (mean median grain size: between 350 and 500 µm (Coates *et al.*, 2010). The average residual water transport is oriented to the northeast (Ruddick & Lacroix, 2006).

The two offshore wind farms are situated along the eastern border of the BPNS, relatively close to each other (about 25 km) and almost equidistant from the gravel grounds of the Hinder banks (about 15 km; see Houziaux *et al.*, 2008). Various shipwrecks are found in the area (see Zintzen, 2007).

3.2.2. Sample collection and processing

A monitoring programme was set up to sample the hard substrata associated with the wind turbines (Kerckhof *et al.*, 2008). The first sampling took place in late summer 2008 at C-Power and in winter 2010 at Belwind (Figure 1). Samples were collected from a selected set of wind turbines: wind turbines D5 and D6 (GBF) (Figure 2) at the C-Power farm and wind turbines C2, B8 and C8 (steel monopile) at the Belwind farm (see Brabant *et al.*, 2011 for the geographic positions of the considered wind turbines).

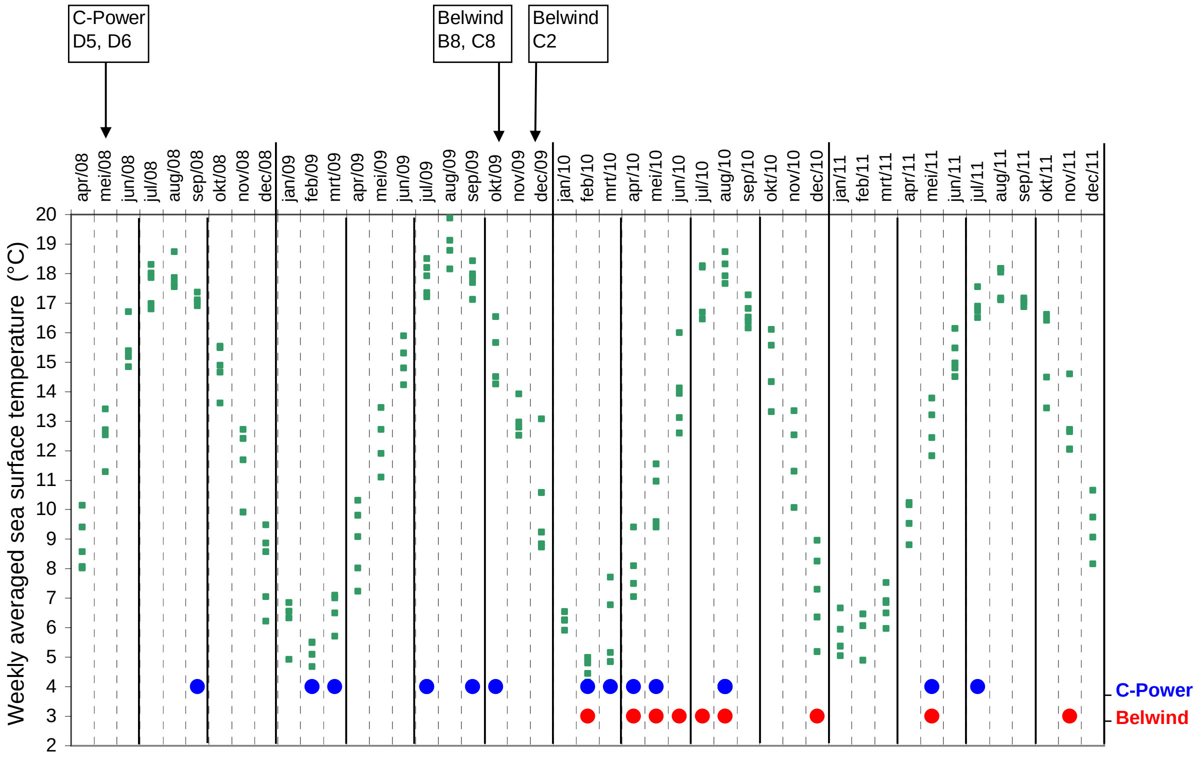


Figure 1. Temporal distribution of sampling occasions and weekly-averaged temperature at the C-Power (blue bullets) and Belwind (red bullets) wind farms, grouped on a monthly basis. Upper arrows indicate the installation time of the considered wind turbines.

Figure 2. Concrete C-Power gravity based foundation (D5) with epifouling of blue mussels *Mytilus edulis* attached to a piece of rope, two colour forms of the sea-anemone *Metridium senile*, the white calcareous tubes of *Pomatoceros triqueter*, sea urchins *Psammechinus miliaris*, sea stars *Asterias rubens* and empty and living specimens of the barnacle *Balanus perforatus*.

Subtidal samples were collected by scraping the fouling organisms with a putty knife from a sampling surface area of 0.25 m x 0.25 m. All scraped material was collected in plastic bags that were sealed under water and transported to the laboratory for processing (entailed fixation in 5% formaldehyde – seawater solution, sieving (1 mm mesh sieve), sorting and preservation in 75% ethanol), identification and quantification. Only the fraction >1 mm was considered in this study. Two to six replicates were collected during each sampling event.

Organisms were identified to the lowest possible taxonomic level and, where possible, counted. Densities were expressed as the number of individuals per m². Identifications were based on the most recent systematic literature and we followed the World Register of Marine Species (WoRMS) for the nomenclature and taxonomy at species level. Video footage collected by the divers was used to determine to what extent the scrape samples represent the actual fauna and to identify a number of rare, large and/or mobile invertebrate species that are otherwise not (adequately) represented in the scrape samples. A total of 111 replicate scrape samples collected at 36 stations were considered for this analysis.

3.2.3. Data processing

Prior to analysis and depending on the taxonomic level of identifications as well as their accuracy, some taxa were lumped to reach consistency in taxonomic resolution throughout the data. Records of skeletal parts, tubes, eggs, spat and larvae were removed from the data set to focus only on juveniles and adults of macrobenthic species that were alive at the time of sampling, the counts of which were summed. For colonial sessile species, which cannot be enumerated otherwise than with semi-quantitative estimates (e.g. SACFOR scale; Connor & Hiscock, 1996), records were converted to presence/absence data because not all data could yet be appropriately processed. The species were flagged according to their enumeration mode (two values: density or presence) and their living habits (two values: sessile or mobile) for further data selection and aggregation.

The seasons were arbitrarily defined depending on the yearly temperature pattern (Figure 1), with winter considered to comprise January, February and March. Sea surface temperature (SST) data were acquired from Bundesamt für Seeschifffahrt und Hydrographie, (Germany; see Loewe, 2003). Weekly-averaged values computed from a variety of field measurements (data source: http://www.bsh.de/aktdat/mk/nordsee/Digdat/) were interpolated in the 5 x 5 km grid of the MIRO&CO-3D model (Lacroix *et al.*, 2007) to obtain average values representative for the Belwind and C-Power wind farms (Figure 1).

As the installation of the wind turbines did not take place simultaneously in the two farms, two time lines were created to chronologically analyze species composition: one time line of elapsed days since installation for each site and one calendar-based time line starting on April 1, 2008 (early spring; Figure 1).

3.2.4. Analysis strategy

Univariate as well as multivariate (statistical) analyses were carried out on replicate samples for various subunits of the data set: sessile species (presence/absence), mobile species (densities) and all enumerable species (densities). One sample was removed from the data-set due to out-of-range density (~10^6 ind/m²) of the amphipod *Jassa herdmani*, apparently linked to the lower sampling depth of this sample (6 m instead of 15 m). The frequency of occurrence of the species in the samples was also considered. For comparison purposes, the C-Power data were further reduced to 800 days (calendar-based time line) and 510 days (time elapsed after installation).

Presence/absence transformation of the whole species range was not done since only few colonial species are frequent. Several univariate diversity indices were computed on a replicate sample basis (species richness, S; total abundance, N; Shannon-Wiener evenness based on log10; taxonomic breadth based on presence-absence data; taxonomic diversity based on density-weighed data) and were averaged per season and year. Multivariate analysis was carried out with the PRIMER-E software (version 6; Clarke and Warwick, 2001; Clarke and Gorley, 2006). The Bray-Curtis similarity matrix was computed on fourth-root transformed densities or on presence-absence data and similarities were plotted through multidimensional scaling (MDS). The same analyses were repeated after removal of the super-abundant *J. herdmani*. Cluster tree analysis with SIMPROF permutation test, as well as the SIMPER procedure were applied to identify species most contributing to within-group similarities. An ANOSIM permutation test was further carried out to detect significant differences between samples on the basis of wind farm, season and season + year.

A principal component analysis (PCA) was used to determine the main structuring variables in the datasets (ter Braak and Prentice, 1988) and was performed on a selection of species (only species occurring in more than 25% of all samples from both farms) to tentatively track the chronological path of species compositions within samples through seasons and years on the basis of sampling event-averaged centroids for PCs 1 and 2.

3.3. Results

3.3.1. General faunistic composition: taxonomic abundance and richness

We considered a total of 78 taxa for analysis of the scrape samples. None of these were new to the BPNS. Eight taxa were exclusively present at Belwind and 29 taxa exclusively at C-Power, while 41 taxa were shared. The taxon richness was lower at Belwind, with a total of 49 taxa compared to 70 observed at C-Power (64 after 800 days). However, it is equally distributed at the Ordo level in both farms (Figure 3). About 50% of all species were either polychaetes or decapods.

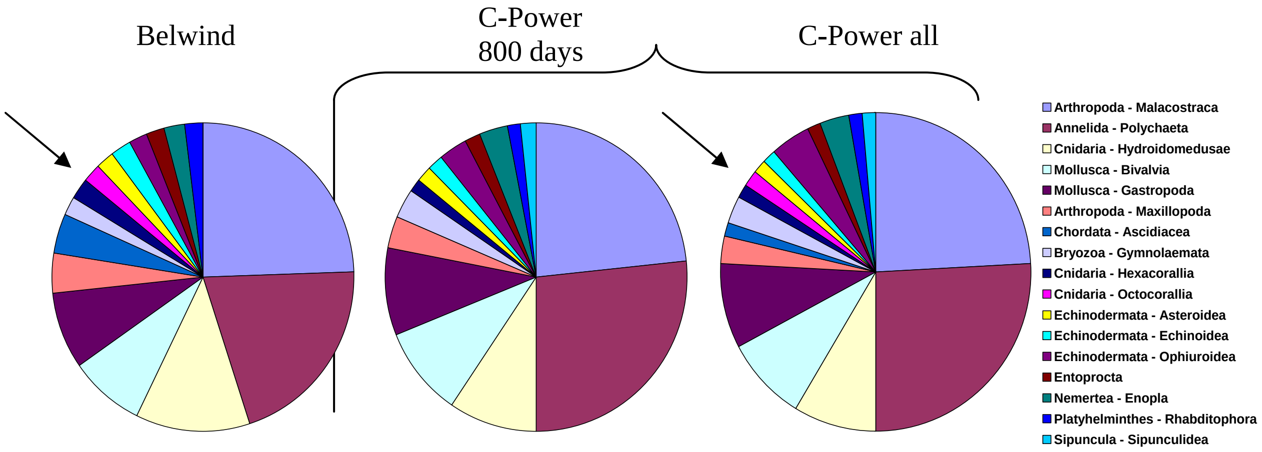


Figure 3: Relative taxon richness in the two wind farms, including the first 800 days at C-Power. Hexacorallia, Chordata, Entoprocta, Nemertea, Platyhelminthes, Sipuncula were lumped at the level of Ordo and their taxon richness most likely is underestimated. The arrows point to the octocorallian *Alcyonium digitatum*.

The samples from both wind farms displayed a wide taxonomic breadth but were strongly numerically dominated by one superabundant species, the amphipod *J. herdmani*, which caused the group Crustacea - Malacostraca to represent on average 67 and 76% of the enumerable sample content at Belwind and C-Power, respectively (Figure 4). When this species was removed, the species assemblage displayed a much larger proportion of barnacles (Arthropoda - Maxillopoda) at Belwind, representing about 50% of enumerable species, even when the data set was reduced to the first 800 days at C-Power. In the latter site, mobile crustaceans still numerically dominated the assemblages, while barnacles rapidly declined. Noticeably, the relative abundance of starfish (mostly represented by *Asterias rubens*) strongly increased in spring 2011. Octocorallians were represented by small colonies of dead man’s fingers *Alcyonium digitatum*, that appeared at both farms at the same moment.

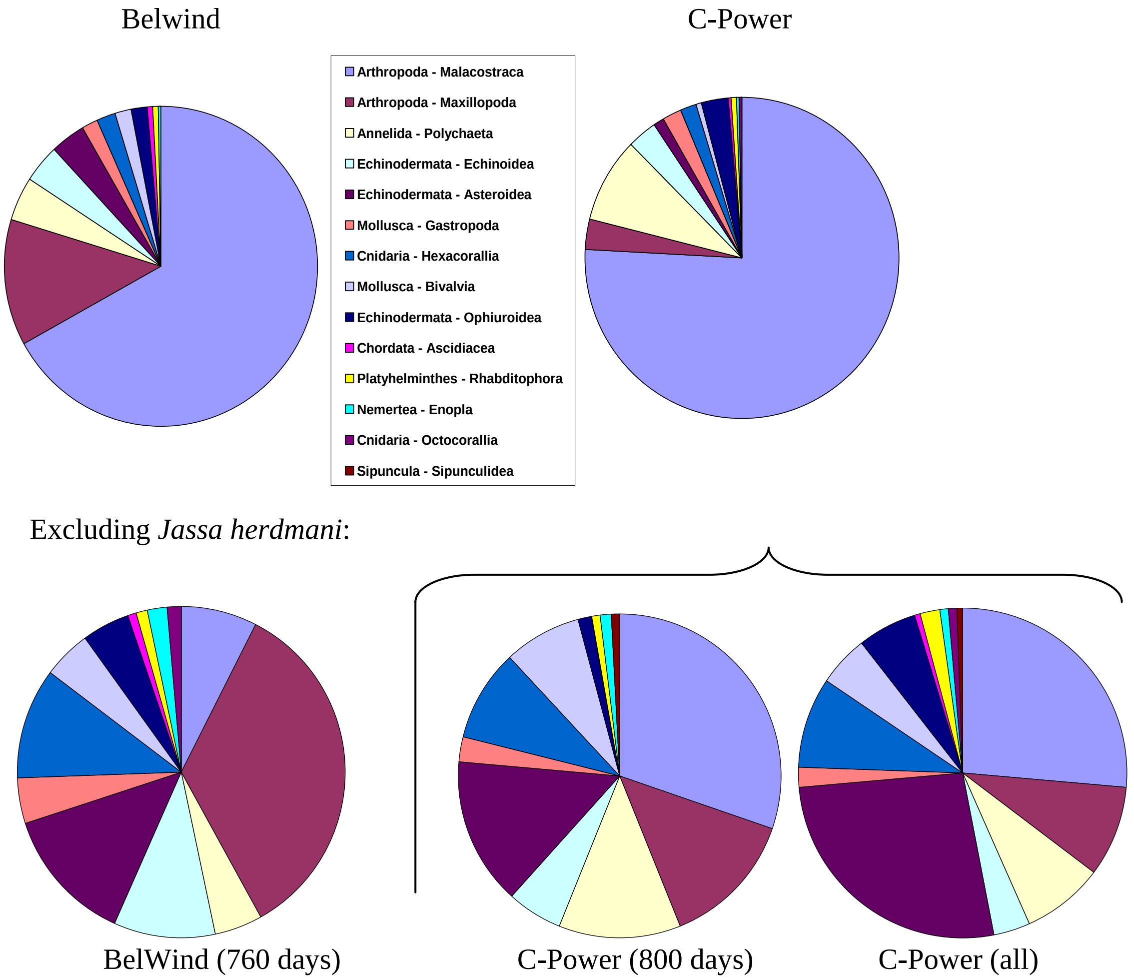


Figure 4. Above: relative abundance of enumerable taxa in the samples collected at Belwind and C-Power. Below: relative abundance of taxa after removal of the superabundant amphipod *Jassa herdmani* at Belwind (overall period: 760 days since installation) and C-Power (800 days since installation and overall period 2008-2011).

3.3.2. Temporal evolution of the biofouling community at both wind farms

3.3.2.1. Univariate diversity indices

Species richness displayed similar trends at both farms during the first year after installation: in the first summer, a high number of taxa colonized the newly available substratum (Figure 5). This happened at C-Power in summer 2008 and at Belwind in spring and summer 2010. However, the first winter sampling at Belwind, eleven weeks after installation, yielded no macro fauna. In both cases, a very fast colonization of the substratum as well as a large variability in taxon richness is observed. The richness dropped in the first fall and winter, reflecting winter mortalities, and increased again in the next spring and summer as new recruits colonize the substratum.

During the second fall, the average taxon richness abruptly decreased at Belwind (from 12 to 5 taxa/sample), while at C-Power it remained relatively constant after fall 2009 (~18 taxa/sample). When split into their mobile and sessile components, the situations differed more markedly: at C-Power the number of sessile species quickly stabilized and did not follow the initial oscillation displayed by the more numerous mobile species. The richness of sessile taxa varied more at Belwind, where it showed a dramatic drop in the second fall (2011). The average richness of sessile taxa is low at the two locations and ranged from 2 (Belwind – fall 2011) to 12 taxa (C-Power - summer 2008).

In summer 2010, the two wind farms had very similar levels of taxon richness (Figure 6), even though the communities were at different stages. Belwind generally had lower levels compared to C-Power, whatever the time elapsed since wind turbine installation, although the variability among the individual samples was important. The taxonomic breadth (sDelta+; not shown) displayed exactly the same trends as the species richness.

When abundances are taken into account (total density; Shannon-Wiener evenness, H'), the amphipod *J. herdmani* exerted a strong influence (Figure 6) by its numerical dominance. The species was virtually absent from the first samples at C-Power, although juveniles were observed in the fine fraction (< 1 mm). Afterwards, *J. herdmani* densities always numerically dominated the rest of the species. At Belwind, the species had the opportunity to colonize the substratum in the first spring and densities comparable to that observed at C-Power were already measured in summer 2010.

When *J. herdmani* was removed, the total density of benthic species showed a similar increasing trend at C-Power except in fall 2009 and winter 2010. Within these periods, the population size of *J. herdmani* kept increasing while the total density of other species decreased, causing the Shannon-Wiener evenness to reach a minimum value. However, after removal of *J. herdmani* from the index the evenness remained fairly stable after the first winter at both farms, even in winter, indicative of a more even distribution of the relative abundances of the other species. The total densities of other species were lower at Belwind except during the first summer, but the Shannon-Wiener evenness was similar.

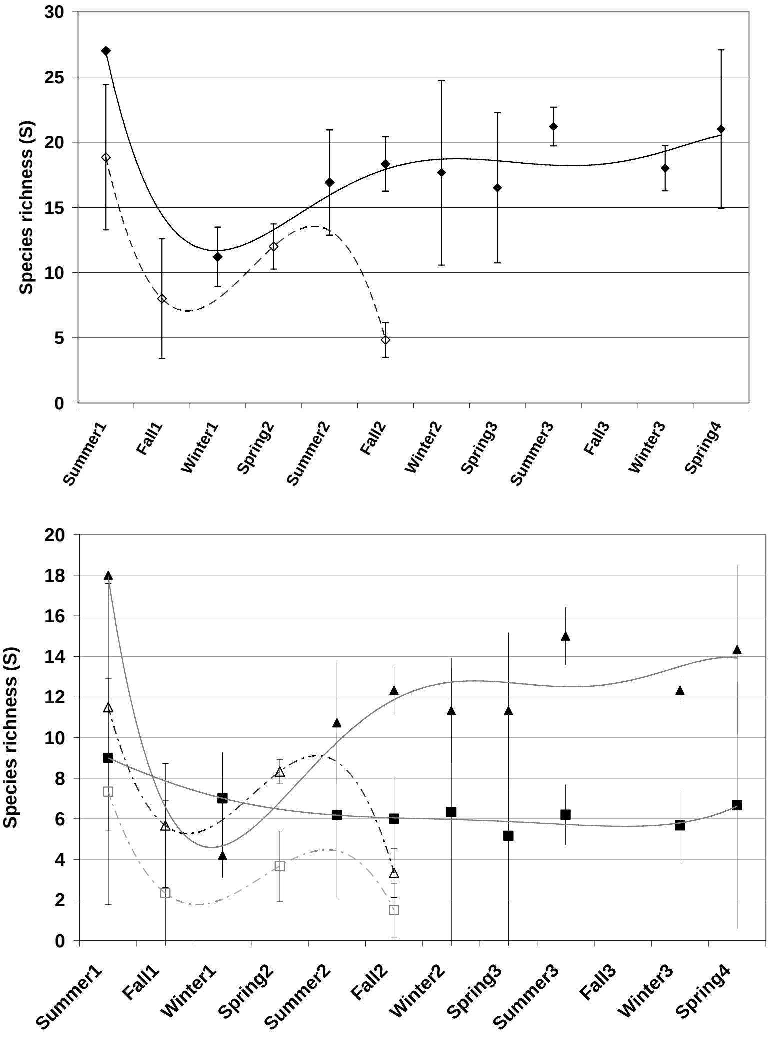


Figure 5. Temporal evolution of species richness since wind turbine installation in the two farms. Above: total species richness; below: sessile and mobile species richness. Open symbols: Belwind; Plain symbols: C-Power. Squares: sessile species; triangles: mobile species. Error bars are standard deviations.

Overall, when not overly influenced by *J. herdmani* abundance, the diversity indices point to a systematically lower biological diversity at the Belwind farm. This difference is further illustrated by a systematically lower frequency of occurrence of individual taxa at the Belwind farm, even when the C-Power data set was reduced to the first 800 days since installation. However, when the dataset was reduced to 510 days at C-Power (Figure 7), the taxon richness was more similar (54 taxa at Belwind versus 49 taxa at C-Power).

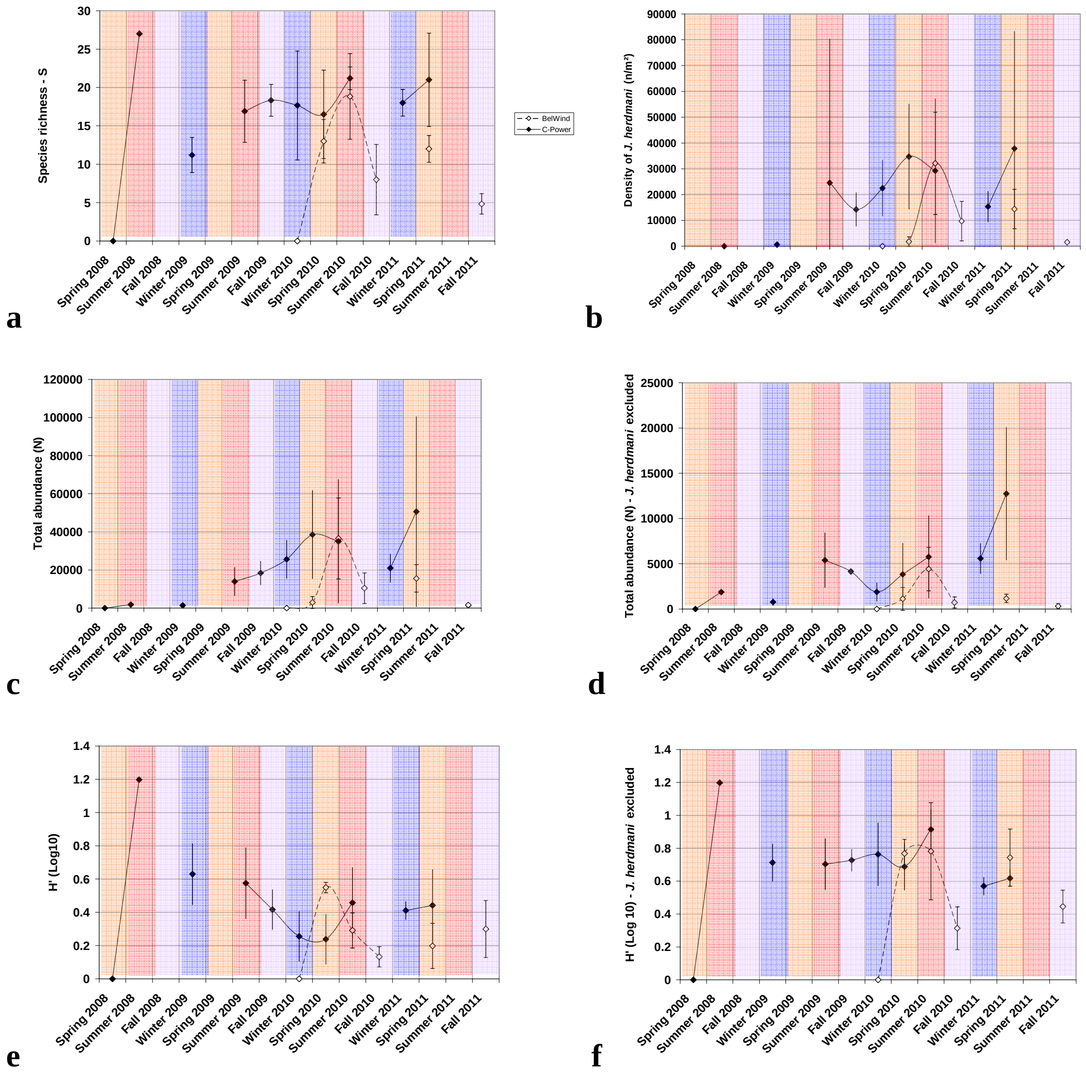


Figure 6. Temporal evolution of diversity indices averaged per wind farm, season and year since spring 2008; (a) taxon richness, (b) density of the amphipod *Jassa herdmani*, (c) total abundance, (d) total abundance, *J. herdmani* excluded, (e) Shannon-Wiener evenness ($\log_{10}$), (f) Shannon-Wiener evenness, *J. herdmani* excluded. Error bars are standard deviations.

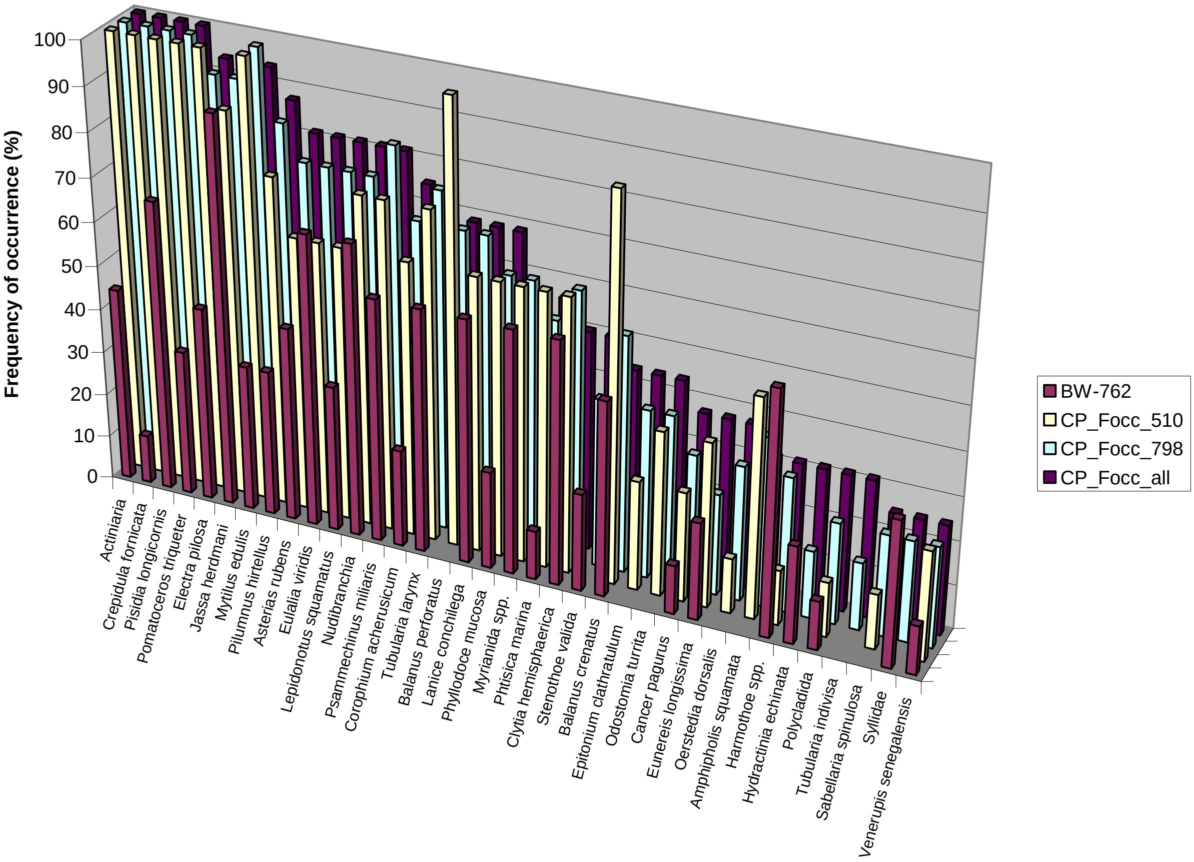


Figure 7. Frequency of occurrence of the species found in more than 50% of the samples at C-Power (510, 800 days since installation and overall) and their occurrence at Belwind.

3.3.2.2. Temporal trends in species composition

Over time, a progressive increase of the total density takes place as well as an increase of the *J. herdmani* population. However, as shown in Figures 5 and 6, the species richness remained relatively constant after the second summer at C-Power. Contrary to Belwind, some coastal soft bottom species were present in the C-Power assemblage. Of the 29 species found exclusively at C-Power, three species, *Abra alba* and *Mysella (Kurtiella) bidentata* (two bivalves) and *Pectinaria koreni* (a polychaete worm) are normally exclusively found in coastal muddy sands. They are among the numerically dominant species of the "*Abra alba* community" (Van Hoey *et al.*, 2004, 2005).

At the C-Power wind farm, some of the species that colonized the substratum in spring/summer 2008 remained present in all samples after summer 2009, indicating that they have established permanent populations (Figure 8). The amphipod *Corophium acherusicum* became particularly abundant in 2010 and 2011, even in winter, while it remained rare at Belwind. Summer 2010 coincided with the early colonization phase at Belwind, characterized by a peak of total density and species richness and a larger abundance of a typical pioneer species, the barnacle *Balanus crenatus*.

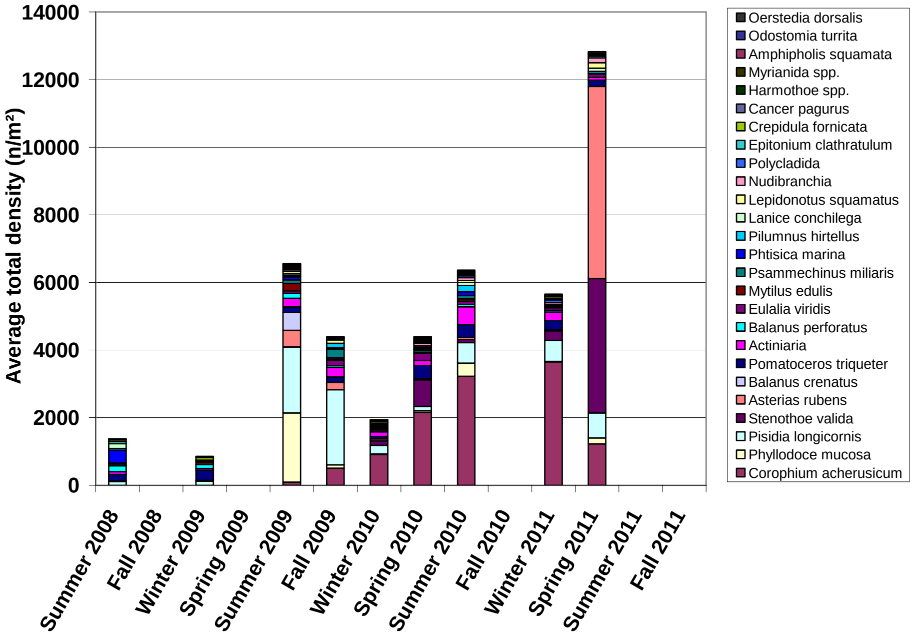


Figure 8a. Trends in densities of species that occur in more than 25% of the stations, excluding the superabundant *J. herdmani*, at C-Power and Belwind

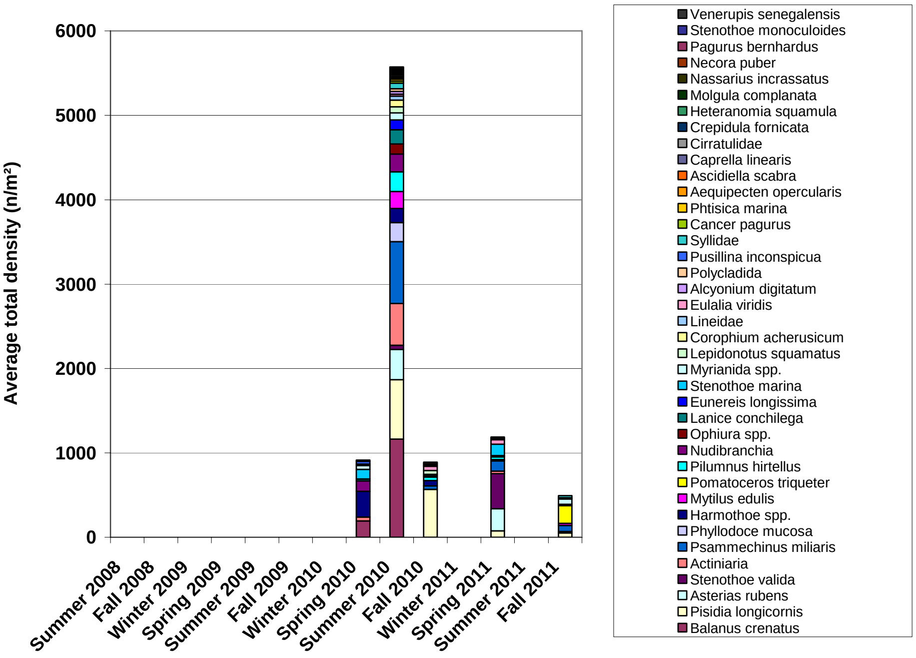


Figure 8b. Trends in densities of all species, excluding the superabundant *J. herdmani*, at C-Power and Belwind

In spring 2011, at C-Power, next to *J. herdmani* two species numerically dominated the samples, namely the starfish *Asterias rubens* and the amphipod *Stenothoe valida*. Exactly the same observation can be made at Belwind in this period, although their average density was lower.

The total number of colonial species was low at both wind farms (Figure 5). Most species were relatively rare, except for the hydroids *Tubularia* spp. and *Clytia hemisphaerica* and the bryozoan *Electra pilosa* (Figure 7). These species were early colonizers in both farms. Next to these colonial species, sea anemones (Actiniaria), represented by *Metridium senile*, *Sagartia troglodytes* and *Urticina felina*, were most frequent as well.

Fourteen sessile (colonial and solitary) species were found at both farms, nine species were found only at the C-Power farm, six only at Belwind. Most non-shared species are found only occasionally, except for *Balanus perforatus* and *Obelia* spp. that were frequently observed at C-Power. Of the shared species pool, ten species displayed consistent temporal distribution patterns at both farms, being found throughout the sampling period, at its beginning or at its end.

At both farms, the number of sessile taxa which could establish permanent populations is low, explaining the pattern of Figure 5. Most of the variability in sessile taxon richness could be explained by occasional species. In fall 2011, when diversity indices dropped at Belwind, only four of the most frequent species were yet observed in these samples (Actiniaria, *Pomatoceros triqueter*, *Lanice conchilega*, *C. hemisphaerica*) along with *Mytilus edulis*. Sea slugs (Nudibranchia) that mostly feed on sessile species were noticeably frequent (Figure 7) and found throughout the sampling period. The barnacle *B. perforatus* was observed throughout the sampling period, in contrast to its close relative *B. crenatus*, found only in the earliest stages of the colonization process at both farms.

The overall taxon composition is analyzed focusing on all enumerable species (sessile and mobile). The non-metric MDS ordination of the Bray-Curtis similarity matrix shows a separation between the samples of both farms (Figure 9). The superabundant *J. herdmani*, which masks patterns explained by other taxon's densities, is a characteristic species of almost all samples. Obviously, when *J. herdmani* is excluded from the data set, a larger dispersion is observed, but the relative positions of samples in the MDS plot are fairly conserved. This species hence does not drive the multivariate pattern but is clearly installing permanent and superabundant populations. It can be removed from the data set to investigate the evolution of the rest of the species assemblage.

Table 1: List of the sessile species thus far determined in the samples of Belwind and C-Power farms and their patterns of occurrence in the samples: "T": Throughout the sampled period; "E": Early colonist; "L": Late colonist; "O": occasional species; "-": absent.

Species	Belwind	C-Power
Actiniaria	T	T
Tubularia larynx	T	T
Lanice conchilega	T	T
Clytia hemisphaerica	T	E
Electra pilosa	T	T
Pomatoceros triqueter	L	T
Mytilus edulis	O	T
Aequipecten opercularis	E	E
Balanus crenatus	E	E
Balanus perforatus	-	T
Alcyonium digitatum	E	O
Obelia sp.	-	E
Molgula complanata	O	O
Pedicellina nutans	O	O
Heteranomia squamula	O	O
Venerupis senegalensis	O	O
Abra alba	-	O
Callopora dumerilii	-	O
Chaetopterus variopedatus	-	O
Mysella (Kurtiella) bidentata	-	O
Pectinaria koreni	-	O
Sabellaria spinulosa	-	O
Tubularia indivisa	-	O
Ascidiella scabra	O	-
Hydrallmania falcata	O	-
Pusillina inconspicua	O	-
Sarsia tubulosa	O	-
Sertularia cupressina	O	-
Verruca stroemia	O	-

The grouping of the C-Power samples after summer 2009 confirmed the trend towards a more similar species composition than the early samples. At Belwind, the spatial variability induced by sampling different foundations seems to play a role in the dispersion of the data. The species compositions are different between years 1 (2010) and 2 (2011). A similar distinction was found for C-Power between 2008 and 2009. The centre of the MDS plot coincides largely with years 2010 and 2011 at C-Power. An ANOSIM procedure resulted in very highly significant discrimination (p<0.001) between samples on the basis of farm, season as well as season + year. A cluster analysis of species compositions at C-Power (not shown) yielded clusters which indeed largely correspond to samples grouped by year, season and farm.

The distribution of the samples of C-Power in the multivariate space seemed to be driven by a seasonal oscillation and a yearly evolution, but these trends were yet unclear. A principal component analysis (PCA) on a selection of species (species occurring in more than 25% of the stations) yielded a seemingly clearer trajectory of samples through time and seasons, but less than 50% of the variance was explained by the two or even three first principle components. In such case, caution is advised by Clarke and Warwick (2001) in data interpretation, especially if the pattern is not conserved in the non-metric Bray-Curtis similarity matrix. This poor result indicates that at C-Power species are still subject to an important turnover after four years, even though the overall species richness stabilized after the second summer (Figures 5 and 6). In addition, the replicate samples display some variability (Figure 9), pointing to a high patchiness of the species composition.

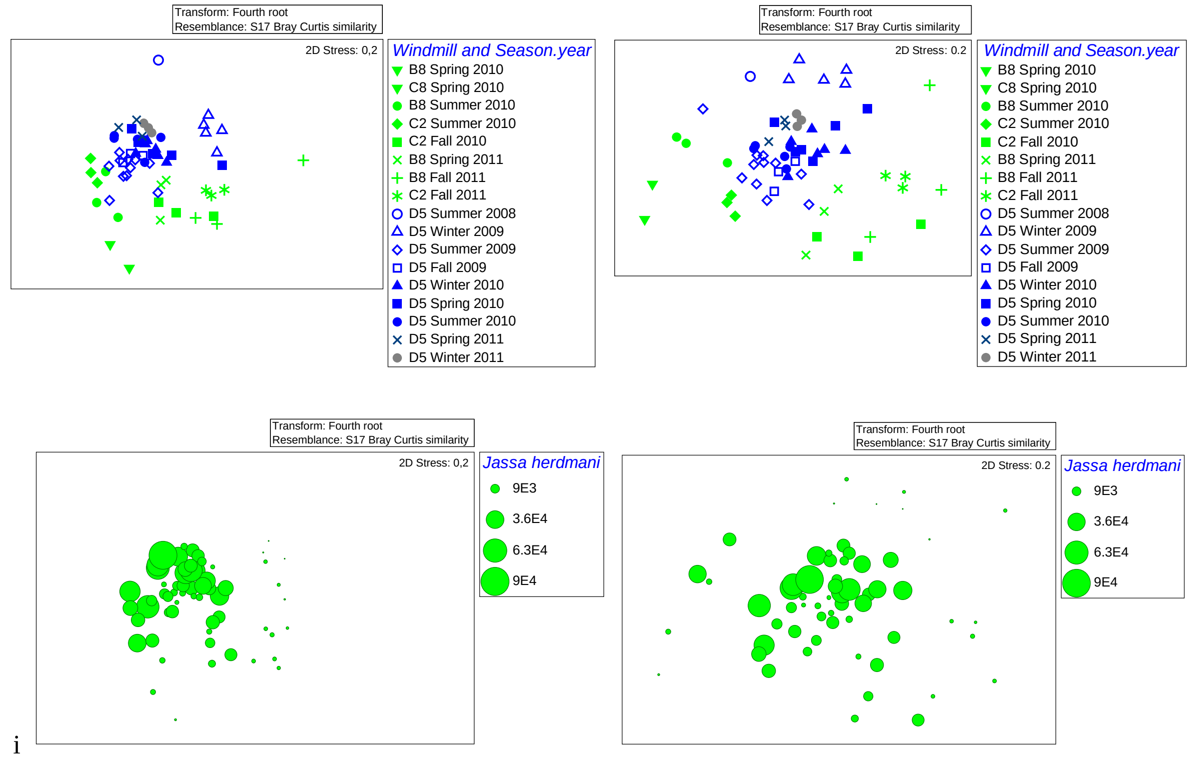


Figure 9. MDS plots of the Bray-Curtis similarity matrix calculated between samples based on densities of enumerable species (after fourth root transformation). Left: including *Jassa herdmani*; Right: excluding *J. herdmani*. In both cases, scaled bubbles were superimposed for (i) abundance of *J. herdmani* in the samples, (ii) total abundance of enumerable species N (excluding *J. herdmani*) and (iii) total taxon richness S.

Figure 9. Continued

3.4. Discussion

3.4.1. General colonization patterns

As could be expected (e.g. Horn, 1974; Connell & Slatyer, 1977), both farms showed a high diversity of fouling organisms during the early settlement phase, despite the fact that the investigated wind turbines were installed at different moments of the year (i.e. winter and late spring). The two wind farms display differences in the frequency of occurrence of some species, even when the C-Power data were reduced to the first 510 and 800 days since installation to account for a comparable time period or number of summers. The numerically dominant species are the same and the overall taxonomic composition is similar, reflecting the background species composition.

The concrete foundations of C- Power, installed by late spring, were colonized by species that settle in late summer and early autumn such as the barnacle *B. perforatus*. Nevertheless, the highest species richness was measured during the first summer in both cases. Many of these species did not survive the winter. In the next spring and summer, the remnant species pool was further altered by new settlers. The data of summer 2010 show that many new spring settlers are the same in both farms, indicating that the species pool of potential species is shared at least to a large extent.

All of the recorded species are typical colonisers of hard substrates and are regularly found on man made surfaces in Belgian waters (Zintzen, 2007) and elsewhere (Bouma & Lengkeek, 2009; Leonhard & Pedersen, 2006; van Moorsel *et al*. 1991; van Moorsel & Waardenburg, 2001). Below the infralitoral mussel zone, the foundations became dominated by echinoderms, anemones (predominately *Metridium senile*) barnacles, hydroids and the tube forming worm, *P. triqueter*. This community occupied the entire surface of the monopiles from the zone below the mussels to the sea floor, indicating that fouling organisms display a wide bathymetric tolerance. A similar habitat and zonation pattern in the subtidal has been reported on artificial hard substrata in the intertidal zone and on other wind farms in the North Sea (e.g. EMU, 2008; Whomersley & Picken, 2003; Joschko *et al*., 2008; Bouma & Lengkeek, 2009; Leonhard & Pedersen, 2006).

The numerically dominant taxa are those found abundantly on other hard substrata of the BPNS thus far studied (shipwrecks: Zintzen, 2007; gravels: Houziaux *et al.,* 2008). Compared to the potential species pool of the area, the sessile component on the foundations, i.e. the vertical component of both wind farms appears to be impoverished. Sponges, branching bryozoans, tunicates, some tubicolous polychaetes as well as many branching hydroid species have not been found or only in low numbers. The data indicate that only few species were able to install permanent populations, while most other species are yet occasional in the samples. Noticeably, the dead man's fingers *A. digitatum*, abundant on the nearby gravels but mostly as tiny colonies, appeared in 2010 at both wind farms, while it was absent at C-Power in the previous years. Some of these "missing" taxa can thus expected to show up in the next years as the communities evolve to a more mature stage, provided that they encounter suitable environmental conditions, room for settlement, as well as a predation pressure that is reasonably low.

However, our data indicates that there is still a large turnover of species and the whole fouling assemblage is still immature. We would expect the initial fauna of opportunistic species (r-strategists) to gradually be replaced by less opportunistic and more long-lived, slowly reproducing species (K-strategists). It is however possible that this will never happen and that these communities will remain in an improvised state as has been observed on other artificial hard substrates, such as wrecks (Hiscock *et al.,* 2010; Zintzen, 2007). This would lead to a community that is distinctly different to natural stony reefs. In such case, the artificial hard substrates provided by the wind farms cannot be considered a replacement for damaged or destroyed natural stony habitats.

The exact combination of species and their relative abundance varies between farms and even replicates and is dependent on environmental characteristics and upon species interactions. Not surprisingly, the multivariate analyses revealed a major influence of seasons and years on the temporal evolution of the species composition at C-Power over a longer period. At Belwind, it seems yet to be too early to analyze the species succession. However, C-Power results, obtained on various wind turbines, highlight the patchiness of the colonization process.

We attempted to attribute the fouling community to one of the biotopes of the JNCC Marine Habitat Classification (Connor *et al.*, 2004). Although for circalittoral rock some fouling communities are mentioned specially for new artificial hard substrata we were not able to find a full match. The *Metridium senile* biotope (CR.FCR.FouFa), typical for artificial hard substrates in the North Sea (Whomersley & Picken, 2003) and identified elsewhere on wind turbines in the North Sea, e.g. EMU, 2008; van Moorsel *et al.,* 1991 is the closest mach. Especially the abundance of echinoderms both *Psammechinus miliaris* and *A. rubens* differentiated our communities from the ones mentioned in the classification. The abundant presence of echinoderms seems to be a feature of the early phase of the colonization and is apparently also found on other wind farms in the southern North Sea (e.g. Bouma & Lengkeek, 2009). This situation may be a transitional one to the *Metridium* dominated biotope. Bare patches left after the passage of *Psammechinus* and *Asterias* will easily be colonized by anemones (mainly *M. senile*) and (tubes of) the small crustacean *J. herdmani.* There are indications that the dominance by echinoderms, especially sea urchins, takes place in the early colonization phase. *Psammechinus miliaris,* for instance, was very abundant on the wreck of the HMS Scilla in the first years, while it disappeared in a later successional stage (Hiscock *et al*., 2010). Characteristic is a high degree of patchiness, with bare areas and areas where one particular species dominated e.g. *Hydractina echinata* (Figure 10), *Tubularia* spp., *P. miliaris* or *M. senile*, a phenomenon also noted by van Moorsel *et al.* 1991 on wrecks off the Dutch coast and called by them "associations", with the *M. senile* association then being the final stage. Offshore shipwrecks might thus be indicative of a mature state of the subtidal epibiotic assemblages on such artificial substrates, namely a specific sub-assemblage of the local pool of hard substratum species. Noticeable, the calcareous tube building *P. triqueter*, despite being always present, did not manage to form reef like structures as is sometimes the case on other natural and artificial substrates.

A number of species typical for the soft sediment "*Abra alba* community" (Van Hoey *et al.*, 2004, 2005) such as *Abra alba* and *Mysella (Kurtiella) bidentata* (two bivalves) and *Pectinaria koreni* (a polychaete worm) were found exclusively at C-Power. These species are normally found living in coastal muddy sands. Two of these species have also been recorded on ship wrecks: *M. bidentata* (found on two wrecks in the BPNS) and *P. koreni* on wrecks in the BPNS and Dutch waters (Zintzen, 2007; van Moorsel et al., 1991). All three species have further been found in the fouling community on buoys (Kerckhof, unpublished data). They obviously take advantage of the presence of the muddy turf, largely as the result of the tube-building activities of the amphipods *Jassa* and *Corophium*, to settle.

Figure 10. Detail of the fouling assemblage on the C-Power D5 with a prominent colony of *Hydractinia echinata* (centre) overgrowing some empty barnacles Balanus perforatus. In the right part of the picture a slipper limpet *Crepidula fornicata*, partly covered by Jassa - turf is visible, two sea urchins *Psammechinus miliaris* and a young starfish *Asterias rubens*, in the left corner a young plumose sea anemone *Metridium senile* can be seen.

3.4.2. Species interactions and community dynamics

Predation is an important factor that controls the occurrence and succession of sessile species. Within the circalittoral zone information on biological interactions is not easily available, but sea urchins and starfish both have the potential to function in keystone roles. Some abundant grazers / predators on the sessile fauna, such as the sea urchin *P. miliaris* and the starfish *A. rubens*, may have prevented the dominance of barnacles – common in the very early phase – and mussels (never present in large numbers). *Psammechinus miliaris* is an omnivorous and voracious species feeding on a broad range of sessile organisms (Kelly & Cook, 2001) and it has been demonstrated that sea urchins can reduce the diversity of the biota by intense grazing (e.g. Mitchell *et al.*, 1983) or prevent the development of the normal invertebrate community (e.g. Sebens (1985 a, b). *Asterias rubens* also an omnivorous and voracious predator (Vevers, 1949) is also considered important in clearing space on rock by grazing barnacles, mussels and ascidians (Menge, 1982). *Asterias* reduced the cover of sessile species on settlement panels in Sweden to 20%, compared to 100% when they were excluded (Lundälv & Christie, 1986). Additionally, sea slugs (nudibranch molluscs) that were frequently and abundantly observed tend to be more specialized. Of the species thus far identified, *Facelina bostoniensis* and *Cuthona gymnota* are known to feed on *Tubularia* spp., *Eubranchus* spp. on *Obelia* spp., *Onchidoris bilamellata* on barnacles and *Onchidoris muricata* on *Electra pilos*a (Picton & Morrow, 1994). Also these species may hence play an important role in the evolution of the fouling community. In spring 2010 at the Belwind farm, for example we could witness the total disappearance of a dense *Tubularia indivisa* cover under the predatory activities of high densities of nudibranchs within no more than five weeks.

The yearly pool of species able to settle onto the artificial hard substrates is dependent upon the species' population dynamics on the surrounding hard substrata, substratum composition, available space for settlement, predation pressure and the prevailing hydrodynamics conditioning their dispersal paths. It seems that sessile species better resist winter conditions at C-Power compared to Belwind, where their richness tends to drop and frequencies of occurrence in the samples remain lower. It seems that the steel foundations are more prone to vibrations induced by wind and / or waves and current (as indicated by the increased underwater noise levels observed in Norro, 2011), a parameter which could exert influence on the settlement and survival of certain species on the vertical surface next to differences in substratum roughness (painted steel *versus* concrete). Wetzel et *al.,* (2011) also concluded that roughness was a key driver to species composition on artificial hard substrates. As the fouling process will continue over the next few years, this pattern will probably become more pronounced.

At the C-Power wind farm, the data collected over a period of four year showed that the mobile component is still subject to an important turnover, with an apparent alternation of numerically dominant species through time. The increased presence of *A. rubens* and *S. valida* in both wind farms during summer 2010 also points to a large influence of the fluctuation in the population dynamics of the species at a larger scale. The sessile component stays dominated by few species, while many other species appear to be occasional or rare. This pattern is also typical for the surrounding natural hard substrata, where most species are found in low densities (Francis Kerckhof, pers. obs.).

The tube dwelling amphipod *J. herdmani* is the most abundant species on both wind farms. Maximum densities of 200.000 ind/m² (only specimens larger than 1 mm), and even 1 million ind/m² in one shallow sample were observed at C-Power in July 2009. This high abundance was also observed in other studies dealing with artificial hard substrata in the southern North Sea such as shipwrecks (e.g. Zintzen, 2007) and other wind farms (e.g. Leonhard and Pedersen, 2006; Orejas *et al.*, 2005). In a German study, even higher densities were recorded (max. 1.317.045 ind/m²; Orejas *et al.,* 2005). This species was one of the earliest colonists and appears to be most successful in taking advantage of the new hard substrate habitats. Although it is a short-lived species, it has almost year-round reproduction and high fecundity (Nair and Anger, 1980). Consequently, juveniles were found in nearly all seasons. *Jassa herdmani* builds tubes and constructs mats that smother underlying species such as barnacles, in addition to making the surface less suitable for the settlement of other species. It seems that the abundance of this species may negatively impact the sessile component, but it was recorded abundantly at both locations. Another tube dwelling amphipod, *C. acherusicum*, was also common but appeared later in the succession. It is remarkable to find both tube dwelling amphipods living side by side on the same substratum. Despite the offshore location of the farms, under the influence of clear English Channel water (Kerckhof *et al.*, 2009), there seems to be enough fine sediment in the water for these species to build their tubes. On the other hand, *S. valida*, a free-living amphipod typically associated with *Tubularia* mats, was also abundantly observed in spring and summer 2011. These species, together with the also abundant porcelain crab *P. longicornis*, constitute an important food source for fish species which aggregate in these wind farms (Reubens *et al.,* 2010).

3.4.3. Taxonomic composition

A large diversity of phyla was able to settle down on these hard substrata explaining the large taxonomic breadth and huge species richness. We used 78 taxa in our analysis, but a total number of 85 invertebrate species was identified in our study. Zintzen (2006) found 99 macrofaunal invertebrate species in the scrape samples of the epifaunal assemblages of two shipwrecks of the BPNS. van Moorsel & Waardenburg (2001) recorded 44 macrofaunal invertebrates on an artificial reef off Noordwijk, The Netherlands, nine years after installation. On the FINO 1 research platform in the German Bight, a total of 44 species was found in the scrape samples during the first 2 years (Orejas et al., 2005). The number of species found on the recently installed wind turbines of the BPNS is thus remarkably high. However diversity remains much lower than that of natural gravels in the region such as the Hinder banks (Houziaux *et al.*, 2008) and the Dover Strait (Foveau *et al.*, 2008).

In addition to the conspicuous large species the biofouling community includes a diverse cryptofauna of small organisms such as nemerteans, polychaete worms and small (or juvenile) molluscs and crustaceans that live more or less hidden amongst the larger sessile fauna which also provides shelter and food for them. It is important to remind the reader that analyses were focused on the fraction larger than 1 mm. Some species reaching sizes slightly larger than 1 mm when mature such as *Pusilina inconspicua* or *Odostomia turrita*, as well as juveniles of larger species were regularly observed in the fraction <1 mm of the samples (Francis Kerckhof, unpubl. data). For these species, the real frequency of occurrence will thus be larger than measured within the larger size fraction. Secondly, in this study we did not address the fauna of the stones of the scour protection. The barnacle *Verruca stroemia*, for instance, seems more abundant on this substratum (Francis Kerckhof, unpubl. data). It can as well be expected that larger crustaceans which need shelters such as those provided by cobble patches (e.g. the lobster *Homarus gammarus* or the edible crab *Cancer pagurus*), may be found more abundantly there. A different assemblage is thus expected to be found on this part of the wind turbine infrastructure, due to the larger size of the three-dimensional structure.

Zintzen *et al.* (2008) showed that the communities associated to shipwrecks display differences in their community composition in function of prevailing environmental drivers. Thus, differences were found between wrecks located in coastal and offshore waters. The two wind farms are located very close to each other, and clearly under the predominant influence of English Channel water (Otto *et al.*, 2006; Zintzen, 2007; Kerckhof *et al.*, 2009). The C-Power wind farm, closer to the coast, can be expected to be more under influence of coastal waters, depending on the prevailing hydro-meteorological conditions (see Van den Eynde *et al.*, 2010). On the other hand, the weekly averaged temperatures (grouped per month in Figure 1) do systematically point to a larger temperature amplitude of about 0.5-1 °C in winter and summer at C-Power compared to Belwind. Environmental differences between the two farms do reflect an onshore-offshore gradient, which is nevertheless weak. The differences in species richness and frequency of occurrence are thus unlikely to originate from natural environmental differences.

3.4.4. Introduced species

Only one introduced species was found thus far in the subtidal zone namely the slipper limpet *Crepidula fornicata*. This is in contrast to the intertidal zone where the newly introduced hard substrata of the offshore wind farms proved to play an important role in the establishment and the expansion of the population size of non-indigenous species including introduced ones (Kerckhof *et al.*, 2011). *Crepidula fornicata* is present in the BPNS since 1911 (Adam & Leloup, 1934) and constitutes now an important member of the local fauna due to its ability to colonize both soft en hard substrata (Kerckhof *et al.*, 2007). Although *C. fornicata* does occur in the intertidal zone on groins along the Belgian coast (Francis Kerckhof, pers. obs.) it was not present in the intertidal samples of the wind farms studied (Kerckhof *et al.*, 2011). In the subtidal, it was amongst the first colonists at the C-Power wind farm (juveniles were already present in the first sampling in autumn 2008) and the species was subsequently present in most of the samples, often as large specimens and even chains of two individuals. However, *C. fornicata* was only found in one Belwind sample (one juvenile specimen in august 2010). The virtual absence of *C. fornicata* in the Belwind farm cannot be ascribed to the different substratum composition, i.e. concrete versus steel or the position further offshore of the Belwind wind farm because *Crepidula* has been observed abundantly on – even remote – offshore buoys and on ships' hulls (Kerckhof *et al.*, 2007; Francis Kerckhof pers. obs.).

3.5. Conclusions

As observed in many other studies, the biofouling of the bare substratum has been very fast in the two studied wind farms. The results on sessile species do suggest that concrete foundations may offer a better settlement surface for hard substratum species, but the variability of overall species composition (turnover) induced by seasonal oscillation and yearly varying environmental conditions and predation is high and prevents robust conclusions at this point in the succession.

On the other hand, the biofouling on the two wind farms exhibits direct influence by the same pool of species originating from the surrounding artificial and natural hard substrata. As for shipwrecks, specific features of the studied structures such as verticality, substratum composition or lower depths may explain specific patterns of taxonomic composition such as an impoverished sessile epifauna compared to the surrounding natural hard substrata. The predatory activities are an important factor controlling the succession of sessile species on the substratum.

The tube dwelling amphipod *J. herdmani*, and to a lesser extent its relatives *C. acherusicum* and *S. valida*, strongly take advantage of the newly available habitats. Together with the porcelain crab *P. longicornis*, these species have been shown by Reubens (2010) to be important prey items for the ichtyofauna that aggregates around the structures.

Apart from the slipper limpet *C. fornicata*, no introduced species have been observed thus far in the permanently submerged part of the wind turbine foundations. This observation contrasts with the intertidal and splash zone, where many introduced species were present in the biofouling community.

3.6. Acknowledgements

This research was financed by C-Power and Belwind, in fulfilment of the environmental monitoring requirements of their environmental permits. Field work could not have been completed without the help and smooth operation provided by the officers and crew of the RV Belgica, owned by the Belgian Ministry of Science Policy, and the RV Zeeleeuw, property of the Flemish government. The sampling could not have been completed without the help of (in alphabetical order): K. Deneudt, F. Francken, G. Jones, G. Lacroix, C. Mahieu, D. Marsham, F. Pasotti, R. Picavet, J. Pire, G. Rooms, A. Simon, H. Tourneur, M. Vanespen, I. Vosselman and V. Woit, M. Dekee, V. Morris, F. Tourneur and L. Verhaeghe are thanked for their contribution to sample analysis. R. Dekker (NIOZ) provided assistance in the identification of nudibranch species. G. Lacroix and J. Ozer (MUMM) are acknowledged for extracting and providing sea surface temperature data series.

3.7. References

Adam, W.; Leloup, E. (1934). Sur la présence du gastéropode *Crepidula fornicata* (Linné, 1758) sur la côte belge Bull. Mus. royal d'Hist. Nat. Belg./Med. Kon. Natuurhist. Mus. Belg. 10(45): 1-6

Bouma, S.; Lengkeek, W. (2009). Development of underwater flora- and fauna communities on hard substrates of the offshore windfarm Egmond aan Zee (OWEZ). Report Bureau Waardenburg nr 08-220.

Brabant, R., Degraer, S. & Rumes, B., (2011). Offshore wind energy development in the Belgian part of the North Sea & anticipated impacts: an update. *In:* Degraer, S., Brabant., R. & Rumes, B. (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences. Management Unit of the North Sea Mathematical Models. Marine Ecosystem Management Unit. Chapter 2: 9-16.

Cattrijsse, A.; Vincx, M. (2001). Biodiversity of the benthos and the avifauna of the Belgian coastal waters: summary of data collected between 1970 and 1998. Sustainable Management of the North Sea. Federal Office for Scientific, Technical and Cultural Affairs: Brussel, Belgium. 48 pp.

Clarke, K.R.; Warwick, R.M. (2001). Changes in marine communities: an approach to statistical analysis and interpretation. 2nd ed. PRIMER-E Ltd, Plymouth Marine Laboratory, UK, 172 p.

Clarke, K.R.; Gorley, R. N. (2006). PRIMER v6: PRIMER-E Ltd, Plymouth Marine Laboratory, UK.

Coates, D. & Vincx, M., (2010). Monitoring the effects of offshore wind farms on the soft substratum macrobenthos: Year-1 Bligh Bank and Year-2 Thorntonbank. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2010). Offshore wind farms in the Belgian part of the North Sea: Early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 7: 83-103.

Connell, S.D. (2001). Urban structures as marine habitats: an experimental comparison of the composition and abundance of subtidal epibiota among pilings, pontoons and rocky reefs. Marine Environmental Research 52: 115-125.

Connell, J.H.: Slatyer R.O. (1977). Mechanisms of succession in natural communities and their role in community stability and organization. The American Naturalist 111, No. 982. 1119- 1144

Connor, D.W.; Hiscock, K. (1996). Data collection methods (with Appendices 5 - 10). In: Marine Nature Conservation Review: rationale and methods, ed. by K. Hiscock, 51-65, 126- 158. Peterborough, Joint Nature Conservation Committee. (Coasts and seas of the United Kingdom. MNCR series.)

Connor, D.W.; Allen J. H.; Golding, N., Howell, K.L.; Lieberknecht, L.M.; Northen, K.; Reker, J.B. (2004). The Marine Habitat Classification for Britain and Ireland Version 04.05. Joint Nature Conservation Committee, Peterborough.

EMU (2008a). Kentish Flats Offshore Windfarm turbine foundation faunal colonisation Diving Survey. Report nr. 08/J/1/03/1034/0839/AMB/Nov 2008.

EN ISO 19493 (2007). Water quality - Guidance on marine biological surveys of hard-substrate communities. 32 pp.

Foveau, A., Desroy, N., Dewarumez, J.-M., Dauvin, J-C., Cabioch, L. (2008). Long-term changes in the sessile epifauna of the Dover Strait pebble community. Journal of Oceanography, Research and Data, 1: 1-11.

Van den Eynde, D., Brabant, R., Fettweis, M., Francken, F., Van Lancker, V., Sas, M. & Melotte, J. (2010) Monitoring of hydrodynamic and morphological changes at the C-Power and Belwind offshore windfarm sites – A synthesis. *In:* Degraer, S., Brabant, R. & Rumes, B. (Eds.) (2010) Offshore wind farms in the Belgian part of the North Sea: Early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter3: 19-36

Fettweis, M.; Nechad, B.; Van den Eynde, D.; Francken, F.; Van Lancker, V**.** (2010). Monitoring en modellering van het cohesieve sedimenttransport en evaluatie van de effecten op het mariene ecosysteem ten gevolge van bagger- en stortoperatie (MOMO): activiteitsrapport (1 januari 2010 - 30 juni 2010). Beheerseenheid van het Mathematisch Model van de Noordzee (BMM): Brussel. 32 + appendices pp

Fettweis, M.; Baeye, M.; Lee, B.J.; Chen, P.; Yu, J.C.S. (2012). Hydro-meteorological influences and multimodal suspended particle size distributions in the Belgian nearshore area (southern North Sea) Geo-Mar. Lett. 32(2): 123-137. dx.doi.org/10.1007/s00367-011-0266-7

Havermans, C.; De Broyer, C.; Mallefet, J.; Zintzen, V. (2007). Dispersal mechanisms in amphipods: a case study of Jassa herdmani (Crustacea, Amphipoda) in the North Sea Mar. Biol. (Berl.) 153(1): 83-89. dx.doi.org/10.1007/s00227-007-0788-8

Hiscock, K.; Sharrock, S.; Highfield, J.; Snelling, D. (2010). Colonization of an artificial reef in south-west England-ex-HMS 'Scylla'. J. Mar. Biol. Assoc., UK. 90(1): 69-94.

Hiscock, K.; Smith, A.; Jenkins, S.; Sewell, J.; Hawkins, S. (2005). Development of a hard substratum Benthic invertebrate Water Framework Directive compliant classification tool. Report to the Environment Agency and the Joint Nature Conservation Committee from the Marine Biological Association. Plymouth: Marine Biological Association. JNCC Contract F90-01-790. 54 pp + Annexes.

Houziaux, J.-S.; Kerckhof, F.; Degrendele, K.; Roche, M.F.; Norro, A. (2008). The Hinder banks: yet an important area for the Belgian marine biodiversity? Belgian Science Policy: Brussel. 248 pp

Horn, H.S. (1974). The Ecology of Secondary Succession Annual Review of Ecology and Systematics, 5: 25-37.

Joschko, T.J.; B.H., Buck, L.; Gutow A.; Schröder (2008). Colonisation of an artificial hard substrate by Mytilus edulis in the German Bight, Marine Biology Research, 4(5), 350-360.

Kelly, M.S.; Cook E.J.; (2001). The ecology of *Psammechinus miliaris*. In: Lawrence JM (ed) Edible sea urchins: biology and ecology. Elsevier Science, Amsterdam, p 217–224

Kerckhof, F.; Haelters, J.; Gollasch, S. (2007). Alien species in the marine and brackish ecosystem: the situation in Belgian waters Aquatic Invasions 2(3): 243-257.

Kerckhof, F.; Norro A.; Jacques T.G. (2009). Early colonisation of a concrete offshore windmill foundation by marine biofouling on the Thornton Bank (southern North Sea). *In*: Degraer, S. & Brabant, R. (Eds.) (2009). Offshore windfarms in the Belgian part of the North Sea: State of the art after two years of environmental monitoring. Royal Belgian Institute of Natural

Sciences, Management Unit of the North Sea Mathematical Models. Marine Ecosystem Management Unit. Chapter 4: 39-51.

Kerckhof, F.; Rumes, B.; Jacques, T.; Degraer, S.; Norro, A. (2010). Early development of the subtidal marine biofouling on a concrete offshore windmill foundation on the Thornton Bank (southern North Sea): first monitoring results Underwat. Technol. 29(3): 137-149.

Kerckhof, F., Rumes, B., Jacques, T., Degraer, S. & Norro, A. (2010). Early development of the subtidal marine biofouling on a concrete offshore windmill foundation on the Thornton Bank southern North Sea): first monitoring results. International Journal of the Society for Underwater Technology, Vol. 29, No 3, pp. 137-149.

Kerckhof, F.; Degraer S.; Norro, A. Rumes, B. (2011). Offshore intertidal hard substrata: a new habitat promoting non-indigenous species in the Southern North Sea: an exploratory study. *In:* Degraer, S., Brabant., R. & Rumes, B. (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences. Management Unit of the North Sea Mathematical Models. Marine Ecosystem Management Unit. Chapter 4: 27-37.

Lacroix, G.; Ruddick, K.; Park, Y.; Gypens, N.; Lancelot, C. (2007). Validation of the 3D biogeochemical model MIRO&CO with field nutrient and phytoplankton data and MERIS-derived surface chlorophyll a images. Journal of Marine Systems, 64 (1-4), p. 66-88.

Leonhard, S.B.; Pedersen J. (2006). Benthic communities at Horns Rev before, during and after construcytion of Horns Rev Offshore Windfarm. Final report 2005. Udarbejdet af Bio/consult as for ELSAM Engineering 96 pp.

Loewe, P. (2003). Weekly North Sea SST Analyses since (1968), in: Hydrographie, O.d.a.h.b. Bundesamt für Seeschifffahrt und Hydrographie, Hamburg (Ed.), D-20305 Hamburg, Germany.

Lundälv, T.; Christie, H. (1986). Comparative trends and ecological patterns of rocky subtidal communities in the Swedish and Norwegian Skagerrak area. Hydrobiologia 142, 71-80.

Menge, B.A. (1982). Effects of feeding on the environment: Asteroidea. In, Echinoderm Nutrition (eds M. Jangoux & J.M. Lawrence), Rotterdam: A.A. Balkema. 521-551p.

Mitchell, R.; Earll, R.C.; Dipper, F.A. (1983). Shallow sublittoral ecosystems in the Inner Hebrides. In: The natural environment of the Inner Hebrides, ed. by J.M. Boyd & D.R. Bowes, Proceedings of the Royal Society of Edinburgh. Series B: Biological Sciences, 83: 161-184.

Nair, K.K.C.; Anger, K. (1980). Seasonal variation in population structure and biochemical composition of *Jassa falcata* (Crustacea, Amphipoda) off the Island of Helgoland (North-Sea). Estuarine and Coastal Marine Science 11:505–513.

Norro, A.; Rumes, B.; Degraer, S. (2011). Characterisation of the operational noise, generated by offshore wind farms in the Belgian part of the North Sea. *In:* Degraer, S., Brabant., R. & Rumes, B. (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences. Management Unit of the North Sea Mathematical Models. Marine Ecosystem Management Unit. Chapter 3: 17-26.

Orejas, C.; Joschko, T.; Schröder, A.; Dierschke, J.; Exo, M.; Friedrich, E.; Hill, R.; Hüppop, O.; Pollehne, F.; Zettler, M. L.; Bochert, R. (2005). Ökologische Begleitforschung zur Windenergienutzung im Offshore-Bereich auf Forschungsplattformen in der Nord- und Ostsee (BeoFINO), AP2 Prozesse im Nahbereich der Piles Nordsee. 161 – 234.

Reubens, J.T.; Degraer, S.; Vincx, M. (2010). The importance of marine windfarms, as artificial hard substrates, for the ecology of the ichyofauna. *In*: Degraer, S., Brabant, R. & Rumes, B. (Eds.) (2010). Offshore windfarms in the Belgian part of the North Sea: Early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine Ecosystem Management Unit. Chapter 6: 69-82.

Sebens, K.P. (1985a). Community ecology of vertical rock walls in the Gulf of Maine, U.S.A.: Small-scale process and alternative community states. In, The Ecology of Rocky Coasts (eds P.G. Moore & R. Seed). pp 346-371. Sevenoaks: Hodder & Stoughton.

Sebens, K.P. (1985b). The ecology of the rocky subtidal zone. American Scientist 73, 548-557.

Petersen, J.K.; Malm, T. (2006). Offshore windmill farms: threats to or possibilities for the marine environment. AMBIO: A Journal of the Human Environment, 35: 75-80.

Picton, B.E.; Morrow, C.C. (1994). A field guide to the nudibranchs of the British Isles. IMMEL Publishing: London. 142 pp.

Ruddick, K.; Lacroix, G. (2006). Hydrodynamics and meteorology of the Belgian coastal zone, in: Rousseau, V. et al. (Ed.) (2006). Current status of eutrophication in the Belgian coastal zone. pp. 1-15.In: Rousseau, V.; Lancelot, C.; Cox, D. (Ed.) (2006). Current status of eutrophication in the Belgian coastal zone. Presses Universitaires de Bruxelles: Bruxelles. 122 pp

Schröder, A.; Orejas, C.; Joschko, T. (2006). Benthos in the vicinity of the piles: FINO 1 (North Sea) In Offshore Wind Energy. Research on Environmental Impacts pp. 185-200. Ed. by J. Köller, J. Köppel & W. Peters. Springer Verlag Heidelberg, Berlin. 371 pp.

ter Braak, C.J.F.; Prentice, I.C. (1988). A theory of gradient analysis. Advances in Ecological Research 18: 271-317.

Van Hoey, G.; Degraer, S; Vincx, M. (2004). Macrobenthic community structure of soft-bottom sediments at the Belgian Continental Shelf. Est Coast Shelf Sci 59: 601-615

Van Hoey, G.; Vincx, M.; Degraer, S. (2005). Small- to large-scale geographical patterns within the macrobenthic *Abra alba* community. Est Coast Shelf Sci 64(4): 751-763

van Moorsel, G.W.N.M., Waardenburg, H.W.; van der Horst, J. (1991). Het leven op en rond scheepswrakken en andere harde substraten in de Noordzee (1986 tot en met 1990) - een synthese. Bureau Waardenburg bv, Culemborg, Rapport 91.19, 49 pp

Vevers, H.G., (1949). The biology of *Asterias rubens* (L.): growth and reproduction. Journal of the Marine Biological Association of the United Kingdom, 28: 165-187.

van Moorsel, G.W.N.M.; Waardenburg, H.W. (2001). Kunstmatige riffen in de Noordzee in 2001. De status 9 jaar na aanleg. Bureau Waardenburg bv, Culemborg Rapport 01-071, 35 pp.

Wetzel, M.A.; Wiegmann M.; Koopa, J.H.E. (2011). The ecological potential of geotextiles in hydraulic engineering. Geotextiles and Geomembranes Vol. 29, 4, August 2011, Pages 440–446.

Whomersley, P.; Picken, G.B. (2003). Long-term dynamics of fouling communities found on offshore installations in the North Sea J. Mar. Biol. Ass. U.K. 83(5): 897-901

Zintzen, V. (2007). Biodiversity of shipwrecks from the Southern Bight of the North Sea. PhD Thesis. Université Catholique de Louvain/Institut Royal des Sciences Naturelles de Belgique: Louvain-la-Neuve, Belgium. 343.

Zintzen, V.; Norro, A.; Massin, C.; Mallefet, J. (2008). Spatial variability of epifaunal communities from artificial habitat: shipwrecks in the Southern Bight of the North Sea Est., Coast. and Shelf Sci. 76(2): 327-344. dx.doi.org/10.1016/j.ecss.2007.07.012

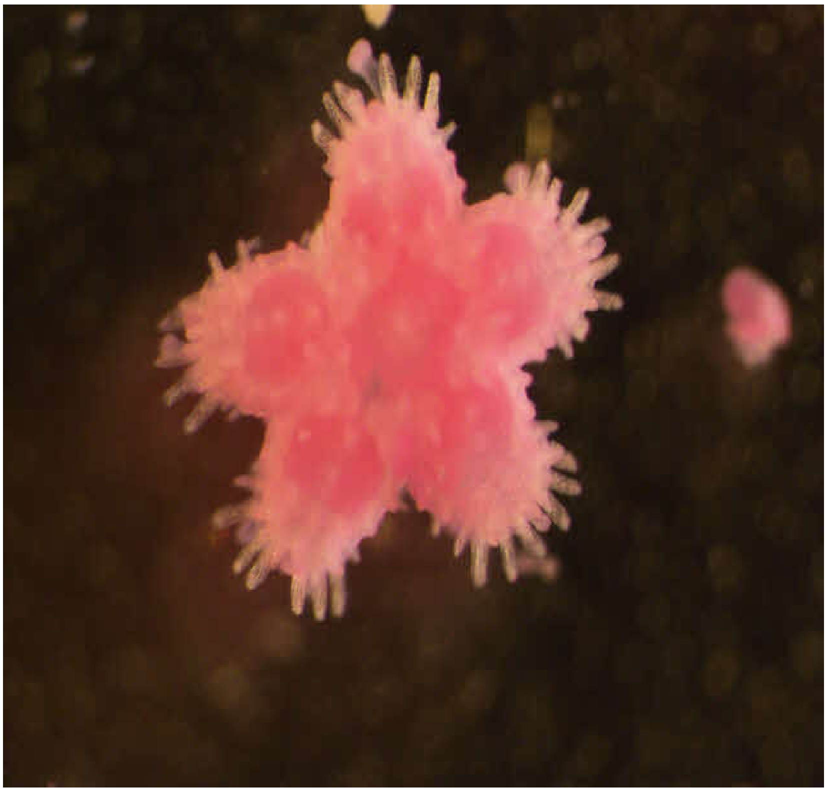
Chapter 4. Enrichment of the soft sediment macrobenthos around a gravity based foundation on the Thorntonbank

---

D. Coates[*], J. Vanaverbeke & M. Vincx

Ghent University, Department of Biology, Marine Biology Research Group, Krijgslaan 281(S8), Ghent

[*] *Corresponding author: Delphine.coates@ugent.be*



Juvenile starfish *Asterias rubens* Photo UGent / Marbiol

Abstract

Belgium initiated a large scale monitoring programme in 2005 to determine the effects of offshore wind farms on the soft sediment macrobenthos. Up till now, the natural, temporal variability of the macrobenthos was detected without any apparent effects from the foundations (Coates & Vincx, 2010; Reubens *et al.*, 2009). In 2011, the soft-sediment macrobenthos in close vicinity to one gravity based foundation on the Thorntonbank was analysed. Bottom samples taken at 15, 25, 50, 100 and 200 meters from the scour protection system of the foundation and in four gradients around the turbine (Southwest, Southeast, Northwest and Northeast), showed a clear distinction in densities. The median grain size showed a declining trend at stations closer to the turbine. An accumulation of juvenile starfish (Asteriidae juv.) together with the polychaetes *Spio* sp. and *Spiophanes bombyx* was obvious at stations <50 meters and mostly on the Southwest gradient. Lower current speeds and changing granulometric characteristics appear to be creating a substantial change in the macrobenthic community on the Southwest gradient. The soft sediment macrobenthic community around the foundation is highly dynamic and far from reaching an equilibrium.

Samenvatting

In 2005 werd een grootschalig monitoringsprogramma opgestart in België om de effecten van offshore windmolenparken op het zachte substraat macrobenthos. Tot nu toe werd de natuurlijke, temporele variabiliteit van het macrobenthos waargenomen zonder duidelijke effecten van de turbine zelf (Coates & Vincx, 2010; Reubens *et al.*, 2009). In 2011 werd het zacht substraat macrobenthos bemonsterd rondom één gravitaire fundering op de Thorntonbank. Een duidelijk verschil in densiteiten werd waargenomen in de bodemstalen genomen op 15, 25, 50, 100 en 200 meter van de erosiebeschermingslaag (stortstenen) en in de vier gradiënten (Zuidwest, Zuidoost, Noordwest en Noordoost) rondom de turbine. Een verlaging in mediane korrelgrootte werd gemeten dichtbij de turbine. Bovendien werd een accumulatie van juveniele zeesterren (Asteriidae juv.) en de Polychaeten *Spio* sp. en *Spiophanes bombyx* op stations <50 meter geobserveerd en dit voornamelijk op de Zuidwest gradiënt. Lagere stroomsnelheden en veranderende granulometrische karakteristieken vormen een substantiële verandering in de macrobenthische gemeenschap op de Zuidwestelijke gradiënt. De macrobenthische gemeenschap rondom de turbine is zeer dynamisch en is nog ver van een equilibrium verwijderd.

4.1. Introduction and objectives

As many offshore renewable installations are planned in Europe during the coming decade, it is essential to investigate the effects they could cause at a large and global scale. A large scale monitoring programme of the offshore wind farms in the Belgian part of the North Sea was therefore initiated in 2005. On the Thorntonbank, both higher densities of fish around the turbines (Reubens *et al.*, 2011) and a clear and rapid colonization of the hard substrate of the first gravity based foundations by epifauna (Kerckhof *et al.*, 2011; Kerckhof *et al.*, 2010) was recorded. However, large scale impacts on the macrobenthos of soft substrates during the first years after construction of the gravity based foundations (Coates & Vincx, 2010; Reubens *et al.*, 2009) could not be detected. Within the Belwind offshore wind farm, macrobenthos sampling was commenced in autumn 2011 (Kapasakali, 2012).

In 2010, a pilot study was carried out to investigate the soft sediment macrobenthic community at a small scale around one gravity based foundation on the Thorntonbank to detect any short term effects. Several trends were already observed in close vicinity of the turbine including a decrease in median grain size, an increase in species density and an accumulation of essential macrobenthic species such as the ecosystem engineer *Lanice conchilega*. The macrobenthic community illustrated a dynamic and evolving system away from the natural occurring *Nephtys cirrosa* community (Coates & Vincx, 2010). As macrobenthic communities can be modified due to changes in granulometric characteristics, organic matter content and hydrographic regimes (currents) (Pearson & Rosenberg, 1978; Wilhelmsson & Malm, 2008), the need to understand these processes around offshore wind turbines is critical.

The main objective of the small scale study in 2011 is to further investigate if the sediments and soft sediment macrobenthic communities are affected by organic enrichment and changing hydrodynamic conditions around the turbine. In a later phase, the observed effects from the targeted monitoring can be extrapolated to investigate possible large scale and global impacts.

4.2. Materials and Methods

The soft sediment macrobenthos was sampled in the close vicinity of the fifth gravity based turbine on the Thorntonbank (D5). Sampling was performed at the end of spring (Table 1) when both densities of the hard substrate epifauna and the deposition of the organic material (phytoplankton) start to increase (Vanaverbeke *et al.*, 2004). Sampling could not be repeated in autumn as access to the wind farm was not possible due to the construction works by C-Power.

4.2.1. Sampling

Sediment samples were obtained along four gradients around the D5 turbine (Southwest, Southeast, Northwest and Northeast). Along each gradient, samples were collected at 15, 25, 50, 100 and 200 meters from the scour protection layer (boulders) (Figure 1). The samples closest to the boulders (< 15m) were situated in the depression formed during construction. Only one macrobenthic sample at 1 meter from the boulders (Southwest gradient) could be collected by divers (operating from the RV Belgica). Sampling on the Northeast gradient was limited to 100 and 200 meters from the scour protection system due to the presence of cables on the seabed.

Table 1. Sampling locations for the Dive (1 & 7m) and Van Veen samples (15, 25, 50, 100 and 200m) around turbine D5 in 2011.

Dive samples 25/05/2011			**Van Veen samples 30/05/2011**
	Abiotic factors	Macrobenthos	Abiotic & macrobenthos
Northeast	/	/	2 stations, 3 replicates
Southeast	/	/	5 stations, 3 replicates
Southwest	Sample at 1 & 7m	Sample at 1m	5 stations, 3 replicates
Northwest	/	/	5 stations, 3 replicates
Total amount of samples	**2 samples**	**1 sample**	**51 samples**

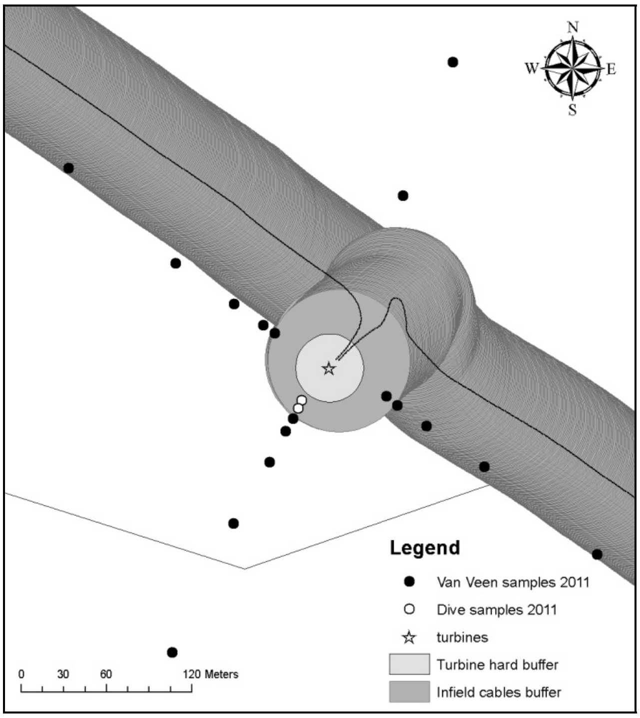


Figure 1. Sampling locations of the 2011 targeted monitoring campaign around turbine D5. Stations close to the turbine (White) were taken by divers, stations further away (Black) were taken with a small Van Veen grab.

Divers employed an airlift suction device (surface area 0.1026m^2) to sample the soft sediment macrobenthos at one meter on the Southwest gradient (no replication). Cores (diameter 27mm) were taken by divers along the Southwest gradient (1 and 7m) to measure median grain size and total organic matter in the sediment. A small Van Veen grab ($0.0247\ \text{m}^2$) was operated from the vessel Geosurveyor IV (GEO.XYZ bvba) to sample stations further away from the turbine (15, 25, 50, 100, 200m). At all stations, three replicate grabs were collected. One core sample (diameter 27mm) was obtained from each grab for physical-chemical analyses. Sediments from both the airlift suction device and the Van Veen grab were sieved (1mm) and fixed in an 8% formaldehyde-seawater solution.

4.2.2. Analyses

4.2.2.1. Abiotic analysis

The grain size partition was determined with a Malvern Mastersizer 2000G, hydro version 5.40. The Mastersizer utilizes a laser diffraction method and has a measuring range of 0.02 – 2000µm. The median grain size and proportions of the Wentworth fractions can therefore easily be determined. Fractions are given as volume percentages with a range from fine clay (< 4µm) to coarse gravel/shell material (>1600µm).

For every sample, the total amount of organic material (TOM %) was determined according to: TOM% = [(DW - AW) / (DW - CrW)] x 100.

The dry weight (DW) was determined after 48 hours at 60°C and the ash weight (AW) after 2h20min at 550°C. For every sample, the used crucible (laboratory weighing pan which can withstand very high temperatures) was weighed (CrW) in order to determine the TOM % (Heiri *et al.*, 2001).

4.2.2.2. Biotic analysis

Samples were stained with 1% Rose Bengal. The macrobenthic organisms were removed from all debris, identified to species level and counted. Due to the high quantity of juvenile starfish these organisms could only be separated by decantation of the sample. If the species level could not be defined, a higher taxonomic level was permitted. Nematoda, pisces and rare species (all species found in maximum three samples, with a maximum of two individuals per sample) were excluded from all analyses as they are not efficiently sampled with a Van Veen grab or they do not belong to the standard remains on a 1 mm sieve. After analysis, organisms were stored per species and per sample (Reference collection) in a 4% neutralized formaldehyde solution at the Marine Biology Research Group (Biology Department, Ghent University). The species list is given in Annex 1 – Systematic species list of the soft-substrate macrobenthos. The most recent systematic-taxonomic literature as well as species lists for the Belgian part of the North Sea were consulted (Adema, 1991; Appeltans W, 2012; Bick *et al.*, 2010; d'Udekem d'Acoz, 2004; De Bruyne, 1994; Degraer *et al.*, 2006; Fiege *et al.*, 2000; Fish & Fish, 1996; Hartmann-Schröder, 1996; Hayward & Ryland, 1995; Jones, 1976; Lincoln, 1979; Naylor, 1972; Tebble, 1966)

4.2.3. Data analysis

The following data were collected per sampling station: date, location, sediment composition, total organic matter content, macrobenthic species list, number of individuals per species and total biomass per species. The number of individuals per sample and per species were standardised to the number of individuals per m² (abundance). Data are stored in the Belgian Marine Data Centre (BMDC).

Statistical analyses were carried out with programmes R 2.14.2 (www.r-project.org) and Primer v6 with PERMANOVA add-on software (Anderson *et al.*, 2008; Clarke & Gorley, 2006). Distribution figures were created with the programme ArcGIS and graphs were created with STATISTICA 7 and Excel 7. Differences were statistically tested using two-way ANOVA, after compliance with the classic assumptions. When significant differences were observed, the post-hoc Tukey HSD test was applied to identify significant differences ($p<0.05$) between pairs of groups. If the assumptions for parametric analyses were not fulfilled, a two-way permutational ANOVA (Permanova) was applied. Significant differences between pairs of groups were further analysed by applying a pair-wise comparison test among factor levels. If the unique possible permutations were too low to create meaningful p-values (Pperm) in the pair-wise comparisons, the interpretation was assigned to the Monte-Carlo p-values P(MC) (Anderson *et al.*, 2008).

For the multivariate community analysis, data was square-root transformed and Bray-Curtis similarity matrices were used to build up non-metric multidimensional scaling (MDS) plots. MDS plots give information on relationships between data points. The stress values indicate how well the relationships are represented. Only results with a stress value lower than 0.2 were accepted (Clarke, 1993). Two-way crossed SIMPER analysis allowed for the identification of those species having important contributions to the within group similarity. The detection of which species contribute to the distinctness of certain communities is found from similarity and dissimilarity percentages.

4.3. Results

4.3.1. Abiotic analysis

The median grain size ranged between 274µm and 437µm. Median grain size was significantly affected by the Gradient x Distance interaction (Two-way ANOVA: $F_{9,\ 34}= 3.8547$; $p=0.0019$). The two samples at one and seven meters on the Southwest gradient were excluded from statistical analyses (only one replicate available). However, it is clear that the overall lowest median grain sizes were recorded at these two stations (Figure 2). The mean median grain size from the other stations, showed higher values with increasing distance from the scour protection on the Northwest and Southwest gradients (Figure 2, Left). The minimum value on the Northwest gradient was measured at 15 meters (310 ± 6 µm) and significantly lower (Tukey HSD test, $p=0.0363$) than the maximum at 200 meters (402 ± 18 µm). On the Southwest gradient, a minimum of 274µm at one meter and a maximum of 399 ± 17µm at 100 meters from the scour protection system were measured. The

Southeast gradient illustrated an opposite trend with a minimum value of 312 ± 6 µm at 100 meters and a maximum of 356 ± 20 µm at 25 meters. Furthermore, the mud content (fractions <64µm) was zero for all stations.

No significant differences for the mean total organic matter content were observed for the factors distance (15 – 200m), gradient (SW, SE, NW, NE) or the interaction effect Gradient x Distance (Two-way ANOVA). However, a peak of 2.0 ± 1.26% at 25 meters on the Northwest gradient (Figure 2, Right) was obvious.

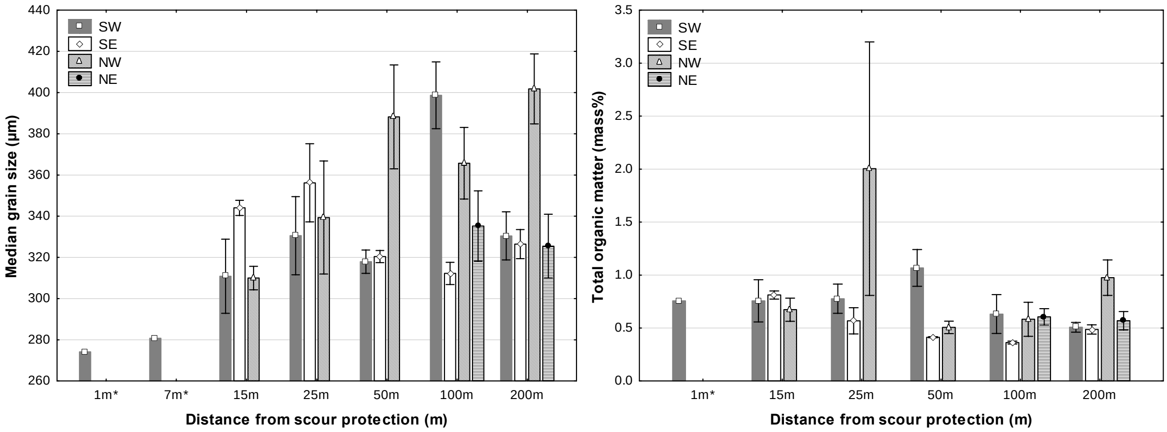


Figure 2. Median grain size (µm) (Left) and Total organic matter content (Mass %) (Right), from one to 200 meters from the scour protection along the four gradients (SW, SE, NW, and NE). Three replicates for every station except for 1 & 7m (Southwest gradient). *Samples at 1 & 7m taken during diving activities.

4.3.2. Biotic variables

4.3.2.1. Macrobenthic density

The total density ranged from 1781 ± 304 ind./m² (SW, 200m) to 62132 ± 54450 ind./m² (SW, 25m). No significant differences (Two-way ANOVA) were measured for the total density of samples (Figure 3 Left, Table 2A). The total density was highly dominated by juvenile starfish (Asteriidae juv.) in a number of samples increasing the mean total density considerably on the Southwest gradient. The highest densities of juvenile Asteriidae were recorded on this gradient, creating a peak in mean total density at 25 meters (62132 ind./m²) from the scour protection (Figure 3, Left). Due to the extreme outliers, juvenile Asteriidae were excluded from Figure 3 (Right) and Table 2B to illustrate the apparent trends in distance from the scour protection and around the turbine on the four gradients.

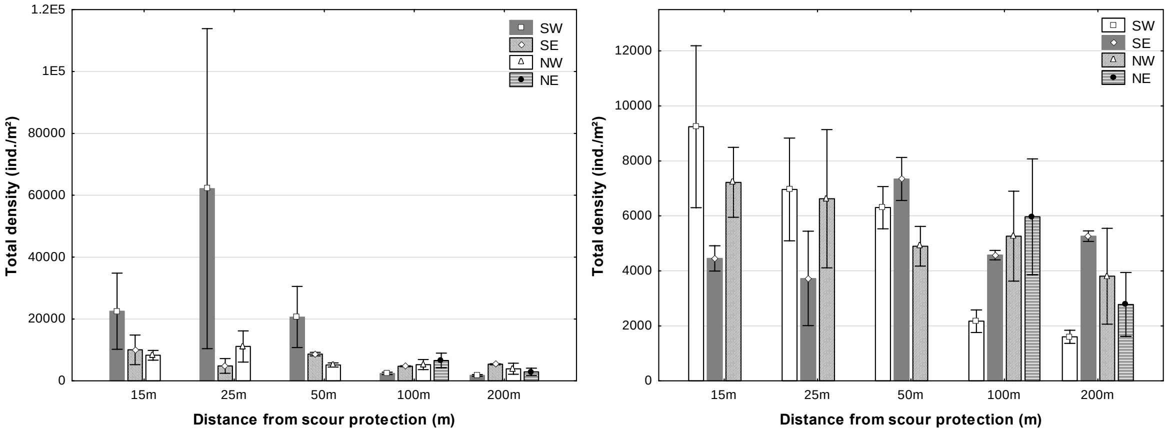


Figure 3. Total density (ind./m²) from one to 200 meters from the scour protection along the four gradients (SW, SE, NW, NE). Left: Asteriidae juv. included Right: Asteriidae juv. excluded due to extreme outliers.

Table 2. Total density (ind./m²) and standard error (SE) for all gradients around the D5 turbine (SW, SE, NW and NE) with increasing distance from the scour protection (1 – 200m). A: Asteriidae juv. included; B: Asteriidae juv. excluded.
* Different sampling method employed at 1m (Air-lift suction device).

A	**Southwest**		**Southeast**		**Northwest**		**Northeast**	
	ind./m²	SE	ind./m²	SE	ind./m²	SE	ind./m²	SE
1m*	2729							
15m	22524	12954	10040	5020	8286	1669		
25m	62132	54450	4845	2487	11147	5306		
50m	20648	10400	8650	605	5169	710		
100m	2443	412	4737	169	5277	1720	6613	248
200m	1781	304	5385	176	3941	1893	2915	1264
B								
1m*	1852							
15m	9244	3099	4453	486	7220	1338		
25m	6964	1964	3725	1807	6626	2645		
50m	6302	808	7341	823	4899	759		
100m	2173	431	4575	183	5263	1721	5965	2218
200m	1606	252	5263	204	3806	1834	2780	1222

After excluding the juvenile Asteriidae (Table 2B), a decreasing density with increasing distance was recorded on the Southwest and Northwest gradients (Figure 3 Right). The mean total density was however only significantly affected by the Distance factor (Two-way Permanova: Pseudo-F= 2.9432; p= 0.0197) with a significant difference measured between 15m and 100-200m together with 50m and 100-200meters (pair-wise comparison; Table 3).

Table 3. Significant (P<0.05) Two-way Pair-wise comparisons (Permanova) of the factor Distance for the total density (ind./m²) around the D5 turbine, excluding Asteriidae juv.

Factor Distance		**T**	**P(perm)**
15m	100m	2.259	0.0245
15m	200m	2.547	0.0132
50m	100m	2.413	0.0224
50m	200m	2.725	0.0109

4.3.2.2. Diversity indices

The mean species richness (N_0) ranged from 9.7 ± 2.3 (NW, 200m) to 23 ± 3 species (SW, 25m) (Figure 4). N_0 was significantly affected by factor Distance (Two-way Permanova: Pseudo-F= 2.6362; p= 0.0408). Pair-wise comparisons revealed significantly higher species richness at 15 m compared to 100m and 200 m (Figure 4; Table 4). No significant differences were detected for species diversity (H') and Taxonomic Diversity (Δ) (Two-way Permanova). The Taxonomic Distinctness (Δ^*) revealed a significant difference for both factors Gradient (Two-way Permanova: Pseudo-F= 7.5385; p= 0.0011) and Distance (Two-way Permanova: Pseudo-F= 4.0111; p= 0.0092). Pair-wise comparisons within factor Gradient revealed significant differences in Taxonomic Distinctness at the Southwest gradient with that at the Northwest and Southeast gradients (Table 4). A significant difference between the Southeast and Northeast gradient was also observed. Pair-wise comparisons within factor Distance described a significant difference in Taxonomic Distinctness between 15 and 200 meters (Table 4). Furthermore, a significant difference was observed for the Taxonomic Distinctness measured at 25 meters with that at 50, 100 and 200 meters from the scour protection (Figure, 4, Table 4).

Table 4. Significant (P<0.05) Two-way pair-wise comparisons (Permanova) of the factor Distance for Species richness (N_0) and of both factors Distance and Gradients for Taxonomic Distinctness (Δ^*) of stations around the D5 turbine.

Species richness (N_0) Pair-wise test Distance		**t**	**P(perm)**
15m	100m	2.4196	0.0185
15m	200m	2.3329	0.0258
Taxonomic Distinctness (Δ^*)			
Pair-wise test Distance			
15m	200m	3.6817	0.0031
25m	50m	2.8029	0.0104
25m	100m	2.6735	0.0151
25m	200m	4.7522	0.0002
Pair-wise test Gradient			
NE	SE	2.5617	0.0225
SW	NW	3.7592	0.0011
SW	SE	4.7614	0.0003

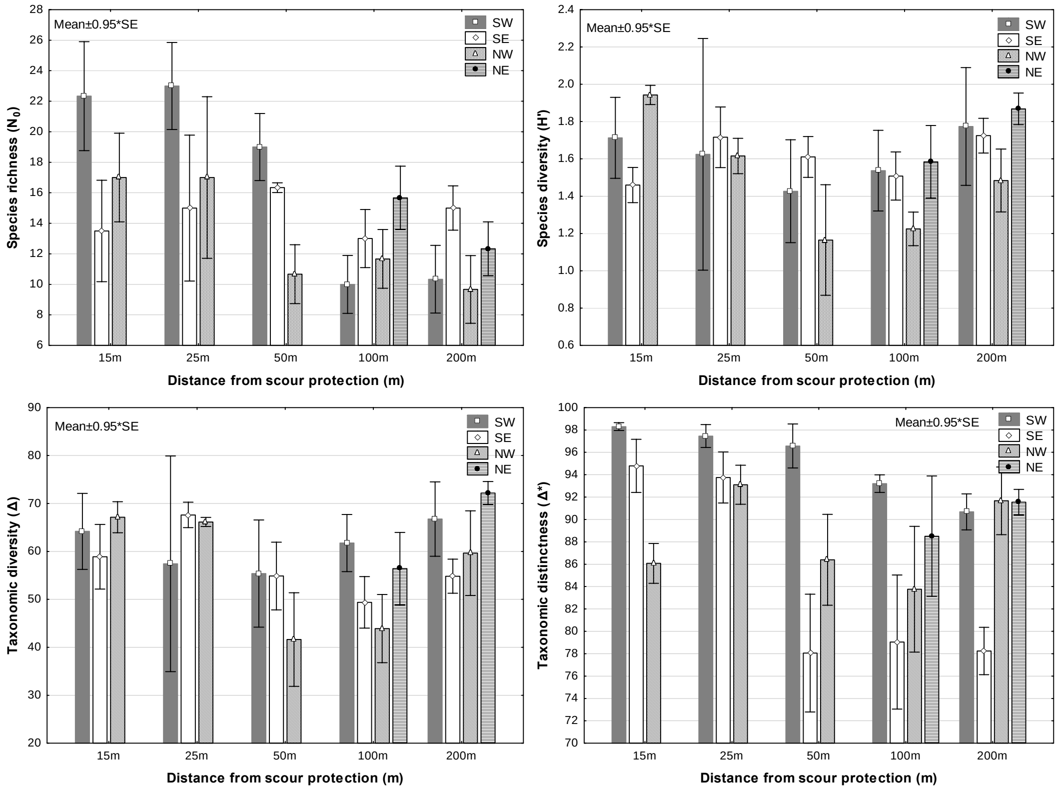


Figure 4. Species richness (N_0), species diversity (H'), taxonomic diversity (Δ) and taxonomic distinctness (Δ^*) for stations sampled from one to 200 meters from the scour protection, along the four gradients (SW, SE, NW, NE).

4.3.2.3. Macrobenthic community composition

The macrobenthic community composition expressed as the species composition based on density and identity of each species, was significantly affected by the Distance x Gradient interaction (Figure 5, Two-way Permanova: Pseudo-F= 1.4782; p= 0.0233). Pair-wise comparisons revealed significant differences between macrobenthic communities sampled at 15m and those sampled at 100m and 200m at both the Northwest and Southwest gradient (Table 5). At the Southwest gradient, additional differences between macrofaunal communities sampled at 25m and 100m from the scour protection were observed (Table 5). At the Southeast gradient, significant differences were observed between the communities from stations located at 50 and 200m (Table 5). Significant differences between stations from different gradients were only observed at 15m distance, and this from the Northwest and Southeast gradients (P(MC)=0.0206).

Table 5. Significant (P(MC) < 0.05) pair-wise comparisons of the interaction effect (Distance x Gradient) with factor Distance (15m) and factor Gradient (SW, NW, SE) for macrobenthic community composition. Unique perms was 10 for every test.

Within factor level				P(perm)	P(MC)
Pair-wise test Distance					
15m	NW	SW	2.3123	0.1064	0.0206
Pair-wise test Gradient					
Southwest	15m	100m	2.2961	0.0982	0.0194
	15m	200m	2.1104	0.0975	0.025
	25m	100m	1.9609	0.1019	0.045
Northwest	15m	100m	2.1641	0.0989	0.0334
	15m	200m	2.1471	0.095	0.0287
Southeast	50m	200m	1.8912	0.101	0.0439

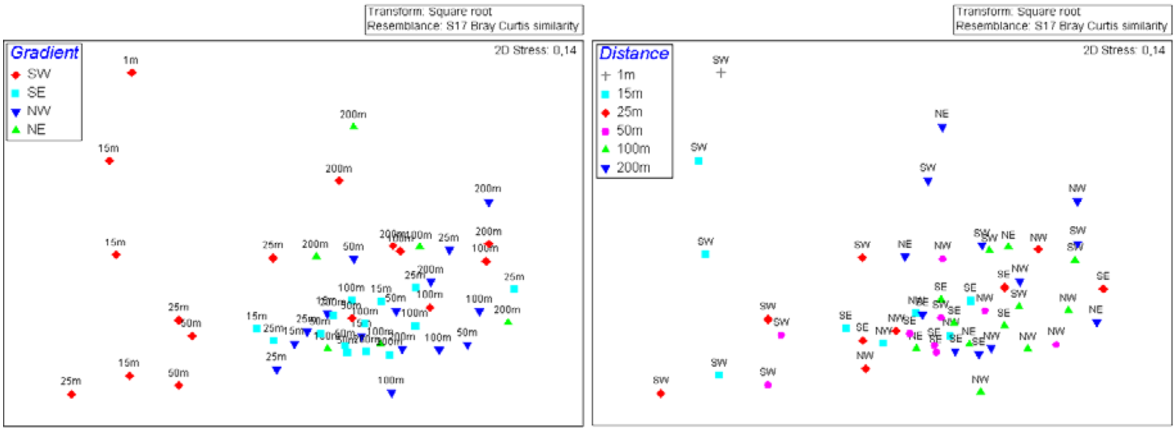


Figure 5. MDS plot based on macrobenthic densities around the D5 gravity based turbine for every gradient (Left) and distance (Right) from the scour protection (1, 15, 25, 50,100 and 200m).

Two-way crossed SIMPER analysis between the Southwest and Northwest gradients revealed a high contribution to the total similarity for juvenile Asteriidae (21.76%) and *Spio* sp. (22.36%) on the Southwest gradient and *Spio* sp. (35.36%) on the Northwest gradient (Annex 2). Both the juvenile Asteriidae and *Spio* sp. showed a high contribution to the average similarity at 15, 25 and 50 meters while Asteriidae juv. declined at 100 meters providing a higher contribution of *Spio* sp. to the average similarity (Annex 2). Furthermore, *Spio* sp., *Spiophanes bombyx* and *Nephtys cirrosa* showed a high contribution to the average similarity at 200 meters (Annex 2). These trends are illustrated in Figure 6 with high mean densities of juvenile Asteriidae at the Southwest gradient. Asteriidae juv. and *Spiophanes bombyx* illustrate high mean densities from 15 to 50 meters while a peak in mean density at 25 meters is revealed for both *Spio* sp. and Nemertea sp.

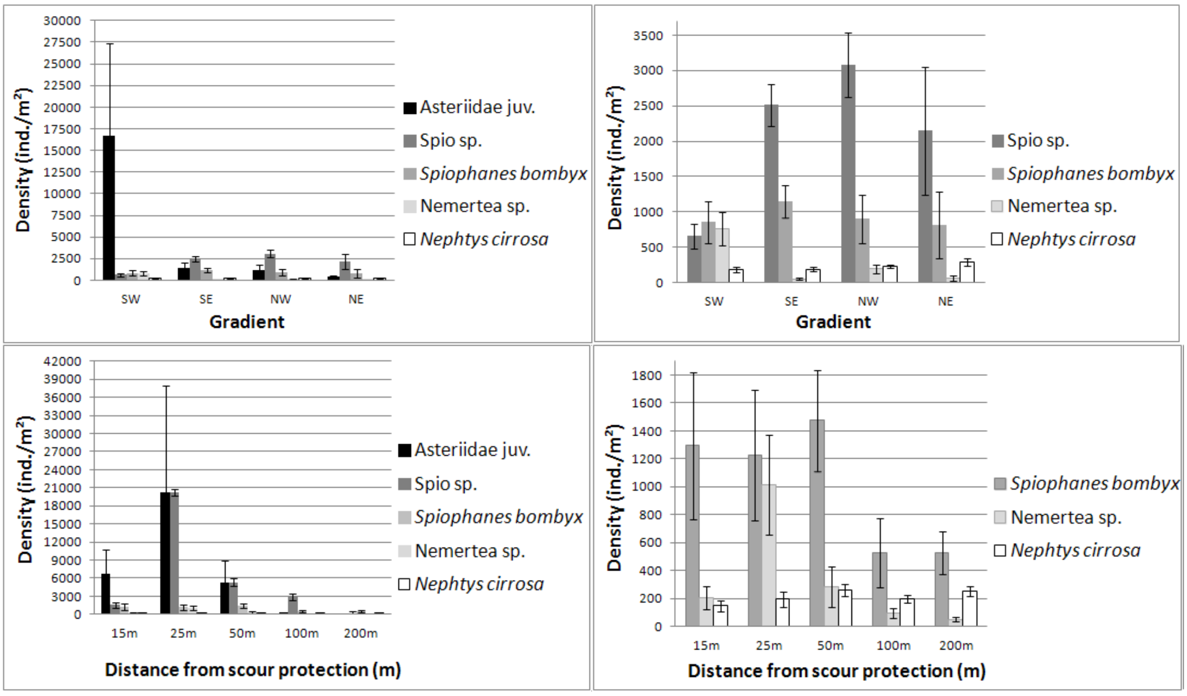


Figure 6. Mean Densities (ind./m²) of the 5 most dominant macrobenthic species on every gradient (SW, SE, NW, NE) and every distance from the scour protection (15-200m).

4.4. Discussion

Our results indicate that the presence of a gravity based wind turbine on the Thorntonbank causes changes in the sandbank habitat, thereby inducing changes in macrofaunal communities. Close to the turbine, sediments were finer and macrofaunal community composition and diversity aspects were changed in comparison to the patterns observed further away.

Significant sedimentological changes were observed around the gravity based foundation. In 2010, a lower median grain size was measured at one and seven meters from the scour protection system and this on the Southwest and Northeast gradients (Coates *et al.*, 2011). However, no clear decrease was observed further away from the turbine. One year later, the median grain size appeared to be decreasing further on the Southwest and Northwest gradients with a significant difference observed between 15 and 200 meters on the Northwest gradient. The lowest value of 274µm was measured at one meter on the Southwest gradient. The lower median grain size at one and seven meters detected on the Northeast gradient in 2010 could not be confirmed due to the limitations in dive sampling during 2011. The total organic matter content did not show any significantly higher concentrations in close vicinity to the turbine. A peak at 25 meters on the Northwest gradient can be explained by small scale patchiness occurring after the spring bloom.

Total densities (excluding Asteriidae juv.) significantly increased closer to the foundation and mainly on the Southwest and Northwest gradients; correlating with a finer grain size (Van Hoey *et al.*, 2004). Furthermore, the species richness and taxonomic distinctness also showed significantly higher values at 15 and 25 meters from the scour protection, indicating an increasing taxonomic diversity closer to the foundation. In 2010, an apparent enrichment and change in community composition was first observed in comparison to previous studies (Coates & Vincx, 2010; De Maersschalck *et al.*, 2006; Reubens *et al.*, 2009) and this in close vicinity to the scour protection. An accumulation of *Lanice conchilega* and *Spiophanes bombyx* was observed in 2010 on the Southwest and Northeast gradients at one and seven meters (<15m). The Northwest and Southeast gradients however showed a dominance of the amphipod species *Monocorophium acherusicum* and *Jassa herdmani* at stations closer than 15 meters (Coates *et al.*, 2011). Again, the lack of dive samples obstructed confirmation of these observations. However, one year later, the clear observations of a changing macrobenthic community already extended up to 50 meters from the scour protection system. A comparable accumulation was mainly observed on the Southwest gradient and from 15 to 50 meters, this time of the juvenile Asteriidae. Across all gradients, *Spio* sp., *Spiophanes bombyx* and Nemertea sp. also showed higher abundances closer to the turbine. Densities of *Nephtys cirrosa,* a typical species on the Thorntonbank (Coates & Vincx, 2010), remained stable throughout the progression away from the turbine. Even though *Lanice conchilega* was not as abundant in comparison to 2010, it is certainly noteworthy that the abundance of this species increased on the Southwest gradient at 15 and 25 meters from the scour protection (cfr. SIMPER analysis Annex 2).

The sedimentological characteristics and macrobenthic communities appear to be primarily affected on the Southwest gradient around the gravity based foundation both in 2010 and 2011 (Figure 6). Tidal currents in this area are mainly directed to the Northeast and Southwest creating areas of low current speed in the wake of the large gravity based foundation (Coates *et al.*, 2011; Van den Eynde, 2005). Therefore, an increased larval settlement in these areas could enhance the recruitment of key macrobenthic species to the wind farm areas such as the tube forming *Lanice conchilega* (mainly observed in high densities in June 2010) and the echinoderm *Asterias rubens* (now in high densities in May 2011).

The Southwest gradient appears to have sand pits created during dredging activities before installation of the gravity based foundation (Van den Eynde *et al.*, 2010). Together with a decreased current flow on this gradient, an ideal situation is created for the settlement of macrobenthic larvae (such as the juvenile Asteriidae) and organic material from the hard substrate (scour protection and foundation) onto the seabed. In 2010, the highest chlorophyll *a* concentration was measured on the Southwest gradient (Coates *et al.*, 2011), suggesting a higher food supply which in turn positively effects the juvenile growth of starfish (Guillou *et al.*, 2012). The higher abundances of juvenile starfish around the gravity based foundation during 2011, can also be linked to the large annual variations in recruitment intensity of *Asterias rubens* (Guillou *et al.*, 2012). Therefore, this observation must be followed up to determine if a yearly recruitment of juvenile starfish or other species will have a long term effect on the soft-sediment macrobenthic community around offshore turbines.

4.5. Conclusions

Up till now, this targeted monitoring study has revealed a very young and dynamic macrobenthic community around the fifth gravity based foundation on the Thorntonbank, increasing key macrobenthic species in this area. The community is evolving away from the original *Nephtys cirrosa* community (Van Hoey *et al.*, 2004) and is far from reaching a stable equilibrium, stressing the need to continue monitoring regularly around the gravity based foundation in the future. With a decreasing median grain size and increasing macrobenthic abundance and diversity the community might possibly be moving towards a variant of the rich *Abra alba – Mysella bidentata* community, normally found in near-shore shallow muddy sands (Van Hoey *et al.*, 2004).

As a substantial increase of marine renewable energy in Northwest Europe is planned during the coming years it is essential to enhance the knowledge of any effects they will create on the soft-sediment macrobenthos. Effects recorded on small scale can help predict cumulative, large scale and global effects.

4.6. Acknowledgements

We would like to thank the company's n.v. C-Power, n.v. Belwind, n.v. Northwind and the Management Unit of the North Sea Mathematical Models (MUMM) for assigning this research project to the Marine Biology Research Group, their financial support and coordination of the project. The Flanders Marine Institute (VLIZ), the diving team and the crews of the research vessels RV Belgica, RV Zeeleeuw and Geosurveyor (GEO.XYZ bvba) are thanked for their technical support. Furthermore, several colleagues from the Marine Biology Research Group are also thanked for their technical support during the sampling campaigns and in the laboratory.

4.7. References

Adema JPHM (ed) (1991) De krabben van Nederland en België (Crustacea, Decapoda, Brachyura). Nationaal Natuurhistorisch Museum Leiden.

Anderson M, Gorley R, Clarke KR (2008) PERMANOVA+ for Primer: Guide to Software and Statistical Methods.: PRIMER-E: Plymouth, UK.

Appeltans W BP, Boxshall GA, Fauchald K, Gordon DP, et al., editors (2012) World Register of Marine Species (WORMS - www.marinespecies.org).

Bick A, Otte K, Meissner K (2010) A contribution to the taxonomy of Spio (Spionidae, Polychaeta, Annelida) occurring in the North and Baltic Seas, with a key to species recorded in this area. Marine Biodiversity 40(3): 161-180

Brey T. (2001) Population dynamics in benthic invertebrates. A virtual handbook. Version 01.2. http://www.thomas-brey.de/science/virtualhandbook.

Clarke K, Gorley R (2006) PRIMER v6: User Manual/Tutorial. PRIMER-E, Plymouth.: 190

Clarke KR (1993) NONPARAMETRIC MULTIVARIATE ANALYSES OF CHANGES IN COMMUNITY STRUCTURE. Australian Journal of Ecology 18(1): 117-143

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:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 6: 47-63.

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

d'Udekem d'Acoz C (2004) The genus Bathyporeia Lindström, 1855, in western Europe (Crustacea: Amphipoda: Pontoporeiidae): Nationaal Natuurhistorisch Museum

De Bruyne RH (ed) (1994) Schelpen van de Nederlandse kust: Stichting Jeugdbondsuitgeverij, Stichting Uitgeverij KNNV, Utrecht. Tweede druk.

De Maersschalck V, Hostens K, Wittoeck J, Cooreman K, Vincx M, Degraer S. (2006) Monitoring van de effecten van het Thornton windmolenpark op de benthische macro-invertebraten en de visfauna van het zachte substraat. p. 136.

Degraer S, Wittoeck J, Appeltans W, Cooreman K, Deprez T, Hillewaert H, Hostens K, Mees J, Vanden Berghe W, Vincx M (eds) (2006) De macrobenthosatlas van het Belgische deel van de Noordzee. : Federaal Wetenschapsbeleid. D/20051191/5.

Fiege D, Licher F, Mackie ASY (2000) A partial review of the European Magelonidae (Annelida : Polychaeta): Magelona mirabilis redefined and M-johnstoni sp nov distinguished. Journal of the Marine Biological Association of the United Kingdom 80(2): 215-234

Fish JD, Fish S (eds) (1996) A student's guide to the seashore. Second Edition.: Cambridge University Press, Cambridge.

Guillou M, Joly-Turquin G, Leyzour S, Pernet P, Dubois P (2012) Factors controlling juvenile growth and population structure of the starfish Asterias rubens in intertidal habitats: field and experimental approaches. Journal of the Marine Biological Association of the United Kingdom 92(2): 367-378.

Hartmann-Schröder G (ed) (1996) Die Tierwelt Deutschlands und der angrenzenden Meersteile nach ihren Merkmalen und ihrer Lebensweise. 58. Teil: Annelida - Borstenwürmer - Polychaeta. 2., neubearbeitete Auflage: VEB Gustav Fischer Verlag Jena.

Hayward PJ, Ryland JS (eds) (1995) Handbook of the marine fauna of North-West Europe.: Oxford University Press, 800 pp.

Heiri O, Lotter AF, Lemcke G (2001) Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. Journal of Paleolimnology 25(1): 101-110

Jones NS (ed) (1976) Synopses of the British Fauna (N.S.) 7: British cumaceans (Arthropoda: Crustacea). Keys and notes for the identification of the species.: The Linnean Society of London, Academic Press London and New York.

Kapasakali D-A (2012) Ecological effects of offshore wind farms on the soft-sediment macrofauna: The case of the Belwind offshore wind farm in the Belgian Part of the North Sea (Bligh Bank, BPNS). Faculty of Sciences, Department of Biology, Section Marine Biology, Ghent University,

Kerckhof F, Degraer S, Norro A, Rumes B (2011). Offshore intertidal hard substrata: a new habitat promoting non-indigenous species in the Southern North Sea: and exploratory study. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 4: 27-38.

Kerckhof F, Rumes B, Norro A, Jacques TG, Degraer S (2010). Seasonal variation and vertical zonation of the marine biofouling on a concrete offshore windmill foundation on the Thornton Bank (southern North Sea). *In:* Degraer, S., Brabant, R., Rumes, B., (Eds.) (2010). Offshore wind farms in the Belgian part of the North Sea. Early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 5: 184.

Lincoln RJ (ed) (1979) British marine amphipoda: Gammaridae: British museum (Natural History), London., 658pp

Naylor E (ed) (1972) Synopsis of the British Fauna (N.S.) 3: British Marine Isopods. Keys and notes for the identification of the species.: The Linnean Society of London, Academic Press London and New York.

Pearson TH, Rosenberg R (1978) Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanog Mar Biol Ann Rev 16: 229-311

Reubens J, Degraer S, Vincx M (2011) Spatial and temporal movements of cod (*Gadus morhua*) in a wind farm in the Belgian part of the North Sea using acoustic telemetry, a VPS study. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 5: 39-46.

Reubens J, Vanden Eede S, Vincx M (2009) Monitoring of the effects of offshore wind farms on the endobenthos of soft substrates: Year-0 Bligh Bank and Year-1 Thorntonbank. *In:* Degraer, S. & Brabant, R. (Eds.) (2009). Offshore wind farms in the Belgian part of the North Sea: State of the art after two years of environmental monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 6: 61-91.

Tebble N (ed) (1966) British bivalve seashells. A handbook for identification: Trustees of the British Museum (Natural History), London. Alden Press Osney Mead, Oxford.

Van den Eynde D. (2005) Numerieke modellering van het sedimenttransport ter hoogte van de Thorntonbank. p. 23.

Van den Eynde D, Brabant R, Fettweis M, Francken F, Melotte J, Sas M, Van Lancker V (2010) Monitoring of hydrodynamic and morphological changes at the C-Power and the Belwind offshore wind farm sites - A synthesis. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2010). Offshore wind farms in the Belgian part of the North Sea: early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. Chapter 3: 19-36.

Van Hoey G, Degraer S, Vincx M (2004) Macrobenthic community structure of soft-bottom sediments at the Belgian Continental Shelf. Estuarine, Coastal and Shelf science 59: 599-613.

Vanaverbeke J, Steyaert M, Soetaert K, Rousseau V, Van Gansbeke D, Parent JY, Vincx M (2004) Changes in structural and functional diversity of nematode communities during a spring phytoplankton bloom in the southern North Sea. journal of Sea Research 52(4): 281-292.

Warwick RM, Clarke KR (1995) New 'biodiversity' measures reveal a decrease in taxonomic distinctness with increasing stress. Marine Ecology-Progress Series 129(1-3): 301-305

Wilhelmsson D, Malm T (2008) Fouling assemblages on offshore wind power plants and adjacent substrata. Estuarine Coastal and Shelf Science 79(3): 459-466

Chapter 5. Monitoring the effects of offshore wind farms on the epifauna and demersal fish fauna of soft-bottom sediments

---

S. Vandendriessche[*], J. Derweduwen & K. Hostens

Institute for Agricultural and Fisheries Research (ILVO-Fisheries), Bio-Environmental Research Group. Ankerstraat 1, 8400 Ostend

[*] *Corresponding author: sofie.vandendriessche@ilvo.vlaanderen.be*

Green sea urchin *Psammechinus miliaris* Photo K. Van Ginderdeuren / ILVO

Abstract

One year after the construction of 55 monopiles on the Bligh Bank, changes within the soft-substratum epibenthos and fish were observed, both on ecosystem component level and on species level. Analyses were conducted to discriminate between effects of the presence of turbines and effects as a result of changing activities in the vicinity of the wind farms (fringe effects). The results showed a decrease in total demersal fish densities and an increase in epibenthos densities within the wind farm. The changes in demersal fish may have resulted from the absence of fisheries in the area or local changes in sedimentology and infaunal communities. For commercially important flatfish, we observed higher densities (turbot, sole) and/or changes in length-frequency distribution (turbot, plaice). This may signal a refugium effect, but bearing in mind that large flatfish such as sole do not stay within a wind farm for longer periods, this effect will be limited. The increase in epibenthos probably resulted from the presence of hard substrates and their fouling communities and from the absence of fisheries. The increase, however, was mainly seen for dominant, scavenging species such as echinoderms and hermit crabs. Signs of recovery of populations of long lived species vulnerable to trawling were not yet observed in autumn 2011 at the Bligh Bank. Some differences between fringe stations and reference stations were described but they cannot straightforwardly be linked to fringe effects resulting from changing activities in the close vicinity of the wind farm concession.

Samenvatting

Een jaar na de constructie van 55 turbines op de Bligh Bank werden reeds duidelijke verschillen waargenomen binnen het epibenthos en de demersale visfauna van het windmolengebied, zowel op het niveau van ecosysteemcomponent als op soortsniveau. De uitgevoerde analyses werden ontworpen om onderscheid te kunnen maken tussen effecten ten gevolge van de aanwezigheid van turbines en effecten ten gevolge van veranderende activiteiten in de omgeving van het windmolenpark. De resultaten toonden een reductie van het aantal demersale vissen en een stijging van het aantal epibenthische ongewervelden binnen het windmolenpark. De veranderingen in demersale visfauna kunnen het gevolg zijn van het verdwijnen van visserijactiviteiten in het gebied of van lokale veranderingen in de bodemsamenstelling en bijgevolg ook van de infauna. Van commercieel interessante platvissoorten hebben we hogere densiteiten waargenomen (tarbot, tong) en/of veranderingen in de lengte-frequentieverspreiding (tarbot, pladijs). Deze waarnemingen kunnen duiden op een refugium effect, maar dergelijke grote platvissen blijven niet lang in een zelfde gebied, dus dat effect is waarschijnlijk beperkt. De stijging in aantallen epibenthos is waarschijnlijk het gevolg van de aanwezigheid van turbines en hun begroeiing, en van de afwezigheid van bodemvisserij. Stijgingen werden echter enkel waargenomen bij reeds dominante aaseters zoals stekelhuidigen en heremietkreeften. Er werden op de Bligh Bank nog geen tekenen waargenomen van herstel van populaties van langlevende soorten die gevoelig zijn voor bodemvisserij. Ten slotte werden er enkele verschillen waargenomen tussen referentiestations en ‘fringe’ stations, maar deze kunnen nog niet eenduidig worden in verband gebracht met veranderende activiteiten in de buurt van het windmolenpark.

5.1. Introduction

The monopiles currently present on the Bligh Bank were constructed in the period September 2009-February 2010 (Brabant *et al.*, 2011). In September-October 2011, samples of epibenthos and demersal fish were taken within the concession zone for the first time since construction. One year after construction, changes within the soft-substratum epibenthos and fish were expected as a result of (1) depletion of phytoplankton by high densities of filtrating organisms on and around the turbine, (2) input of organic material from organisms associated with the turbines, as well as entrapment of material by the turbines, (3) predation by fish and crabs associated with the turbines, and (4) a reef effect (Andersson *et al.*, 2009; Wilhelmsson *et al.*, 2009). Additionally, epibenthos and fish may be influenced by the exclusion of fisheries activities from wind farms and their safety buffers (e.g.

Jaworski *et al.,* 2006), and by local reallocation of fishing effort in the area surrounding the wind farm (Berkenhagen *et al.,* 2010). Changes in and around wind farms were already observed in the vicinity of the six gravity based foundations on the Thorntonbank (two years after construction; see Vandendriessche *et al,* 2011a) and included (1) generally larger individuals of the swimming crab *Liocarcinus holsatus* and the brown shrimp *Crangon crangon* at the impact station, which may reflect either increased growth due to a high food availability or increased predation pressure eliminating smaller individuals; (2) higher autumn densities of small whiting *Merlangius merlangus* at the impact station. Additionally, an increase in fisheries intensity of the Belgian fleet and recreational fisheries in the area north of the Thorntonbank concession presumably caused the absence of the smallest size classes of sole, via increased indirect fishing mortality (such as discards) or via changes in the local benthic community.

Changes within the soft substratum fauna resulting from turbine construction and exploitation on the Bligh Bank are expected to differ from the ones observed on the Thorntonbank since (1) monopiles are substantially different structures than gravity based foundations, (2) the Bligh Bank has a different sandbank topography and includes deeper areas than the Thorntonbank, (3) there are important differences within the species assemblages of epibenthos and demersal fish, especially at the most offshore situated stations (Derweduwen *et al.*, 2010), and (4) the Bligh Bank area was less intensively fished prior to construction than the Thorntonbank area (Vandendriessche *et al.*, 2011b).

This chapter reports on the condition of demersal fish, benthopelagic fish and epibenthos in the concession zones and reference zones of the Bligh Bank wind farm in the second year after the construction of 55 monopiles, and on the occurrence of fringe effects concerning these ecosystem components. The results were compared with post-construction observations at the Thorntonbank.

5.2. Material and Methods

For the baseline monitoring in 2011, 12 stations were sampled within the Bligh Bank concession area and the adjoining reference areas (figures 1 and 2). On these track locations, demersal fish fauna and macro-epibenthos were sampled from the research vessel Belgica with an 8-meter shrimp trawl (stretched mesh width 22 mm in the cod end) equipped with a bolder-chain (no tickler chains to minimize environmental damage). The net was dragged over the bottom during 15 minutes at an average speed of 4 knots. Data on time, start and stop coordinates, trajectory and sampling depth were noted to enable a correct conversion towards sampled surface units. The fish tracks were positioned following depth contours that run parallel to the coastline, thereby minimizing the depth variation within a single track. After each fish track, a photograph was taken of the net content prior to the processing of the catch. All fish, except gobies, were identified, measured and/or counted or wet weighed on board. In the case of small catches , the epibenthos (including gobies) was processed on board as well; in the case of a large catch, a subsample of 6 litres was frozen for further laboratory analyses. Rare or peculiar species/individuals were stored for further reference or investigation. The net contents were divided into ‘bentho-pelagic fish’,‘demersal fish’ and ‘epifauna’. For these ecosystem components density, biomass (epibenthos only), diversity and length frequency distribution were analysed. The number of individuals per sample and per species was converted to number of individuals per 1000m² (abundance). Biomass was expressed as grams of wet weight (WW) per 1000m² and diversity was evaluated based on Hill’s diversity indices N0 and N1 and on the variable Expected Number of Species (ES(n)).

The evolution of these parameters was investigated on an ecosystem component level (benthopelagic fish, demersal fish, epibenthos). Density was analysed in further detail for a selection of species. This selection was based on the species’ weight in discriminating components in the community composition analyses concerning the wind farm concession area and the adjoining reference areas, as conducted by Derweduwen *et al.* (2010).

For these analyses, stations were pooled based on their classification as wind farm reference station, wind farm impact station, impact fringe station or reference fringe station. The analyses were designed to trace two possible sources of environmental change:

- Impact of the presence of turbines: the Belwind turbines are located both on top of the Bligh Bank and in the surrounding gullies. Stations WBB05 and WBB07 were considered as impact stations for the gullies; WBB06a and WBB06b were considered as impact stations for the sandbank top. The latter were sampled for the first time since construction, in autumn 2011. Most effects were expected at these stations since they are situated between the turbine rows, while the gully impact stations are situated at a distance along the outer rows of turbines.
- Fringe effects in the vicinity of the wind farm concessions: impacts of changing activities, mainly fisheries, can be expected in the fringes of the concession, since VMS analyses and visual observations already indicated similar changes in the Thorntonbank area (Vandendriessche et al., 2011). Stations WBB08 and WBB04 were considered fringe stations, other gully stations outside the concession were treated as references.

Figure 1. Sampling stations visited in 2011 in the framework of the wind farm monitoring activities: Bligh Bank concession and references.

Figure 2. Sampling inside the Bligh Bank wind farm

5.3. Results

In function of visual representation, the results were split up into sandbank "top" results and "gully" results.

5.3.1. Sandbank tops

The data concerning the Bligh Bank top were very discontinuous throughout the years, especially in spring. Hence, only autumn data were represented as time series graphs and discussed.

At the ecosystem component level (benthopelagic fish, demersal fish, epifauna), the fluctuations of total density in impact and reference stations were considerable, especially in benthopelagic fish (figure 3). In autumn 2011, no benthopelagic fish were caught at the impact top stations and densities at the reference top stations were very low (av. < 1 ind/1000m²) compared to samples of preceding years. For demersal fish, the total density at impact stations in autumn was substantially lower (av. 91 ind/1000m²) than at reference stations (av. 192 ind/1000m²). For epibenthos, the situation was the other way round: densities were substantially higher at impact stations (20 ind/1000m²) compared to reference stations (5 ind/1000m²). The reference values of epibenthos density were quite stable throughout the period 2008-2011 (figure 3), while impact values were not. Epibenthos biomass showed a pattern similar to the one of density, with higher values at the impact stations (av. 52 g ADW/1000m²) compared to the references (av. 18 g ADW/1000m²). This difference between impact and reference stations had, however, already been observed in autumn 2009, when construction activities at the Bligh Bank had just been started. The average species richness fluctuated between 1 and 2 spp. for benthopelagic fish, between 10 and 15.5 spp. for demersal fish and between 8.5 and 14.5 spp. for epibenthos (figure 3). No clear patterns could be observed in species number and diversity (Expected Number of Species or ES) for benthopelagic fish. In demersal fish, the evolution of diversity was very similar at impact and reference stations. Only for epibenthos, there was a diversity difference between impact and reference stations: the species number was higher at the impact stations (av. 12.5 spp.) than at the references (av. 10 spp.). The ES measure showed the same trend, only more pronounced.

Benthopelagic fish species such as whiting (*Merlangius merlangus*), horse mackerel (*Trachurus trachurus*), sprat (*Sprattus sprattus*) and pouting (*Trisopterus luscus*) were only sporadically found in low numbers at the Bligh Bank top, and seldomly at impact and reference stations at the same time. Similar sporadical observations were also done for the demersal species hooknose (*Agonus cataphractus*), and brill (*Scopthalmus rhombus*), and for the epibenthic species *Crangon allmani*, and dogwhelk (*Buccinum undatum*). The density evolution was similar at impact and reference stations for dab (*Limanda limanda*), shrimp (*Crangon crangon*), common and reticulated dragonet (*Callionymus lyra & C. reticulatus*), swimming crab (*Liocarcinus holsatus*), the ophiuroid *Ophiura ophiura* and gobies (*Pomatoschistus* sp.). Differences in density evolution patterns (figure 4) can roughly be divided into two trends:

- Higher or similar values at the impact stations compared to the references in 2008-2009 and lower values in 2011. This was observed for the species solenette (*Buglossidium luteum* - av. 0.1 vs. 0.2 ind/1000m² in autumn 2011), lesser weever (*Echiichthys vipera* - av. 44 vs. 94 ind/1000m²), and scaldfish (*Arnoglossus laterna* - av. 0.1 ind/1000m vs. 0.6 ind/1000m²).
- Lower or similar values at the impact stations compared to the references in 2008-2009 and higher values in 2011. This was observed for the squid *Allotheutis subulata* (av. 0.3 vs 0 ind/1000m²), the common seastar (*A. rubens*, av. 3.8 vs. av. 0.3 ind/1000m²), the ophiuroid *Ophiura albida* (av. 3.4 vs 0.2 ind/1000m²), the urchin *Psammechinus miliaris* (av. 0.3 vs 0 ind/1000m²), the hermit crab (*Pagurus bernhardus* – av. 8 vs. 3 ind/1000m²), sole (*Solea solea* - av. 0.1 vs 0 ind/1000m²), and turbot (*Psetta maxima* - av. 0.1 vs 0 ind/1000m²).

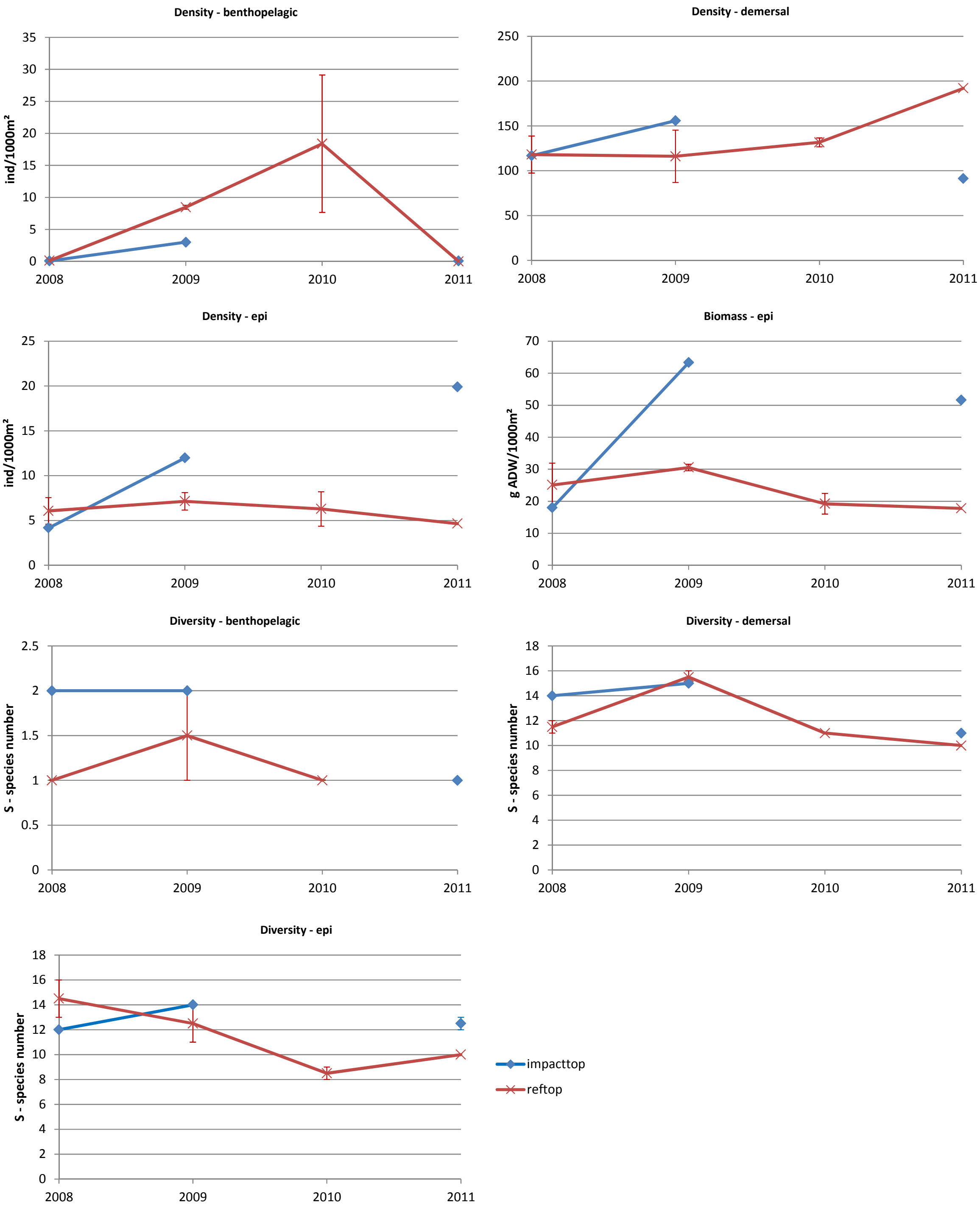


Figure 3. Charts representing differences between sandbank top impact stations and reference stations concerning autumn density, biomass and diversity of benthopelagic fish, demersal fish and epibenthos. Whiskers represent standard error.

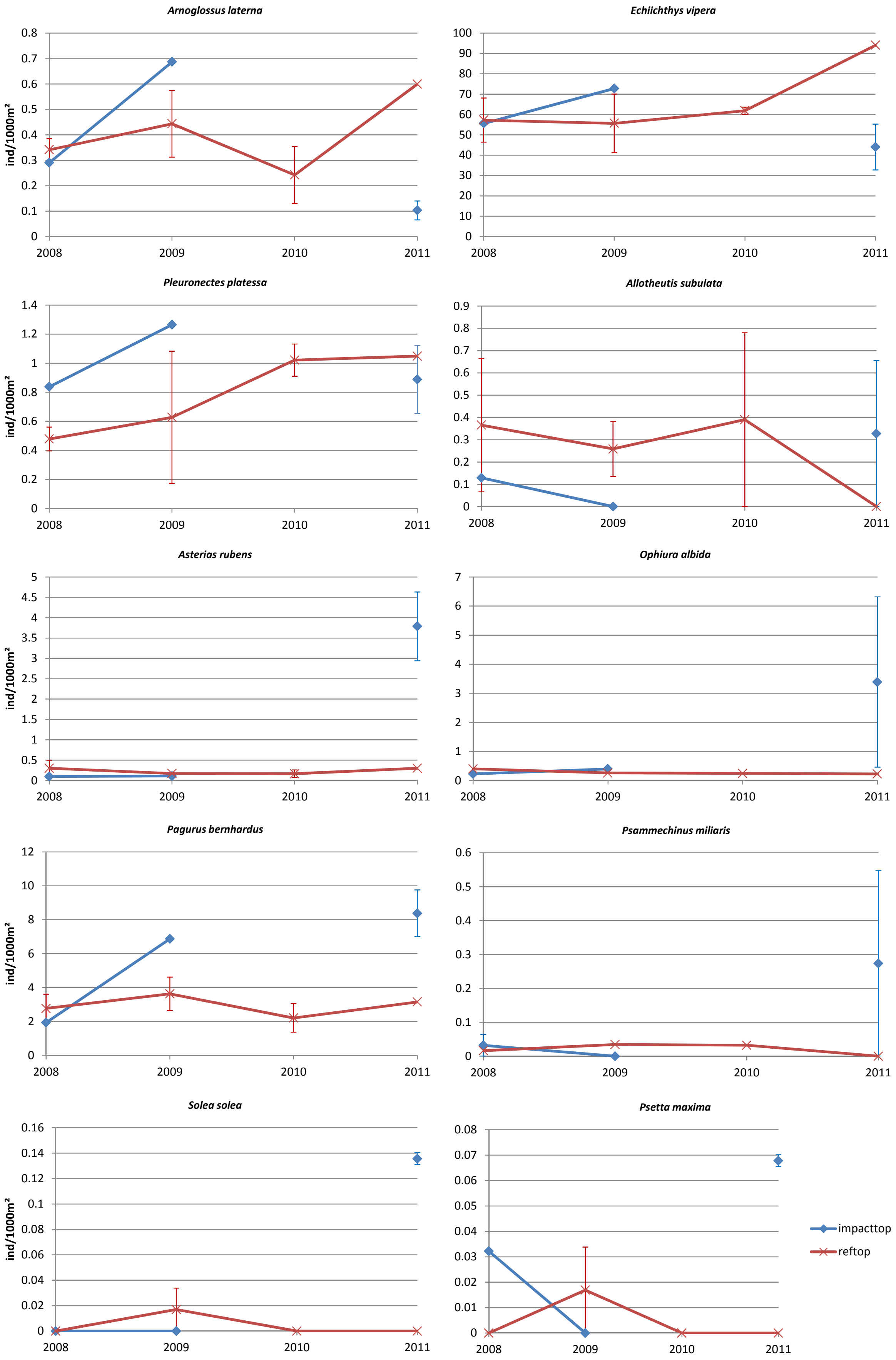


Figure 4. Charts representing differences in autumn density for a selection of species. Whiskers represent standard error

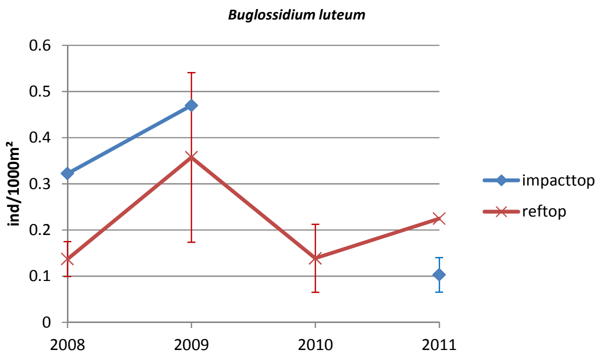


Figure 4. Continued

In the analyses of length-frequency distributions of nine abundant species in 2011 (figure 5), differences between impact top stations and reference top stations included:

- The presence of a full unimodal curve with dominant size class 7cm for the reticulated dragonet (*C. reticulatus*) at the impact stations, while only individuals of 6 to 8 cm were found in the reference samples (dominant size class 6 cm)
- The presence of a few relatively large turbots (*P. maxima*) at the impact stations in autumn 2011
- The presence of larger plaice (*P. platessa*) at the impact stations (dominant size class 23 cm) compared to the references (dominant size class 18 cm) in autumn, and the virtual absence of plaice smaller than 20 cm at the impact stations.

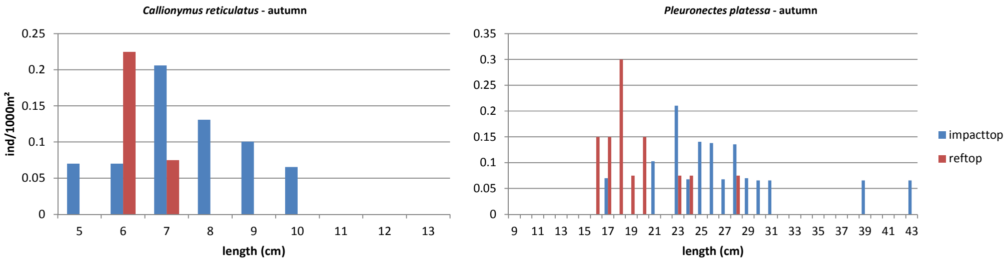


Figure 5. Charts representing differences in length frequency distribution of reticulated dragonet (*Callionymus reticulatus*) and plaice (*Pleuronectes platessa*).

5.3.2. Sandbank gullies

At the ecosystem component level (benthopelagic fish, demersal fish, epifauna), the fluctuations of total density in impact, fringe and reference stations were less outspoken than for the sandbank top samples. For demersal fish in autumn 2011, however, reference values were considerably higher (av. 73 ind /1000m²) compared to impact values (av. 34 ind/1000m²) and fringe values (34 ind/1000m²). Epibenthos densities at impact stations were quite high compared to reference stations, both in autumn (av. 63 vs. 36 ind/1000m²) and spring (av. 59 vs. 25 ind/1000m²). Fringe stations showed similar values as references (figure 6). The evolution of biomass values at the three station groups was quite similar in autumn. In spring 2011, values from impact stations were higher than those from reference stations (av. 140 vs. 78 g ADW/1000m²). The species richness and the Expected Number of Species measure showed no clear differences between impact, fringe and reference stations for the different ecosystem components in both spring and autumn (figure 6).

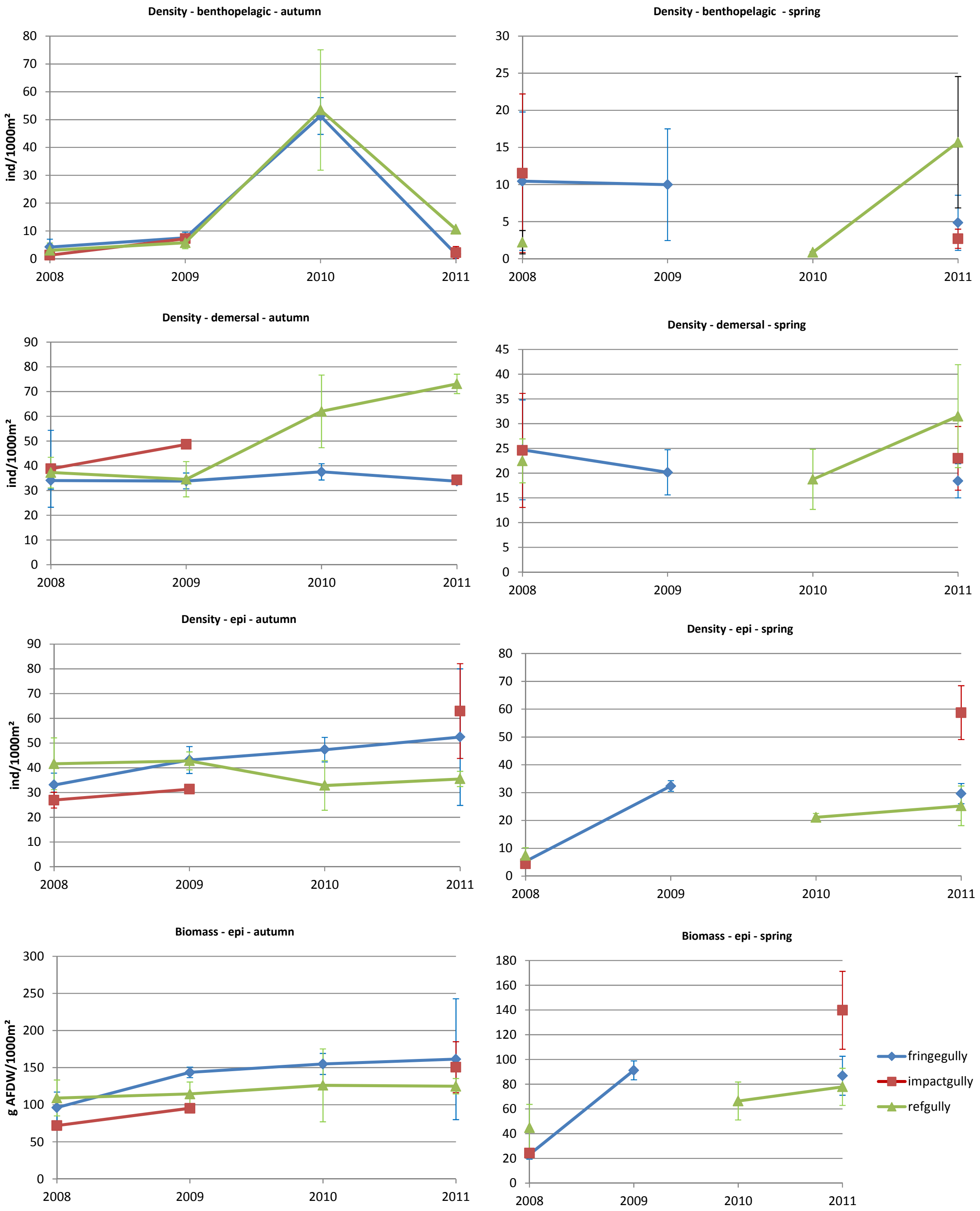


Figure 6. Charts representing differences between gully impact stations, reference stations and fringe stations concerning autumn density, biomass and diversity of benthopelagic fish, demersal fish and epibenthos. Whiskers represent standard error.

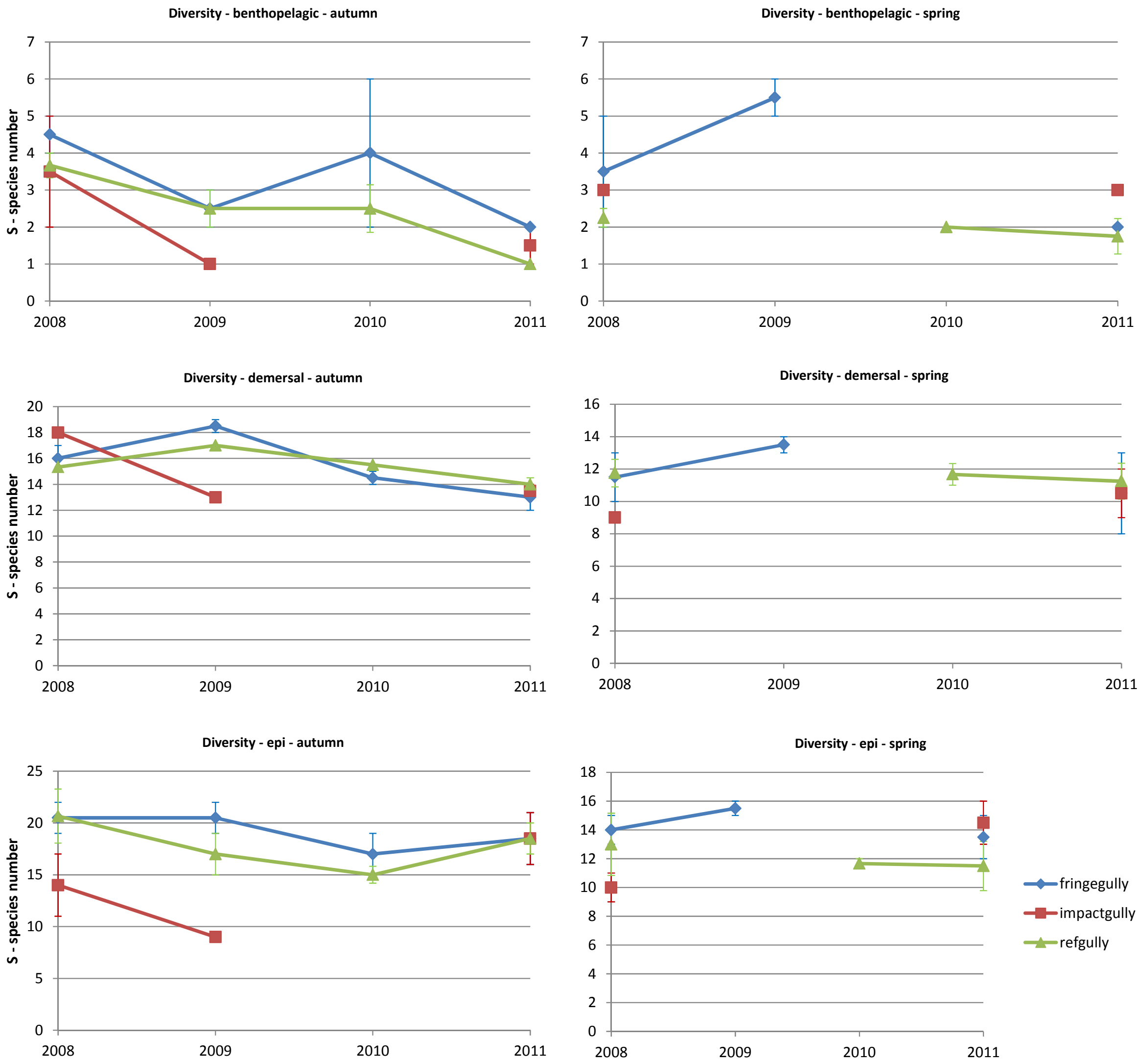


Figure 6. Continued

Benthopelagic fish species such as cod (*Gadus morhua*), sprat (*S. sprattus*) and pouting (*T. luscus*) were only sporadically found at low numbers in the Bligh Bank gullies. Whiting (*M. merlangus*) and horse mackerel (*T. trachurus*) were more abundant than at the sandbank top, especially in autumn. The species hooknose (*A. cataphractus*), dogwhelk (*B. undatum*), the common sea star (*A. rubens*), solenette (*B. luteum*), the common dragonet (*C. lyra*), dab (*L. limanda*), ophiuroids (*O. ophiura & O. albida*), hermit crabs (*P. bernhardus*), plaice (*P. platessa*), sea urchin (*P. miliaris*) and sole (*S. solea*) were more abundant in the gullies than on the sandbank top. Differences between impact stations and reference stations at species level (figure 7) were observed for the species:

- Common starfish (*A. rubens*): density evolved collinearly in autumn and spring until 2010 but then increased substantially at the impact stations compared to the references (spring: av. 10 vs 1 ind/1000m²; autumn: av. 11 vs. 3 ind/1000m²).
- Lesser weever (*Echiichthys vipera*): although reference densities were similar or lower than impact densities in autumn 2008 & 2009, densities were higher in 2011 (29 vs. 14 ind/1000m²).
- Plaice (*Pleuronectes platessa*): density evolved collinearly in reference and impact stations in autumn until 2011, when densities were lower at the impact stations (1.5 ind/1000m²) compared to the references (3 ind/1000m²).

- Urchin (*Psammechinus miliaris*): density evolved collinearly in autumn and spring until 2011 but then increased substantially at the impact stations compared to the references (spring: av. 7 vs 0.2 ind/1000m²; autumn: av. 12 vs. 0.2 ind/1000m²).

In the analyses of length-frequency distributions of nine abundant species in 2011 (figure 8), differences between impact gully stations and reference gully stations included:

- Very low numbers of dab individuals older than one year (>14 cm) at the impact stations compared to the reference and fringe stations in autumn 2011.
- The absence of swimming crabs smaller than 30mm at the impact stations, while these were more abundantly present at the reference stations.
- The presence of a 38cm turbot at one of the impact stations.

Differences between fringe stations and reference stations at species level (figure 7) were observed for the species:

- Squid (*Allotheutis subulata*): densities were similar in autumn 2008 but diverged afterwards, with a divergence of av. 0.4 vs av. 0.07 ind/1000m² between fringe stations and reference stations in 2011.
- Common starfish (*A. rubens*): density evolved collinearly in autumn until 2010 but then increased substantially at the fringe stations compared to the references (av. 10 vs. 2.7 ind/1000m²).
- Lesser weever (*Echiichthys vipera*): densities were similar in autumn 2008 and 2009 but diverged afterwards, with a divergence of av. 14 vs av. 29 ind/1000m² between fringe stations and reference stations in 2011.
- Swimming crab (*Liocarcinus holsatus*): while autumn reference values were more or less stable in the period 2008-2011, values from fringe stations decreased. In 2011, the difference was substantial (av. 0.2 ind/1000m² at fringes vs av 1.2 ind/1000m² at references)
- Plaice (*Pleuronectes platessa*): density evolved collinearly in reference and fringe stations in autumn until 2011, when densities were lower at the fringe stations (1.5 ind/1000m²) compared to the references (2.8 ind/1000m²).
- Whiting (*Merlangius merlangus*): autumn densities in 2010 were considerably higher at the fringe stations compared to the references (av. 1.6 ind/1000m² vs. 0.2 ind/1000m²). In preceding years and in 2011, the values were all very similar.

In the analyses of length-frequency distributions of nine abundant species in 2011, differences between fringe stations and reference stations included:

- Higher numbers of first-year dab (3-8cm) in autumn 2011 compared to the references.

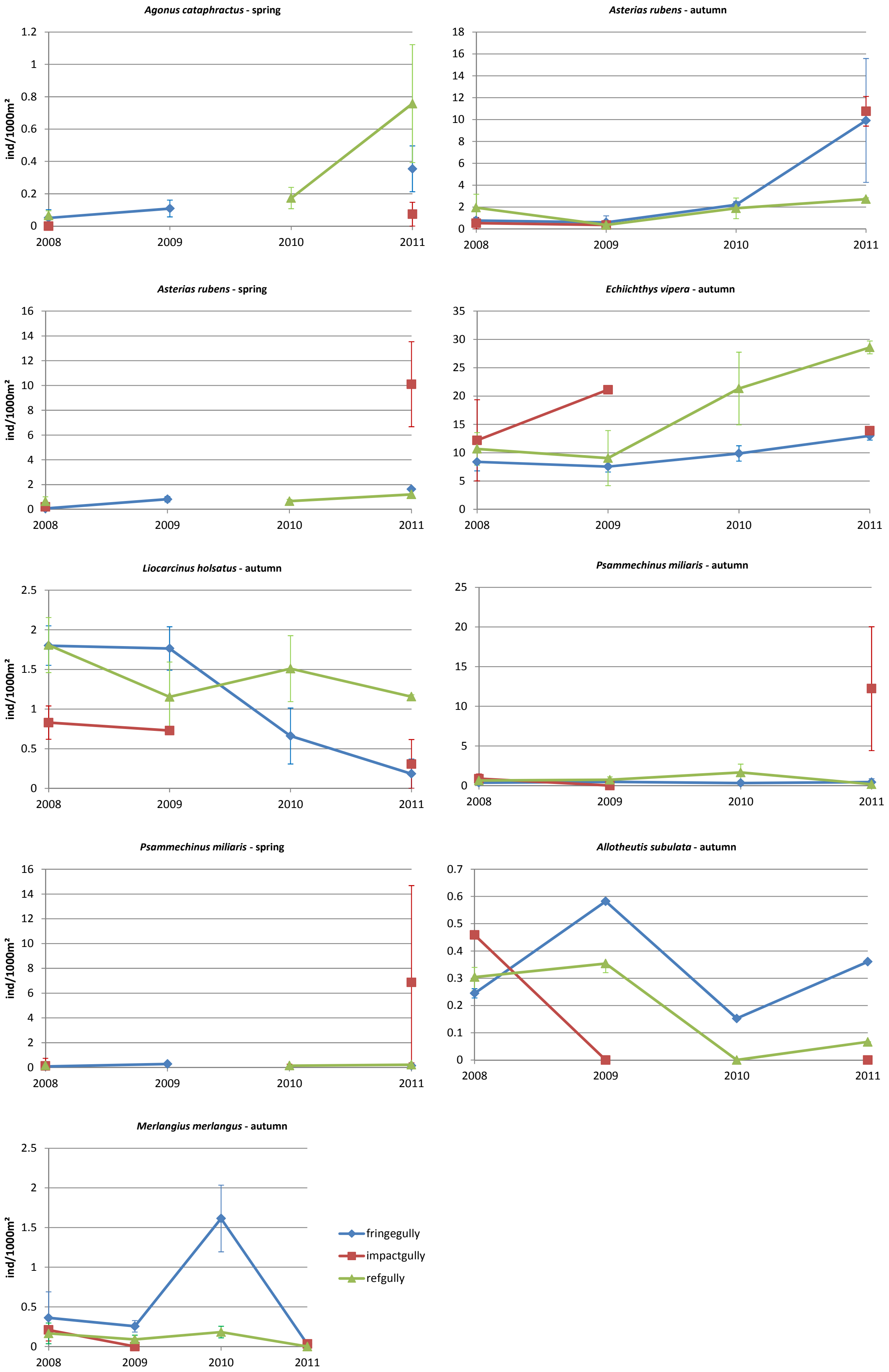


Figure 7. Charts representing differences in density for a selection of species. Whiskers represent standard error.

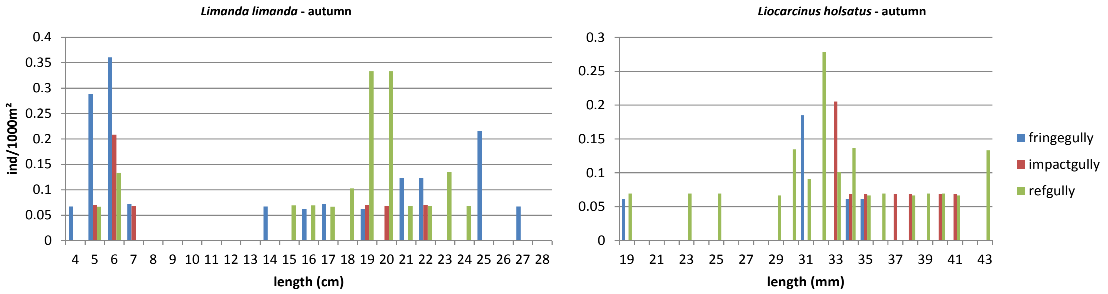


Figure 8. Charts representing differences in length frequency distribution for dab (*Limanda limanda*) and swimming crab (*Liocarcinus holsatus*).

5.4. Discussion

Variations of total densities of benthopelagic fish, demersal fish and epibenthos at the Bligh Bank top showed a virtual absence of benthopelagic fish at all stations, a decrease of demersal fish at the impact stations compared to the references and an increase in epibenthos. These trends are mainly reflections of differences at species level: the decrease in demersal fish reflects the evolution of lesser weever (*E. vipera*), scaldfish (*A. laterna*) and solenette *(B. luteum*), and the increase in epibenthos reflects the evolution of echinoderms (*A. rubens*, *O. ophiura*, *O. albida*, *P. miliaris*) and hermit crabs (*P. bernhardus*). In terms of diversity, differences were only seen for epibenthos, with higher values at the impact stations than at the references. In the gullies, differences in density, biomass and diversity between impact stations, fringe stations and reference stations were less outspoken than on the sandbank tops.

The current observation of a decrease in density of lesser weever within the wind farm in both seasons agrees with the observations of Lindeboom *et al.* (2011). In the current study, similar trends were observed for solenette and scaldfish. The increase of sole (*S. solea*) observed by Lindeboom *et al* (2011), however, was not seen at the Thorntonbank (Vandendriessche *et al.*, 2011a) and was not outspoken at the Bligh Bank wind farm, in which a moderate increase was only seen on the sandbank top in autumn. Lindeboom *et al* (2011) found no clear explanation for why these demersal species reacted this way, but since these species are closely associated with the sediment, it can be hypothesized that their reaction is linked with changes in sediment composition and/or changes within the resident macrobenthos.

More large plaice were found in the impact top stations compared to the references, and a very limited number of plaice smaller than 20cm. Additionally, several large plaice were again observed during diving operations in the Belwind park on 22-23th November 2011, for the first time since that type of monitoring started (J. Reubens, pers. comm.). This indicates either reduced recruitment, increased natural mortality or emigration in the smallest size class, or improved growth. Observations from the Danish Horns Rev wind farm suggest that the generally strong currents within this wind farm and the reduction of the amount of fine sediment may have resulted in low numbers of young fish: plaice depends on the access to sediments that allow the fish to bury themselves in all stages of life. Smaller individuals prefer finer grains while larger individuals prefer more coarse sediments. A lack of finer sediments can hence cause a reduced density of young plaice and a higher abundance of older plaice (Spanggaard, 2006). On the Bligh Bank, small-scale and less-exposed sandy habitats are formed between the rocks and stones that are used to protect the wind turbines. These may provide perfect refugia for plaice and other soft-bottom benthic species. On the other hand, the change in size distribution of plaice could also result from a "closed area" effect. Due to the exclusion of fisheries, a decrease in fisheries-induced mortality of the smaller size classes is expected within wind farms. An increase in undersized fish, however, was not observed in the field. Reactions of plaice populations on fisheries exclusion have been studied in the Dutch "Plaice Box". In this closed area, it was noted that juveniles tended to aggregate near the border of the box since disturbance by trawlers outside the box increased food availability for these small fish (Pastoors *et al.*, 2000). Although the wind farm area is very small compared to the "Plaice Box", and has only been closed for fisheries for a few years, the possible occurrence of similar effects should be considered. Based on the data collected so far, we can state that larger fish may find a refuge within wind farm area. Additional data on length-frequency of plaice in the coming years will provide a better insight in the effect on the smaller size classes.

This refuge hypothesis is also applicable to turbot (*P. maxima*). Although the time series analysis for this flatfish species was based on very few specimens, comparisons with catches elsewhere on the BPNS suggest that wind farms might influence the density and size of this species: 4 out of 13 specimens caught at the BPNS in 2011 originated from inside the wind farm. These four turbots had an average length of 34 cm, while the average length of all other specimens was 23 cm.

The increase in densities of echinoderms and hermit crabs at the impact stations (top and gully) is striking. High numbers of young ophiuroids and starfish, and clusters of urchins have also been observed on and near the turbines. An urchin density of 112 ind/m² was observed on the hard substrates in autumn 2011 (F. Kerckhof, unpubl. data), so the observations from the surrounding soft substrates are probably the direct result of the presence of hard substrates. Especially for the urchin *P. miliaris*, which feeds predominantly on seaweed, hydroids, bryozoans and barnacles, the presence of hard substrates is of great importance. The increased densities, especially in the gullies, may result from dislodgment from the turbines and from the presence of coarse sediments around the wind turbines, which is the preferred habitat for green sea urchins. Additionally, urchins are prone to physical damage by trawling (Lokkeborg, 2005), so this species profits from the absence of beam trawl fisheries within the wind farm.

At the Thorntonbank wind farm, the individuals of the swimming crab *Liocarcinus holsatus* were generally larger at the impact station in 2010 compared to the reference stations (Vandendriessche *et al.*, 2011a). A similar observation was done in 2011 in the Bligh Bank impact gully stations. This may reflect either increased growth due to a high food availability or increased predation pressure eliminating smaller individuals. Small swimming crabs have been found in stomachs of pouting (*Trisopterus luscus*) in the Thorntonbank wind farm (Reubens *et al.*, 2011), so changes in their length-frequency distribution might signal their importance as prey item supporting ‘reef’ fishes.

In 2010, high autumn densities of small whiting (*M. merlangus)* were recorded at the Thorntonbank impact station. Similar high values were also seen at the Bligh Bank fringe stations in 2010 (no impact samples available), but did not persist in 2011. Consequently, these high densities may be a result of the species’ demography rather than a wind farm effect.

Other differences between fringe stations and reference stations were seen for the common starfish (*A. rubens*) and the squid (*A. subulata*), showing higher densities at the fringes, and for lesser weever (*E. vipera*), swimming crab (*L. holsatus*) and plaice (*P. platessa*), showing lower densities at the fringes. Whether these observations are the effect of changing activities near the Bligh Bank wind farm after construction has to be confirmed by VMS data analyses for the period 2010-2011. The data from 2006-2009, however, indicate that the area was not an important fishing ground for the Belgian fleet (Vandendriessche *et al.*, 2011b).

5.5. Conclusion

One year after the start of construction activities at the Bligh Bank wind farm, some remarkable differences between impact and reference stations were observed. In general, we saw a decrease in demersal fish and an increase in epibenthos. The changes in demersal fish may have resulted from the absence of fisheries in the area or local changes in sedimentology and infaunal communities. For commercially important flatfish, we observed higher densities (turbot, sole) and/or changes in length-frequency distribution (turbot, plaice). This may signal a refugium effect, but bearing in mind that large flatfish such as sole do not stay within a wind farm for longer periods (Lindeboom *et al.*, 2011), this effect will be limited. The increase in epibenthos probably resulted from the presence of hard substrates and their fouling communities and from the absence of fisheries. The increase, however, was mainly seen for dominant, scavenging species such as echinoderms and hermit crabs. Signs of recovery of populations of long lived species vulnerable to trawling, as was seen for *Ostrea edulis* and *Sertularia cupressina* at Horns Rev (Anonymous, 2006), were not yet observed in autumn 2011 at the Bligh Bank. Some differences between fringe stations and reference stations were described but they could not straightforwardly be linked to fringe effects resulting from changing activities in the close vicinity of the wind farm concession.

5.6. References

Andersson M.H., Berggren M., Wilhelmsson D., Öhman M.C., (2009). Epibenthic colonization of concrete and steel pilings in a cold-temperate embayment - a feld experiment. Helgoland Marine Research 63: 249-260.

Berkenhagen J.; Döring R.; Fock H.; Kloppmann O., Ferdinand, M.H.; Pedersen S.A.; Schulze T., (2010). Decision bias in marine spatial planning of offshore wind farms: Problems of singular versus cumulative assessments of economic impacts on fisheries. Marine policy, Band 34 (3), 733-736.

Anonymous, (2006). Danish Off shore Wind – Key Environmental Issues. Published by DONG Energy, Vattenfall, The Danish Energy Authority and The Danish Forest and Nature Agency. 144pp

Brabant R., Degraer S. & Rumes B., (2011). Offshore wind energy development in the Belgian part of the North Sea & anticipated impacts: an update. pp. 9-16. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. 157 pp. + annex.

Derweduwen J.; Vandendriessche S.; Hostens K., (2010). Monitoring the effects of the Thorntonbank and Bligh Bank wind farms on the epifauna and demersal fish fauna of soft bottom sediments. Thorntonbank: status during construction (T2). Bligh Bank: status during construction (T1). *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2010). Offshore wind farms in the Belgian part of the North Sea: Early environmental impact assessment and spatio-temporal variability. pp. 105-132.

Lindeboom, H.J., Kouwenhoven, M.J., Bergman, N., Bouma, S., Brasseur, S., Daan, R., Fijn, R.C., de Haan, D., Dirksen, S. , van Hal, R., Lambers, H.R., ter Hofstede, R., Krijgsveld, K.L., Leopold, M., Scheidat, M., (2011). Short-term ecological effects of an offshore wind farm in the Dutch coastal zone; a compilation. Environmental Research Letters. 6 035101.

Jaworski, A., Solmundsson, J. & Ragnarsson, S. A., (2006). The effect of area closures on the demersal fish community off the east coast of Iceland. ICES Journal of Marine Science, 63: 897-911.

Lokkeborg S. (2005). Impacts of trawling and scallop dredging on benthic habitats and communities. FAO Technical Fisheries Paper No 472. Rome. FAO. 58pp

Pastoors M. A., Rijnsdorp A. D. & Van Beek F. A. (2000). Effects of a partially closed area in the North Sea (‘‘plaice box’’) on stock development of plaice. – ICES Journal of Marine Science, 57: 1014–1022.

Reubens J.T.; Degraer S.; Vincx M., (2011). Aggregation and feeding behaviour of pouting (*Trisopterus luscus*) at wind turbines in the Belgian part of the North Sea. Fish. Res. 108(1): 223-227.

Spanggaard (ed) (2006) EIA Report Fish, Horns Rev 2 Offshore Wind Farm. 68pp

Vandendriessche S., Derweduwen J. & Hostens K., (2011a). Monitoring the effects of offshore windmill parks on the epifauna and demersal fish fauna of soft-bottom sediments: baseline monitoring. pp. 65-81. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. 157 pp. + annex.

Vandendriessche S., Hostens K., Courtens W. & Stienen E.W.M., (2011b). Monitoring the effects of offshore wind farms: evaluating changes in fishing effort using Vessel Monitoring System data: targeted monitoring results. pp. 83-92. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management

Unit of the North Sea Mathematical Models. Marine ecosystem management unit. 157 pp. + annex.

Wilhelmsson D., (2009). Aspects of offshore renewable energy and the alterations of marine habitats. PhD Thesis, University of Stockholm, 56pp.

Chapter 6. The diet of demersal and semi-pelagic fish in the Thorntonbank wind farm: tracing changes using stomach analyses data

J. Derweduwen[*], S. Vandendriessche, T. Willems
& K. Hostens

Institute for Agricultural and Fisheries Research (ILVO-Fisheries), Bio-Environmental Research Group. Ankerstraat 1, 8400 Ostend

[*] *Corresponding author: jozefien.derweduwen@ilvo.vlaanderen.be*

Herring stomach

Photo Karl Van Ginderdeuren / ILVO-Fisheries

Abstract

This chapter focuses on the diet of six demersal and semi-pelagic fish species (dab, solenette, dragonet, lesser weever, whiting and horse mackerel) in the Thorntonbank wind farm and its surrounding areas. Stomach analyses were conducted to unravel changes in feeding patterns and to discriminate between effects of the presence of the turbines and effects as a result of fisheries displacement at the border of the wind farm concession area. Differences were observed between impact, fringe and reference stations. The fullness index indicated that fish had a fuller stomach close to the wind turbines and at the borders of the concession area. This might be an indication of a higher food availability around the wind turbines. The stomach content of dab revealed more amphipods and especially hard substratum species (e.g. *Phtisica marina*) in the impact area compared to the reference area. However, the most abundant hard substratum species present on the turbines (i.e. the amphipod *Jassa herdmani* and the crab *Pisidia longicornis)* were not found in any of the dab stomachs. This can probably be linked to the sampling distance (500-1500m) or to the prey preferences of dab.

In general, differences in feeding patterns between sampling stations were observed. Whether these differences originated from the wind turbine presence or from changes in fisheries activities can only be unequivocally confirmed by replication within and between the wind farm(s) and by an optimisation of the sampling strategy.

Samenvatting

Dit hoofdstuk behandelt de dieetsamenstelling van zes demersale en semi-pelagische vissoorten (schar, dwergtong, pitvis, kleine pieterman, wijting en horsmakreel) afkomstig uit het Thorntonbank windmolenpark en de omringende gebieden. Om mogelijke veranderingen in voedingspatronen te ontrafelen en om onderscheid te maken tussen effecten van de aanwezigheid van de turbines en effecten die het resultaat zijn van een verschuiving van visserijactiviteiten aan de rand van het windmolenconcessiegebied, werd gebruik gemaakt van maaganalysedata. Verschillen werden waargenomen tussen impact-, ‘fringe’- en referentiestations. De fullness-index toonde aan dat vissen een vollere maag hadden dicht bij de windmolens en aan de rand van het concessiegebied. Dit kan wijzen op de beschikbaarheid van meer voedsel in de buurt van de windmolens. In de maag van schar vonden we meer amfipoden en vooral hardsubstraatsoorten (vb. *Phtisica marina*) in het impactgebied dan in het referentiegebied. Toch waren de meest abundante hardsubstraatsoorten op de turbines (*Jassa herdmani* en *Pisidia longicornis*) niet terug te vinden in de magen van schar. Dit kan te wijten zijn aan de staalnameafstand (500-1500m) of aan de prooivoorkeur bij schar. Algemeen werden verschillen in voedingspatronen tussen de stations waargenomen. Of deze verschillen veroorzaakt zijn door de aanwezigheid van windmolens of door een verschuiving van visserijactiviteiten, kan enkel bekrachtigd worden door replicatie binnen en tussen het/de windmolenpark(en) en door een optimalisering van de staalnamestrategie.

6.1. Introduction

Artificial hard substrates introduced in wind farms are covered by hard substratum fauna (Petersen & Malm, 2006). Consequently, we expect some demersal and semi-pelagic fish species to forage on hard substrate prey species in the vicinity of the Thorntonbank wind turbines. In the North Hoyle wind farm (GB), large schools of juvenile whiting (*Merlangius merlangus*) fed on the tube building amphipod *Jassa falcata*, which was dominantly present on the turbines (May, 2005). Stomach content analyses of pouting (*Trisopterus luscus*) aggregating around the turbines on the Belgian Thorntonbank also indicated a preference for hard substrate preys (e.g. *Jassa herdmani* and *Pisidia longicornis*) (Reubens *et al.* 2011).

Some fish species may exhibit changes in their diet due to competition with fish species which are new in the area or with fish of which the density drastically increased since the construction of the wind farm. Recent observations on the Bligh Bank (Derweduwen *et al.,* 2012), for example, showed more large and less small individuals of plaice *(Pleuronectes platessa)*. Other species, such as solenette (*Buglossidium luteum*), may profit from the absence of that smaller size fraction, resulting in a reduced interspecific competition for prey (Schückel *et al.*, 2012). Changes in diet composition of demersal and semi-pelagic fish can also result from the exclusion of fisheries activities from wind farms and their safety buffers, and from a local reallocation of fishing effort (Berkenhagen *et al.*, 2010), so called 'fringe effects'.

Shifts in feeding pattern of locally abundant species can be verified by means of stomach analyses. In the current study, the feeding patterns of demersal and semi-pelagic fish in the reference areas were compared with the feeding patterns of the fish in the impact areas (in the vicinity of the turbines) and the fringe areas (at the borders of the concession area) to answer the following questions:

(1) Is the presence of the wind turbines affecting the diet of fish?

(2) Are there any fringe effects with regard to the diet composition at the border of the concession area?

6.2. Material and Methods

6.2.1. Sampling

In spring 2009 and autumn 2010, samples for stomach analyses were collected at 4 stations at the Thorntonbank using an 8m shrimp trawl (Vandendriessche *et al.*, 2012; this volume). The impact station and its adjoining reference station are referred to as IMP and REF1, respectively. The fringe station (at the border of the concession area) and its adjoining reference station are called FRI and REF2, respectively.

Per station, a number of individuals for the following species were collected and injected with formaldehyde (35 %) for preservation: lesser weever (*Echiichthys vipera*), horse mackerel (*Trachurus trachurus*), solenette (*Buglossidium luteum)*, dragonet (*Callionymus sp.*), dab (*Limanda Limanda*) and whiting (*Merlangius merlangus*).

The specimens were stored in formaldehyde (8 %) until analysis.

6.2.2. Laboratory treatment

The intact stomachs were removed under a stereoscopic microscope, by cutting above the oesophagus and below the large intestine. An incision was made along the longitudinal axis and the contents were emptied onto a Petri dish with a few drops of deionised water. All prey items encountered in the stomachs, were counted and identified. If possible, prey items were identified to species level but most of them were classified into a higher taxonomic level (e.g. order) due to fragmentation or partial digestion. Both fish and stomach contents were placed into separate vials for potential further investigation and subsequent drying. After identification, the stomach contents were placed in pre-weighed aluminium foil cups, dried at 110°C for 5 hours, weighed, incinerated in ceramic cups at 550°C for 15 minutes and cooled to room temperature in a desiccator for 2 hours before weighing in order to obtain Ash Free Dry Weight.

6.2.3. Stomach content analysis

For the quantitative analysis of the stomach contents, the fullness index (FI) was used:

$$FI = \frac{S_i}{W_i} x100$$

where S_i is the ash-free dry weight (ADW) of the stomach content in milligram (mg) and W_i is the ash-free dry weight (ADW) of the fish (mg). In addition to the average fullness index, the percentage of empty stomachs was calculated for each fish species, season, station and length category.

Furthermore, frequencies of occurrence and numerical percentages of prey items were calculated to characterise the stomach contents (Hyslop, 1980). The frequency of occurrence (*FO%*) calculates the percentage of the total number of stomachs in which the specific prey species occurs:

$$FO\% = \frac{FO_i}{FO_t} x100$$

where FO_i is the number of stomachs in which the species '*i*' occurs, and FO_t is the total number of full stomachs.

The diet composition was expressed as a numerical percentage (*N%*):

$$N\% : \frac{number\ of\ individuals\ of\ prey\ type\ i}{total\ number\ of\ ingested\ prey\ items} x100$$

Per fish species, season, year and station, a length-frequency analysis was carried out, during which fish were assigned to three length classes (small, medium, large; defined per species based on minimum and maximum length). Length classes with less than five fish (empty stomachs included) were omitted from all analyses. Stomach data were compared for two station pairs based on the evaluated impact (cf. Vandendriessche *et al,* 2011a):

- Station pair 1: REF1 (reference) – IMP (impact)
  → Evaluated impact: wind turbine presence (sandbank top)
- Station pair 2: REF2 (reference) – FRI (fringe)
  → Evaluated impact: fringe effects (gullies)

Fringe effects -due to the presence of wind turbines- could not be examined due to the fact that no wind turbines were present yet during the sampling period (spring 2009 - autumn 2010). However, it may be possible to detect fringe effects arising from altered fisheries activities.

Special attention was paid to the occurrence of hard-substrate prey species (e.g. *Phtisica marina, Jassa herdmani, Pisidia longicornis*) in the vicinity of the wind turbines. This information was subsequently integrated in the results and discussion.

Only in the case of dab, data of two seasons (spring 2009 and autumn 2010) were available. However, the number of data from spring 2009 was too limited to analyse the diet composition.

The limited number of sampled stations and the fact that not all station pairs covered enough individuals of a certain length category obstructed the analysis of the data. This was the case for horse mackerel (*Trachurus trachurus*), for which only one station (IMP) yielded enough specimens.

6.3. Results

- Dab (*Limanda limanda*)

We distinguished several differences in diet composition between impact, fringe and reference stations. The diet of dab generally consisted of amphipods, decapods and polychaetes with a frequency of occurrence of 57 %, 35 % and 30 % respectively (Table 1). It strongly differed at the reference station REF1 (mainly mysids) compared to the impact station IMP (dominantly amphipods) (Fig. 1A). However, the frequency of occurrence of mysids (4%) (Table 1) indicated that only a single/few individual(s) of dab consumed a high percentage of mysids. The dwarf swimming crab *Liocarcinus pusillus* and the hard substratum amphipod *Phtisica marina* occurred in the stomachs of dab originating from IMP but were not found at REF1. Other hard substratum species (e.g. *Jassa herdmani* and *Pisidia longicornis*) were not found.

When comparing the reference station REF2 and the fringe station FRI (Fig. 1A), we observed that more different prey taxa (e.g. ophiurids, copepods and cumaceans) were consumed at reference station REF2. *Phtisica marina* occurred in the stomachs originating from station FRI but did not appear in the stomachs from station REF2.

The fullness index showed higher values for the samples originating from station FRI (0.28) compared to the samples from station REF2 (0.17) and for the IMP samples (0.15) compared to those from REF1 (0.05) (Table 2 & Fig. 2). The medium sized dab from autumn 2010 however, had a fuller stomach at station REF2 (0.28) compared to station FRI (0.18) (Table 2 & Fig.2). So, fish generally had a fuller stomach at the fringe station and the impact station than at the reference stations.

Other differences between reference and fringe stations and between small and medium sized dab were of less significance.

- Solenette (*Buglossidium luteum*)

The diet composition of solenette was not uniform and differed between stations and length categories. With a frequency of occurrence of 67 %, the diet particularly consisted of amphipods, followed by cumaceans (FO% = 37) and decapods (FO% = 22) (Table 1).

In the stomachs originating from station REF2, amphipods were the most dominantly present taxon and decapods were absent (Fig. 1B). In the FRI stomach samples however, mainly cumaceans and decapods were found. When we look into more detail at the different length categories of solenette at station FRI, it is notable that the large individuals were foraging on cumaceans and decapods only, whereas the medium sized individuals also ingested amphipods, polychaetes and copepods.

The fullness index for solenette showed the lowest values (0,01-0,05) from all examined fish species (Table 2) and was higher at the fringe station FRI than at the reference station REF2, both for medium and large sized individuals (Fig. 2). This can partially be linked to the high percentages of empty stomachs (33-71%) (Table2).

- Dragonet (*Callionymus* sp.)

The food preference of small dragonet particularly went out to amphipods and decapods (Fig. 1C). With a frequency of occurrence of 50 % (Table1), amphipods and decapods were equally important in the diet of dragonet.

The number of prey taxa was higher at station FRI compared to station REF2. Bivalves and cumaceans were only preyed upon at station REF2, whereas copepods, gastropods, mysids and polychaetes were also present in the stomachs of dragonet originating from station FRI. There were also more *P. marina* individuals found in the stomachs from FRI compared to the stomachs originating from REF2.

Dragonet individuals had a fuller stomach at REF2 (FI = 0,56) compared to FRI (FI = 0,2) (Table 2 & Fig.2), with no empty stomachs at REF2 and 59% of empty stomachs at FRI (Table 2).

- Lesser weever (*Echiichthys vipera*)

In the stomachs of lesser weever, mysids were numerically the most important prey item (Fig. 1D). Especially the species *Gastrosaccus spinifer* was frequently found. With a frequency of occurrence of 56% (Table 1) however, fish was the most frequently encountered taxon in the stomachs of lesser weever, whereas the frequency of occurrence of mysids was only 44%. Decapods and mysids were more preyed upon at REF2, whereas fish was more important in the diet of lesser weever at FRI (Fig. 1D).

The fullness index was higher at station FRI (1,3) than at station REF2 (0,87). The percentage of empty stomachs was 39% at the FRI station and 47% at the REF2 station (Table 2).

- Whiting (*Merlangius merlangus*)

In general, decapods (especially *Crangon crangon*) occurred most frequently in the diet of whiting (FO% = 79), followed by copepods and fish (FO% = 46) (Table 1). Comparing the stations REF2 and FRI, the stomach content was rather similar but the proportion in terms of percentage however, differed quite remarkably (Fig. 1E). Copepods were of less significance at station FRI, whereas decapods, amphipods and fish contributed more to the diet of whiting at this station. Cumaceans only occurred in the stomachs of whiting originating from REF2.

With no empty stomachs found at the FRI station, whiting individuals had a fuller stomach (FI = 1,23) than at the REF2 station (FI = 0,78) (Table 2).

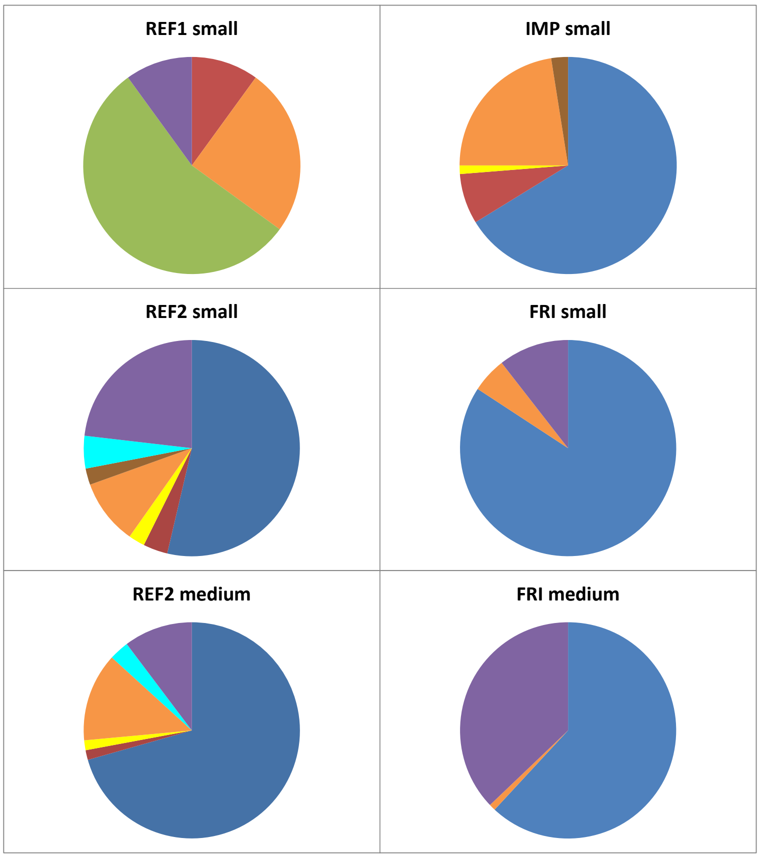
- Horse mackerel (Trachurus trachurus)

With a frequency of occurrence of 89%, the diet of horse mackerel mainly consisted of copepods (i.e. Calanoida). Amphipods and decapods (FO% = 78) were the second most important prey taxa found in the stomachs of horse mackerel (Table 1). Numerically however, the contribution of amphipods and decapods was almost negligible in the diet compared to the contribution of copepods (Fig. 1F). The fullness index for horse mackerel was 0,98 and the percentage of empty stomachs was 0%.

Table 1: Frequency of occurrence (%FO) of several prey taxa for each analysed fish species.

	dab	solenette	dragonet	lesser weever	whiting	horse mackerel
Algae	0	0	0	0	0	0
Amphipods	56	67	50	22	21	78
Bivalvia	6	0	0	0	0	0
Bryozoa	2	4	0	0	0	6
Copepoda	19	19	6	7	46	89
Cumacea	15	37	6	4	8	0
Decapoda	35	22	50	33	79	78
Gastropoda	7	0	11	0	4	0
Hydrozoa	2	0	0	0	0	0
Isopoda	0	0	0	0	0	0
Mysida	4	4	11	44	13	17
Ophiurida	7	0	0	0	0	0
Pisces	20	0	0	56	46	0
Polychaeta	30	19	6	4	8	0

A. Dab (*Limanda Limanda*)



B. Solenette (*Buglossidium luteum*)

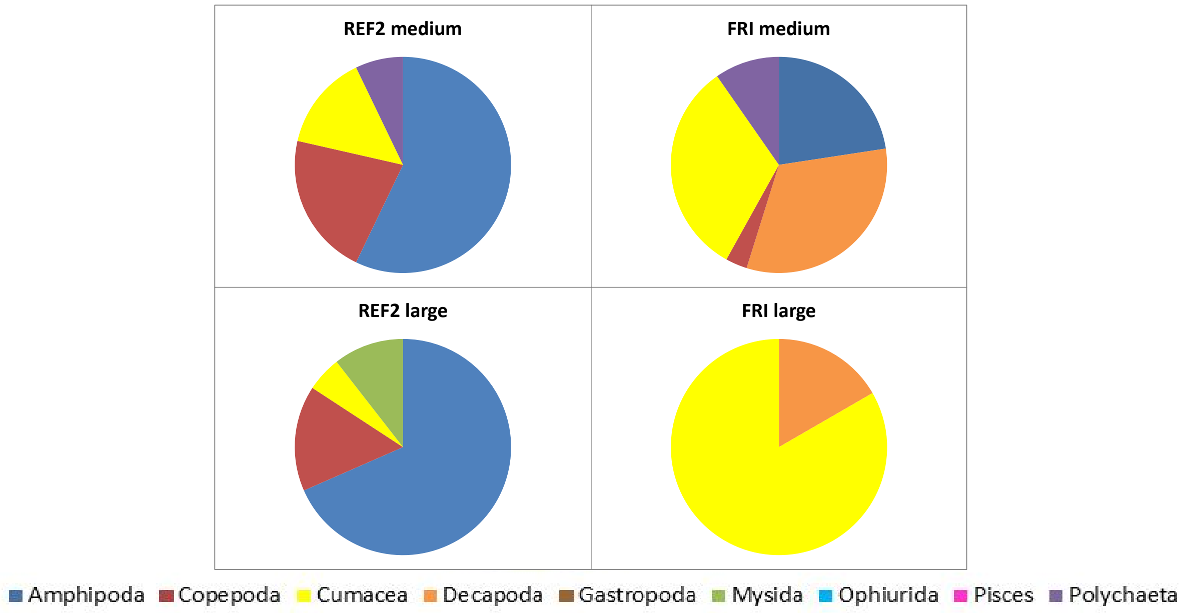
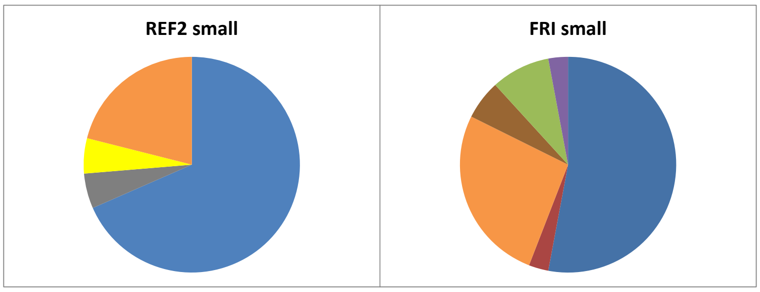
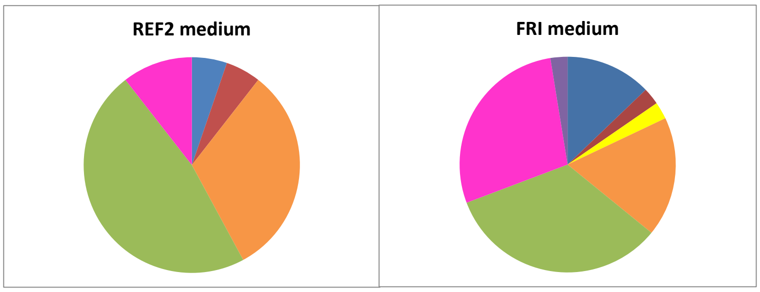


Figure 1. Pie charts representing the diet composition based on numerical percentages (N%) of prey items, for several fish species in autumn 2010 for the stations REF1, IMP, REF2 and FRI.

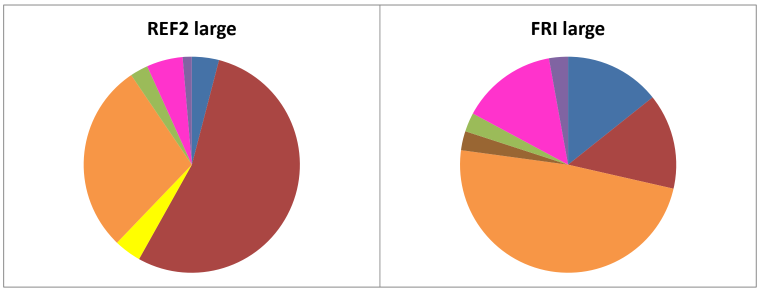
C. Dragonet (*Callionymus* sp.)



D. Lesser weever (*Echiichthys vipera*)



E. Whiting (*Merlangius merlangus*)



F. Horse mackerel (*Trachurus trachurus*)

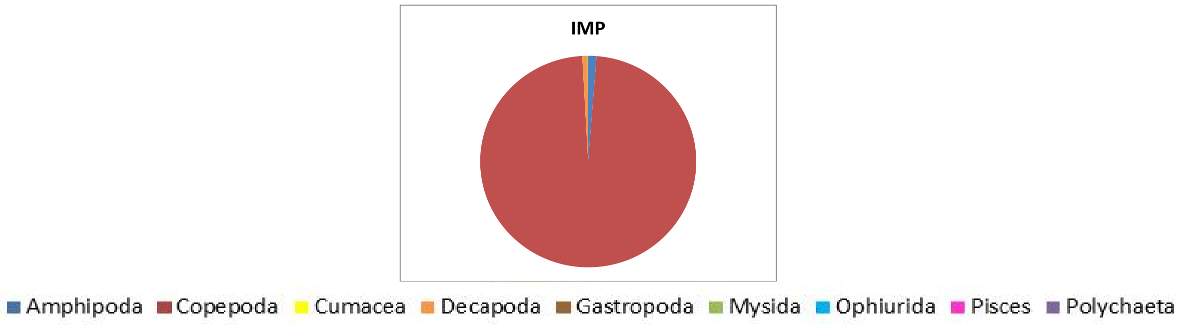


Figure 1. Continued.

Table 2: Fullness Index (FI) and percentage of empty stomachs per species, season, station and length category.

			small		medium		large	
			FI	%empty	FI	%empty	FI	%empty
dab	spring 2009	REF1			0.07	33		
		IMP			0.24	14		
	autumn 2010	REF1	0.05	15				
		IMP	0.15	9				
		REF2	0.17	0	0.28	0		
		FRI	0.28	0	0.18	0		
solenette	autumn 2010	REF2			0.02	50	0.01	33
		FRI			0.05	37	0.02	71
dragonet	autumn 2010	REF2	0.56	0				
		FRI	0.20	59				
lesser weever	autumn 2010	REF2			0.87	47		
		FRI			1.30	39		
whiting	autumn 2010	REF2					0.78	12
		FRI					1.23	0

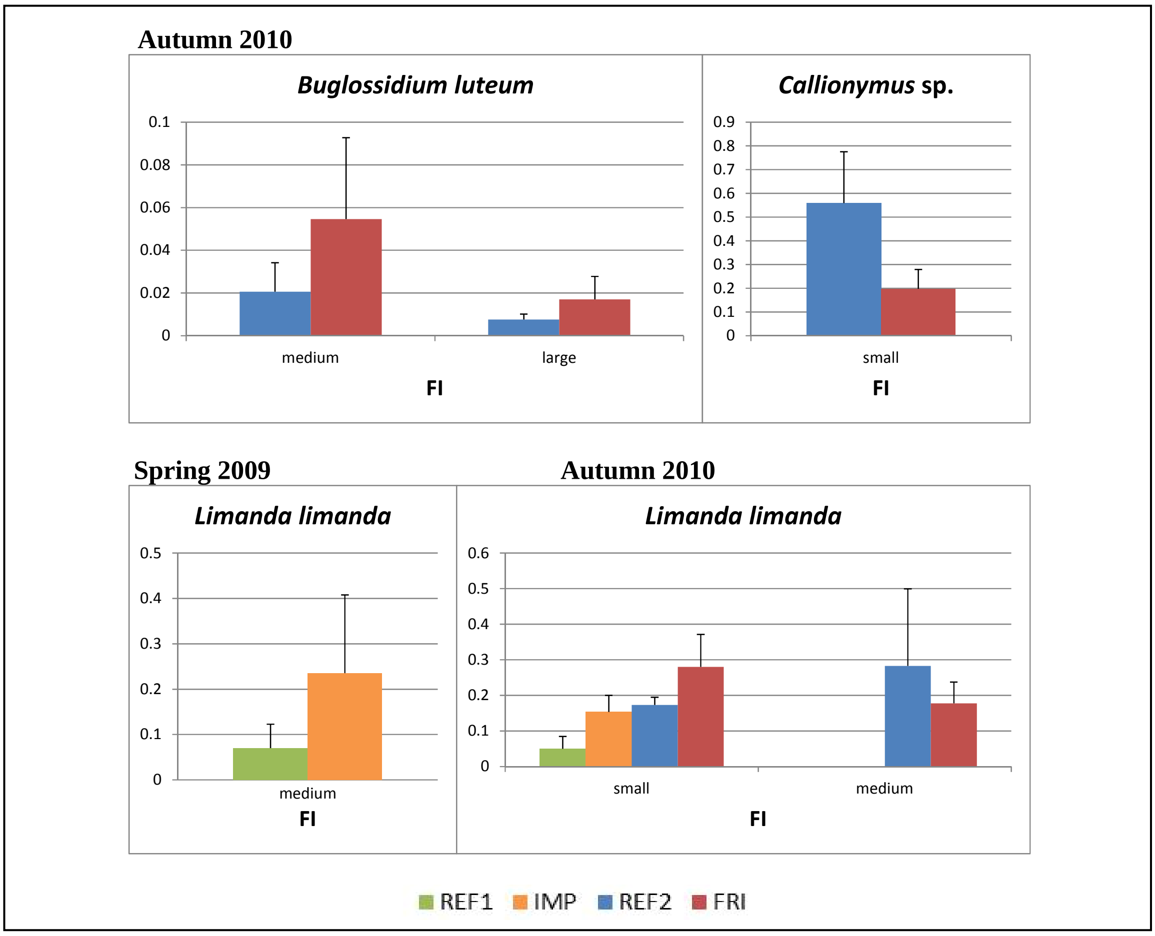


Figure 2: Fullness Index (FI) of a selected number of species per season and length category (+ SE).

6.4. Discussion

This study aimed to reveal possible changes in feeding patterns of six fish species in the vicinity of the Thorntonbank wind farm. Stomach analyses were conducted to (1) investigate whether the presence of the wind turbines had an effect on the fish's diet, and (2) investigate whether there were fringe effects with regard to the diet composition at the border of the concession area.

Despite some practical constraints (e.g. identification of partially digested prey, adequate number of specimens per station pair, seasonal differences in abundance), we observed several differences in diet composition between impact, fringe and reference stations.

The diet of dab strongly differed in the reference station (mainly mysids) compared to the impact station (mainly amphipods). The dwarf swimming crab *Liocarcinus pusillus* -which likes coarser sediments- and the hard substratum amphipod *Phtisica marina* only occurred in the stomachs of dab originating from the impact station but were not found at the reference station. Other hard substratum species (e.g. *Jassa herdmani* and *Pisidia longicornis*) were not found, although they are abundantly present on the wind turbines (Kerckhof *et al.*, 2010) and were the most abundant prey types in the stomachs of pouting (*Trisopterus luscus*) caught in the vicinity of the turbines (Reubens *et al.*, 2011). This probably can be linked to (1) the fact that pouting was caught nearby the turbines, whereas the fish in this study were sampled at a distance of 500-1500m from the turbines or (2) to different prey preferences of pouting and the fish species in this study. The differences between reference station and fringe station were rather insignificant.

Considering the numerical percentages, lesser weever mostly foraged on mysids (in accordance with Vasconcelos *et al.*, 2004). However, when taking into account the frequency of occurrence, fish (56%) contributed more to the diet than mysids (44%), especially in the fringe station. This might be an indication of the higher availability of small fish at the fringe station. However, Vandendriessche *et al.* (2011a) noted an absence of the smallest size classes of sole (*Solea solea*) at the fringe station, which could be attributed to an increased indirect fishing mortality (discards) or to changes in the local benthic community.

Whiting's diet principally consisted of decapods, mainly the brown shrimp *Crangon crangon*. However, there is no direct link between the consumption and the availability of brown shrimp because densities were virtually identical in the reference and fringe stations (Vandendriessche *et al.,* 2011a). We were not able to confirm the findings of May (2005), who described large shoals of juvenile whiting foraging on *Jassa falcata*, due to the lack of samples in the wind farm area.

The fullness index mostly showed higher values in the fringe and impact station than in the reference station. In other words, fish had a fuller stomach close to the wind turbines and at the borders of the concession area. This might be an indication of a higher food availability around the wind turbines. For Danish wind farms, Leonhard & Pederson (2006) estimated that the availability of food for fish around the turbine sites directly increased by a factor of approximately 50 after the introduction of hard substrates, in comparison with the former sandy area. There also may be more food available at the border of the Thornton bank concession area since beam trawl activities (Belgian fleet) and subsequent discard mortality have increased in that area between 2008 and 2009. This increase was more distinct at the fringe station than at the reference station (Vandendriessche *et al.*, 2011b).

We encountered a great percentage of empty stomachs, especially for the species solenette in autumn 2010. The feeding activity of solenette peeks in summer and drastically declines in winter (Schuckel *et al.,* 2011). So, the sampling period (spring and autumn) could possibly explain the high percentage of empty stomachs.

The high percentage of empty stomachs found in lesser weever (*E. vipera*) is in accordance with Quiniou (1978), Dauvin (1988) and Creutzbert & Witte (1989) and is probably due to the burrowing behaviour of this species, the discontinuous prey captures as suggested by Quiniou (1978) and its nocturnal activity (Lewis, 1976; Wheeler, 1978 in Vasconcelos *et al.,* 2004).

Another factor that also might explain the high percentage of empty stomachs in general, is the type of prey. Highly digestible preys (such as bivalve siphons) are more difficult to identify. A recommendation for future research is to implement genetic prey identification and stable isotope analyses. These techniques are able to identify assimilated (Pitt *et al.*, 2009) and digested sources and to define the position of a species in the food chain, so the problem of empty stomachs or stomachs with an undefinable content could hence be tackled.

6.5. Conclusion

The feeding guild structure of six fish (dab, solenette, dragonet, lesser weever, whiting and horse mackerel) species has been examined in the Thorntonbank wind farm and its surrounding areas. Differences were observed between impact, fringe and reference stations. The stomach content of dab revealed more amphipods and especially hard substratum species (e.g. *Phtisica marina*) in the impact area compared to the reference area. However, the most abundant hard substratum species *Jassa herdmani* and *Pisidia longicornis* present on the turbines (Kerckhof *et al.,* 2010; Reubens *et al.,* 2011) could not be found in the stomachs of dab. This probably can be linked to the sampling distance or to different prey preferences of dab. The fullness index indicated that fish had a fuller stomach close to the wind turbines and at the borders of the concession area. This might be an indication of a higher food availability around the wind turbines. Differences between fringe and reference stations were found but could not directly be linked to the higher fisheries activities in that area (Vandendriessche *et al.,* 2011b).

Whether these differences originate from the wind turbine presence, changes in fisheries activities or natural variation can only be unequivocally confirmed by replication within (Thorntonbank) and between the wind farm(s) (Thorntonbank, Bligh Bank and eventually also the Lodewijkbank) and by an optimisation of the sampling strategy. Based on the results of this study and other monitoring activities, we propose to focus on dab, lesser weever and whiting. Genetic and stable isotope analyses (Pitt *et al.*, 2009) are also recommended for future feeding guild structure research.

6.6. References

Berkenhagen, J., Döring, R., Fock, H.O., Kloppmann, M.H.F., Pedersen, S.A. & Schulze, T. (2010). Decision bias in marine spatial planning of offshore wind farms: Problems of singular versus cumulative assessments of economic impacts on fisheries. Marine policy, Band 34 (3), 733-736.

Creutzberg, F., Witte, J.I.J. (1989). An attempt to estimate the predatory pressure exerted by the lesser weever, Trachinus vipera Cuvier, in the southern North Sea. J. Fish Biol. 34, 429-449.

Dauvin, J.C. (1988). Rôle du macrobenthos dans l'alimentation des poissons démerseaux vivant sur les fonds de sediments fins de la Manche occidentale. Cah. Biol. Mar. 29, 445-467.

Derweduwen J., Hostens K., Reubens J. & Vandendriessche S. (2012). Do windmill parks function as a refugium? In Mees J. & Seys J. (Ed.) (2012) Book of abstracts – VLIZ Young Scientists' Day. Brugge, Belgium, 24 february 2012. VLIZ Special Publication, 55. xi, pp. 28.

Hyslop, E.J. (1980). Stomach contents analysis – a review of methods and their application. J. Fish Biol. 17, 411-429.

Kerckhof, F., Rumes, B., Norro, A., Jacques, T.G. & Degraer, S. (2010). Seasonal variation and vertical zonation of the marine biofouling on a concrete offshore windmill foundation on the Thornton Bank (southern North Sea). *In:* Degraer, S., Brabant, R. & Rumes B. (Eds.) (2010). Offshore wind farms in the Belgian Part of the North Sea: Early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 5: 53-68.

Leonhard, S.B., and Pederson, J. (2006). Benthic communities at Horns Rev before, during and after construction of Horns Rev Offshore Wind Farm Vattenfall Report number: Final Report/Annual report 2005, p. 134.

Lindeboom, H.J., Kouwenhoven, H. J., Bergman, M.J.N, Bouma, S., Brasseur S., Daan R., Fijn, R.C., de Haan, D., Dirksen, S., van Hal, R., Hille Ris Lambers, R., ter Hofstede, R., Krijgsveld, K.L., Leopold, M. and Scheidat, M. (2011). Short-term ecological effects of an offshore wind farm in the Dutch coastal zone; a compilation. *Environ. Res. Lett.* 6 035101

May, J. (2005). Post-construction results from the North Hoyle offshore wind farm. Paper for the Copenhagen offshore wind international conference, Project Management Support Services Ltd, 10pp.

Petersen, K.J. & Malm T. (2006). Offshore windmill farms: threats to or possibilities for the marine environment. Ambio 35 (2): 75-80.

Pitt, K.A., Connolly, R.M., Meziane, T. (2009). Stable isotope and fatty acid tracers in energy and nutrient studies of jellyfish: a review. Hydrobiologia 616: 119-132.

Quiniou, L. (1978): Les poissons démersaux de la baie de Douarnenez. Alimentation et écologie. Thèse 3e cycle, Océanogr. Biol., Univ. Brest, Brest, France, 222p.

Reubens, J.T.; Degraer, S.; Vincx, M. (2011). Aggregation and feeding behaviour of pouting (*Trisopterus luscus*) at wind turbines in the Belgian part of the North Sea. Fisheries Research 108(1): 223-227.

Schückel, S., Sell, A., Kröncke, I., Reiss, H. (2011) Diet composition and resource partitioning in two small flatfish species in the German Bight. Journal of Sea Research 66 (3): p. 195 (10).

Schückel, S., Sell, A., Kröncke, I., Reiss, H. (2012) Diet overlap among flatfish species in the southern North Sea. Journal of Fish Biology doi: 10.1111/j.1095-8649.2012.03309.x

Vandendriessche, S., Derweduwen, J. & Hostens, K. (2011a). Monitoring the effects of offshore windmill parks on the epifauna and demersal fish fauna of soft-bottom sediments; baseline monitoring. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 7: 65-81.

Vandendriessche, S., Hostens, K., Courtens, W. & Stienen, E.W.M. (2011b). Monitoring the effects of offshore wind farms: evaluating changes in fishing effort using Vessel Monitoring System data: targeted monitoring results. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 8: 83-92.

Vandendriessche, S., Derweduwen, J. & Hostens, K. (2012). Monitoring the effects of offshore wind farms on the epifauna and demersal fish fauna of soft bottom sediments. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2012). Offshore wind farms in the Belgian part of the North Sea: Heading for an understanding of environmental impacts. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. Chapter 5: 55-71.

Vasconcelos, R., Prista, N., Cabral, H. and Costa, M. J. (2004), Feeding ecology of the lesser weever, *Echiichthys vipera* (Cuvier, 1829), on the western coast of Portugal. Journal of Applied Ichthyology, 20 (2004): 211–216.

Chapter 7. Seabirds & Offshore Wind Farms: Monitoring Results 2011

N. Vanermen[*], E.W.M. Stienen, T. Onkelinx, W. Courtens,
M. Van de walle, P. Verschelde & H. Verstraete

Research Institute for Nature and Forest, Kliniekstraat 25, 1070 Brussels

[*] *Corresponding author: nicolas.vanermen@inbo.be*

Photo INBO

Abstract

‘Seabirds at sea’ count data exhibit extreme spatial and temporal variation, impeding the assessment of the impact of wind turbines on seabird abundance and distribution. We designed a BACI monitoring program to assess the effect of wind farm presence on seabird displacement and used the results of ship-based surveys to simulate a broad range of empirical scenarios. Based upon these, we investigated how the power of detecting a change in seabird numbers is affected by survey length, monitoring intensity and data characteristics. The methodology used for the assessment was revised as compared to the previous reports. The most crucial difference is the application of zero-inflated negative binomial modelling, instead of quasi likelihood estimation. Data on 13 seabird species regularly occurring in the Thorntonbank and Bligh Bank wind farm area were used for the assessment of displacement effects caused by wind turbines.

The impact modelling at the Thorntonbank study area so far only reveals attraction effects, i.e. for Little Gull, Great Black-backed Gull, Black-legged Kittiwake, Sandwich and Common Tern. These findings are highly provisory since at the time of the study, one line of wind mills was present. Nevertheless, this poses some serious conservation concerns, given the high protection status and the fragility of the populations of both tern species and of Little Gull, combined with the raised threat of collision-mortality.

After the turbines were built at the Bligh Bank, numbers of Common Guillemot and Northern Gannet significantly decreased in the wind farm area. In contrast, numbers of Common Gull significantly increased, and the BACI-graphs suggest attraction of Herring Gull as well. Gulls are probably attracted by the wind farm from a sheer physical point of view, with the farm functioning as a stepping stone, a resting place or a reference feature in the wide open sea. During recent surveys in 2012, good numbers of auks and even Harbour porpoises were encountered inside the wind farm. From an ecological point of view, the presence of auks is very interesting, and we wonder if these self-fishing species are already habituating to the presence of the turbines, and if they will profit from a (hypothetical) increase in food availability.

Samenvatting

Een typische zeevogeldataset wordt gekenmerkt door een grote variatie van de waarnemingen in ruimte en tijd, wat het evalueren van de impact van windmolens op de aantallen en verspreiding van zeevogels bemoeilijkt. Teneinde het verplaatsingseffect van windmolens op zeevogels na te gaan, werd een BACI-monitoringprogramma opgesteld en werden de resultaten van scheepstellingen gebruikt om een groot aantal empirische scenario’s te simuleren. Aan de hand hiervan werd onderzocht hoe de power om een verandering in zeevogelaantallen beïnvloed wordt door de lengte en de frequentie van de tellingen en de eigenschappen van de data. De methodiek om de verplaatsingseffecten te detecteren werd enigszins herzien in vergelijking met de vorige rapporten. Het belangrijkste verschil is de toepassing van zero-inflated negative binomial-modellering in plaats van quasi likelihood estimation. Data over aantallen en verspreiding van 13 soorten zeevogels die regelmatig voorkomen in de zone van de windparken op de Thorntonbank en Bligh Bank werden gebruikt om een inschatting te maken van de effecten die windturbines hebben op de aanwezigheid van zeevogels.

Voor de Thorntonbank werden voorlopig enkel aantrekkingseffecten vastgesteld, i.e. voor Dwergmeeuw, Grote Mantelmeeuw, Drieteenmeeuw, Grote Stern en Visdief. Deze resultaten dienen evenwel met grote voorzichtigheid te worden geïnterpreteerd, gezien op het moment van onderzoek slechts één rij van zes windmolens aanwezig was. Niettemin is dit een belangrijk aandachtspunt gezien de hoge beschermingsstatus en de kwetsbaarheid van de populaties van beide sternensoorten en van Dwergmeeuw, gecombineerd met een verhoogde kans op aanvaringen met windmolens.

Nadat de turbines op de Bligh Bank werden geplaatst, werd een significante afname van de aantallen Zeekoeten en Jan-van-Genten in het windparkgebied vastgesteld. Stormmeeuwen waren dan weer abundanter na de bouw van de molens en er zijn indicaties dat ook Zilvermeeuwen worden aangetrokken. Meeuwen worden allicht aangetrokken door het fysieke aspect van het park, waarbij het fungeert als een ‘stepping stone’, als rustgebied of als referentiebaken binnen het open zeegebied.

Tijdens recente scheepstellingen in 2012 werden bovendien vrij grote aantallen alkachtigen en Bruinvissen gezien in het windpark. Hier stelt zich de vraag of deze soorten nu al zijn aangepast aan de aanwezigheid van de turbines en of ze mogelijk kunnen profiteren van een (hypothetische) verhoging van de voedselbeschikbaarheid.

7.1. Introduction

In order to meet the targets set by the European Directive 2009/29/EG on renewable energy, the European Union is aiming at a total offshore capacity of 43 GW by the year 2020. Meanwhile, the offshore wind industry is growing fast and by the end of 2011, 1371 offshore wind turbines were already fully grid-connected in European waters, totalling 3.8 GW (European Wind Energy Association, 2011). The Belgian government has reserved a concession zone comprising almost 7% of the waters under its jurisdiction for wind farming (an area measuring 238 km²). In 2008, C-Power installed six wind turbines (30 MW) at the Thorntonbank, located 27 km offshore, and in 2009, Belwind constructed 55 turbines (165 MW) at the Bligh Bank, 40 km offshore. In the first coming years at least 175 more turbines will be installed in this part of the North Sea (MUMM, 2011).

Possible effects of offshore wind farming on seabirds range from direct mortality through collision, to more indirect effects like habitat change (including positive effects of increased food availability and resting opportunities), habitat loss and barrier-effects (Exo *et al.*, 2003; Langston & Pullan, 2003; Fox *et al.*, 2006; Drewitt & Langston, 2006; Stienen *et al.*, 2007). Whereas several studies investigated the effects of offshore turbines on migrating or local seabird communities (Desholm, 2005; Petterson, 2005; Petersen *et al.,* 2006; Larsen & Guillemette, 2007), only a few papers focussed on the monitoring protocol to assess these effects (Maclean *et al.,* 2006 & 2007; Pérez-Lapeña *et al.,* 2010 & 2011).

The Research Institute for Nature and Forest (INBO) is in charge of monitoring the effects of these wind farms on the local seabird distribution. Therefore, it designed a BACI monitoring program and delineated impact and control areas for both wind farm projects. INBO performs monthly seabird surveys in these areas, and developed an impact assessment methodology accounting for the statistical problems inherent to ‘seabirds at sea’ (SAS) data.

7.2. Methodology

Based on a peer review we revised our methodology (as compared to the one presented in Vanermen *et al*., 2011), the most crucial difference being the application of zero-inflated negative binomial modelling, instead of quasi likelihood estimation. We performed power analyses to investigate how the power of our impact study is affected by survey length, monitoring intensity and data characteristics. Lastly, we applied the proposed methodology for assessing seabird displacement effects caused by the early presence of the C-Power and Belwind wind farms.

7.2.1. BACI monitoring set-up

Stewart-Oaten & Bence (2001) reviewed several approaches for environmental impact assessment, differing in goals and time series available. When ‘before’ data are available and the inclusion of a suitable control is possible, BACI is the suggested approach. While the importance of temporal replication in BACI assessments is widely recognized, there is disagreement on the role of spatial replication, i.e. inclusion of several control locations (Bernstein & Zalinski, 1983; Stewart-Oaten *et al*., 1986; Underwood, 1994; Underwood & Chapman, 2003; Stewart-Oaten & Bence, 2001). In a ‘seabirds at sea’ (SAS) context, including more than one control area is unfeasible, considering the obvious logistic and financial limitations. However, Stewart-Oaten & Bence (2001) argue that when the goal of the assessment is to detect a particular change at a non-random place (e.g. the Thorntonbank wind farm), variation among control sites is irrelevant to the assessment problem. The authors conclude that multiple controls are not needed, but can be useful for insurance, model checking and causal assessment.

Migrating birds show deflections in flight orientation from up to a distance of 1 to 5 km (Petterson, 2005; Petersen *et al.*, 2006), but little is known on the avoidance of swimming birds. Yet, a significant post-construction decrease in densities of divers, scoters and Long-tailed Ducks was shown by Petersen *et al.* (2006) out to a distance of 3 km away from the Nysted wind farm in Denmark. Considering this, we applied a buffer zone of 3 km around the future wind farms to define the 'impact area' (Figure 1), being the zone where effects of turbine presence can be expected. Next, an equally large control area was delineated, harbouring comparable numbers of seabirds, showing similar environmental conditions, and enclosing a high number of historical count data (Vanermen *et al.*, 2010). Considering the large day-to-day variation in observation conditions and seabird densities, the distance from the control to the impact area was chosen to be small enough to be able to survey both areas on the same day by means of a research vessel. As a result, control and impact area are only 1.5 km apart, equalling half the mean distance sailed during a ten-minute transect count (the applied unit in our seabird database).

Considering the fact that the construction of the wind farms is far from completed (55 out of 110 turbines at the Bligh Bank and 6 out of 54 turbines at the Thorntonbank at the time of data collection), the impact area regarded at this stage is limited to the zone where turbines are already present, surrounded by a buffer zone of 3 km (Figure 1). Also, data collected during the construction periods are not included for impact assessment. During construction activities, access to the wind farm areas was often restricted, hampering adequate monitoring. Moreover, construction activities may cause other effects to occur than the ones during the operational phase. Recently, access to the wind farms has greatly improved, e.g. during construction of phase 2 & 3 of the C-Power wind farm.

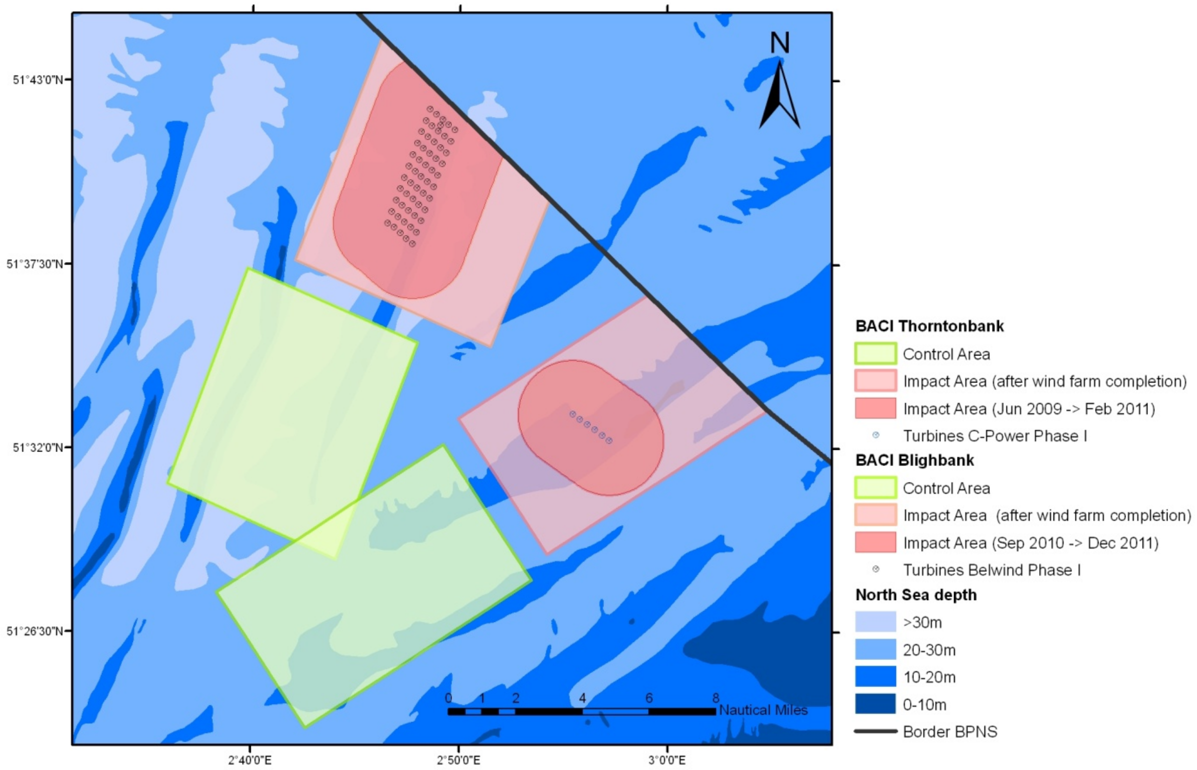


Figure 1. BACI set-up for the monitoring at the Thorntonbank & Bligh Bank wind farms.

The first turbines at the Thorntonbank were erected in 2008, and the reference period includes all data collected up until March 2008. INBO started monthly monitoring of the study area in 2005, but has data available dating back to 1993. In total, 64 surveys were included in the reference dataset - with two counts per area per survey this results in a sample size (N) of 128. Construction activities continued until May 2009, and meanwhile access to the area was restricted. Impact data hence include all observations collected from June 2009 to February 2011 (after which construction activities for phase 2 took place), totalling 33 impact surveys (N=66).

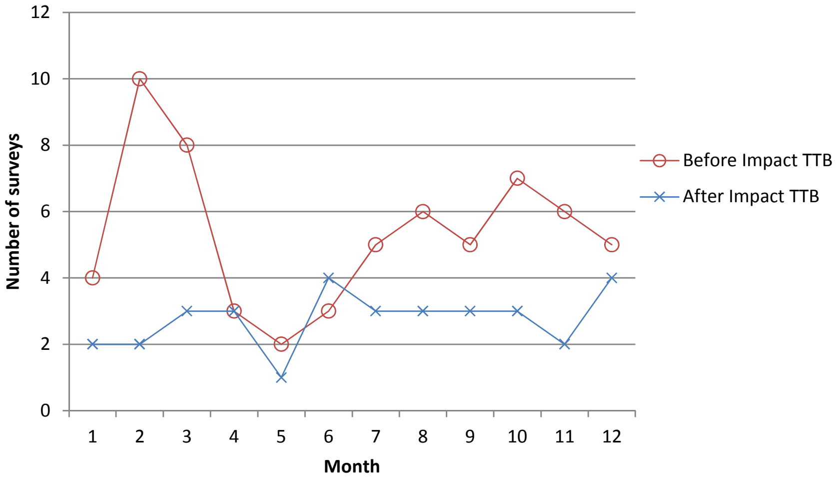


Figure 2. Count effort at the Thorntonbank study area, with indication of the number of surveys performed before and after the construction of the first turbines.

At the Bligh Bank construction activities started in September 2009, prior to which INBO performed 73 reference surveys (N=146). The last of 55 turbines was built in September 2010, and from that month on, impact monitoring was performed inside the wind farm. The impact period includes all data collected from September 2010 to December 2011 (totalling 16 surveys – N=32).

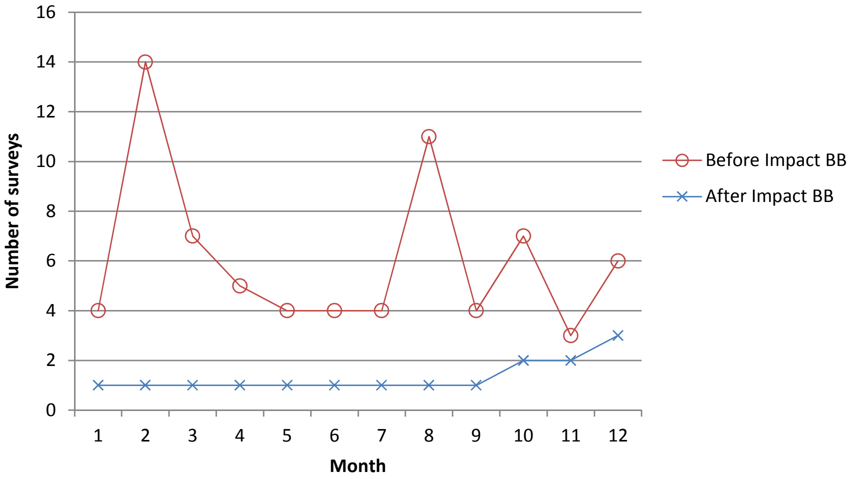


Figure 3. Count effort at the Bligh Bank study area, with indication of the number of surveys performed before and after the construction of the first turbines.

7.2.2. Ship-based seabird counts

Both in the impact and control areas, monitoring was performed through ship-based seabird counts. These are conducted according to a standardized and internationally applied method (Tasker *et al*., 1984; Komdeur *et al*., 1992). While steaming, all birds in touch with the water (swimming, dipping, diving) located within a 300 m wide transect along one side of the ship's track are counted ('transect count'). For flying birds, this transect is divided in discrete blocks of time. During one minute the ship covers a distance of approximately 300 m, and right at the start of each minute we count all birds flying within a quadrant of 300 by 300 m inside the transect ('snapshot count'). Taking into account the distance travelled, these count results can be transformed to seabird densities. The applied count unit in our seabird database is the result of so-called 'ten-minute tracks'.

Stewart-Oaten *et al.* (1986) state that in BACI-assessments, any information gained from replicates taken at the same time is not useful, and that it is better to consider one summarised value (observation X_{ijk}) for each time (t_{ij}), in period i (Before/After) and at place k (Control/Impact). Accordingly, we summed our transect count data per area (Control/Impact) and per monitoring day, resulting in day-totals. This way, we avoided pseudo-replication, and minimized overall variance. It is also advised to take samples in the impact and control area simultaneously (Stewart-Oaten *et al.*, 1986), and so we included only those days at which both areas were visited, minimizing variation due to short-term temporal changes in seabird abundance and in weather and observation conditions. Today, the monitoring routes always include both of these areas, but this was not always the case in our historical data.

We used data on thirteen seabird species occurring regularly in the Thorntonbank and Bligh Bank wind farm areas (see Table 1).

Table 1. Species included in the assessment of displacement effects caused by wind turbines.

Species	**Thorntonbank**	**Bligh Bank**
Northern Fulmar (*Fulmarus glacialis*)	X	X
Northern Gannet (*Morus bassanus*)	X	X
Great Skua (*Stercorarius skua*)		X
Little Gull (*Hydrocoloeus minutus*)	X	X
Common Gull (*Larus canus*)	X	X
Lesser Black-backed Gull (*Larus argentatus*)	X	X
Herring Gull (*Larus fuscus*)	X	X
Great Black-backed Gull (*Larus marinus)*	X	X
Black-legged Kittiwake (*Rissa tridactyla*)	X	X
Sandwich Tern (*Sterna sandvicensis*)	X	
Common Tern (*Sterna hirundo*)	X	
Common Guillemot (*Uria aalge*)	X	X
Razorbill (*Alca torda*)	X	X

7.2.3. Data-analysis: Reference modelling

The data collected prior to the construction of the turbines were modelled during the so-called 'reference modelling'. There are several ways in which SAS-data can be modelled, using generalized linear models (Leopold *et al.*, 2004; Maclean *et al.*, 2006 & 2007), quasi-likelihood estimation (McDonald *et al.*, 2000), generalized additive models (Clarke *et al.*, 2003; Karnovsky *et al.*, 2006; Huettmann & Diamond, 2006; Certain *et al.*, 2007), or combining one of these with geostatistics (Pebesma *et al.*, 2000; Pérez-Lapeña *et al.*, 2010 & 2011). When a counted subject is randomly dispersed, count results correspond to a Poisson-distribution (McCullagh & Nelder, 1989). However, as seabirds often occur strongly aggregated, we applied a negative binomial (NB) distribution, being the standard parametric model used to account for over-dispersion (Potts & Elith, 2006). Another common problem in ecological data is an excess in zero counts (Fletcher *et al.*, 2005). We tested if our data were in fact zero-inflated, and performed preliminary tests to compare the performance of a NB model with a zero-inflated NB model (ZINB), both in terms of predictive value as of resulting power (Zeileis *et al.*, 2008; Wenger & Freeman, 2008). Zero-inflated models consist of two components, a count component modelling the positive count data (in this case according to a negative binomial distribution), and a zero-component modelling the excess of zeros.

Despite the data aggregation to day totals, it seemed that for several species the count data were still zero-inflated. Preliminary tests learned that in this case, the ZINB models performed better compared to NB models, both in terms of the predicted model probability as in terms of power. On the other hand, when comparing the ZINB with NB model results for non-zero-inflated data, coefficient estimates and corresponding P-values are highly similar, and power results are unaffected by the choice of model (further illustrated in the §7.3.1.2, and Figure 6). During this explorative part of the study (reference modelling, data simulation and power analyses) we therefore chose to apply one type of model, being the zero-inflated type, as a base for all data simulations and consequent power calculations, making it easier to compare and interpret the obtained results.

Whether counts were performed in the control or impact area is defined in the count component of the models by the factor variable 'CI' (Control-Impact). We also added seasonality as an explanatory variable since seabird occurrence is subject to large seasonal fluctuations. Seasonal patterns can be described through a sine curve, which can be modelled as the linear sum of a sine and a cosine term (Stewart-Oaten & Bence, 2001; Onkelinx *et al.*, 2008), including 'month' as a continuous variable. We did not allow for interaction between area (CI) and seasonality since differences in seasonal patterns are not likely to occur at such a small scale.

As described above, the response variable equals the total number of birds observed (inside the transect) during one monitoring day in either the control or impact area. To correct for varying monitoring effort, the number of km² counted is included in the model as an offset-variable. The count component of the ZINB model is thus of the following form:

$$\log(response) = offset(\log(km^2)) + a_1 + a_2.\sin\left(2\Pi\frac{month}{12}\right) + a_3.\cos\left(2\Pi\frac{month}{12}\right) + a_6.CI \qquad \text{(Eq. 1)}$$

In Eq.1, seasonality is modelled as a sine curve with a period of 12 months. Several migratory species however show two peaks in density per year. For these species another sine curve with a period of 6 months is added, and the reference model can thus be written as:

$$\log(response) = offset(\log(km^2)) + a_1 + a_2.\sin\left(2\Pi\frac{month}{12}\right) + a_3.\cos\left(2\Pi\frac{month}{12}\right) + a_4.\sin\left(2\Pi\frac{month}{6}\right) + a_5.\cos\left(2\Pi\frac{month}{6}\right) + a_6.CI$$

(Eq. 2)

Lastly, the zero-component of the ZINB model is built up solely from an intercept (b_1), linked to response by a logit-function. Back-transformation of this intercept results in the additive chance of encountering a zero-value (e.g. an intercept of 1 corresponds to a chance of 73.1%).

The resulting reference model is selected through backward model selection, first testing for the area-effect CI, and then testing for the seasonality-effect, considering an ANOVA test-statistic, and comparing the AIC-values of the different models.

7.2.4. Power analysis

The power analysis as presented in this report is based on the reference data collected in the Thorntonbank study area (see also §2.1). The power is estimated by simulating random datasets with pre-defined characteristics, e.g. the model parameters as found during the reference modelling (§7.2.3), and imposing a hypothetical change on the post-construction numbers. This change in numbers is supposed to occur throughout the impact area, immediately after the impact, and to persist as long as turbines are present ('press disturbance' – Underwood, 1992; Underwood & Chapman, 2003).

The model to determine a turbine impact is a simple extension of the count component of the selected reference model:

$$response \sim Seasonality + CI + BA + BA:CI \qquad \text{(Eq. 3)}$$

Or – when the factor variable CI was already rejected from the reference model – the impact model looks somewhat different:

$$response \sim Seasonality + BA + T \qquad \text{(Eq. 4)}$$

In both equations, 'Seasonality' is the sine wave described earlier and the two-level factor variable BA stands for Before/After the impact. In Eq.3, a turbine effect is indicated by the amount of interaction between BA with CI, while in Eq.4, this effect is indicated by factor T (which stands for turbine presence versus absence).

7.2.4.1. Power analysis: effect of model parameters

To be able to isolate the effect of the several model parameters, we first modelled the reference data applying the same reference ('base') model for all species (Eq. 1). This revealed empirical ranges of the intercept (a_1), the amplitude of seasonality (=$\sqrt{a_2^2 + a_3^2}$), the CI-effect (a_6), the amount of zero-inflation (b_1) and theta (θ). The latter is part of the variance function of a negative binomial distribution:

$$V(\mu) = \mu + \frac{\mu^2}{\theta} \qquad \text{(Eq. 5)}$$

Next, we varied all of these coefficient values within the given ranges, and calculated the power for each scenario. At this stage, the monitoring set-up is held constant, with a reference and impact period of both 5 years, one survey per month (with an effort of 10 km² per area), a decrease in numbers of 50% and a significance level of 10%. This significance level represents the chance of wrongly concluding that the turbines are causing an impact, while in fact they are not ('type I error'). Each scenario is simulated 1000 times, and the power thus equals the percentage of times the z-test reveals a P-value less than 10% for the BA:CI or T-term, indicating a turbine effect.

7.2.4.2. Power analysis: effect of survey duration and degree of seabird displacement

In a second step we calculated powers based on species-specific reference models (as explained in §7.2.3), varying monitoring set-up characteristics, i.e. the decrease in numbers in the impact area to be detected (25, 50 & 75%) and the monitoring period (5 years before versus 1, 3, 5, 7, 9, 11, 13 & 15 years after impact).

7.2.5. Data-analysis: Impact modeling

During the impact modelling we analysed all collected count data to investigate whether the presence of wind turbines is causing seabird displacement. As outlined in §7.2.4, the applied impact model is a simple extension in the count component of the reference model (Eq. 3 & 4). While we applied a ZINB model for all species during the explorative phase, we now considered each species separately to decide whether to use the ZINB or NB model. Two criteria can be used to do so:

- The P-value of the zero-component intercept: the null hypothesis of the z-test testing for the effect of the intercept is that b_1 equals zero. Back-transformation of an intercept value of zero however corresponds to a chance of 50%, which can be classified as a high degree of zero-inflation.
- A Vuong test (Vuong, 1989): a test that compares non-nested models, as is the case here with a NB model and its zero-inflated analogue. The sign (+/-) of the test-statistic indicates which model is superior over the other in terms of probability. However, in most cases, the corresponding P-value appeared to be indecisive.

Hence, none of these two options gave satisfactory results. Therefore, we defined our own criterion and calculated the lower boundary of the confidence interval of the zero-component intercept: when this lower boundary exceeds -2.2 (corresponding to an additive chance of 10% to encounter zero birds), we decided to hold on to the ZINB model. The choice made as such largely corresponds to what one would expect based on the sign (+/-) of the Vuong test-statistic.

7.2.6. Statistics

All modelling was performed in R.2.14.0 (R Development Core Team 2011), making use of the following packages:

- MASS (Venables & Ripley, 2002)
- pscl (Zeileis *et al.*, 2008; Jackman, 2011)

7.3. Results

7.3.1. Reference modelling & Power analyses

7.3.1.1. Base modelling: coefficient estimates

First, we applied the same 'base model' (Eq. 1) to all species, providing us with empirical coefficient ranges. Based upon these, we defined unique coefficient combinations, which are applied in the 'test models'. As such, the intercept a_1 of the count component was varied stepwise from -4 to 0. The amplitude was varied by setting a_3 to zero and varying a_2 from 1 to 4, again in discrete steps of one unit. Figure 4 displays the empirical model coefficients, as well as the ones used for the 'test models'. In order to be able to fully exclude the effect of seasonality, we also combined an amplitude of 0 with an intercept varying from -4 to 2.

Next, we defined an empirical range for theta, as well as for b_1, indicating zero-inflation. The base modelling revealed an interaction between the theta-value and the amount of zero-inflation. For data showing no zero-inflation ($b_1 < -5$), theta was small, varying between 0.18 and 0.66, while in data subject to zero-inflation ($b_1 > 0.5$), theta-values were clearly higher, ranging from 0.48 to 1.40. This is interesting, because it suggests that in the latter case, over-dispersion is (at least partly) captured by the zero-component. Thus we combined a b_1-value of -10 (zero-inflation=0%) with a theta varying by 0.2, 0.4 & 0.6, and a b_1-value of 1 (zero-inflation=±75%) with a theta varying by 0.6 & 1.2.

Combining all of these parameters, we end up with 135 theoretical scenarios. This enables us to isolate and explore the effect of the different model parameters on the power of our impact analysis, given a certain monitoring set-up (i.e. to detect a decrease in numbers of 50% after 10 years of monitoring, i.e. 5 year before and 5 years after the impact).

Until now, the area-coefficient a_6 was fixed at zero, but the base models showed this coefficient to vary between -1.02 and 1.25. As a last step, we calculated the effect of the CI-factor on the resulting power by varying a_6 with -1, 0 and 1.

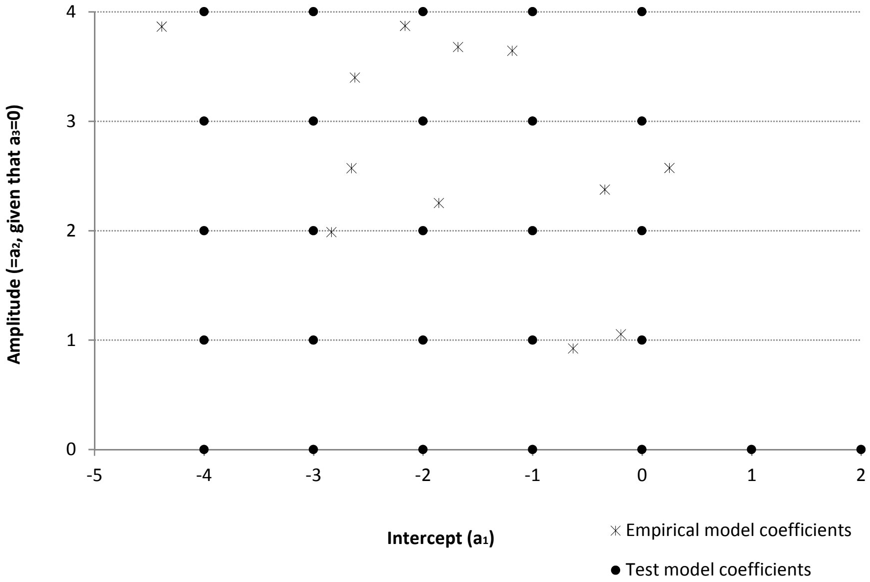


Figure 4. Values for the intercept (a_1) and amplitude (equalling a_2 as a_3 is set to zero) as used in the test models, and indication of the empirical values as found in the reference data collected in the Thorntonbank study area.

Since all of these model coefficient values are linked to the response variable by a logarithmic link function, they are difficult to interpret. Therefore we visualize the corresponding predicted densities for 8 unique combinations of intercept and amplitude (Figure 5).

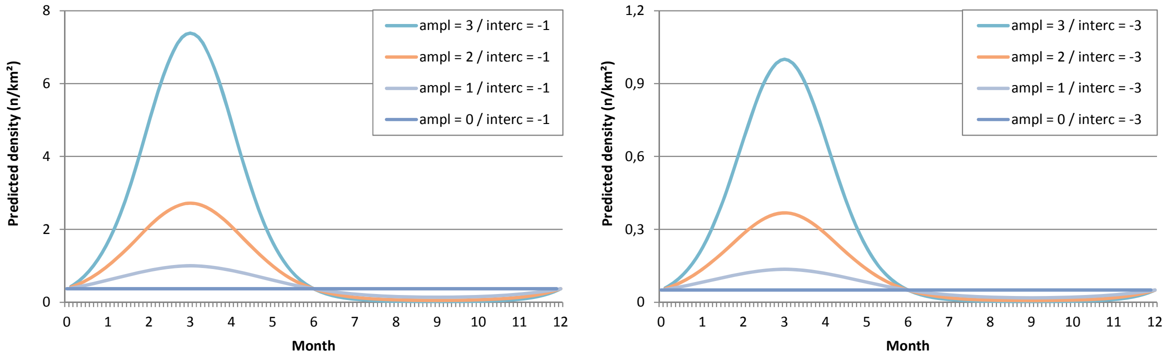


Figure 5. Predicted densities (n/km²) when applying to 8 unique combinations of intercept and amplitude values as used in the test models (see also Figure 4).

7.3.1.2. Power analysis: effect of model parameters

We calculated the power for 135 scenarios with varying intercept, amplitude, theta and amount of zero-inflation, as determined in §7.3.1.1.

Zero-inflation has a clear negative effect on the power of the impact study (Figure 6). It is also shown that when non-zero-inflated data are simulated (intercept of the zero-component = -10), equal powers are obtained when comparing NB and ZINB models. When we do include zero-inflation in the data simulation (b_1=0 or b_1=1, corresponding to a zero-inflation of 50 & 73%), the ZINB model clearly performs better. We hypothesise that this is due to fact that over-dispersion can now be captured by the zero-component, instead of being fully absorbed by the theta value.

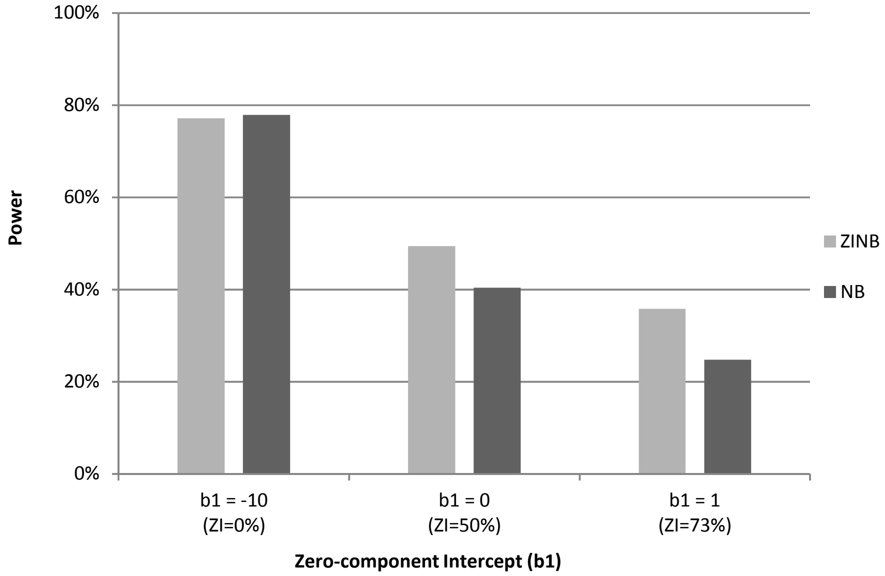


Figure 6. Comparison of the power to detect a 50% decrease in numbers based on a negative binomial (NB) and a zero-inflated model (ZINB), for several levels of zero-inflation (a_1=-1, a_2=1, a_3=0, a_6=0, θ=0.5).

The results show that θ is another important parameter influencing the power of our impact analysis (Figure 7). A theta of 0.2 or less inevitably results in low power after five years of post-impact monitoring, and assuming no zero-inflation is present, a value of 0.4 is needed to obtain a power of 80%.

Base modelling showed that for some species, the reference data combine a seemingly favourable theta with a certain amount of zero-inflation. The power-curve "θ=0.6 / ZI=73%" in Figure 7 shows that all benefits gained from a favourable theta are lost due to zero-inflation. As θ continues to rise, power results start to catch up ("θ=1.2 / ZI=73%"), but still do not exceed the powers found for the scenarios "θ=0.2 / ZI=0%" and "θ=0.4 / ZI=0%".

Based on Figure 7, we also see that the intercept is positively correlated with resulting power, which is particularly true for intercepts ranging from -4 to 0. Increase in power levels off when the intercept exceeds zero, corresponding to a seabird density of 1 bird/km². Due to strong seasonality, the intercepts estimated for our reference data were in fact all below or around zero (Figure 4).

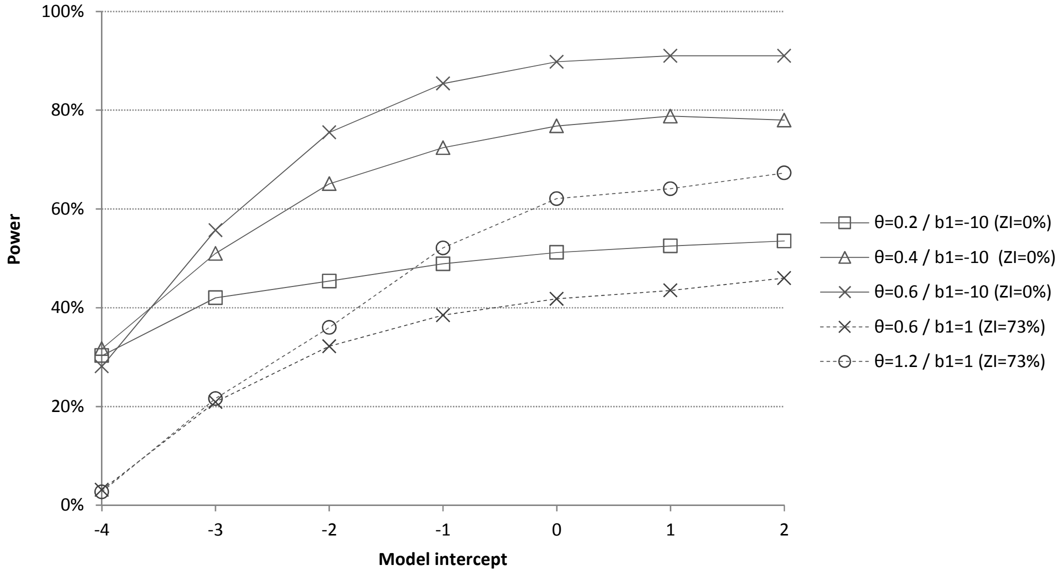


Figure 7. Effect of the model intercept, theta (θ) and the amount of zero-inflation (ZI) on the power of the impact analysis (for test models with a seasonal amplitude equalling zero).

The amplitude of the modelled seasonality pattern appears to have a rather limited effect on the power to detect a change in numbers. We found a positive correlation between the amplitude and power in case of very low intercepts (<-3), and a slightly negative correlation in case of higher intercepts (Figure 8).

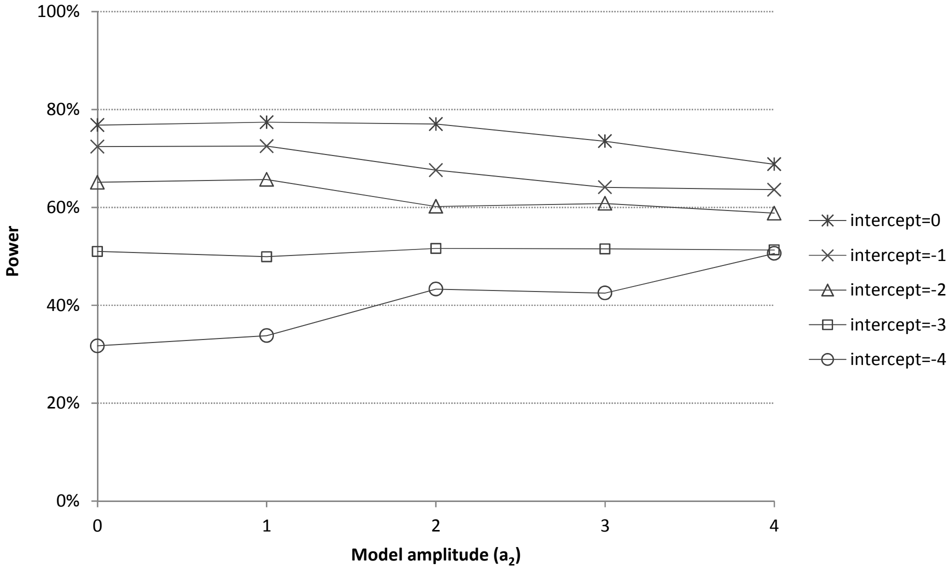


Figure 8. Effect of the seasonal amplitude (equalling a_2 as a_3 is set to zero) and the model intercept (a_1) on the power of the impact analysis (when θ=0.4).

Finally, we investigated the effect of the area factor (CI). For the same relative decrease in numbers (50%), we simulated datasets with varying CI-coefficients a_6 (-1, 0 & 1), and calculated the power based on two different types of impact models. One model takes in account the imposed CI effect (see Eq. 3), while the other one ignores it (Eq. 4). Figure 9 shows the importance of including the CI-factor into the model. When doing so, the power results are much more stable (and hence reliable) compared to the results when the CI-effect is ignored. Of course, when the CI-factor does not attribute significantly to the reference model (P>0.10), it can and should be excluded, as the resulting gain in 2 degrees of freedom will always reflected by better power.

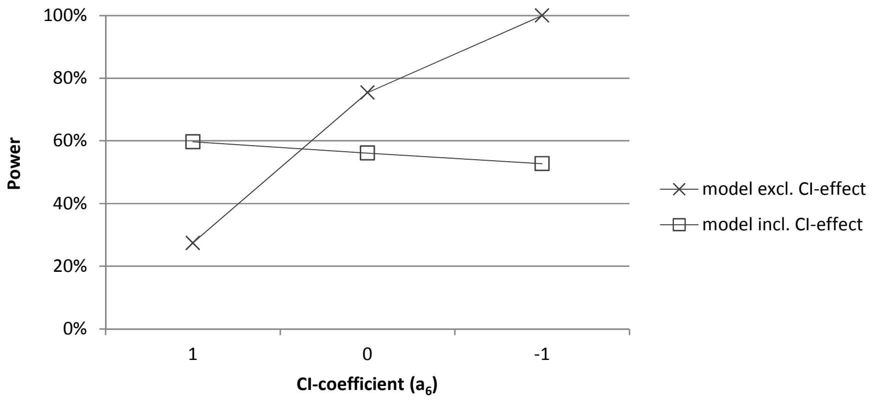


Figure 9. Comparison of power results for two types of models (including or excluding an area effect – see Eq. 3 & 4) for several levels of CI-coefficient a_6 (a_1=-1, a_2=1, a_3=0, θ= 0.5).

7.3.1.3. Species-specific reference models (Thorntonbank)

We built species-specific reference models (as set out in §7.2.4.2) and Table 2 shows all estimated coefficients. Considering their specific seasonal occurrence in the study area, we used a double sine curve to explain seasonal variation in numbers for four species, i.e. Northern Gannet, Little Gull, Sandwich Tern and Common Tern. The occurrence of all other species was described by using a single sine curve. In only two out of twelve species, we retained a significant area-effect i.e. for Common Gull (a_6=1.26) and Black-legged Kittiwake (a_6=-0.87).

Back-transformation of the intercept values b_1 of the model's zero component (IntZero) shown in Table 2 learns that zero-inflation occurs in the data of Northern Fulmar (54.0%), Sandwich Tern (52.2%) and Common Tern (74.8%). For the two latter species, theta values are high (3.68 & 11.05), suggesting that most of the over-dispersion is captured by the zero-component. In all other species zero-inflation is very close to 0%. Figure 10 displays the seasonally varying model predictions for all 12 seabird species.

Table 2. Model coefficients of the selected reference models at the Thorntonbank.

	IntCount	Sin (1yr)	Cos (1yr)	Sin (1/2yr)	Cos (1/2yr)	CI	IntZero	θ
Northern Fulmar	-0.83	-1.08	0.17				0.16	0.27
Northern Gannet	-0.82	-0.65	0.26	-0.60	-0.54		-10.55	0.37
Little Gull	-3.35	1.67	3.75	-1.28	-0.84		-3.46	0.22
Common Gull	-4.39	2.00	3.30			1.26	-10.85	0.21
Lesser Black-backed Gull	0.07	1.09	-2.33				-11.09	0.22
Herring Gull	-2.75	1.77	0.78				-7.70	0.20
Great Black-backed Gull	-1.52	-0.30	2.30				-10.19	0.18
Black-legged Kittiwake	-0.36	-1.10	2.13			-0.87	-12.94	0.26
Sandwich Tern	-8.90	0.48	-11.00	1.18	-6.39		0.09	3.64
Common Tern	-10.54	-1.25	-13.61	-0.93	-7.24		1.09	11.03
Common Guillemot	-1.29	0.56	3.63				-11.59	0.65
Razorbill	-2.50	-0.16	3.39				-11.12	0.32

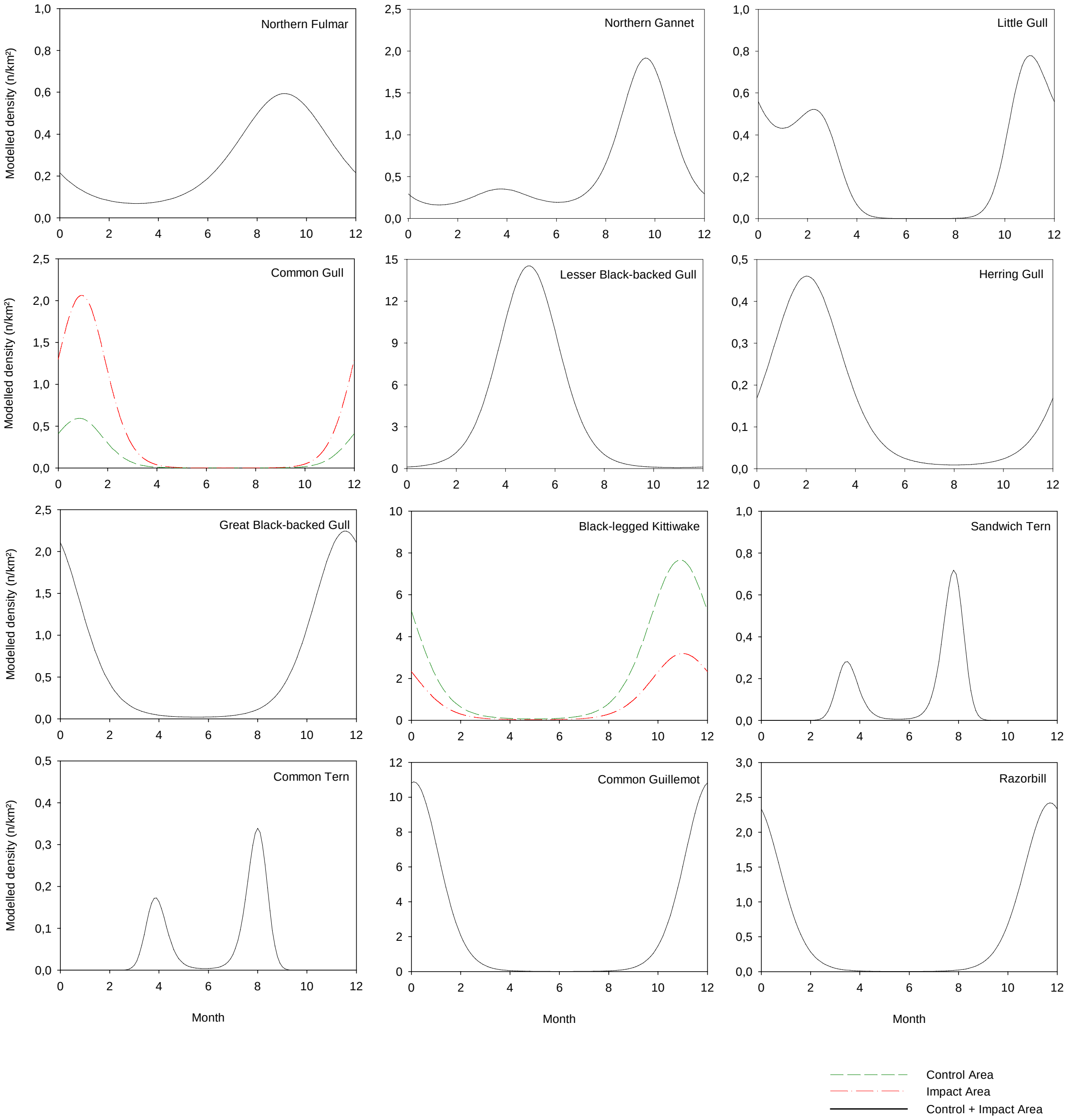


Figure 10. Modelled densities of 12 seabird species, based on data collected at the Thorntonbank study area prior to the construction of the wind farm.

7.3.1.4. Power analysis: effect of survey duration and degree of seabird displacement

Based on the selected reference models, we studied how power is related to survey duration (Figure 11). We found that for none of the 12 seabird species under study, we will be able to detect a change in numbers of 25% with a power of more than 55%, not even after 15 years of impact monitoring. In contrast, a change in numbers of 50% should be detectable within less than 10 years with a chance of >90% in two seabird species i.e. Northern gannet and Common guillemot. Within the same time frame we will be able to detect a decrease of 75% with a power >90% in all species except for Common Gull.

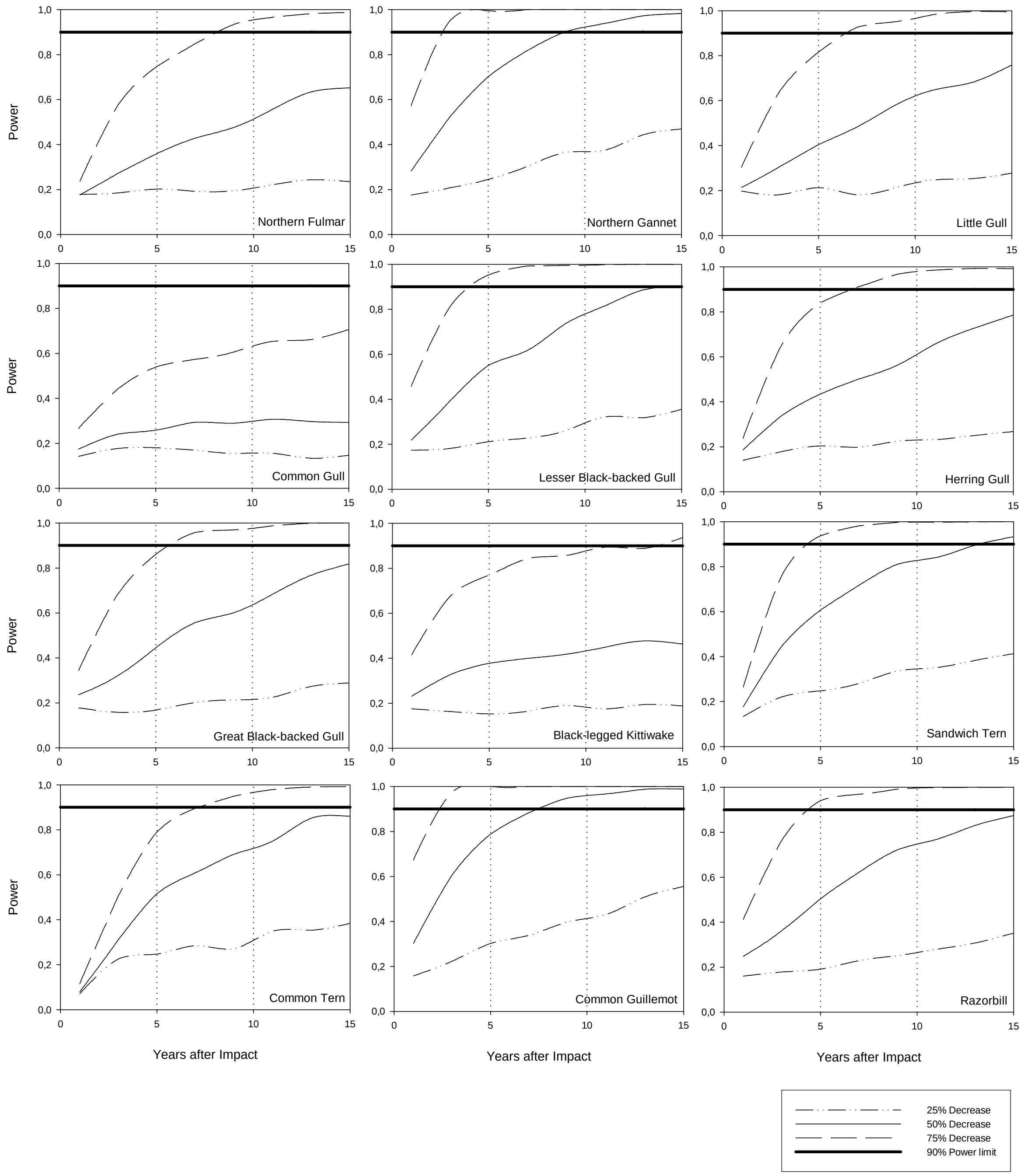


Figure 11. Power results for 12 seabird species for an impact study with a monitoring intensity of one survey of 10km² per month per area, and 5 years of reference monitoring (significance level = 0.10).

7.3.2. Impact modelling

7.3.2.1. Thorntonbank

The impact modelling at the Thorntonbank study area only reveals attraction effects, i.e. for Little Gull, Great Black-backed Gull, Black-legged Kittiwake and both tern species.

Figure 12 shows typical BACI-graphs displaying 4 geometric mean density values. These graphs give a first indication of attraction or avoidance effects, but these might as well be hidden. For example, based on the BACI-graphs, it is relatively obvious that there must have been an effect on the occurrence of Little Gull, Sandwich Tern & Common Tern. However, this is much less obvious based on the graphs of Great Black-backed Gull and Black-legged Kittiwake, showing that the impact modelling process reveals effects that otherwise could be hard to detect.

Table 3. Impact modelling results for the Thorntonbank wind farm.

		T – effect		BA:CI – effect	
		Coeff	P-Value		
Northern Fulmar	ZINB	-13,63	0,986		
Northern Gannet	NB	-0,71	0,127		
Little Gull	NB	**1,22**	**0,084.**		
Common Gull	NB			-1,43	0,101
Lesser Black-backed Gull	NB	-0,13	0,809		
Herring Gull	NB	0,37	0,566		
Great Black-backed Gull	NB	**1,49**	**0,023***		
Black-legged Kittiwake	NB			**2,01**	**0,005***
Sandwich Tern	ZINB	**2,43**	**0,001****		
Common Tern	ZINB	**2,42**	**0,028***		
Common Guillemot	NB	-0,17	0,710		
Razorbill	NB	0,43	0,480		

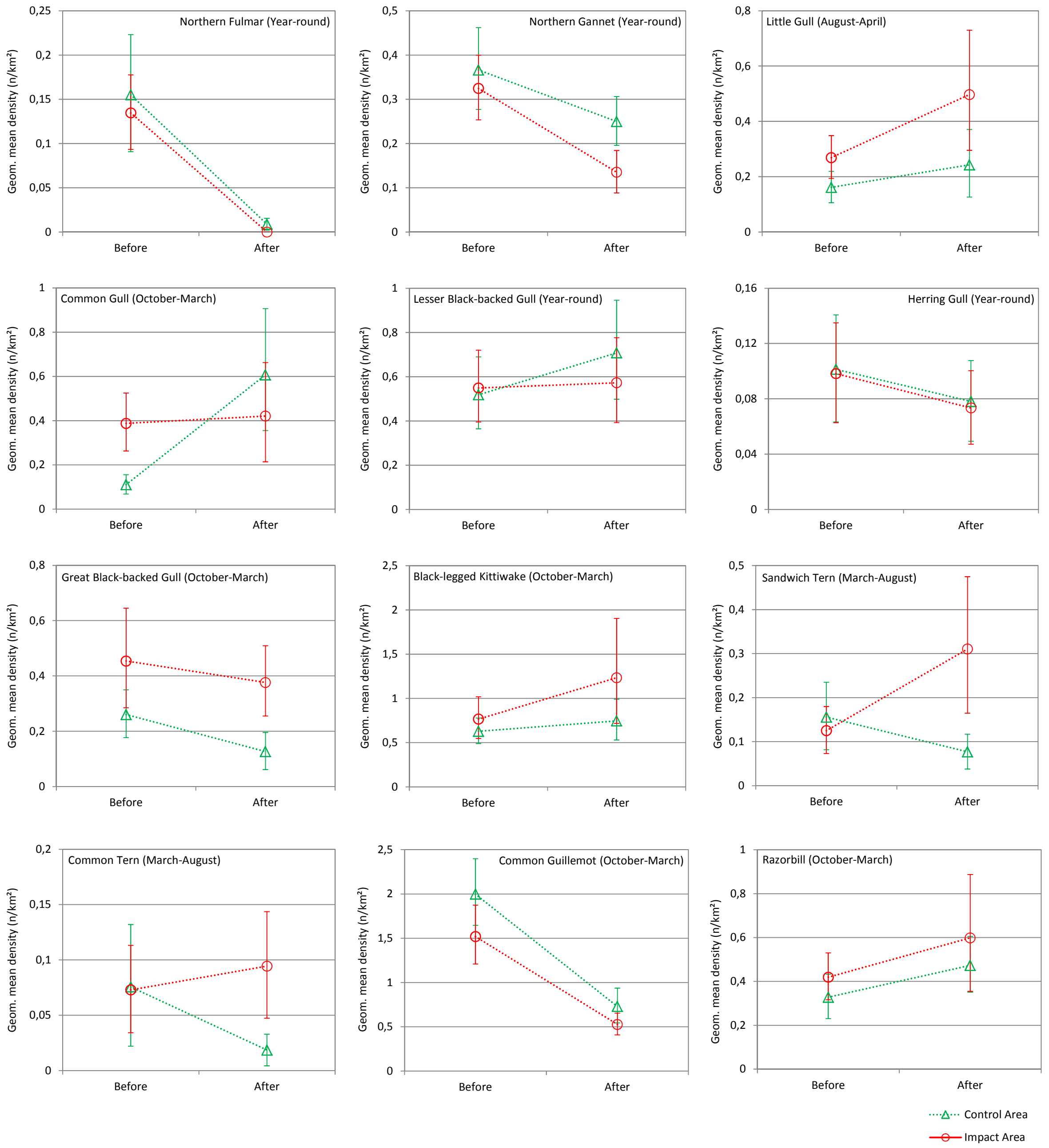


Figure 12. Geometric mean seabird densities (+/- std. errors) in the reference and impact area before and after the turbines were built at the Thorntonbank.

7.3.2.2. Bligh Bank

Reference modelling revealed a significant area effect for three species, i.e. Little, Common and Great Black-backed Gull. All three showed higher densities in the impact area compared to the reference area. The data of Great Skua, Little Gull and Common Gull appear to be zero-inflated (75-80%). As in the reference data at the Thorntonbank, a positive intercept in the zero-component is accompanied with a high theta value in the count component, suggesting that overdispersion is captured by the zero-component of the model. For the non-zero-inflated data, theta varies between 0.10 and 0.58. Analogous to the reference data at the Thorntonbank, the two most favourable theta values are found in the count data of Common guillemot (0.58) and Northern Gannet (0.40), while the least favourable theta (0.10) is put away for Great Back-backed Gull. The only species where we modelled a double-peaked seasonality is Northern Gannet (Figure 13).

Table 4. Model coefficients of the selected reference models at the Bligh Bank.

	IntCount	Sin (1yr)	Cos (1yr)	Sin (1/2yr)	Cos (1/2yr)	CI	IntZero	θ
Northern Fulmar	-1.71	0.94	0.84				-8.23	0.14
Northern Gannet	-1.50	-0.16	1.50	0.01	-0.96		-10.13	0.40
Great Skua	-1.88						1.09	4.76
Little Gull	-12.30	11.26	-1.09			1.83	1.29	1.63
Common Gull	-3.24	1.24	2.82			0.71	1.44	97828.37
Lesser Black-backed Gull	-1.08	0.52	-0.67				-9.48	0.17
Herring Gull	-4.58	2.51	1.42				-7.34	0.33
Great Black-backed Gull	-2.80	1.64	1.73			2.24	-9.90	0.10
Black-legged Kittiwake	-1.13	0.18	2.56				-11.21	0.27
Common Guillemot	-1.69	1.15	3.00				-11.32	0.58
Razorbill	-4.07	1.79	3.45				-7.99	0.29

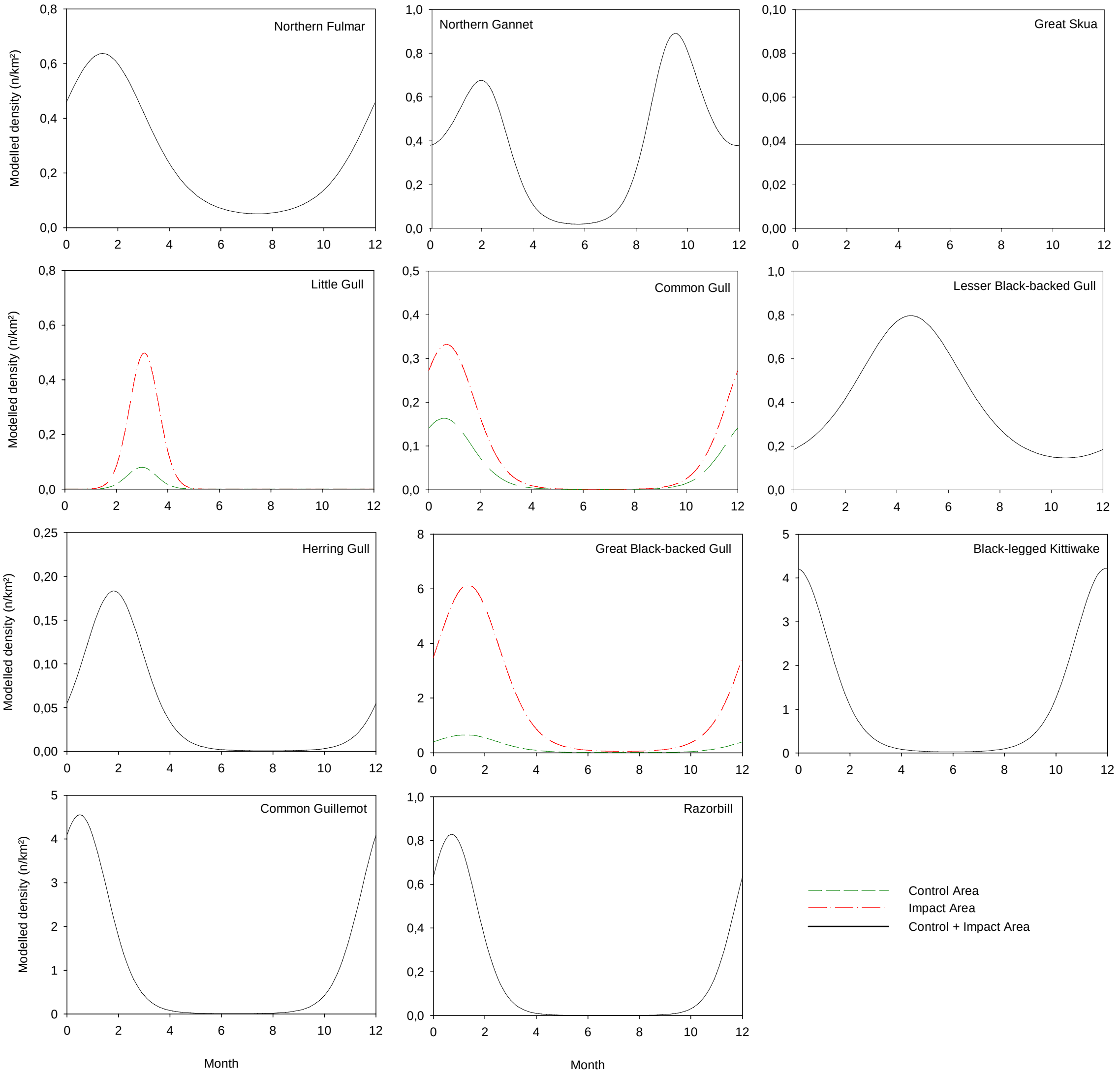


Figure 13. Modelled densities of 11 seabird species, based on data collected at the Bligh Bank study area prior to the construction of the wind farm.

In the impact data, zero-inflation persisted in the count results of Great Skua and Common Gull, while this was no longer the case for Little Gull. On the other hand, we did use a ZINB model for Herring Gull, since a NB model was unable to fit.

After the turbines were built, numbers of Common Guillemot and Northern Gannet significantly decreased in the wind farm area, while numbers of Common Gull increased. These trends are also obvious when looking at the BACI-graphs in Figure 14. Based on the BACI-graph of Herring Gull, we could have expected a positive turbine effect, but this was not detected by our statistical modelling (P=0.209).

Table 5. Impact modelling results for the Bligh Bank wind farm.

		T – effect		BA:CI – effect	
		Coeff	P-Value	Coeff	P-Value
Northern Fulmar	NB	-28.60	1.000		
Northern Gannet	NB	-1.50	0.016*		
Great Skua	ZINB	-14.86	0.995		
Little Gull	NB			-0.79	0.643
Common Gull	ZINB			3.04	0.026*
Lesser Black-backed Gull	NB	0.14	0.871		
Herring Gull	ZINB	1.34	0.209		
Great Black-backed Gull	NB			-0.55	0.653
Black-legged Kittiwake	NB	0.56	0.444		
Common Guillemot	NB	-1.15	0.046*		
Razorbill	NB	-1.29	0.127		

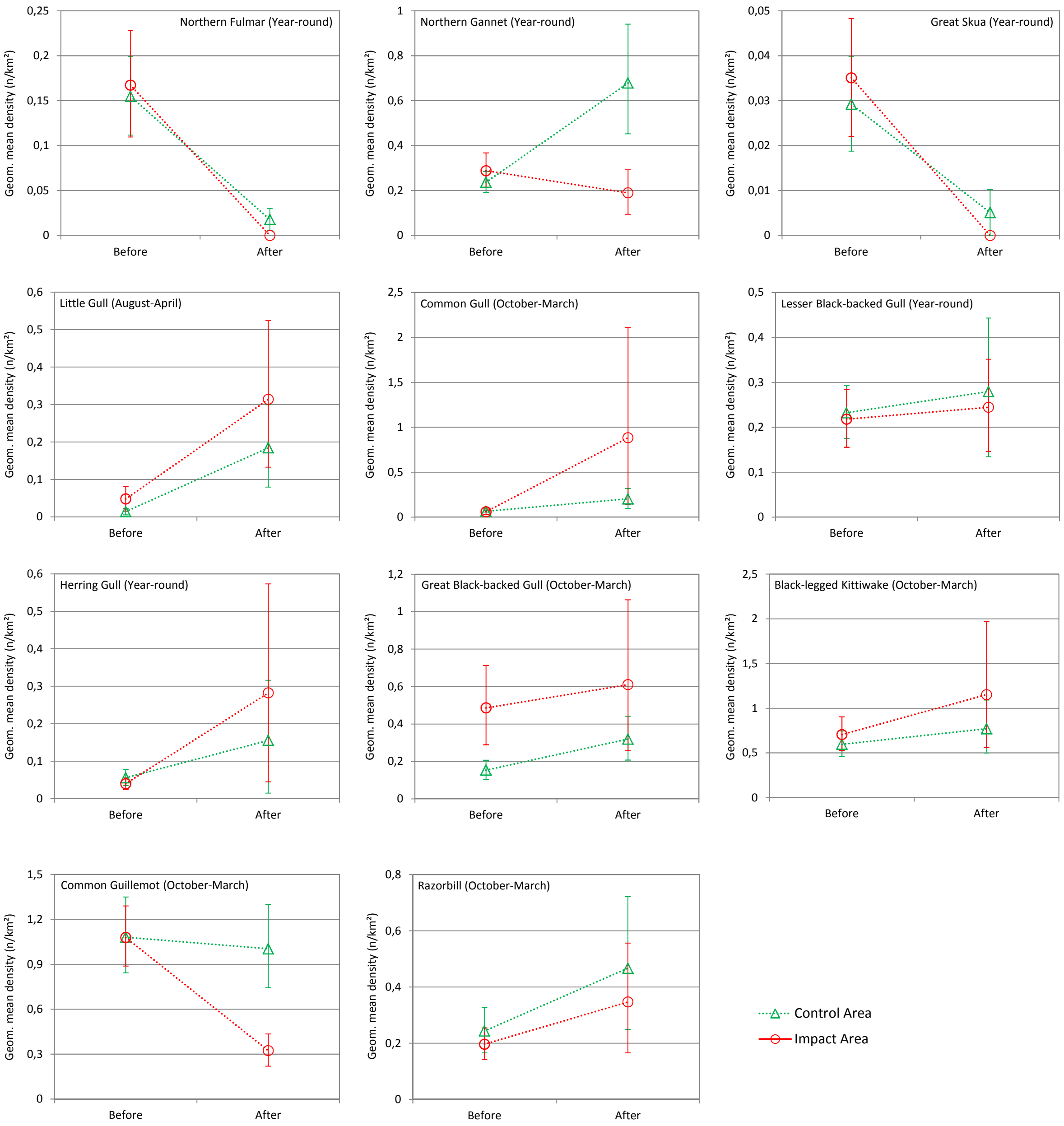


Figure 14. Geometric mean seabird densities (+/- std. errors) in the reference and impact area before and after the turbines were built at the Bligh Bank.

7.4. Discussion

7.4.1. Impact assessment

The impact modelling at the Thorntonbank study area only reveals attraction effects, i.e. for Little Gull, Great Black-backed Gull, Black-legged Kittiwake and both tern species. These findings are highly provisory since it is mathematically impossible to count *inside* a one dimensional wind farm (i.e. one line of wind mills). At best, any conclusions drawn from the study presented here are valid *for a wind farm buffer zone* (in this study set to 3 km).

At the OWEZ wind farm in the Netherlands, Little Gulls are rarely seen inside the wind farm and seemed to avoid the area between the turbines, and the same was concluded for Sandwich Tern (Leopold *et al*., 2010). At the Horns Rev wind farm in Denmark, Petersen *et al*. (2006) found slightly increased (non-significant) post-construction numbers of Little Gull inside the wind farm, and a significant increase in numbers just outside its boundaries (up to 2 km). The same authors found a total absence of Common Tern inside the wind farm, avoidance up to 1 km outside its boundaries, but a clear post-construction increase in numbers in the immediate vicinity of the farm (1 to 8km). This is in correspondence to what was found in this study, and meanwhile, the findings at Horns Rev stress the need to perform separate analyses for the wind farm and the buffer zone around it!

Nevertheless, if the attraction effects as found now should persist during the following wind farm phases, this is of serious conservational importance. Both tern species as well as Little Gull are included on the Annex I list of the Birds Directive (EC/2009/147), and high proportions of the biogeographical populations of all three species migrate through the Southern North Sea (Stienen *et al*., 2007).

After the turbines were built at the Bligh Bank, numbers of Common Guillemot and Northern Gannet significantly decreased in the wind farm area. In correspondence, avoidance by gannets and auks is reported by Petersen *et al*. (2006) at the Horns Rev wind farm in Denmark, and by Leopold *et al*. (2010) in the OWEZ wind farm in the Netherlands.

In contrast, numbers of Common Gull significantly increased, and the BACI-graphs suggest attraction of Herring Gull as well. While gulls are known at least not to avoid the wind farms, attraction effects could not be proven during the Danish and Dutch monitoring program (Petersen *et al*., 2006; Leopold *et al*., 2010). Spatial distribution of gulls is strongly influenced by fishery activities, which makes it very difficult to discern and correctly interpret any changes in distribution patterns. In this respect, the main effect of wind farms on gull distribution patterns is likely to result from the prohibition for trawlers to fish inside their boundaries (Leopold *et al.,* 2010).

Nevertheless, despite the absence of beam trawlers, all gull species were regularly observed between the turbines. Gulls are probably attracted by the wind farm from a sheer physical point of view, with the farm functioning as a stepping stone, a resting place or a reference feature in the wide open sea. During recent surveys in 2012, good numbers of auks and even Harbour porpoises were encountered inside the wind farm. From an ecological point of view, the presence of auks is very interesting, and we wonder if these self-fishing species are already habituating to the presence of the turbines, and if they will profit from a (hypothetical) increase in food availability (Degrear *et al*., 2011).

7.4.2. Data handling

Traditionally, the applied count unit in SAS-research is the result of a 5- or 10-minute track, geo-referenced in the middle point (following Tasker *et al*., 1984; Komdeur *et al*., 1992). However, when collected during the same day, these rather short transect counts are likely to be pseudo-replicates which are not independent (Stewart-Oaten *et al*., 1986; Pebesma *et al*., 2000; Karnovsky *et al*., 2006). Therefore we condensated our transect count data to day totals per area.

Based on these binned data, we applied a negative binomial (NB) distribution to predict seabird densities in the study area. In case of highly over-dispersed data, the use of a NB distribution is to be preferred over a quasi-poisson distribution, as used in Vanermen *et al*. (2010) (Zuur *et al*., 2009). Moreover, simulating a (continuous) quasi-poisson distribution, implies the simulation results to be rounded to the nearest integer, which in the end may result in false power results. Seasonal variation was modelled by fitting a sine curve to our data, enabling us to include ‘month’ as a continuous variable in the models. This method performed much better compared to the inclusion of ‘month’ as a factor variable, which splits the data in twelve subsets, resulting in highly unreliable coefficient estimates. In order to explain spatial variation in seabird distribution and abundance, environmental variables are often included in the assessment modelling (e.g. Garthe, 1997; Pebesma *et al*., 2000; Karnovsky *et al*., 2006; Huettmann & Diamond, 2006; Maclean *et al.*, 2006 & 2007; Oppel *et al*., in press). However, in this study, any variation in seabird numbers induced by environmental gradients is excluded through the aggregation of our data per day and per area, while the difference between both areas is described by a two-level factor variable (‘CI’). The last challenge in the modelling process was dealing with zero-inflation, as SAS-data – and ecological data in general – are often characterised by an excess in zero-counts (Fletcher *et al*., 2005). We investigated if this was also the case in our data by fitting a zero-inflated model (ZINB), built out of a negative binomial count component (predicting abundance given that birds are present) and a logistical zero component (predicting presence/absence). Due to the data condensation overall variance was lowered, but still few species showed zero-inflated count data. In this case, we strongly recommend using the ZINB model. It was shown that for data subject to an excess in zero-counts, the ZINB model results in better power compared to the NB model.

7.4.3. Statistical power

Modelling the reference data collected in the Thorntonbank study area resulted in empirical ranges of model coefficients. Based upon these we defined numerous scenarios, varying model parameters as well as monitoring set-up characteristics. For each scenario we performed 1000 simulations, allowing us to investigate how the different model parameters affect the power of detecting a change in numbers. Each of these parameters appears to interact with one another, so unambiguous conclusions are difficult to draw. Nevertheless, it could be shown that for the given monitoring set-up (5 years before / 5 year after the impact with a survey effort of 10 km² per month per area), count data subject to zero-inflation and/or characterised by a low theta (<0.4) will hardly be of any value in impact monitoring. Ideally, the data show no zero-inflation ($b_1<-5$), a positive intercept ($a_1>0$), a favourable theta (>0.4) and no significant area effect.

Clearly, after binning the data to day totals, the nature and characteristics of the count results can no longer be changed, but still there are some ways to enhance the power. By far the easiest way to do so is to apply a higher significance threshold (alpha). In this context, a higher alpha increases the chance of wrongly concluding that the turbines are causing an impact, while in fact they are not ('type I error'). However, a stringent significance level goes at the expense of the power, resulting that certain impact effects may go unnoticed (Underwood & Chapman 2003). Most impact studies are meant to function as an early warning system, in order to detect potential negative effects as soon as possible. For decision-making, ecological studies commonly set the probability of a type I error (α) to 0.05, and the probability of a type II error (β) to 0.20. However, this choice tends to be arbitrary and such values imply that the acceptable risk of committing a type II error is four times higher than the risk of a type I error (Pérez-Lapeña *et al.*, 2011). In this paper, we use 90% as a boundary for 'sufficient' power (β) and the acceptable risk of making a type I error α was set to 10%, thus equalling acceptable levels for both risks (α=β). Nevertheless, it would still be better for these values to be determined by predefined management objectives (Pérez-Lapeña *et al.*, 2011). An approach to set acceptable values for α and β based on costs (in economic, political, environmental and social terms) is proposed by Mapstone (1995).

In a negative binomial distribution the variance function equals $V(\mu) = \mu + \frac{\mu^2}{\theta}$, and so variance is negatively correlated with theta (θ). According to Underwood & Chapman (2003), power is strongly affected by the variability in the measurements. Indeed, we found that power strongly increases with increasing theta. A low theta value depicts over-dispersion, which in this case might arise from year-to-year variation in observed seabird numbers or from strong spatial aggregation of seabirds (e.g. the presence of a fishing vessel inside the study area). It is also closely related to the amount of unexplained data variance, which proves that building a good reference model, i.e. a model explaining as much biologically relevant variation as possible, is of key importance to the final impact assessment results.

Another finding of this study is the importance of selecting a well-considered control area. Ideally, this area hosts highly comparable seabird numbers to the wind farm site, allowing us to perform the impact assessment with more degrees of freedom, reflected by better power.

As was shown, power is strongly enhanced by counting for a longer period of time, due to the increase in sample size (Underwood & Chapman, 2003, Pérez-Lapeña *et al.*, 2011). One could argue that the timeframe needed to reach a certain power can be halved by performing two monitoring surveys each month. This is in fact true, but surveys still need to be sufficiently spread over time to avoid temporal autocorrelation. Contrastingly, doubling the effort by counting 20 km² per survey per area - instead of 10 km² - does not result in enhanced power, at least not in a direct way. However, it can yield more reliable count results, which in turn can influence the parameter estimates. If let's say, doubling the count effort per survey has a positive effect on the theta value, or lowers the amount of zero-inflation, this will inevitably be reflected in a higher power. It would be very interesting to know how this count effort per survey is linked to the variation/robustness in parameter estimates.

As a last step we calculated powers based on species-specific reference models of twelve seabird species, as observed at the Thorntonbank study area prior to the construction of turbines in 2008. To detect a of 50% decrease in numbers, a power of 90% is reached within 10 years for two seabird species only, i.e. Northern Gannet and Common guillemot. Within the same time frame, power to detect a 75% decrease in numbers exceeds 90% for all species, except for Common Gull. Poorest results are seen in Common Gull and Black-legged Kittiwake, both exhibiting a significant difference in abundance between control and impact area during reference years. All of these results are based on a monitoring set-up in which there is one monthly survey, with an effort of 10 km² in both the control and impact area.

Maclean *et al.* (2006 & 2007) conducted a comparable study on long-time series of aerial survey count data of five seabird species (Red-throated Diver, Common Scoter, Sandwich Tern, Lesser & Great Black-backed Gull), collected in the UK North Sea waters. The (hypothetical) monitoring set-up in that study is quite different from the one presented here. The authors calculated the power of detecting changes within a study area of varying size (2x2 km², 5x5 km², etc.), with the hypothetical wind farm located in the centre. The study investigates the effect of the gradient of decline (uniform / gradually), spatial scale, survey intensity, survey duration, inclusion of spatial variables and inclusion of reference areas. Maclean *et al.* (2007) concluded that "the statistical power to detect a 50% change in bird numbers remains low (<85%) for all species irrespective of the length of time over which monitoring is carried out", for a significance level of 0.20. The power results presented here are thus clearly higher. We hypothesize that this is largely due to the binning of data, in which day totals instead of single transect counts were used as a base for modelling.

7.5. Acknowledgements

First of all we want to thank C-Power NV & Belwind NV for financing this research, and the Management Unit of the North Sea Mathematical Models (MUMM) for assigning it to us. MUMM staff members Robin Brabant & Steven Degraer are thanked for their logistic support and cooperation throughout the monitoring program. During the early stages of the statistical processing, my colleagues Dirk Bauwens and Paul Quataert provided great and crucial advice. We thank all volunteers (especially Walter Wackenier) and colleagues who assisted during the seabird counts, as well as the crew of the research vessel 'Zeeleeuw', and not in the least the Flanders Marine Institute (VLIZ), for their extremely valuable logistic support.

7.6. References

Bernstein, B.B. & Zalinski, J., (1983). An optimum sampling design and power tests for environmental biologists. Journal of Environmental Management 16: 35-43.

Certain, G., Bellier, E., Planque, B. & Bretagnolle, V. (2007). Characterising the temporal variability of the spatial distribution of animals: an application to seabirds at sea. Ecography 30: 695-708.

Clarke, E.D., Spear, L.B., McCracken, M.L., Marques, F.F.C., Borchers, D.L., Buckland, S.T., Ainley, D.G. (2003). Validating the use of generalized additive models and at sea surveys to estimate size and temporal trends of seabird populations. Journal of Applied Ecology 40: 278-292.

Degraer, S., Brabant, R. & Rumes, B. (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring. Brussels: Management Unit of the North Sea Mathematical Models – Marine Ecosystem Management Unit.

Desholm, M. (2005). Wind farm related mortality among avian migrants – a remote sensing study and model analysis. Ph. D. University of Copenhagen.

Drewitt, A.L. & Langston, R.H.W. (2006). Assessing the impact of wind farms on birds. Ibis 148: 29-42.

Exo, K.-M., Hüppop, O. & Garthe, S. (2003). Birds and offshore wind farms: a hot topic in marine ecology. Wader Study Group Bull. 100: 50-53.

European Wind Energy Association. (2011). The European offshore wind industry - key 2011 trends and statistics. [online] Available at: <http://www.ewea.org/index.php?id=1861> [Accessed 20 June 2012]

Fletcher, D., MacKenzie, D. & Villouta, E. (2005). Modelling skewed data with many zeros: A simple approach combining ordinary and logistic regression. Environmental and Ecological Statistics 12: 45-54.

Fox, A.D., Desholm, M., Kahlert, J., Christensen, T.K. & Petersen, I.K. (2006). Information needs to support environmental impact assessment of the effects of European marine offshore wind farms on birds. Ibis 148: 129-144.

Garthe, S. (1997). Influence of hydrography, fishing activity, and colony location on summer seabird distribution in the south-eastern North Sea. ICES Journal of Marine Science 54: 566-77.

Huettmann, F. & Diamond, A.W. (2006). Large-scale effects on the spatial distribution of seabirds in the Northwest Atlantic. Landscape Ecology 21: 1089-1108.

Jackman, S. (2011). pscl: A package of classes and methods for R developed in the political science computational laboratory. Stanford University.

Karnovsky, N.J., Spear, L.B., Carter, H.R., Ainley, D.G., Amey, K.D., Ballance, L.T., Briggs, K.T., Ford, R.G., Hunt Jr., G.L., Keiper, C., Mason, J.W., Morgan, K.H., Pitman, R.L. & Tynan, C.T. (2006). At-sea distribution, abundance and habitat affinities of Xantus's Murrelets. Marine Ornithology 38: 89-104.

Komdeur, J., Bertelsen, J., Cracknell, G. (eds) (1992). Manual for aeroplane and ship surveys of waterfowl and seabirds. IWRB Spec. Publ. 19. Slimbridge: IWRB.

Langston, R.H.W. & Pullan, J.D. (2003). Windfarms and Birds: An analysis of the effects of windfarms on birds, and guidance on environmental assessment criteria and site selection issues. T-PVS/Inf(2003)12. Report written by Birdlife International on behalf of the Bern Convention.

Larsen, J.K. & Guillemette, M. (2007). Effects of wind turbines on flight behaviour of wintering common eiders : implications for habitat use and collision risk. Journal of Applied Ecology 44: 516-522.

Leopold, M.F., Camphuysen, C.J., ter Braak, C.J.F., Dijkman, E.M., Kersting, K. & van Lieshout, S.M.J. (2004). Baseline studies North Sea Wind Farms: Lot 8 Marine Birds in and around the future sites Nearshore Windfarm (NSW) and Q7. Alterra-report 1048. Wageningen: Alterra.

Leopold, M.F., Dijkman, E.M., Teal, L. & the OWEZ-team (2010). Local Birds in and around the Offshore Wind Farm Egmond aan Zee (T0 & T1, 2002-2010). Report Number C187/11. Wageningen: IMARES.

McCullagh, P. & Nelder, J.A. (1989). Generalized linear models (2nd edition). London: Chapman and Hall.

McDonald, T.L., Erickson, W.P. & McDonald, L.L. (2000). Analysis of Count Data from Before-After Control-Impact studies. Journal of Agricultural, Biological and Environmental Statistics 5: 262-279.

Maclean, I.M.D., Skov, H., Rehfisch, M.M. & Piper, W. (2006). Use of aerial surveys to detect bird displacement by offshore windfarms. BTO Research Report No. 446 to COWRIE. Thetford: British Trust for Ornithology.

Maclean, I.M.D., Skov, H. & Rehfisch, M.M. (2007). Further use of aerial surveys to detect bird displacement by offshore wind farms. BTO Research Report No. 482 to COWRIE. Thetford: British Trust for Ornithology.

Mapstone, B.D. (1995). Scalable decision rules for environmental impact studies: effect size, type I, and type II errors. Ecological Applications 5: 401-410.

MUMM. (2011). Windmolenparken in zee. [online] Available at: http://www.mumm.ac.be/NL/Management/Sea-based/windmills.php> [Accessed 10 December 2011]

Onkelinx, T., Van Ryckegem, G., Bauwens, D., Quataert, P. & Van den Bergh, E. (2008). Potentie van ruimtelijke modellen als beleidsondersteunend instrument met betrekking tot het voorkomen van watervogels in de Zeeschelde. Report INBO.R.2008.34. Brussels: Research Institute for Nature and Forest.

Oppel, S., Meirinho, A., Ramírez, I., Gardner, B., O'Connel, A.F., Miller, P.I. & Louzao, M. (in press). Comparison of five modelling techniques to predict the spatial distribution and abundance of seabirds. Biological Conservation (2011).

Pebesma, E.J., Bio, A.F. & Duin, R.N.M. (2000). Mapping Sea Bird Densities on the North Sea: combining geostatistics and generalized linear models. In: Kleingeld, W.J. & Krige, D.G. (eds). Geostatistics 2000 Cape Town, Proceedings of the Sixth International Geostatistics Congress held in Cape Town, South Africa, April 2000.

Pérez-Lapeña, B., Wijnberg, K.M., Hulscher, S.J.M.H. & Stein, A. (2010). Environmental impact assessment of offshore wind farms: a simulation-based approach. Journal of Applied Ecology 47: 1110-1118.

Pérez-Lapeña, B., Wijnberg, K.M., Stein, A. & Hulscher, S.J.M.H. (2011). Spatial factors affecting statistical power in testing marine fauna displacement. Ecological Applications 21: 2756–2769.

Pettersson, J. (2005). The impact of offshore wind farms on bird life in Kalmar Sound, Sweden. A final report based on studies 1999-2003. Lund: Lunds Universitet.

Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M. & Fox, A.D. (2006). Final results of bird studies at the offshore wind farms at Nysted and Horns Rev, Denmark. Denmark: National Environmental Research Institute.

Potts, J.M. & Elith, J. (2006). Comparing species abundance models. Ecological Modelling 199: 153-163.

R Development Core Team (2011). R: A language and environment for statistical computing. Vienna: R Foundation for Statistical Computing.

Schneider, D.C. (1982). Fronts and Seabird Aggregations in the Southeastern Bering Sea. Marine Ecology - Progress Series 10: 101-103.

Schneider, D.C. & Duffy, D.C. (1985). Scale-dependent variability in seabird abundance. Marine Ecology - Progress Series 25: 211-218.

Schneider, D.C. 1990. Spatial autocorrelation in marine birds. Polar Research 8: 89-97.

Stewart-Oaten, A., Murdoch, W.W. & Parker, K.R. (1986). Environmental impact assessment: "pseudoreplication" in time? Ecology 67: 929-940

Stewart-Oaten, A., Bence, J.R. (2001). Temporal and spatial variation in environmental impact assessment. Ecological Monographs 71: 305-339.

Stienen, E.W.M., Van Waeyenberge, J., Kuijken, E. & Seys, J. (2007). Trapped within the corridor of the Southern North Sea: the potential impact of offshore wind farms on seabirds. In: de Lucas, M., Janss, G.F.E. & Ferrer, M. (eds). Birds and Wind Farms - Risk assessment and Mitigation: 71-80. Madrid: Quercus.

Tasker, M.L., Jones, P.H., Dixon, T.J. & Blake, B.F. (1984). Counting seabirds at sea from ships: a review of methods employed and a suggestion for a standardised approach. Auk 101: 567-577.

Underwood, A.J. 1994. On beyond BACI: sampling designs that might reliably detect environmental disturbances. Ecological Applications 4: 3-15.

Underwood, A.J. & Chapman, M.G. (2003). Power, precaution, Type II error and sampling design in assessment of environmental impacts. Journal of Experimental Biology and Ecology 296: 49-70.

Vanermen, N., Stienen, E.W.M., Courtens, W. & Van de walle, M. (2006). Referentiestudie van de avifauna van de Thorntonbank. Report IN.A.2006.22. Brussels: Research Institute for Nature and Forest.

Vanermen, N. & Stienen, E.W.M. (2009). Seabirds & Offshore Wind Farms: Monitoring results 2008. *In:* Degraer, S. & Brabant, R. (Eds.) (2009). Offshore wind farms in the Belgian part of the North Sea – State of the art after two years of environmental monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. Chapter 8: 151-221.

Vanermen, N., Stienen, E.W.M., Onkelinx, T., Courtens, W., Van de walle, M. & Verstraete, H. (2010). Monitoring seabird displacement: a modelling approach. *In:* Degraer, S., Brabant, R. & Rumes, B. (Eds.) (2010). Offshore wind farms in the Belgian part of the North Sea – Early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of

Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. Chapter 9: 133-152.

Venables, W. N. & Ripley, B. D. (2002). Modern Applied Statistics with S. Fourth Edition. New York: Springer.

Vuong, Q.H. (1989). Likelihood ratio tests for model selection and non-nested hypotheses. *Econometrica* 57: 307-333.

Wenger, S.J. & Freeman, M.C. (2008). Estimating species occurrence, abundance and detection probability using zero-inflated distributions. Ecology 89(10): 2983-2959.

Zeileis, A., Keibler, C. & Jackman, S. (2008). Regression Models for Count Data in R. Journal of Statistical Software 27 (8).

Zuur, A.F., Ieno, E.N., Walker, N.J., Saveliev, A.A. & Smith G.M. (2009). Mixed Effects Models and Extensions in Ecology with R. New York: Springer.

Chapter 8. Radar research on the impact of offshore wind farms on birds: Preparing to go offshore.

R. Brabant[1*], L. Vigin[1], E.W.M. Stienen[2], N. Vanermen[2] & S. Degraer[1]

[1]*Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models (MUMM), Marine Ecosystem Management Section, Gulledelle 100, 1200 Brussels*
[2]*Research Institute for Nature and Forest (INBO), Ministry of the Flemish Government, Kliniekstraat 25, 1070 Brussels*

[*]*Corresponding author: R.Brabant@mumm.ac.be*

Dual radar system installed on the top deck of the C-Power offshore transformation platform on the Thorntonbank

Photo MUMM / RBINS

Abstract

Wind farms have three possible effects on birds. One of them is the barrier effect. Fox *et al.* (2006) and Krijgsveld *et al.* (2011) both described that birds change their direction of flight in the vicinity of a wind farm. It is unknown if this will also be the case for the offshore wind farms in the Belgian part of the North Sea and what the extent of this effect will be. To study the barrier effect there is a need for a technique that provides continuous data on a large scale. Automated radar systems offer such a tool.

The objectives of this study are (1) to develop an analytical procedure to assess the quality of the radar data and to process the data to effectively remove noise (i.e. data reduction); (2) to develop and test a methodology for radar data analysis, including the influence of co-variables, such as wind direction; and (3) to draft the analytical procedure for future radar research in offshore wind farms (i.e. lessons learnt).

The radar system was tested in the port of Zeebrugge, which holds in important tern breeding colony, to get acquainted with the system and the data processing. Foraging flights of terns are typically in a well-defined direction, as is the case for migrating birds. Therefore the foraging flights can be used as a proxy for the migration flights of migrating birds offshore. Variation in the direction of those foraging flights might be in function of co-variables such as wind direction and wind speed. The barrier effect created by offshore wind farms also makes migrating birds change their direction of flight when they approach a wind farm (Petersen *et al.*, 2006; Krijgsveld *et al.*, 2011). This is thus a co-variable that influences the direction of flight of migrating birds. Both parallels allow us using the data that were recorded near the tern colony as a proxy for the future offshore radar research. Lessons learnt will be directly applicable to the offshore work.

The Zeebrugge case study offered a good opportunity to focus on a specific type of birds and flight behaviour with the radar system. A lot of experience was gained and the methodology was developed and fine-tuned for the future research offshore. It can be concluded that the radar system is an appropriate tool to monitor bird movements. It offers a possibility to show significant patterns in bird movements, even if that pattern is rather small.

Samenvatting

Windmolenparken hebben drie mogelijke effecten op vogels. Een daarvan is het barrière-effect. Fox *et al.* (2006) en Krijgsveld *et al.* (2011) toonden beiden aan dat vogels hun vliegrichting aanpasten in de nabijheid van een windmolenpark. Het is onbekend of dit ook het geval zal zijn voor de offshore windmolenparken in het Belgisch deel van de Noordzee en wat de omvang van dit effect zal zijn. Om dit te bestuderen is er nood aan een techniek die continue data aanlevert op een grote schaal. Automatische radarsystemen beantwoorden aan deze vereisten.

Om vertrouwd te raken met het system en het verwerken van de data, werd het radarsysteem getest in de haven van Zeebrugge. Deze herbergt een belangrijke broedkolonie sternen. Foerageervluchten van sternen zijn typisch rechtlijnig in een bepaalde richting. Dit is ook het geval voor vluchten van migrerende vogels. Daarom kunnen deze foerageervluchten als een proxy worden gebruikt voor migrerende vogels op zee. Wijzigingen in de richting van die foerageervluchten zijn mogelijks in functie van co-variabelen zoals windrichting en –snelheid. Het barrière-effect zorgt ervoor dat migrerende vogels hun richting aanpassen bij het naderen van een windmolenpark (Petersen *et al.*, 2006; Krijgsveld *et al.*, 2011). Dit is dus een co-variabele die de vliegrichting van migrerende vogels beïnvloedt. Beide parallellen laten het toe om de data die verzameld werden in de buurt van de sternenkolonie te gebruiken als een proxy voor het toekomstige offshore radaronderzoek.

De test fase in Zeebrugge bood de mogelijkheid om met het radarsysteem te focussen op een bepaalde soort en het vlieggedrag van die soort. Er werd veel ervaring opgedaan en de methodologie voor het toekomstige onderzoek offshore werd ontwikkeld. Er kan geconcludeerd worden dat het radarsysteem een geschikt middel is om vliegbewegingen van vogels te onderzoeken. Het biedt de mogelijkheid om significante patronen in vliegbewegingen aan te tonen, zelfs indien dit patroon eerder zwak is.

8.1. Introduction

The European directive on the promotion of electricity produced from renewable energy sources imposes upon each Member State a target figure of the contribution of the production of electricity from renewable energy sources. Offshore wind farms are expected to make an important contribution to achieve that target figure. A zone in the Belgian part of the North Sea (BPNS), with a total surface of 238 km², is reserved for the production of electricity. This zone starts at about 20 km from the coast and is orientated perpendicular to the coast. Once the construction of the different wind farms is finished, there will be several hundreds of wind turbines in that area. The UK and the Netherlands are also planning to construct wind farms in the Southern North Sea.

Wind farms have three possible impacts on birds (Exo *et al.*, 2003; Fox *et al.*, 2006; Drewitt & Langston, 2006): (a) collision of birds with the structures (direct impact); (b) the disturbance and alteration of the distribution / behaviour of local birds during foraging and resting, this is called displacement (indirect impact) and (c) a barrier effect, i.e. the disturbance of flying birds by the presence of the wind farms (indirect impact). Collisions of birds with fixed and rotating structures of wind turbines have been recorded in several wind farms on land (Everaert & Stienen, 2006; Barclay *et al.*, 2007; etc.). For obvious reasons it is more difficult to know the number of collision victims from an offshore wind farm. Collision models offer a tool to estimate that number. Several collision models already exist (Bolker *et al.*, 2006; Troost, 2009; Band *et al.*, 2007) and they take certain specifications of the wind farm and wind turbines into account. To make a realistic estimate of the number of collisions, it is necessary to know the flux of birds trough the wind farm. This will be tackled in the future by the vertical radar and visual flux counts (to validate the vertical radar data). Second, displacement of local birds is shown to be highly species specific (Petersen *et al.*, 2006). In the wind farms of Nysted and Horns Rev in Denmark, avoidance behaviour was most notable for divers, scoters, auks and long-tailed ducks (Petersen *et al.*, 2006). Leopold *et al.* (2010) showed a similar response of divers, grebes, gannets, little gulls and auks in the Dutch part of the North Sea. Ship-based visual counts are performed on a monthly base and give insight in the species-specific displacement behaviour. Finally, Fox *et al.* (2006) and Krijgsveld *et al.* (2011) both described a barrier effect of wind farms of birds in Denmark and the Netherlands, respectively. Both studies showed that birds change their direction of flight in the vicinity of a wind farm. It is unknown if this will also be the case for the offshore wind farms in the BPNS and what the extent of this effect will be. An estimated number, based on land based and ship based counts, of 1 to 1.3 million birds migrate through the southern North Sea each year (Stienen *et al.*, 2007). However, little is known about migration at night and intensive migration events at sea. The Southern part of the North Sea has the shape of a bottleneck. Offshore wind farms may act as barriers for birds migrating through that bottleneck. It is unknown if the extent of this barrier effect changes during different circumstances (e.g. at night, during periods with low visibility). In the future, birds will encounter several wind farms during their migration through the southern North Sea and thus, may suffer from the cumulative effect of the encountered wind farms. That cumulative effect of the wind farms along their migration trajectory might affect their energy expenditure, although the impact on populations of long-lived seabirds are probably only marginal (Masden *et al.*, 2009, 2010; Poot *et al.*, 2011).

To study the barrier effect there is a need for a technique that provides continuous data on a large scale. Radars have been used in similar research for several years abroad, for instance in Denmark (Petersen *et al.*, 2006) and the Netherlands (Krijgsveld *et al.*, 2011). They provide continuous data, also during conditions where it is very difficult to gather visual data (e.g. at night, during bad weather conditions, far offshore). The range in which data are gathered depends on the system and settings, but is typically around 3 nm in similar studies. This allows studying patterns of flight movements in a wide range. However, there are also several restrictions to this technique: the recorded radar data have a low taxonomic resolution, quantification of the data is very difficult and the radar also records objects other than birds (e.g. sea surface, ships and rain). All unwanted detections are being referred to as clutter.

The objectives of this study are (1) to develop an analytical procedure to assess the quality of the radar data and to process the data to effectively remove noise (i.e. data reduction); (2) to develop and test a methodology for radar data analysis, including the influence of co-variables, such as wind direction; and (3) to draft the analytical procedure for future radar research in offshore wind farms (i.e. lessons learnt).

8.2. Materials and methods

8.2.1. Radar system

In 2010, MUMM purchased a Merlin radar system from DeTect Inc. (Florida, USA). The system consists of two identical solid state S-band radar antennas, one scanning in the horizontal pane and one in the vertical. The horizontal scanning radar (HSR) provides information on flight tracks and therefore on the possible avoidance behaviour. The vertical scanning radar (VSR) provides data on the flight altitudes and the flux of birds trough the area. The range of the radars can be specified in the system's settings. The radars are usually operated at a range between two and four nautical miles for the HSR and 0.75 nautical miles for the VSR. This type of system records birds continuously year-round and is remotely manageable.

The Merlin software of the radar is designed to record and track moving objects. The objects of interest are in this case birds. When the radar energy reflects on a bird and this is received by the radar antenna, a radar echo then appears on the raw radar screen. If the echo meets certain (plotting) criteria (minimum size, intensity, etc.) it will be plotted on the processed Merlin screen. If the radar detects the same echo in four consecutive scans, it is considered as a confirmed 'track' and will be written to the database, together with its own, unique track identification code. The radar further registers for every record over 40 variables (e.g. time, location, speed, heading, size).

Obviously not only birds are recorded by the radar; this also happens for rain, waves, boats, wind turbines, etc. These unwanted echoes are being referred to as 'clutter'. For offshore studies the biggest source of clutter is the sea surface (further referred to as 'sea clutter') and the clutter created by the high reflectance of the steel surfaces of large vessels (further referred to as 'boat clutter'). This clutter needs to be filtered out of the database (i.e. data processing). Visual observations (further referred to as 'groundtruthing) are helpful in that process as they allow to validate the radar registrations. During the groundtruthing one person is looking at the radar screen and another person is outside to locate the targets visually. If an object is both seen visually and on the radar screen, then it is tagged on screen and visually confirmed information is added. That way it is possible to mark targets as birds (and add the species and number of birds), but also as boats, sea clutter, etc. This renders a separate database of the radar tracks (and all the variables that are recorded by the radar) combined with the visually confirmed information. With such a database it is possible to determine which variables discriminate the most between the groundtruthed classes.

8.2.2. Case study: breeding tern flight patterns as a proxy for offshore migration patterns

The port of Zeebrugge was chosen as a test location for the radar system because it has easy access and it overlooks the sea. The test phase aimed at getting acquainted with the system and the data processing. The site also holds an internationally important breeding colony of terns, nesting on an artificial peninsula on the inside of the eastern port jetty (Figure 1). There is also a wind farm on that jetty, consisting of 14 turbines, causing a high number of collisions (Everaert & Stienen, 2006).

During the breeding season in 2011, 1354 couples of common tern *Sterna hirundo* bred on the artificial peninsula. Also lower numbers of sandwich tern *Sterna sandvicensis* (54 breeding couples) and little tern *Sternula albifrons* (102 breeding couples) bred in Zeebrugge. Hence, 90% of the birds in the area of the breeding colony were common terns.

Figure 1. Radar test location in the port of Zeebrugge.

During the incubation phase and once the chicks have hatched, the adult terns make regular foraging flights to feed themselves or to feed their partners and chicks.

These foraging flights were used as a proxy for the seasonal migration flights to be targeted by the offshore wind farm monitoring programme. There are two parallels between migrating birds, that will encounter the offshore wind farms twice a year during spring and autumn migration and the foraging flights that common terns make during the breeding season. Firstly, radar tracks of foraging common terns are actually very similar to tracks of migrating birds. As mentioned earlier, 1 – 1.3 million birds migrate through the ‘migration bottleneck’ of the Southern North Sea twice a year on their way to and from the breeding colonies and the wintering grounds (Stienen *et al.*, 2007). Foraging flights of terns are typically very directional flights from the breeding colony to the foraging grounds. So, in both cases the birds fly in a well-defined direction and more or less maintain that heading. Therefore the foraging flights can be used as a proxy for the migration flights of birds offshore. Secondly, variation in the direction of those foraging flights might be in function of co-variables such as wind direction and wind speed. The barrier effect created by offshore wind farms also makes migrating birds change their direction of flight when they approach a wind farm (Petersen *et al.*, 2006; Krijgsveld *et al.*, 2011). This is thus a co-variable that influences the direction of flight of migrating birds, similar to co-variables that influence the terns during their foraging flights.

Both parallels allow us using the data that were recorded near the tern colony as a proxy for the future offshore radar research. Lessons learnt will be directly applicable to the offshore work.

8.2.3. Data analysis

8.2.3.1. Data availability

8.2.3.1.1. Groundtruthing

On seven days in the months September, October and November visual counts were done to validate the radar data. Over 500 tracks were classified as birds (identified to species level, whenever possible), vessels, sea clutter or boat clutter.

8.2.3.1.2. Flight patterns

The system was installed on of the eastern jetty of the port of Zeebrugge in early January 2011. After dealing with several technical problems and gradually improving the tracking ability of the radar by changing the system settings, the system performed well and collected data on a continuous base. Before the tern breeding season the radar was moved closer to the colony (Figure 1) and the range was set at 0.5 nm. The radar collected data from May 19th until June 26th. The six weeks of data collection near the colony resulted in a total amount of 76 Gb of data.

In this study, only the data from June were used because this is the only period during which data from the breeding colony were recorded in a consistent manner, with the same system settings, radar range and without technical failures.

8.2.3.2. Analytical procedure to process the data

To effectively remove clutter from the radar data and to prepare the data for further analysis it is necessary to process the data. This also includes separating terns arriving in the colony from the terns that are departing. Otherwise it is impossible to assess the impact of co-variables on the flight direction. An analytical procedure was created for this data processing (Figure 2).

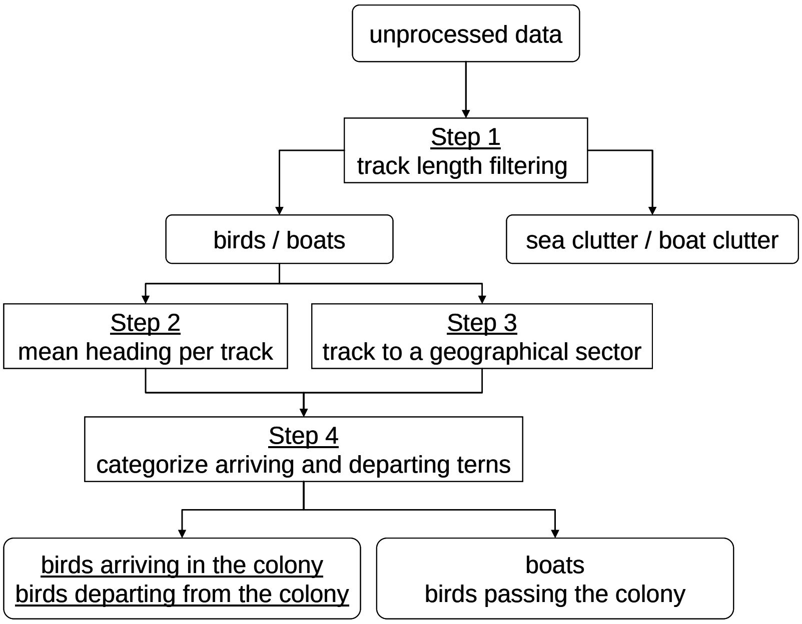


Figure 2. Model to process the radar data

To successfully remove clutter from the database, the data were filtered on track length in the first step of the data processing. All tracks shorter than a certain cut off value were removed. The cut off value was chosen based on the groundtruthed data. This step also removed bird tracks with a track length shorter than this cut off value. After this step, tracks of birds and boats longer than this cut off value were retained.

For further analysis it was necessary to know the mean direction of a track. Therefore, in step 2, the mean direction of every track was calculated, based on the direction values of the individual records of the tracks. To calculate the average value of the heading of a track, every heading value was split in its sin and cos value and the average of the sin and cos was calculated per track. Those values were then converted back to degrees using the arctan2 function.

Step 3 of the model is assigning every track to a certain geographical sector around the colony. To do so, the area around the breeding colony was divided in eight sectors of 45°. The area with the highest breeding density of terns in 2011 was chosen as the center point of this spatial analysis. Sector 1 is the direction straight from the colony to the sea, and thus over the port jetty. The boundaries of the sectors and the heading of the tracks were re-scaled to values between -90 and 90°, with 0° being the direction perpendicular to the coastline, to make the results more comprehensible. The boundaries of sector 1 are therefore 0° to 45° (Figure 3). Using GIS (Spatial Analyst ArcGIS v.9.3), every track's center point was determined, and used to assign the track to a sector.

Step 4 separated birds arriving in the colony from birds flying away from it and from birds passing by the colony. To do so it was necessary to combine the geographical sector and the mean heading of a track. For example, if a bird flew in sector 1 with a mean heading of 180°, the bird is arriving in the colony. A bird in sector 5 with that same heading is departing from the colony. An algorithm was created that categorizes birds as arriving in, departing from or passing by the colony. It combines the mean heading of the track, its sector and *a priori* defined heading intervals for arriving and departing birds. According to that algorithm, a bird is catalogued as departing from the colony when the heading is within the sector boundaries. For instance, tracks in sector 1 with a heading between 0° and 45° are departing from the colony. If their heading is between 180° and 225°, then they are considered as arriving in the colony. All tracks in sector 1, with a heading outside of those intervals are considered as birds passing the breeding colony. For this study, intervals of 45° wide were chosen. If the intervals would be wider, one risks including too much birds that pass by the colony, in the analysis. It is clear that also some arriving birds and birds departing from the colony are catalogued as passing birds, but since this happens randomly in all sectors and since this study is focusing on flight behaviour and is not quantitative, this can be accepted.

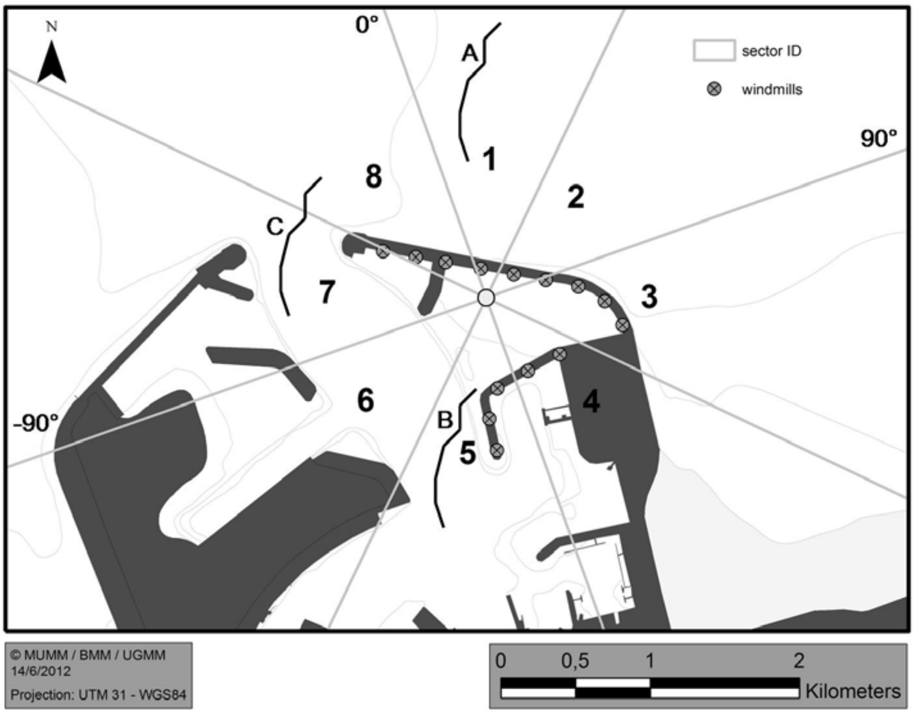


Figure 3. Geographical sectors around the center of the tern colony that were used in step 3 and 4 of the data processing model. The tracks A, B and C are identical and have the same mean heading of 180°. The tracks would, respectively, be catalogued as arriving, leaving and passing by the colony by the algorithm that is explained in step 4.

After those steps it is possible to statistically analyze the processed data to investigate how the co-variables wind direction and velocity influence the direction of flight of arriving and departing birds (dependent variable heading). Wind direction and speed were recorded by the Flemish hydrographic service on the eastern port jetty of the port of Zeebrugge. All tracks were pooled in eight groups according to the wind direction during the time of the recording (N, NE, E, SE, S, SW, W and NW). This way it was possible to assess if the heading of arriving and departing terns differs significantly at different wind directions.

In those analyses only the data from the sectors oriented towards the open sea were retained (sectors 1, 2, 7, 8) because study is focusing on the direction of flight of terns that are foraging out at sea (and not in the port). All statistical analyses were performed in Statistica (v.10).

Since the center of the analysis is the hart of the colony, and not the radar location, the area where the radar recorded is different in the different sectors. Therefore the number of tracks in the different sectors was multiplied by a unique factor, to correct for this size difference.

8.3. Results

8.3.1. Data processing

The groundtruthed data show that the track length of sea and boat clutter is shorter than the recorded tracks for birds and vessels. Sea clutter (ANOVA post-hoc Tukey test, p= 0.0001) and boat clutter (p= < 0.0001) consist of significantly shorter tracks than birds. The mean track length of sea and boat clutter is around five records per track. Tracks of small vessels are significantly longer than bird tracks (p= < 0.0001). Track length of large vessels is similar to those of birds (p= 0.8431; Figure 4a). Based on these results the cut off value of seven records as minimum track length was chosen for the first step of the data processing model.

The speed of birds is significantly different from the speed of sea clutter (ANOVA post-hoc Tukey test, p= <0.0001) and boat clutter (p= <0.0001) (Figure 4b). This means, that also the variable speed makes it possible to discriminate between sea clutter / boat clutter and birds / ships.

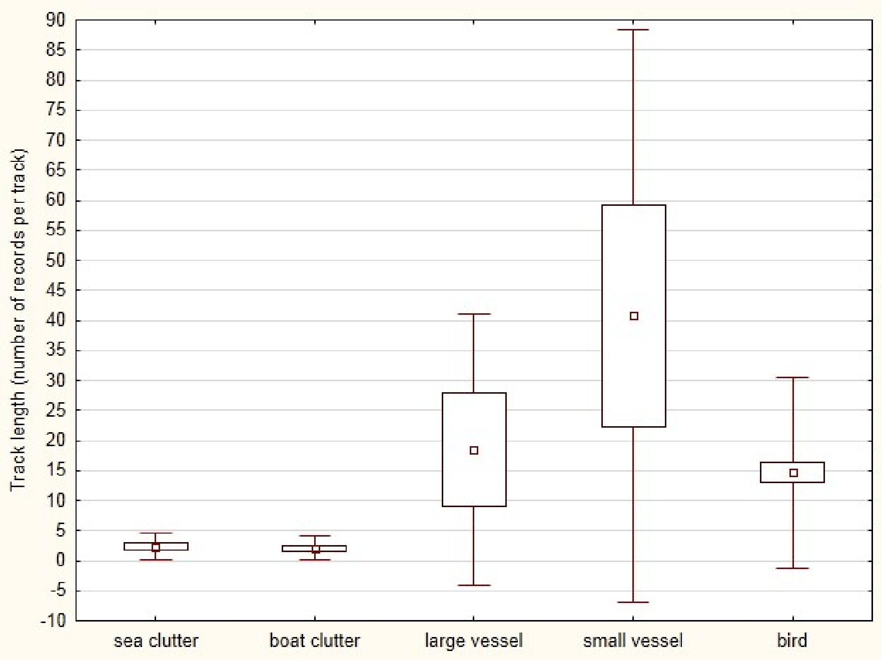


Figure 4a. Track length of groundtruthed tracks assigned to sea clutter, boat clutter, large vessels, small vessels and birds. Mean ± standard deviation (whiskers) and 95% confidence intervals (box).

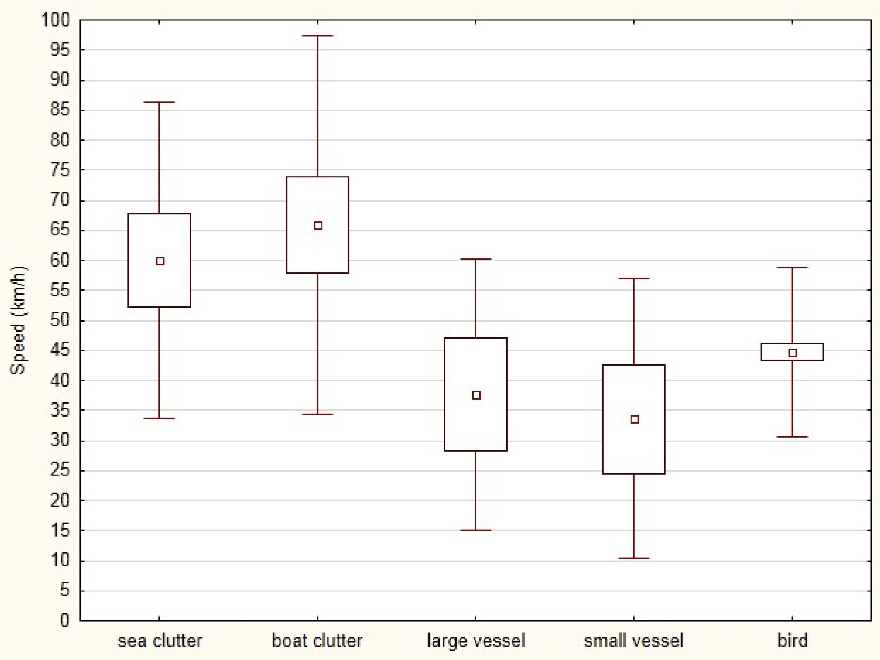


Figure 4b. Speed (km/h) of groundtruthed tracks assigned to sea clutter, boat clutter, large vessels, small vessels and birds. Mean ± standard deviation (whiskers) and 95% confidence intervals (box).

Sea clutter and boat clutter were removed from the dataset based on the track length (step 1 of the data processing). Boats themselves were still in the database after step 1 because they usually have tracks longer than the cut off value of seven records (Figure 4a.). As boats were obviously not sailing towards the breeding colony, they were removed by the algorithm in step 4 of the data processing.

After step 1 of the data processing model the database size was already reduced to approximately 600 MB. This is less than 1% of the Merlin data. After the entire data processing, 50711 arriving and 46117 departing tracks were retained in the four sectors that are facing the sea.

8.3.2. Results on foraging flights:

Significant differences in the headings of bird tracks were found between all wind directions, except for SW-E, NW-N and SE-S for arriving terns and for NW-SE, NW-E and SE-E for departing terns (Table 1, Table 2 and Figure 5).

Table1. Post-hoc Tukey test p-values for arriving common terns

	NE	E	N	NW	S	SW	W	SE
NE		< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001
E	< 0.0001		< 0.0001	< 0.0001	< 0.0001	0.9964	< 0.0001	< 0.0001
N	< 0.0001	< 0.0001		0.5222	< 0.0001	< 0.0001	0.0001	< 0.0001
NW	< 0.0001	< 0.0001	0.5222		< 0.0001	< 0.0001	< 0.0001	< 0.0001
S	< 0.0001	< 0.0001	< 0.0001	< 0.0001		< 0.0001	< 0.0001	0.2047
SW	< 0.0001	0.9964	< 0.0001	< 0.0001	< 0.0001		< 0.0001	< 0.0001
W	< 0.0001	< 0.0001	0.0001	< 0.0001	< 0.0001	< 0.0001		< 0.0001
SE	< 0.0001	< 0.0001	< 0.0001	< 0.0001	0.2047	< 0.0001	< 0.0001	

Table 2. Post-hoc Tukey test p-values for departing common terns

	NE	N	NW	W	S	SE	SW	E
NE		< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001
N	< 0.0001		0.0145	< 0.0001	< 0.0001	< 0.0001	< 0.0001	0.0002
NW	< 0.0001	0.0145		< 0.0001	0.0074	0.9702	< 0.0001	1
W	< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001
S	< 0.0001	< 0.0001	0.0074	< 0.0001		0.0004	< 0.0001	< 0.0001
SE	< 0.0001	< 0.0001	0.9702	< 0.0001	0.0004		< 0.0001	0.9452
SW	< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001		< 0.0001
E	< 0.0001	0.0002	1	< 0.0001	< 0.0001	0.9452	< 0.0001	

In case of S and SE winds the terns arrive with the lowest mean direction, -44.5° and -47.2° respectively. In case of other wind directions the mean heading increases to a maximum when the wind was coming from the N (-25.8°) and NW (-23.6°).

Terns departing from the colony have the lowest mean heading (-45.8°) when the wind was coming from the NE. When the wind was coming from the W, the mean heading was the highest (-9.8°).

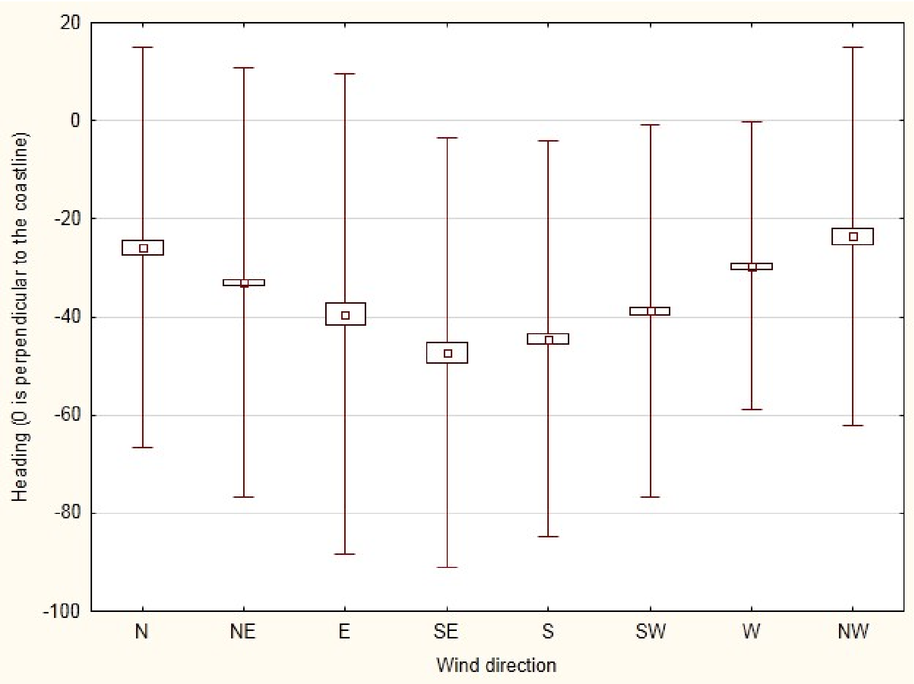


Figure 5a. Heading of terns arriving in the colony, grouped by wind direction. Mean ± standard deviation (whiskers) and 95% confidence intervals (box). A heading of 0° represents a direction perpendicular to the coastline.

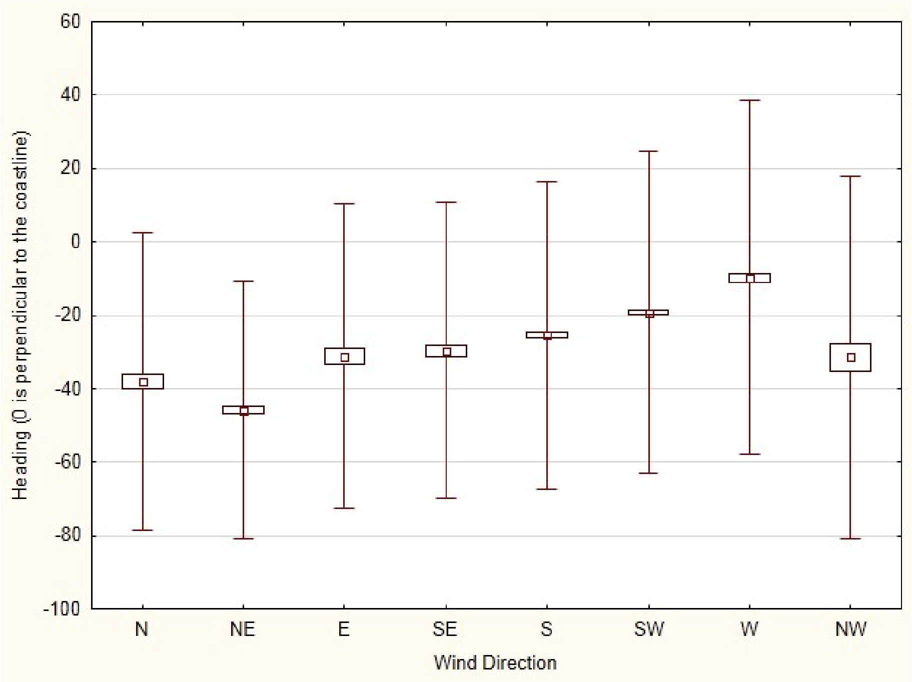


Figure 5b. Heading of terns departing from the colony, grouped by wind direction. Mean ± standard deviation (whiskers) and 95% confidence intervals (box). A heading of 0° represents a direction perpendicular to the coastline.

Both arriving and departing birds are influenced by the wind directions. In June 2011 the wind was coming predominantly from the Southwest (45% of the time) and the Northeast (16% of the time). The wind speeds were not significantly different (p= 0.0769) when the wind was coming from those two directions. Average wind speed was rather low (mean <4m/s) when the wind was coming from the E and the SE (Figure 6).

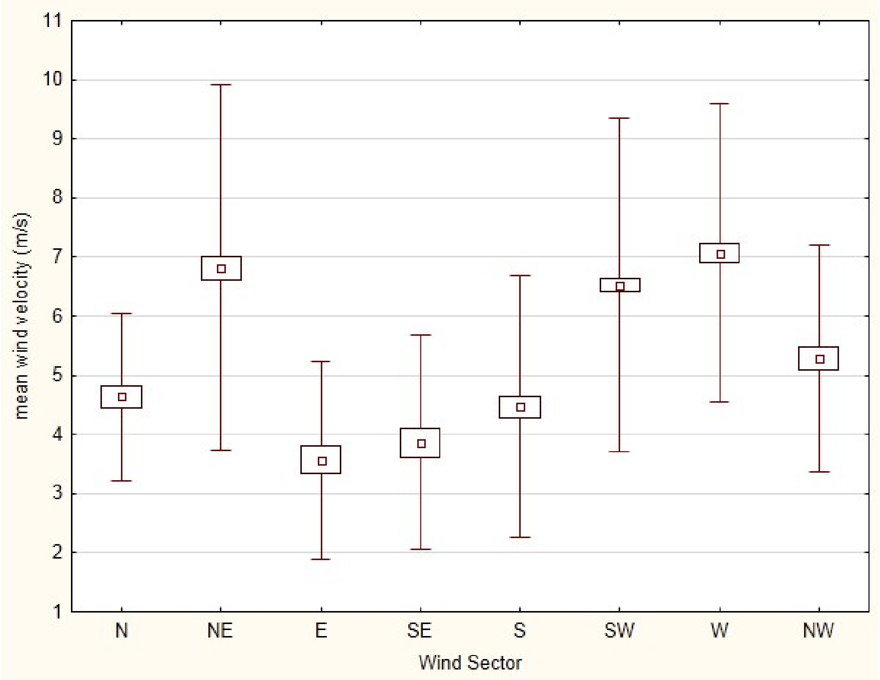


Figure 6. Wind direction and wind speed in the month of June 2011.

In case of southwesterly wind, a large portion of terns (28%) used sector 1 to leave the colony. This was not the case when the wind was coming from the NE. During those periods not a lot of birds were using the sea facing sectors to depart from the colony. However, 23 % of the birds used sector 7 to depart from the colony.

Arriving terns showed a very different pattern from departing terns during southwesterly winds. Few birds used the sea facing sectors to arrive in the colony. When the wind was coming from the NE the terns were arriving very dispersed. A notable portion (16%) was however arriving via sector 1 (Figure 7).

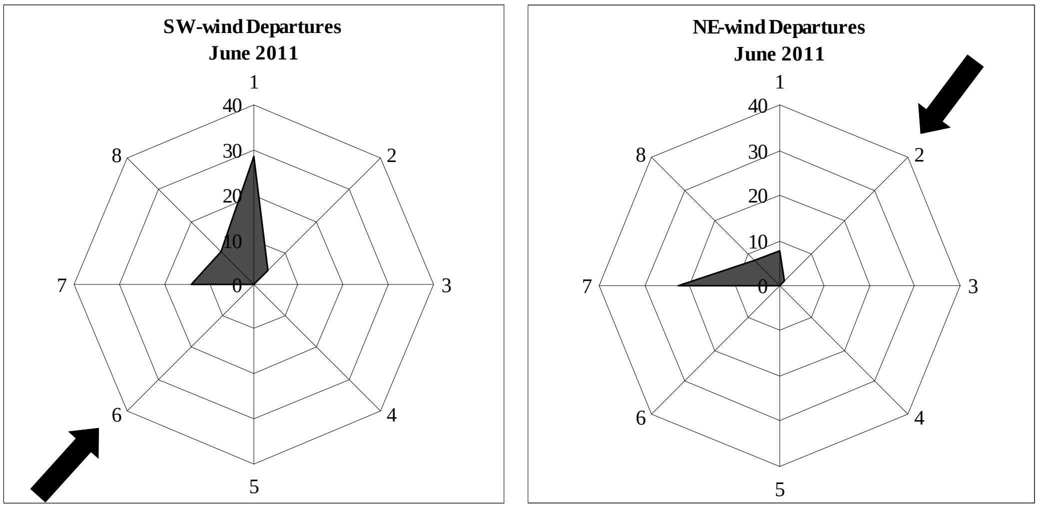


Figure 7a. Percentage of terns departing from the colony via the sea facing sectors when the wind was coming from the SW (left) versus wind coming from the NE (right). The black arrows indicate the wind direction.

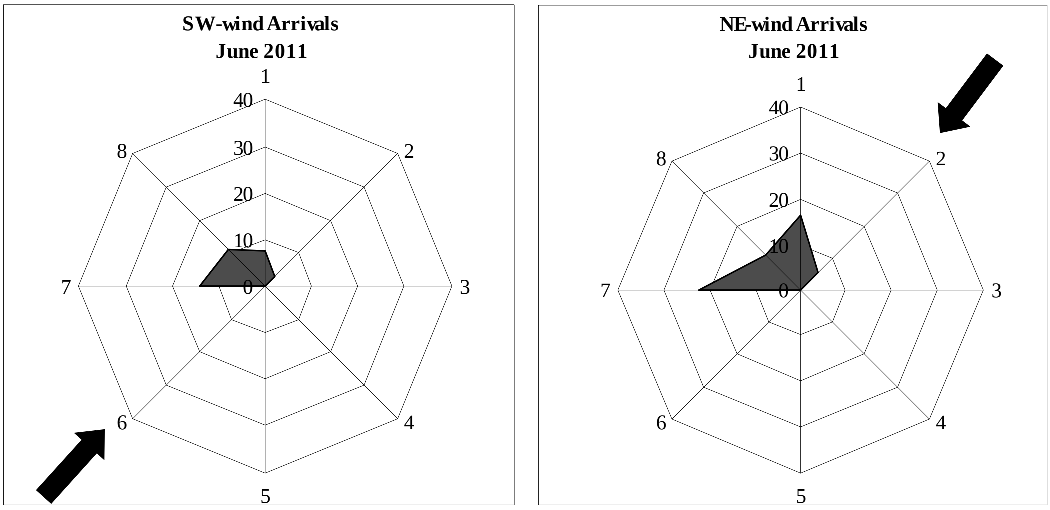


Figure 7b. Percentage of terns arriving in the colony via the sea facing sectors when the wind was coming from the SW (left) versus wind coming from the NE (right). The black arrow indicates the wind direction.

8.4. Discussion

8.4.1. Flight behaviour of foraging common terns during the breeding season.

The areas just around the port of Zeebrugge and especially those to the west of the port (as far as the Wenduinebank), are the most important feeding grounds for the common terns that are breeding in the port of Zeebrugge (Figure 8, Vanaverbeke *et al.*, 2011).

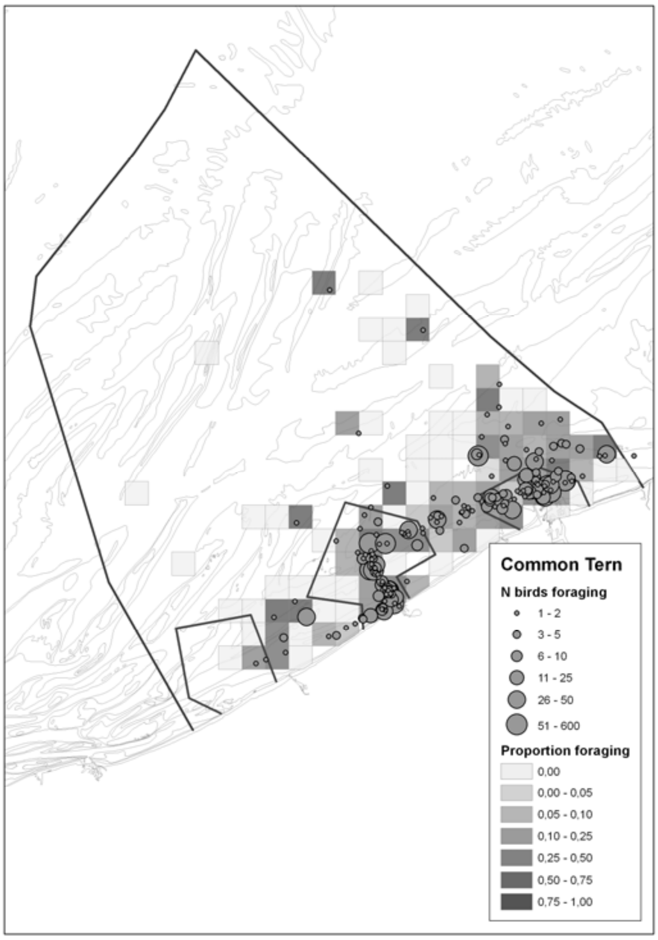


Figure 8. Foraging activity of common tern during the breeding season in 1992-2010. Dots show the numerical occurrence of foraging terns and shaded grids represent foraging frequency (i.e. number of actively foraging individuals/ total numbers). Drawn lines indicate the border of the BPNS and the three special protection areas under the birds directive (Vanaverbeke *et al.*, 2011).

The results of this radar study show that wind direction plays an important role in the foraging behaviour of common terns in Zeebrugge. A portion of the common terns is always foraging in the port itself or is flying via sectors 3 and 6 towards/from the sea, but a large number forages at sea leaving/entering the harbour via sectors 1, 2, 7 or 8. The direction in which these birds arrive in and depart from the colony is clearly influenced by the wind direction and probably also by the wind speed.

Our results suggest that a lot of birds leave the colony with side- or tailwind, then forage with headwind, so they can use the headwind whilst hovering, and come back with side- or headwind. As such it looks as if they make a loop from the colony to the feeding ground and back, as was the case the colonies in Norfolk and Anglesey, United Kingdom (Perrow *et al.*, 2011). The direction in which they fly in that loop depends on the wind direction. Visual observations of arriving and departing terns in Zeebrugge, made by Alvarez del Villar D’Onofrio (2005), indeed show that most of the common terns arrived from the West and the North of the colony and departed towards the West and the Southwest. These observations confirm the radar observations and also support the theory that the common terns make a loop during their foraging.

Unfortunately the radar range was set at only 0.5 nm during the time of recording, because we wanted to register micro-avoidance of common terns around a single turbine. A longer range would have allowed for a better interpretation of the direction of the foraging flights.

8.4.2. Evaluation of parallels between the study of foraging terns and the assessment of the impact of offshore wind farm on birds with an automated radar system.

Although the case of the foraging terns in Zeebrugge is very different from an offshore wind farm site, there are many parallels in the processing and the analysis of the data that were gathered in Zeebrugge and the data that will be gathered offshore. Lessons-learnt here will hence have a direct added value for the future research offshore.

The analytical procedure to process radar data that was developed in this study showed that it is possible to successfully remove clutter and retain high quality data on bird movements. Ground truthed data made it possible to discriminate between different types of targets by using the variables that are logged by the radar (track length, speed, etc.). A large advantage of the case study in Zeebrugge was that one species was very dominant in the breeding colony area. Because all bird tracks in the area belonged to (almost) one species, the filtering of the data could be done in a very straightforward way. This filtering also removed a lot of bird tracks (with track length shorter than seven records), but since this is not a quantitative study, but a study focusing on flight patterns, this is acceptable. The data that will be collected offshore will need to be processed in a similar way as was done with the Zeebrugge data. The species composition offshore is a lot more diverse than in the breeding colony in Zeebrugge. Several seabird species are present in that area and a wide range of species, from geese to small passerines, is migrating through the area twice a year. If the offshore data would be filtered in the same way as we filtered the tern data, one risks loosing a specific segment of bird species and as such bias the data. Since the detection loss of the radar increases with the distance from the radar, especially for smaller birds, a lot of the smaller species would possibly be lost. That is why the track length criterion will need to be re-assessed for offshore conditions.

Instead of using only one variable (as was done with the tern data), a combination of several discriminating variables will be used to filter the offshore radar data into different target groups (e.g. clutter, boats, large gulls, terns, passerines). The quality of that filter will off course depend on the data that were groundtruthed to be used as input to build such a filter. Therefore it will be important to do visual groundtruthing of the radar data on site on a regular base. This will also render the necessary information on the sampling efficiency and the taxonomic resolution (i.e. to what level species (groups) can be separated) of the radar system.

The processed data from this study were further analyzed with GIS, an essential step to be able to interpret the ecological relevance of the data. This will also be necessary for the offshore data. However, instead of assigning the data to sectors around a breeding colony, the offshore data will be assigned to a grid cell of a grid that covers the wind farm and the area around it. This is similar to what was done by Petersen *et al.* (2006) and Krijgsveld *et al.* (2011). Therefore the number of tracks and their heading in grid cells within the wind farm will be compared to the number of tracks and heading in grid cells outside wind farm. As such, avoidance behaviour of birds in response to the wind farm can be quantified. The way the data are handled in GIS and the way the tracks are assigned to a certain grid cell is identical to what was done with the test data. After this step it is also possible to test the importance of co-variables such as wind direction, visibility, etc.

A third parallel between the test phase in Zeebrugge and the offshore study is the use of multiple regression modeling to study the importance of several (explanatory) co-variables that influence the flight behaviour (i.e. response variable) of birds. The most interesting co-variables to include are wind speed and direction, visibility, sea-state and activity of the turbines. For the Zeebrugge test phase we tried to explain the heading of arriving or departing terns (i.e. response variable) by the co-variable wind direction and wind speed. We here encountered the problem that wind direction is an angular variable. This means, for instance, that the values 1 and 359 are almost the same, so this value can not be used in a linear manner. This problem could not be solved so far. Fisher & Lee (1992) describe of solution for this kind of problems, but it was not possible to include this in this study.

An attempt was done to register birds in the vicinity of a single turbine and therefore study micro-avoidance of birds around a turbine. This was the reason why the HSR range was set at only 0.5 nm. This appeared not to be possible due to the high clutter environment created by the concrete jetty. Krijgsveld *et al.* (2011) operated their radar system offshore at 0.75 nm and were able to register birds down to 10 m and less from the turbine. Therefore this should be possible in the offshore study location.

8.5. Conclusions

Wind direction was demonstrated to play an important role in the foraging behaviour of common terns in Zeebrugge. The direction in which these birds arrive in and depart from the colony is clearly influenced by the wind direction and probably also by the wind speed.

The Zeebrugge case offered a good opportunity to focus on a specific type of birds and flight behaviour with the radar system. This was relevant for the future impact assessment of offshore wind farms because the approach and methodology are very similar. With this case study, a lot of experience was gained and the methodology was developed and fine-tuned for the future research offshore. It can be concluded that the radar system is an appropriate tool to monitor bird movements. It offers a possibility to show significant patterns in bird movements, even if the pattern's strength is rather small.

It was not possible to register birds in close vicinity of a single turbine because of the clutter created by the port jetty. However, based on the results of Krijgsveld *et al.* (2011), we are confident that this will be possible in the offshore wind farms.

8.6. References

Alvarez del Villar D'Onofrio, Adriana M. (2005). FORAGING AREAS FOR SANDWICH AND COMMON TERNS IN BELGIAN MARINE WATERS. Thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Ecological Marine Management. 57 pp.

Bolker E.D., Hatch J.J. & Zara C. (2006) Modelling bird passage through a wind farm. http://www.cs.umb.edu/~eb/windfarm

Band, W., Madders, M. & Whitfield, D.P., (2007). Developing field and analytical methods to assess avian collision risk at wind farms. M. de Lucas, G.F.E. Janss & M. Ferrer. Birds and Wind Farms: Risk Assessment and Mitigation. Quercus. Madrid. pp. 259 – 275.

Barclay, R.M.R., Baerwald, E.F. & Gruver, J.C., (2007). Variation in bat and bird fatalities at wind energy facilities: assessing the effects of rotor size and tower height. (Can. J. Zool.) 85, 381 – 387.

Brabant R. & Jacques T.G. (2009) Research strategy and equipment for studying flying birds in wind farms in the Belgian part of the North Sea. *In:* Degraer, S. & Brabant, R. (Eds.) (2009) Offshore wind farms in the Belgian part of the North Sea: State of the art after two years of environmental monitoring. Royal Belgian Institute for Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. pp. 223-235.

De Groote, D. & Roggeman, W. (2006) Gebruik van radarsystemen voor monitoring van de avifauna op de Thorntonbank. 49 pp.

Desholm, M., Fox, A.D., Beasley, P.D.L. & Kahlert, J., (2006). Remote techniques for counting and estimating the number of bird-wind turbine collisions at sea: a review. (Ibis) 148, 76 – 89.

Drewitt, A.L. & Langston, R.H.W., (2006). Assessing the impacts of wind farms on birds. (Ibis) 148, 29 – 42.

Exo, Klaus-Michael, Huppop, O.& Garthe., S., (2003). Birds and Offshore wind farms: a hot topic in marine ecology. Wader Study Group Bulletin 100: 50-53.

Everaert, J. & Stienen, E.W.M., (2006). Impact of wind turbines on birds in Zeebrugge (Belgium). Significant effect on breeding tern colony due to collisions. Biodivers. Conserv. 15(10).

Fisher N.I. & Lee A.J., (1992). Regression Models for an Angular Response. Biometrics 48, 665-677.

Fox, A.D., Desholm, M., Kahlert, J., Christensen, T.K. en Petersen, I.B.K., (2006). Information needs to support environmental impact assessment of the effects of European marine offshore wind farms on birds (Ibis) 148, 129 – 144.

Krijgsveld, K.L., Fijn, R.C., Japink, M., van Horssen, P.W., Heunks, C., Collier, M.P., Poot, M.J.M., Beuker, D. & Dirksen, S., (2011). Effect studies Offshore Wind Farm Egmond aan Zee, Final report on fluxes, flight altitudes and behaviour of flying birds. Bureau Waardenburg. 330 pp.

Langston R.H.W. & Pullan J.D., (2003). Windfarms and birds: An analysis of the effects of windfarms on birds, and guidance on environmental assessment criteria and site selection issues. Report by Birdlife International on behalf of the Bern Convention. Council of Europe T-PVS/Inf (2003) 12. See also Bern Convention 'Draft Recommendation' T-PVS (2003) 11.

Leopold, M.F., Dijkman, E.M., Teal, L. & the OWEZ-Team (2010). Local Birds in and around the Offshore Wind Park Egmond aan Zee (OWEZ) (T-0 & T-1, 2002-2010) IMARES, Wageningen UR Report number: OWEZ R 221 T1 20111220, pp. 269.

Masden, E.A., D.T. Haydon, A.D. Fox, R.D. Furness, R. Bullman & M. Desholm, (2009). Barriers to movement: impacts of wind farms on migrating birds. ICES Journal of Marine Science 66: 746-753.

Masden, E.A., Fox, A.D., Furness, R.W., Bullman, R., & Haydon, D.T. (2010). Cumulative impact assessments and bird/wind farm interactions: developing a conceptual framework. Environmental Impact Assessment Review. *Environmental Impact Assessment Review*. 30:1-7

Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M. & Fox, A.D. (2006). Final results of bird studies at the offshore wind farms at Nysted and Horns Rev, Denmark. NERI Report request. Commissioned by DONG energy and Vattenfall A/S. National Environmental Research Institute. Ministry of the Environment. Department of Wildlife Ecology and Biodiversity. 161 pp.

Perrow, M.R., Skeate, E.R., Gilroy, J.J. (2011) Visual tracking from a rigid-hulled inflatable boat to determine foraging movements of breeding terns. (Journal of Field Ornithology) 82, 68-79.

Poot, M.J.M., M.P. Collier, P.W. van Horssen, R. Lensink & S. Dirksen, 2011. Effect studies Offshore Wind Egmond aan Zee: Cumulative effects on seabirds. A modelling approach to estimate effects on population levels in seabirds. NoordzeeWind Report OWEZ_R_212_20111021_Cumulative_Effects. Bureau Waardenburg report 11-026. Culemborg, Netherlands.

Stienen, E.W.M., Van Waeyenberghe, J. en Kuijken, E., (2007). Trapped within the corridor of the southern North Sea: the potential impact of offshore wind farms on seabirds. In: de Lucas, M., Guyonne, F.E. en Ferrer, M., 2007. Birds and wind farms: risk assessment and mitigation, p. 71 – 80.

Troost, T., (2008). "Estimating the frequency of bird collisions with wind turbines at sea. Guidelines for using the spreadsheet 'Bird collisions Deltares v1-0.xls'." Deltares.

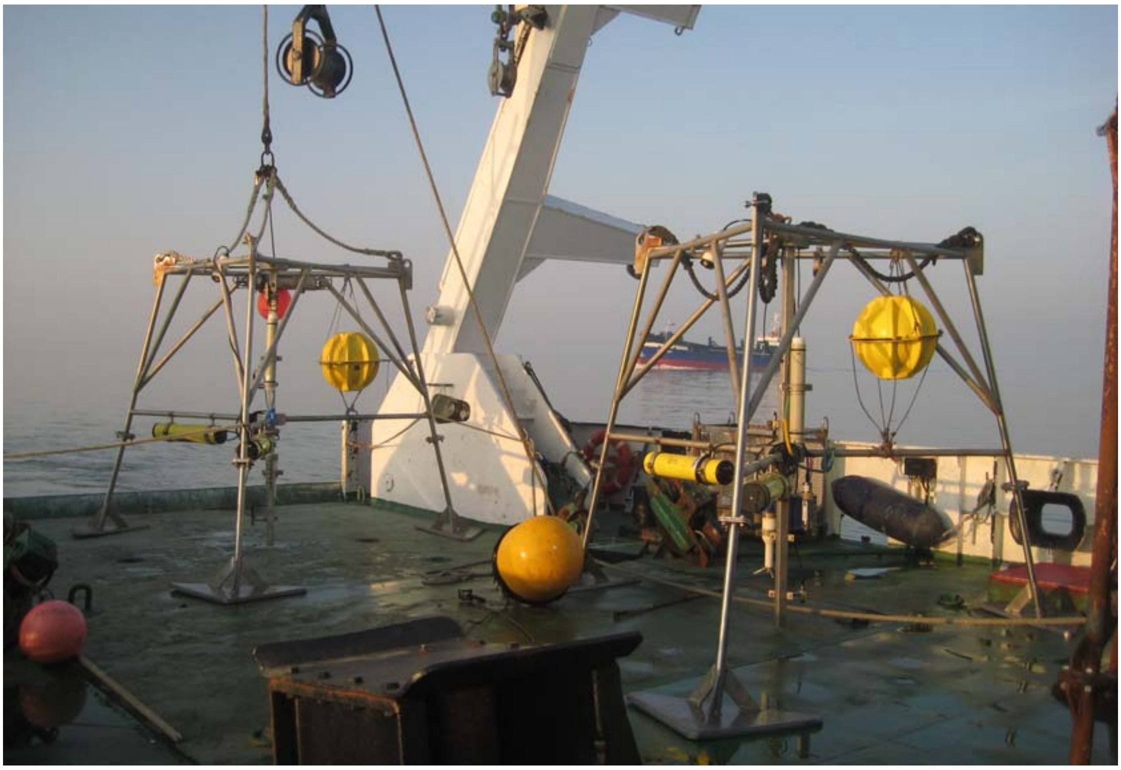
Vanaverbeke J., Braarup Cuykens A., Braeckman U., Courtens W., Cuveliers E., Deneudt K., Goffin A., Hellemans B., Huyse T., Lacroix G., Larmuseau M., Mees J., Provoost P., Rabaut M., Remerie T., Savina M., Soetaert K. , Stienen E.W.M., Verstraete H., Volckaert F. & Vincx M. (2011). *Understanding benthic, pelagic and airborne ecosystem interactions in shallow coastal seas.* "*WestBanks*". Final Report. Brussels: Belgian Science Policy Office 2011 – 82 p. (Research Programme Science for a Sustainable Development)

Chapter 9. The effect of pile driving on harbour porpoises in Belgian waters

J. Haelters[*], W. Van Roy, L. Vigin & S. Degraer

Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models (MUMM), Marine Ecosystem Management Section, 3de en 23ste Linieregimentsplein, 8400 Oostende and Gulledelle 100, 1200 Brussels

**Corresponding author: J.Haelters@mumm.ac.be*



Tripod at the MOW1 reference site after 49 days in the water (right), being replaced by another tripod on 21 March 2011. Both tripods have a PoD attached to their vertical column.

Photo MUMM / RBINS

Abstract

The impact of pile driving on marine mammals is one of the major environmental concerns in offshore wind farm (OWF) construction. We assessed the impact of pile driving for the construction of the C-Power OWF (Thorntonbank, Belgian waters) on the spatial and temporal distribution of harbour porpoises during the season in which they occur in a high density in Belgian waters. We combined data collected by passive acoustic monitoring (PAM) devices, moored inside and outside the project area, with standardised aerial surveys covering the Belgian part of the North Sea. The combination and integration of both techniques proved very useful as PAM results in low spatial but high temporal resolution data, while aerial surveys have a complementary low temporal but high spatial resolution. At the end of March 2011, just before construction activities started, aerial surveys yielded an estimate of 2.5 harbour porpoises/km². Density estimates in mid April 2011, after the start of the construction (piling) activities, had fallen to 1.3 animals/km². Although a decreasing density in Belgian waters towards the end of April should be considered as normal (cf. seasonal migration), changes in the spatial distribution between pre-and post-piling suggested harbour porpoise disturbance. This pattern was further explored with PAM, which indeed showed a clear fine-scale match between acoustic harbour porpoise detections and piling activities. Immediately upon the start of piling activities, harbour porpoise detections at a few km from the piling site fell to virtually zero. After the cessation of piling it took hours to days before new detections were made at this location. Aerial surveys allowed quantifying the distance over which an apparent impact occurred at around 22 km, with a repopulation of part of the area observed after one day with no piling.

Samenvatting

Er heerst bezorgdheid over de mogelijke impact op zeezoogdieren door het heien van palen voor de constructie van offshore windparken. We onderzochten de impact van het heien op bruinvissen *Phocoena phocoena* tijdens de constructie van het C-Power windpark (Thorntonbank), en in het bijzonder tijdens de periode waarin deze dieren in een hoge densiteit voorkomen in Belgische wateren. Voor het bepalen van eventuele effecten op hun temporele en ruimtelijke verspreiding gebruikten we passieve akoestische monitoring (PAM) en surveys vanuit de lucht. De combinatie en integratie van deze technieken bleek zeer nuttig; PAM verschaft immers gegevens in lage ruimtelijke maar hoge temporele resolutie, terwijl surveys vanuit de lucht data verschaffen met een complementaire lage temporele maar hoge ruimtelijke resolutie. Eind maart 2011, net voor de start van de hei-activiteiten, werd de densiteit aan bruinvissen in het survey-gebied geschat op 2,5 dieren/km². Midden april 2011, na de aanvang van de constructie van het windpark op de Thorntonbank, was deze densiteit gedaald tot naar schatting 1,3 dieren/km². Hoewel een afnemende densiteit in de Belgische wateren in april moet worden beschouwd als normaal (cfr. seizoensgebonden migratie), suggereerden veranderingen in de ruimtelijke verspreiding tussen de periodes voor en na het heien een verstoring van bruinvissen. Verder onderzoek van dit patroon door middel van PAM toonde inderdaad een duidelijke verband aan tussen de hei-operaties en de detecties van bruinvissen in het impact-gebied. Onmiddellijk na de start van het heien daalde het aantal detecties van bruinvissen er tot vrijwel nul, en na het stoppen van het heien duurde het uren tot dagen voor nieuwe detecties plaatsvonden. Luchtsurveys lieten toe om een impact aan te tonen tot op ongeveer 22 km afstand. Na een dag zonder hei-activiteiten werd een gedeeltelijke herpopulatie van het impactgebied waargenomen.

9.1. Introduction

During the last decade offshore wind farms (OWFs) have emerged as a new and important industrial activity at sea. In the North Sea, the number of planned, authorised and operational OWFs, and OWFs under construction (OSPAR, 2010) is impressive and the scale of this industrial activity is unprecedented. In the North Sea, OWFs will cover thousands of square km of marine area within the next decade. During construction and operation of OWFs conflicts can arise with marine conservation objectives. Species that may be impacted most are marine mammals and especially for cetaceans concerns exist. As they depend on the emission and reception of sound for orientation and foraging through echolocation and for social interactions, it is obvious that they can be affected by excessive underwater noise originating from construction activities and/or during the operation of wind turbines.

In the North Sea specific concerns exist about the impact of noise on the harbour porpoise *Phocoena phocoena* (OSPAR, 2008; 2009; ASCOBANS, 2009). This is the most common marine mammal in Belgian waters (Degraer *et al.*, 2009) and is considered vulnerable (Reijnders *et al.*, 2009). It is known to be sensitive to disturbance originating from several activities related to OWF construction and operation, such as shipping (Grøn & Buchwald, 1997; Richardson & Würsig, 1997; IMO, 2010), the use of ships' SONAR and seismic survey devices (Bain & Williams, 2006; Eisfeld & Kock, 2006; Nowacek *et al.*, 2007) and especially pile driving (Brandt *et al.*, 2009; 2011; OSPAR, 2009; Tougaard *et al.*, 2009). The harbour porpoise is one of the species most sensitive to excessive noise (Verboom & Kastelein, 2005; Southall *et al.*, 2007).

In 2011 the C-Power windfarm project entered its second phase, with the start of the construction of 48 wind turbines at the Thorntonbank OWF site. This construction phase included the installation of jacket type foundations for wind turbines and a transformer platform. For each foundation, four steel piles were driven into the seafloor. Pile driving is known to generate very high levels of underwater noise (Madsen *et al.*, 2006; Norro *et al.*, 2010; Norro *et al.*, 2012, this volume). Pile driving at other locations in and around the North Sea apparently disturbed harbour porpoises up to distances of tens of km (Brandt *et al.*, 2009; 2011; Tougaard *et al.*, 2009; 2011) and Lucke *et al.* (2007) demonstrated that acute physical effects in the form of temporary hearing threshold shifts (TTS) or even permanent hearing threshold shifts (PTS) possibly occurred in individual animals at close range from a piling site. Of the cetacean species investigated until now, TTS is estimated to be induced at the lowest received energy levels in harbour porpoises (SMRU, 2007).

Our knowledge of the impact of piling is however fairly limited and exists mainly of simulation studies of individual animals in captivity, extrapolations to the marine environment, predictions such as made in environmental impact assessments and data collected during monitoring at a number of construction sites. While some understanding exists about the possible effects on individual harbour porpoises, a quantification and assessment of the effects on a population level is still lacking. The harbour porpoise is particularly difficult to study in the wild because of its elusive nature and technical difficulties related to the environment it lives in. Usually, observations at sea are difficult to make and even more difficult to interpret. While it may be relatively easy to describe, quantify and assess mortality such as originating from incidental catches in fishing gear, it is much more difficult to describe sub-lethal effects at the level of an individual animal and on a population level. In this report we focused on the impact of pile driving on harbour porpoises at the Thorntonbank. We specifically investigated if pile driving at this location did disturb harbour porpoises, over which area and period disturbance occurred and how many animals were affected.

9.2. Material and methods

9.2.1. Piling activities

Piling at the Thorntonbank started on 7 April 2011 and continued until 21 August. For the piling, a IHC Hydrohammer S-800, with a maximum power of 800 kJ was used. The average energy used for the blows was 413 kJ (Norro *et al.*, 2012, this volume). The 196 jacket foundation pinpiles had a diameter of 1.83 m and a length of 21.0 to 49.5 m. The noise levels (peak to peak) measured, normalised to 750 m from the source, ranged from 178 to 195 dB re 1µPa (Norro *et al.*, 2012, this volume). From half an hour before the onset of piling and during each piling activity, a seal scarer (Lofitech), considered as a high-level acoustic harassment device (AHD), was used to scare off any marine mammals in the immediate vicinity of the piling site. The AHD had a source level of 189 dB re 1µPa and emitted noise predominantly in a frequency of around 14 kHz.

9.2.2. Research strategy

The monitoring programme combined passive acoustic monitoring (PAM) with aerial surveys. PAM makes use of the sounds emitted by the animals. It has a very good temporal resolution (continuous data) but a poor spatial coverage (one location per PAM device, but actual survey area unknown). Aerial surveys, used to obtain information about density and distribution over a predefined area, have a high spatial coverage, but a poor temporal resolution (one moment per survey). Whereas PAM data only yield a measure for presence/absence of harbour porpoises, expressed in different temporal resolutions and as such in a semi-quantitative way, aerial surveys provide for the possibility to obtain an absolute estimate of density. The monitoring programme followed a BACI (before-after, control-impact) design, with standardised and dedicated aerial surveys shortly before and after the start of piling and PAM devices moored in the impacted area and in two reference areas prior to and during the piling operations.

9.2.3. Aerial surveys

9.2.3.1. Methodology of the surveys

The methodology used in the aerial surveys was line transect sampling (Buckland *et al.*, 2001). We used a Norman Britten Islander aircraft equipped with two bubble windows. All flight parameters were standardised as much as possible: track lines remained the same during all surveys, with parallel tracks 5 km apart and perpendicular to the coastline, flight altitude (600 ft) and speed (100 kts) were kept constant and surveys were only performed during good observation conditions. For practical purposes, tracks started at 5 km from the coast and included a small part of French waters (Haelters, 2009). GPS positions were automatically recorded.

During the flight, two observers continuously observed the water surface for the presence of marine mammals. For every sighting the species was recorded, together with the number of animals, their activity, the direction of movement and the presence of calves. The angle of the observed animals perpendicular to the aircraft was recorded with a Suunto Clinometer PM-5/360PC. Also environmental conditions, i.e. sea state, cloud cover, glare and haziness, were recorded.

Given the standardisation of the observation and data recording procedure, the data of all flights, including those collected during previous years, were pooled to obtain a harbour porpoise detection probability (vs. distance from the aircraft) function using Distance software (Version 6.0. Release 2; Thomas *et al.*, 2009). The resulting hazard rate function was based on 1018 sightings of in total 1240 harbour porpoises. As not all animals were detectable (e.g. under water) or were actually detected at the surface by the observer, we used 0.45 as a correction factor for g(0) (after Hiby, 2008) for all surveys (as all surveys were conducted during good observation conditions) and did not apply a confidence value (CV) to it. Given our relatively small dataset, we assumed that the detection probability was independent of time of the day, cloud cover, season (although Scheidat *et al.* (2011) demonstrated that these factors could have an influence on detectability) and group size, density of animals and observer. Given the assumptions, the results presented here and especially CVs need to be treated with caution and should not be over-interpreted. The effective (half) strip width, uncorrected for g(0), was 144.1 m (136.9-151.7 m).

For impact assessment we used the results of four aerial surveys: 24-25 March and 29 March (pre-piling phase) and 14 and 16 April (piling phase). The aerial monitoring on 14 April started approximately 23 hours after the cessation of the piling activities and no piling took place during the flight. The survey on 16 April started 29 hours after the cessation of piling and half an hour after piling had restarted. Piling activities continued throughout the flight. Each survey consisted of 13 tracks. Their total length ranged from 367.7 nautical miles (nm) to 379.9 nm and their duration was 3h39’ to 3h44’.

9.2.3.2. From average density to distribution

From the aerial surveys an extrapolation method was used to obtain a density surface estimate of harbour porpoises in Belgian waters at the moment of the surveys. As survey flights tentatively follow the same tracks during every survey, the densities obtained from one survey can be compared to the densities of other surveys performed at another moment: variables considered as possibly affecting the distribution of harbour porpoises and that remain constant between the surveys (such as water depth, shipping traffic, distance from the coast) do not have an effect on changes observed. Changes in between surveys can hence be explained by variables that do not remain constant and that also affect harbour porpoise distribution, such as period of the year and irregular human activities (e.g. pile driving for offshore wind farm construction). The surveys considered here were performed within three weeks, during which changes in the distribution of harbour porpoises due to, for instance, spatial changes in food availability or seasonal movement patterns were presumed limited.

The density surface modelling (DSM) analysis was performed in two stages. In a first stage the survey track and the observations were analysed using Distance software (Thomas *et al.*, 2009), yielding a detection probability and an effective (half) strip width (ESW). Using the g(0) obtained from literature, a density estimate for the whole survey area could be obtained. In a second step, we subdivided every transect into segments of five km using the actual flight track or alternatively using the start- and endpoints of each transect to reconstruct the track. Segments shorter than five km were not used. To the mid-points of each segment a density d_i was attributed using the length of the segment (l_i), the width of the segment (calculated from the detection probability function) and the sum of the harbour porpoises observed in each group (n_i), always on or very near to the segment and corrected with g(0):

$$d_i = \frac{\sum n_i \times g(0)}{(ESW \times 2) \times l_i}$$

The result of this generalisation was a number of geo-localised densities, more or less equally spread throughout the survey area. An inverse distance weighting (IDW) interpolation method to predict values in a regular grid with cells of 800 m by 800 m was applied. IDW estimated values for each grid cell by averaging the values of four sample data points in the vicinity of each cell (using a power of 1). As such, densities were attributed to areas not visited and original densities were smoothed. Because of the process used, the density in areas situated along the edge of the surveyed area and in the 5 km zone near the coastline (not surveyed), has a lower confidence.

The grid in which the densities are presented has a relatively high resolution in comparison to the swimming speed of harbour porpoises, which according to Otani *et al.* (2001) is on average 0.9 m/s and is mostly lower than 1.5 m/s. Maximum recorded speeds vary from 4.3 m/s (Otani *et al.*, 2001) to 6.2 m/s (Leatherwood *et al.*, 1988). However, given the relatively small area studied, a sufficiently high number of grid cells is required for assessing the possible effects of a very localised activity in Belgian waters. Such a resolution is further acceptable, given the short time interval between the surveys and the highly standardised nature of the surveys. However, the results of the high-resolution assessment should not be misused to spatially over-interpret data and conclusions should only be drawn on a courser scale.

9.2.3.3. Impact distance modelling

The DSM maps were finally used to model the pressure response of harbour porpoises to piling activities, as well as a recovery response after cessation of the pressure. Within this analysis the DSM maps of 24-25 March and 29 March were considered as two independent reference situations, while the 16 April and 14 April DSM maps represent the situation during piling and one day after the cessation of the piling activities, respectively. Two response replicates could hence be obtained for the situation during piling (i.e. 16/04 vs. 24-25/03 and 16/04 vs. 29/03) and the situation one day after the cessation of the piling activities (i.e. 14/04 vs. 24-25/03 and 14/04 vs. 29/03). All four response replicates were constructed by calculating the relative change within each grid cell (e.g. -100% = all porpoises disappeared; +100% = the porpoise number doubled), after which the average relative change was calculated for each one-kilometre ring around the piling location (0 to 55 km) using ArcGIS Spatial Analyst. Only grid cells within the Belgian Exclusive Economic Zone (i.e. 12+ nm offshore) were considered, as harbour porpoises are presumed to occur in lower densities in the territorial waters in the eastern part of Belgian waters (Haelters *et al.*, 2011; see 3.1.2), hence maximising the representativeness of the analysis.

The theoretic response model of the impact on harbour porpoise spatial distribution consists of two functions: (1) an exponential escape function, describing the porpoise distribution as they move out of the impact area and (2) a log-normal distribution function, describing the porpoises' (re)distribution with an accumulation at the edge of, and outside the impact area shortly after the start of the impact. As we are particularly interested in the quantification of the impact radius in this study, only the exponential escape-function is considered here. To avoid a bias by the harbour porpoise distribution outside the impact area, the exponential function was calculated on the basis of only those data points between 0 km and the distance corresponding to the maximum average relative increase of porpoises.

The null hypothesis of this analysis (i.e. harbour porpoise distribution is not impacted by the piling activities) is that the average relative change in porpoises shows no relation with distance from the piling location. Its function can then be described as y = constant and corresponds to the relative decrease of the total harbour porpoise population size in the BPNS as a result of the seasonal movement out of the BPNS (Haelters *et al.*, 2011). This constant value was calculated from the aerial survey population size estimates (see 2.3.1).

9.2.4. Passive acoustic monitoring

As PAM devices we used Porpoise Detectors (C-PoDs). These autonomous devices can be moored for months and record *ad hoc* sound event characteristics that can be analysed to identify the sounds produced by different small cetaceans. We used the click train detection algorithm of the dedicated software CPOD.exe Version 2.025 for the analysis. We set the species filter to harbour porpoises and used only high and moderate train quality data (see www.chelonia.co.uk for details). All data were visually inspected to eliminate false detections such as originating from depth sounders or broadband signals. We used two types of quantitative measure to present harbour porpoise presence/absence around the device: detection positive 10 minutes per day (dp10m/d), which is the number of 10 minute blocks per day during which the presence of harbour porpoises was detected, and detection positive minutes per hour (dpm/h). By using dp10m/d the possible differences in sensitivity between C-PoDs is minimised (www.chelonia.co.uk), while for a higher temporal resolution analysis dpm/h was useful.

Two weeks before piling started and up to the end of May 2011, a C-PoD was moored at three locations (Figure 1): (1) an impact site (IS) at the edge of the C-Power construction area on the Thorntonbank (51°35.42'N; 003°00.305'E), (2) a reference site (REF1) at the MOW1 location, 4.5 km off Blankenberge (51°21.5'N; 003°07'E) and (3) a second reference site (REF2) at the Oostdyck W cardinal buoy, 22 km off the western Belgian coast (51°17.15'N; 002°26.32'E). The PoDs at IS and REF1 were moored on a tripod placed on the seafloor, with a PoD vertically mounted on the central column, while the REF2 PoD was hung from the side of a cardinal buoy at a depth of 1.5 to 2 m (Figure 2). As the water in which the PoDs were moored, was very shallow and given the PoD detection range for harbour porpoises of more than 200 m, we assumed that data collected near the seafloor and near the surface can be compared.

Piling started on 7 April at around 6:30h UTC at the Thorntonbank at approximately 5.4 km, 22.7 km and 47.3 km from the PoDs at respectively IS, REF1 and REF2. The piling on 10 April took place at approximately 7.3 km, 22.3 km and 45.2 km from the PoD moorings at respectively IS, REF1 and REF2. Subsequent piling activities between 11 and 19 April took place at a distance of 2.6 to 5.2 km from the IS PoD, 23.0 to 24.2 km from the REF1 PoD and 47.2 to 50.6 km from the REF2 PoD.

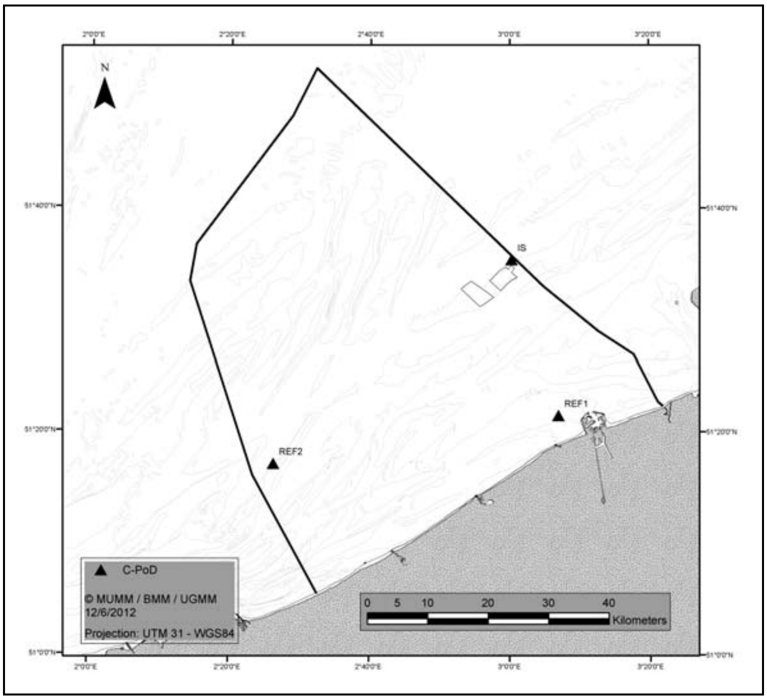


Figure 1. Location of the PoD moorings during 2011.

Figure 2. Oostdyck W buoy upon recovery, with a PoD (REF2) hanging from the side (right).

We used data obtained from the three PoDs between 25 March and 25 May to investigate trends in detection rate (dp10m/d). We also compared the average detection rates (dpm/h) one week prior to the piling with those one week into the piling phase. Therefore, we carried out two-sample parametric t-tests (using STATISTICA, v.10) comparing the data collected between 31 March 5:00h UTC to 7 April 6:00h UTC (1 week pre-piling) and 7 April 6:00h UTC to 14 April 7:00h UTC (1 week piling – post-piling). These periods were covered by PoD moorings at all stations and they presented a compromise between a short period with too few data (given the sometimes irregular detection patterns, especially in a high temporal resolution analysis of detections in a low density area) and a long period possibly including migration effects.

9.3. Results

9.3.1. Results of the aerial surveys

9.3.1.1. Density of harbour porpoises in Belgian waters

During the surveys of 24-25 March, 29 March, 14 April and 16 April respectively, 184, 198, 110 and 85 sightings were made of groups of harbour porpoises, totalling 214, 230, 123 and 113 animals. The estimates of the average density of harbour porpoises over the surveyed area were 2.4 and 2.6 animals/km² on respectively 24-25 March and 29 March (pre-piling phase) and 1.3 animals/km² for the surveys performed on 14 April and 16 April (piling phase; Figure 3).

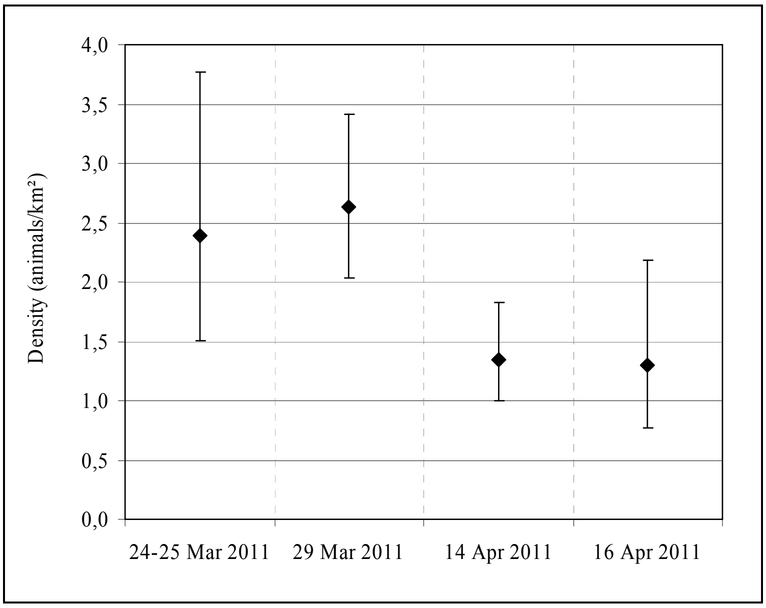


Figure 3. Average harbour porpoise density estimates on 24-25 March, 29 March, 14 April and 16 April 2011 (± 95% confidence interval).

9.3.1.2. Distribution of harbour porpoises before and during piling operations

Density surface mapping indicate that before the piling harbour porpoises were distributed unevenly throughout Belgian waters, with the highest densities in the westerly part and lower densities within territorial waters (Figure 4). A concentration area existed in the north-western part, around the Fairy and Westhinder sandbanks. After the start of the piling activities at the Thorntonbank, densities remained the highest in the westerly part. On 14 April harbour porpoises occurred in a relatively low density within a radius of 10 to 20 km from the piling site, but the density within 10 km from the piling site was very low with virtually no observations. On 16 April, during piling, densities were very low within a radius of slightly over 20 km around the piling site: during the survey virtually no porpoises were observed in this area and in the area to the north-west of the piling location.

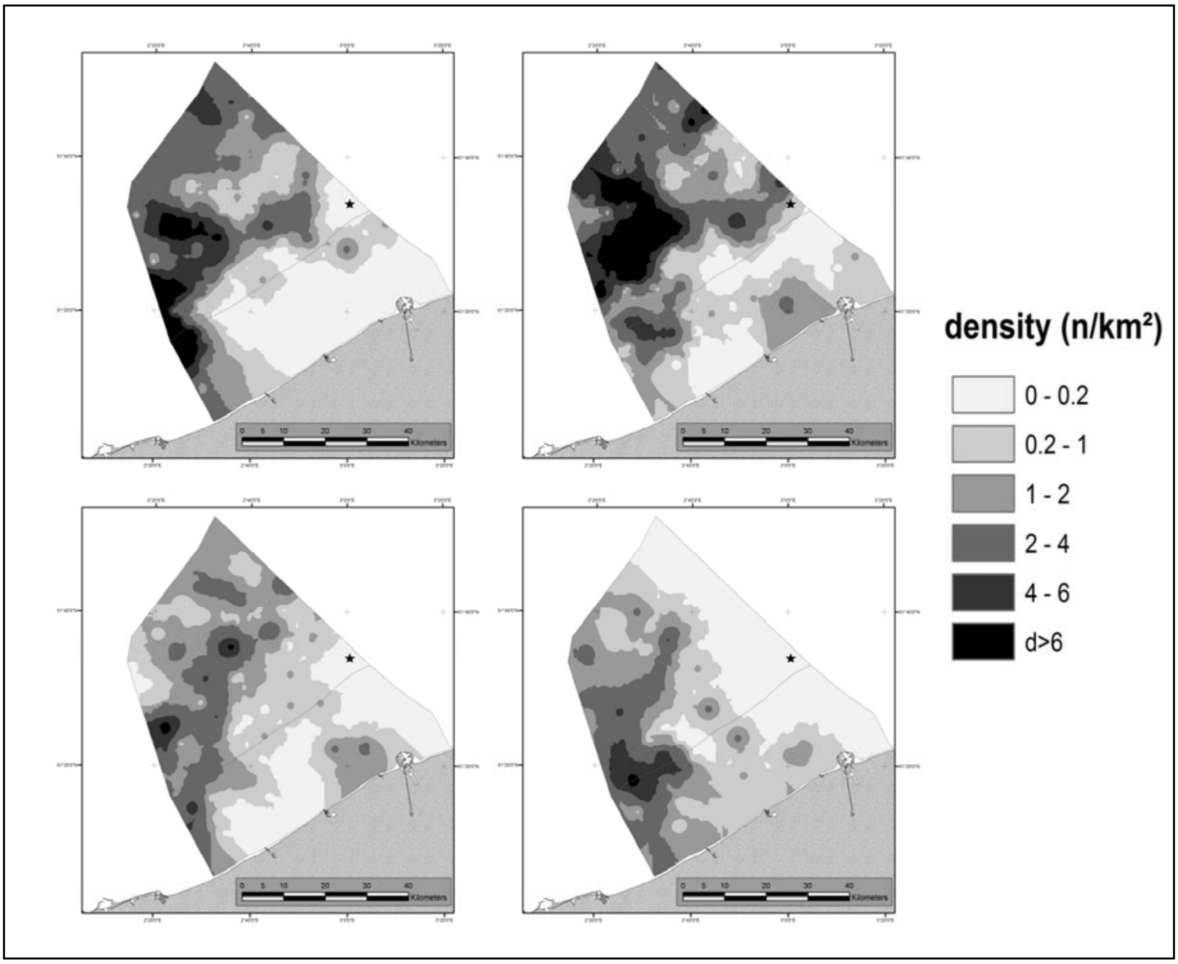


Figure 4. Density of harbour porpoises in the survey area as modelled on the basis of aerial survey data on 24-25 March (top left), 29 March (top right), 14 April (bottom left) and 16 April 2011 (bottom right); the star indicates the piling location.

9.3.1.3. Impact modelling

During piling harbour porpoises were absent within a radius of 17 km around the piling location (Figure 5). The average relative change in density in the pressure response modelling indicate a sharp increase in relative average density from 17 km distance onwards to reach a maximum of +82% at 24 km. One day after the cessation of piling activities, harbour porpoises were absent within a radius of 4 km. The pressure response modelling shows a sharp increase in the average relative change in density from a distance of 5 km to attain a maximum of +90% at a distance of 20 km. At distances further than 24 km (during piling) and 20 km (one day after piling) the relative change in density gradually decreased to an average value of -23% and -5%, respectively.

The logarithmically fitted escape functions cross the function of the average decrease of the total BPNS population size between the reference period and the impact period (y = -52%) at about 22 km (during piling) and at about 13 km (one day after the cessation of piling).

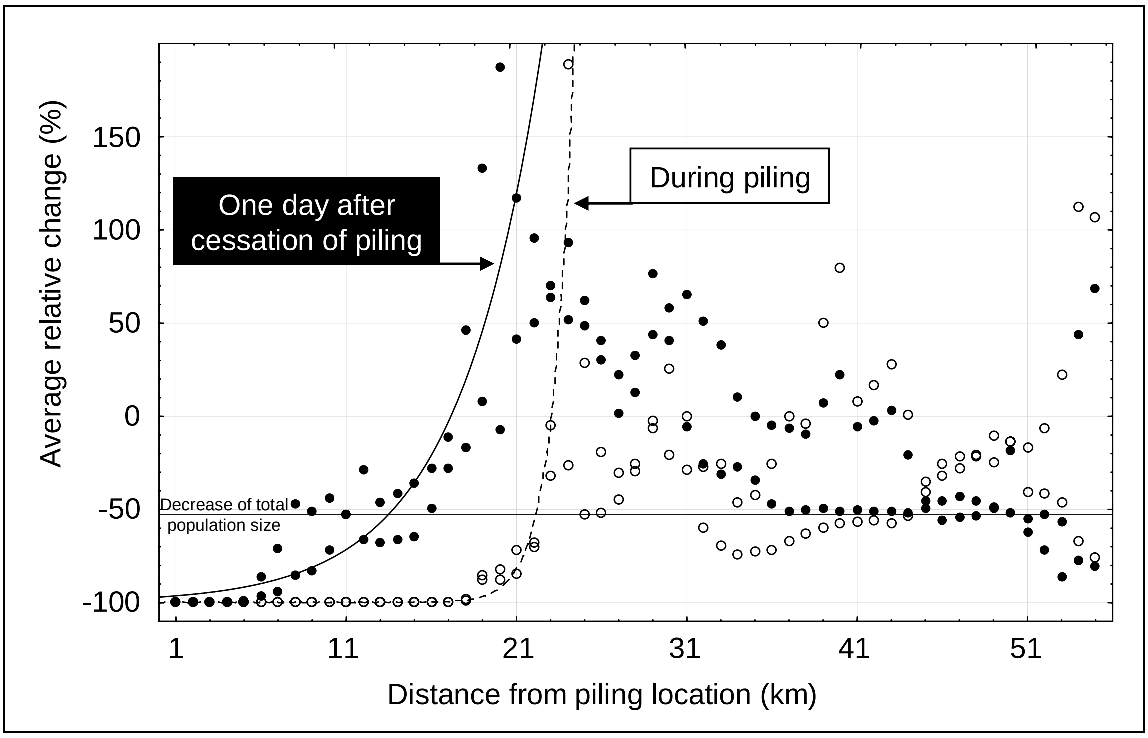


Figure 5. Average relative change in harbour porpoise density as a function of distance from the piling location. ○: response values during piling; ●: response values one day after the cessation of piling. Exponential fits describe the escape function during piling (dashed line) and one day after the cessation of piling (full line).

9.3.2. Results of the passive acoustic monitoring

The harbour porpoise detection rates (dp10m/d) at all mooring stations between 25 March and 25 May showed important day to day fluctuations (Figure 6). However, at all stations a negative trend was detected. The detection rates around late March/early April at the sites relatively far offshore (IS and REF2) were higher than at the inshore site (REF2). While the decline at REF1 and REF2 between early April and mid May was in general fairly gradual, the one at IS was very abrupt and came earlier, with a sudden drop to very low levels around 7 April and virtually no recovery in the subsequent weeks.

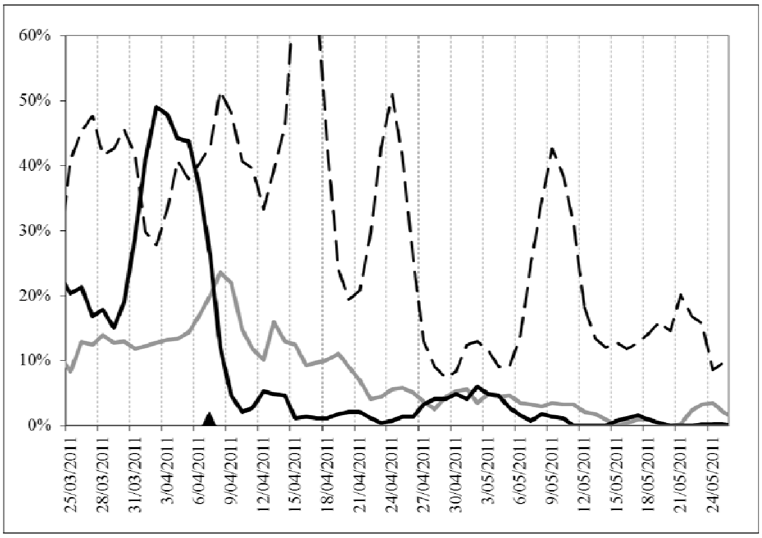


Figure 6. Harbour porpoise detection rate (dp10m/d, expressed as % of the positive 10 minute blocks per day; floating average 3 days) in the PoDs moored at IS (black solid line), REF1 (grey line) and REF2 (dashed line); a black triangle indicates the start of piling activities at the Thorntonbank.

A more detailed view (dpm/h) into the period with the strong and abrupt decline in detection rate at IS indicated a downward trend between 31 March and the onset of piling activities on 7 April (Figure 7), a period followed by a rapid decline to a detection rate of virtually zero from the first piling event onwards. There was only a slight recovery at 12 to 14 hours after the end of the first piling event (with 3 to 14 dpm/h), followed by a decline to zero with the start of the second piling event. The detection rate remained at a very low level afterwards, with maximum 5 dpm/h and on average 1 dpm every 4 hours.

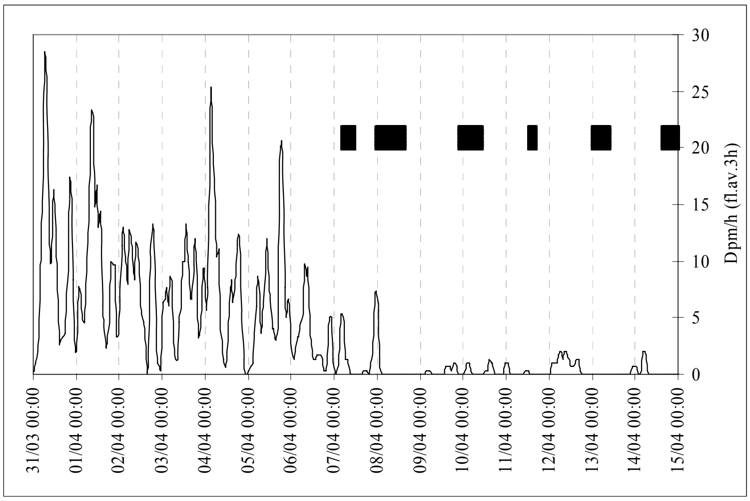


Figure 7. Detection rate (dpm/h; floating average 3 hours; black line) registered by the PoD moored near the Thorntonbank project site (IS) between 31 March and 14 April 2012 (incl.); the black boxes indicate the piling activities.

The average detection rate (dpm/h) at IS during the first week after the onset of piling activities was significantly lower (t-test: $p<0.0001$) than during the week preceding the piling activities. At REF1 there was no significant difference in the average detection rate, while at REF2 a significantly higher ($p=0.0001$) detection rate was observed in the week after the piling started (Figure 8).

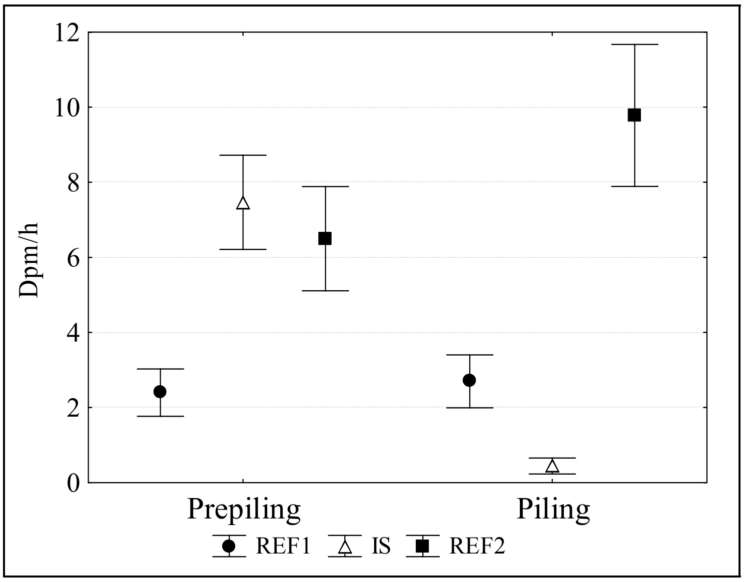


Figure 8. Mean harbour porpoise detection rate (dpm/h; ± CI) one week prior to the onset of piling (‘Prepiling’) and one week starting from the onset of piling (‘Piling’) at IS, REF1 and REF2.

9.4. Discussion

9.4.1. General spatio-temporal patterns

With an average density of around 2.5 animals/km², 8500 harbour porpoises were present in an area equivalent to Belgian waters in the second half of March 2011. This is the highest density ever recorded in these waters and is in line with densities recorded during the same period in Dutch and German coastal waters (ASCOBANS, 2011; Geelhoed *et al.*, 2011). The estimated average density of 1.3 animals/km² during mid April, or in total 4500 harbour porpoises in an area equivalent to Belgian waters, can still be considered as high compared to the surveys performed in the past (Haelters *et al.*, 2010; 2011a; 2011b). The decline in the PAM detection rates (dp10m/d) at all PoD mooring stations between 25 March and 25 May is consistent with high densities of harbour porpoises in Belgian waters during March-April, and low densities during May (Haelters *et al.*, 2011a).

The density of harbour porpoises at the end of March was higher in the western part of Belgian waters than in the eastern part. This could be due to pre-piling activities around the Thorntonbank, but also to better feeding conditions in that area during that period (prey availability, turbidity,…) or less disturbance in the form of shipping. However, also a gradual and general migration of harbour porpoises from easterly to more westerly coastal waters of the southern North Sea during late winter – early spring (Haelters & Camphuysen, 2009) may be at the origin of this phenomenon. During an aerial survey on 16-22 February 2011 higher densities occurred in the eastern part of Belgian waters than in March (MUMM, unpublished data). Such a gradual migration, however, cannot explain on its own the absence of harbour porpoises in a large area in the eastern part of Belgian waters on 16 April 2011 and the virtual absence of acoustic detections from 7 April onwards at the Thorntonbank PoD, phenomena all indicating disturbance in this area.

9.4.2. Effects of pile driving

The downward trend in the dpm/h rate at the impact site PoD between 31 March and the onset of piling activities on 7 April may have been due to an increasing disturbance caused by vessel and platform activity at and around the piling site. There was a clear synchronisation between the harbour porpoise detection rate at the impact site and the onset of piling activities on 7 March, which took place at a distance of 5.4 km from the PoD mooring. The detection rate dropped to zero immediately after the start of piling, and did not recover during the whole period of the spring piling activities. This indicates a rapid decline in harbour porpoise numbers around the piling site and virtually no return in between piling operations. The detection rate pattern recorded by this PoD was very different from those of the other PoDs, which were irregular, but overall showed a more gradual decline towards late April, which can be linked to general seasonal movements of harbour porpoises.

The PAM data were consistent with the results of the aerial surveys. One week into the piling phase of the offshore windfarm construction at the Thorntonbank, the average density of harbour porpoises in the area surveyed had been halved compared to one week prior to the first piling event, probably due to a combination of disturbance over a large part of this area and natural general and seasonal movements. During the aerial survey of 16 April, which was made during pile driving, no harbour porpoises were observed within a radius of slightly over 20 km from the piling site. During the aerial survey of 14 April, that started 23 hours after the cessation of piling, a small number of harbour porpoises were observed within a radius of 10 km around the piling site. This can be explained by the immigration of animals into the now undisturbed area from adjacent areas after piling. These animals were probably exposed to relatively high noise levels at the start of the next piling activity, and driven away again. As piling continued, it is possible the area of exclusion of harbour porpoises became gradually larger; Diederichs *et al.* (2011) linked the area of disturbance with the duration of piling.

The impact range modelled through the comparison of DSM maps before and during piling indicates a range of disturbance of around 20 km. This corresponds well with the impact range of 19 km as predicted by Norro *et al.* (2012, this volume). This theoretical range was based on the noise measurements made around the Thorntonbank during the piling and the distance at which the piling noise would reach a discomfort level (peak to peak) for harbour porpoises, described at Tougaard *et al.* (2011) as 140 dB re 1µPa. Major disturbance can be described as one likely leading to a strong reaction in individual animals, such as through a noise level (peak to peak) of 155 dB re 1 µPa and higher (Bailey *et al.*, 2010); this level would have been reached within a perimeter of 8 km around the pile driving site (Norro *et al.*, 2012, this volume).

After cessation of piling the harbour porpoises do repopulate the impact area. According to the modelling, the apparent impact zone decreased from one with a radius of 22 km to one with a radius of 13 km after one day without piling, or a decrease of 9 km per day. This modelled speed of repopulation of harbour porpoises after the cessation of piling in the direction of the piling location is much slower than their average speed of directional swimming (0.9 m/s; Otani *et al.*, 2001). This would however be consistent with a fairly random movement (dispersal) in the area during that period, a slow displacement because of local foraging activities, general seasonal migration patterns, and possibly also tidal water mass movements.

The consequences of disturbance from an area for harbour porpoises are difficult to assess. Animals that are disturbed may not be able to forage as efficiently and leave for possibly a less suitable area. Which consequences this can have for a small endothermic animal with a limited capacity to store energy is not known. As a harbour porpoise needs to feed every day and its blubber layer functions as an energy reserve as well as thermal insulation, it can be expected that regular disturbance may at least have an influence on its condition, and related to that, its health.

9.4.3. Effects linked to pile driving or acoustic harassment?

It is unlikely that the very low detection rate at the impact site PoD after the onset of piling or the absence of harbour porpoises around a wide radius around the piling site, as observed during the aerial surveys, is due to the deployment of the AHD. While the level of noise produced by this device is much lower than that produced by piling, it is in a frequency range at which harbour porpoises are more sensitive. According to Kastelein *et al.* (2002), the hearing threshold of harbour porpoises at 14 kHz lies around 50 dB lower than at 500 Hz (typical piling noise frequency).

We can use a simple noise propagation model to calculate the theoretical noise levels (L_R) in a radial distance R from the AHD with a source level (L_S) of 189 dB re 1µPa. We assume a transmission loss of -20log$R$ (after Thiele, 2002) and an absorption coefficient α of 1.693 dB/km (sound of 14 kHz in seawater of 10°C; after Ainslie & McColm, 1998). The model used is:

$$L_R = L_S - 20\log(R) - \alpha R$$

In theory the AHD would produce a noise level of 140 dB re 1µPa at around 300 m from the device, and 120 dB re 1µPa at a distance of 2 km. However, Shapiro et al. (2009) demonstrated that in the use of AHDs, unpredictable and rapid variations in the (received) sound level occur which conflict with our classical concept of the occurrence of concentric zones of lower noise levels with increasing distance to the noise source. This led to an uncertainty in the prediction of the sound level at distance. Olesiuk *et al.* (2002) and Hoeschle *et al.* (2011) demonstrated a disturbing effect of the Lofitech seal scarer on harbour porpoises up to a few km from the deployment site.

While the noise produced by the AHD could have scared porpoises away from the piling site up to a few km, and as such may have been an effective way to prevent that they were exposed to potentially (physically) harmful noise, it was much lower than the pile driving noise itself. Therefore, while the use of the seal scarer inevitably complicates the assessment of the effects of piling, it does not have an influence on the main conclusions, being that the piling at the C-Power site apparently scared harbour porpoises away from the piling site up to a distance of around 20 km. Similar conclusions were made by Thomsen *et al.* (2006), Brandt *et al.* (2011) and Lucke *et al.* (2011).

9.5. Conclusions

The main conclusions on the possible effects of pile driving during 2011 can be summarised as follows:

1. Harbour porpoises were, with an average density of 2.5 animals per km² in the survey area, very common in Belgian waters at the end of March 2011. Their density was higher in the western part of Belgian waters than in the eastern part, where the wind farms are located – this does however not mean that this is the case during other periods of the year.
2. By mid April the average density in the survey area had halved, probably due to a combination of disturbance by pile driving over a large part of this area and the onset of a general seasonal migration out of Belgian waters. However, an average density of 1.3 animals per km² can still be considered as high.
3. The apparent distance from the pile driving location over which harbour porpoises were disturbed during pile driving on 16 April extended to 22 km, as estimated by the escape function – it is most likely that similar effects occurred in Dutch waters. On 14 April, after one day without pile driving, a small number of harbour porpoises were observed within a radius of 10 km around the pile-driving site, but acoustic detections at a distance of a few km from the piling site remained very low, indicating a very low density in that area.
4. Given the distances where hardly any harbour porpoises were observed and the source levels of the noise produced by pile driving and the AHD deployed prior to and during pile driving, we can safely assume that the piling and not the AHD was at the origin of the disappearance of harbour porpoises in an area of hundreds of km² around the piling site.
5. Assuming a radius of disturbance of 22 km (escape function), and an average density of 1.3 to 2.5 harbour porpoises per km², as taken from the aerial surveys, piling would have affected between 2000 and 3800 harbour porpoises. However, as this animal shows seasonal migrations, a turn-over of harbour porpoises travelling through the southern North Sea can be assumed. As piling lasted for many months, the total number of harbour porpoises disturbed could hence be many times higher.
6. The physical consequences for the animals of this disturbance remain unknown.

9.6. Acknowledgements

We thank the pilots and observers for their assistance during the aerial surveys. The mooring of the PoDs would have been impossible without the cooperation of the crew of RV BELGICA, colleagues of MUMM Meetdienst Oostende and MUMM Brussels, and the much appreciated support of the Flemish Community (Hydrographic Service). Our colleagues Serge Scory and José Ozer assisted in the analysis of the aerial surveys. We acknowledge the financial support of, and cooperation with C-Power, Belwind and Northwind.

9.7. References

Ainslie, M.A. & McColm, J.G., 1998. A simplified formula for viscous and chemical absorption in sea water. Journal of the Acoustical Society of America 103(3): 1671-1672.

ASCOBANS, 2009. Resolution on the adverse effects of underwater noise on marine mammals during offshore construction activities for renewable energy production. Resolution no. 2 adopted at the 6th Meeting of the Parties of the Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas, UN Campus, Bonn, Germany, 16-18 September 2009.

ASCOBANS, 2011. Summary Record of the 18th Meeting of the Advisory Committee. UN Campus, Bonn, Germany, 4-6 May 2011.

Bailey, H., Senior, B., Simmons, D., Rusin, J., Picken, G. & Thompson, P.M., 2010. Assessing underwater noise levels during pile-driving at an offshore windfarm and its potential effects on marine mammals. Marine Pollution Bulletin 60: 888-897.

Bain, D.E. & Williams, R., 2006. Long-range effects of airgun noise on marine mammals: responses as a function of received sound level and distance. IWC Scientific Committee, IWC Annual Meeting, 1-13 June, St. Kitts, Doc. SC/58/E35.

Brandt, M.J., Diederichs, A. & Nehls, G., 2009. Harbour porpoise responses to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. Final report to DONG Energy. Husum, Germany, BioConsult SH.

Brandt, M.J., Diederichs, A., Betke, K. & Nehls, G., 2011. Responses of harbour porpoises to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. Marine Ecology Progress Series 421: 205–216. doi: 10.3354/meps08888

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L. & Thomas, L., 2001. Introduction to Distance Sampling: estimating abundance of biological populations. Oxford University Press.

Degraer, S., Braeckman, U., Haelters, J., Hostens, K., Jacques, T., Kerckhof, F., Merckx, B., Rabaut, M., Stienen, E., Van Hoey, G., Van Lancker, V. & Vincx, M., 2009. Studie betreffende het opstellen van een lijst van potentiële Habitatrichtlijngebieden in het Belgische deel van de Noordzee. Eindrapport in opdracht van de Federale Overheidsdienst Volksgezondheid, Veiligheid van de Voedselketen en Leefmilieu, Directoraat-generaal Leefmilieu. Brussel, België. 93 pp.

Diederichs, A., Brandt, M.J. & Nehls, G., 2011. Duration of ramming procedure has a clear effect on displacement of harbour porpoises. 25th Conference of the European Cetacean Society, Book of Abstracts, 21–23 March 2011, Cádiz, Spain.

Eisfeld, S.M. & Kock, K.-H., 2006. Expert paper to the ASCOBANS conservation plan for harbour porpoises (*Phocoena phocoena L.*) in the North Sea. 93 p.

Geelhoed, S., Scheidat, M., Aarts, G., van Bemmelen, R., Janinhoff, N., Verdaat, H. & Witte, R., 2011. Shortlist masterplan wind aerial surveys of harbour porpoises on the Dutch Continental Shelf. IMARES, Wageningen Report number C103/11.

Grøn, P. & Buchwald, E., (1997). Effekter af hurtigfærgesejlads på havfugle, sæler og marsvin. Slutrapport. Skov- og Naturstyrelsen. Carl Bro & Bio/consult, Copenhagen (Effects of highspeed ferries on seabirds, seals and harbour porpoises).

Haelters, J., (2009). Monitoring of marine mammals in the framework of the construction and exploitation of offshore windfarms in Belgian marine waters. *In:* Degraer, S. & Brabant, R. (Eds.) (2009). Offshore windfarms in the Belgian part of the North Sea: State of the art after two years of environmental monitoring. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. Chapter 10: 237-266.

Haelters, J. & Camphuysen, K., (2009). The harbour porpoise in the southern North Sea: abundance, threats and research- & management proposals. Royal Belgian Institute of Natural Sciences (RBINS/MUMM) and the Royal Netherlands Institute for Sea Research (NIOZ); report commissioned by the International Fund for Animal Welfare (IFAW), 56 p.

Haelters, J., Kerckhof, F., Jacques, T.G. & Degraer, S., (2010). Spatio-temporal patterns of the harbour porpoise *Phocoena phocoena* in the Belgian part of the North Sea. *In:* Degraer, S., Brabant, R. & Rumes, B. (2010) Offshore wind farms in the Belgian part of the North Sea: early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. Chapter 10: 153-163.

Haelters, J., Kerckhof, F., Jacques, T.G. & Degraer, S., (2011a). The harbour porpoise *Phocoena phocoena* in the Belgian part of the North Sea: trends in abundance and distribution. Belgian Journal of Zoology, 141: 75-84.

Haelters, J., Kerckhof, F., Vigin, L. & Degraer, S., (2011b). Offshore windfarm impact assessment: monitoring of marine mammals during 2010. *In:* Degraer, S., Brabant, R. & Rumes, B. (Eds.) (2011). Offshore wind farms in the Belgian part of the North Sea: selected findings from the baseline and targeted monitoring. Royal Belgian Institute of Natural Sciences, Management

Unit of the North Sea Mathematical Models, Marine ecosystem management unit. Chapter 10: 131-146.

Hiby, L., (2008). Effective strip half-width estimates from aerial survey data. In: SCANS II, 2008. Small Cetaceans in the European Atlantic and North Sea (SCANS II). Final Report to the European Commission, Appendix D3.1.

Hoeschle, C., Brandt, M.J., Diederichs, A., Betke, K. & Nehls, G., (2011). Testing the effects of an acoustic harassment device on the behaviour of harbour porpoises (*Phocoena phocoena*). 25th Conference of the European Cetacean Society, 21-23 March 2011, Cádiz, Spain, book of abstracts.

IMO, (2010). Noise from commercial shipping and its adverse impact on marine life. International Maritime Organisation (IMO), Marine Environmental Protection Committee (MEPC), document 61/19.

Kastelein, R.A., Bunskoek, P., Hagedoorn, M., Au, W.W.L. & de Haan, D., (2002). Audiogram of a harbor porpoise (*Phocoena phocoena*) measured with narrow-band frequency-modulated signals. Journal of the Acoustical Society of America 112(1): 334-344.

Leatherwood, S., Reeves, R.R., Perrin, W.F. & Evans, W.E., (1988). Whales, dolphins, and porpoises of the Eastern North Pacific and adjacent arctic waters. A guide to their identification. Dover Publications, New York.

Lucke, K., Lepper, P.A., Blanchet, M. & Siebert, M., (2007). Testing the auditory tolerance of harbour porpoise hearing for impulsive sounds. Poster presented at the International Conference on the Effects of Noise on Aquatic Life, Nyborg, Denmark, August 13-17, 2007.

Lucke, K., Dähne, M., Peschko, V., Müller, S., Adler, S., Gilles, A., Krügel, K., Brandecker, A., Sundermeyer, J., Verfuss, U., Benke, H. & Siebert, U., (2011). StUKplus: Investigations in addition to the effects monitoring – marine mammals. Meeresumwelt-Symposium, Hamburg, Proceedings, 61-62.

Madsen, P.T., Wahlberg, M., Tougaard, J., Lucke, K. & Tyack, P.L., (2006). Wind turbine underwater noise and marine mammals: implications of current knowledge and data needs. Marine Ecology Progress Series 309: 279-295.

Norro, A., Haelters, J., Rumes, B. & Degraer, S., (2010). Underwater noise produced by the piling activities during the construction of the Belwind offshore wind farm (Bligh Bank, Belgian marine waters). *In:* Degraer, S., Brabant, R. & Rumes, B. (Eds) (2010). Offshore wind farms in the Belgian part of the North Sea. Early environmental impact assessment and spatio-temporal variability. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. Chapter 4: 37-51.

Norro, A., Rumes, B. & Degraer, S., (2012). Differentiating between underwater construction noise of monopile and jacket foundation wind turbines: A case study from the Belgian part of the North Sea. *In:* Degraer, S., Brabant, R. & Rumes, B., (Eds.) (2012). Offshore wind farms in the Belgian part of the North Sea: Heading for an understanding of environmental impacts. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. Chapter 10: 145-155.

Nowacek, D.P., Thorne, L.H., Johnston, D.W. & Tyack, P.L., (2007). Responses of cetaceans to anthropogenic noise. Mammal Review 37(2): 81-115.

Olesiuk, P.F., Nichol, L.M., Sowden, M.J., Ford, J.K.B., 2002. Effect of the sound generated by an acoustic harassment device on the relative abundance and distribution of harbor porpoises (*Phocoena phocoena*) in retreat passage, British Columbia. Marine Mammal Science 18(4) : 843–862; doi: 10.1111/j.1748-7692.2002.tb01077.x

OSPAR, 2008. Assessment of the environmental impact of offshore windfarms. OSPAR Biodiversity Series, publication 385.

OSPAR, 2009. Overview of the impacts of anthropogenic underwater sound in the marine environment. OSPAR Biodiversity series, publication 441.

OSPAR, 2010. OSPAR database on offshore wind-farms; data 2009/2010 (updated in 2010). OSPAR Biodiversity Series.

Otani, S., Naito ,Y., Kato, A. & Kawamura, A., 2001. Oxygen consumption and swim speed of the harbor porpoise *Phocoena phocoena*. Fishery Science 67(5): 894–898.

Reijnders, P.J.H., Donovan, G.P., Bjørge, A., Kock, K.-H., Eisfeld, S., Scheidat.M. & Tasker, M.L., 2009. ASCOBANS Conservation Plan for harbour porpoises (*Phocoena phocoena L.*) in the North Sea. Report presented to the 16th ASCOBANS Advisory Committee meeting, Bruges, Belgium, 20-24 April 2009, AC16/Doc.21.

Richardson, W.J. & Würsig, B., 1997. Influences of man-made noise and other human actions on cetacean behaviour. Marine and Freshwater Behaviour and Physiology 29 (1-4), Special Issue: Underwater bioacoustics: behavioral, environmental and evolutionary perspectives; 183-209.

Scheidat, M., Tougaard, J., Brasseur, S., Carstensen, J., van Polanen Petel, T., Teilmann, J. & Reijnders, P., 2011. Harbour porpoises (*Phocoena phocoena*) and wind farms: a case study in the Dutch North Sea. Environmental Research Letters 6, 025102. doi: 10.1088/1748-9326/6/2/025102.

Shapiro, A.D., Tougaard, J., Jørgensen, P.B., Kyhn, L.A., Balle, J.D., Bernardez, C., Fjälling, A., Karlsen, J. & Wahlberg, M., 2008. Transmission loss patterns from acoustic harassment and deterrent devices do not always follow geometrical spreading predictions. Marine Mammal Science 25(1): 53–67. doi: 10.1111/j.1748-7692.2008.00243.x

SMRU, 2007. Assessment of the potential for acoustic deterrents to mitigate the impact on marine mammals of underwater noise arising from the construction of offshore windfarms. Report of the Sea Mammal Research Unit, St.-Andrews, UK, Commissioned by COWRIE Ltd (project reference: DETER-01-07).

Southall, B.L., Bowles, A.E., Ellison, W.T., Finneran, J., Gentry, R., Green, C.R., Kastak, C.R., Ketten, D.R., Miller, J.H., Nachtigall, P.E., Richardson, W.J., Thomas, J.A. & Tyack, P.L., 2007. Marine Mammal Noise Exposure Criteria. Aquatic Mammals 33: 411-521.

Thiele, R., 2002. Propagation loss values for the North Sea. Handout Fachgespräch: Offshore-Windmillssound emissions and marine mammals. FTZ-Büsum, 15 January 2002.

Thomas, L., Laake J.L., Rextad, E., Strindberg, S., Marques, F.F.C., Buckland, S.T., Borchers, D.L., Anderson, D.R., Burnham, K.P., Burt, M.L., Hedley, S.L., Pollard, J.H., Bishop, J.R.B. & Marques, T.A., 2009. Distance 6.0. Release 2. Research Unit for Wildlife Population Assessment, University of St. Andrews, UK. http://www.ruwpa.st-and.ac.uk/distance/

Thomsen, F., Lüdemann, K., Kafemann, R. & Piper, W., 2006. Effects of offshore wind farm noise on marine mammals and fish. BIOLA, Hamburg, Germany, on behalf of COWRIE Ltd.

Tougaard, J., Henriksen, O.D. & Miller, L.A., 2009. Underwater noise from three types of offshore wind turbines: estimation of impact zones for harbor porpoises and harbor seals. J. Acoust. Soc. Am. 125: 3766–3773.

Tougaard, J., Kyhn, L.A., Amundin, M., Wennerberg, D. & Bordin, C., 2011. Harbour porpoise are deterred by simulated pile driving noise. 25th Conference of the European Cetacean Society, Book of Abstracts, 21–23 March 2011, Cádiz, Spain.

Verboom, W.C. & Kastelein, R.A., 2005. Some examples of marine mammal discomfort thresholds in relation to man-made noise. Proceedings UDT 2005, Amsterdam.

Chapter 10. Differentiating between underwater construction noise of monopile and jacket foundation wind turbines: A case study from the Belgian part of the North Sea

A. Norro*, B. Rumes & S. Degraer

Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models (MUMM), Marine Ecosystem Management Section, Gulledelle 100, 1200 Brussels

**Corresponding author: A.Norro@mumm.ac.be*

Piling of a pinpile on the Thorntonbank Photo MUMM / RBINS

Abstract

Offshore wind farms generate underwater noise during construction, operation and decommissioning. Different foundation designs are in use for offshore wind farms. Steel monopiles, jackets made of four steel pinpiles and gravity-based foundations are applied in the Belgian part of the North Sea. Both monopiles and jacket pinpiles are hammered into the bottom and as such excessive noise is generated during their installation. This paper aims at comparing the emitted underwater noise generated during the piling activities of two wind farms, each using a different type of pile foundation: steel monopiles were used in the Belwind wind farm (Bligh Bank), while jacket pinpiles were chosen for the C-Power project (phases 2 and 3) on the Thorntonbank.

Underwater noise was measured from a rigid hull inflatable boat at various distances from the pile driving location during the installation of steel monopiles and pinpiles. Analysis focused on the quantification of the discontinuous impulsive pile driving-generated underwater noise. Zero to peak sound pressure level (Lz-p), unweighted sound exposure level (SEL), cumulative SEL and 1/3 octave spectra were computed in order to quantify and compare the underwater noise. Lz-p and SEL were normalized to 750 m distance from the piling location. Piling of the monopiles generated a Lz-p between 179 and 194 dB re 1µPa. Although piling of the smaller pinpiles was expected to be less noisy, no significant differences with monopiling could be demonstrated (pinpiling Lz-p: 172-189 dB re 1 µPa). Similarly, also SEL, varying between 145 and 168 dB re 1 µPa²s, exerted no statistical difference between mono- and pinpiling and furthermore near identical spectra were measured for both types of piling.

The rather wide variability within and broad overlap between both types of piling can possibly be attributed to the (small scale) differences in bottom type as well as in the setup used. However, the radius of discomfort in the harbour porpoise *Phocoena phocoena* was estimated at 30 km and 19 km for monopiles and pinpiles, respectively. Furthermore, the pinpiling (four pinpiles per foundation) took about 2.5 times more time than the monopiling and will as such have a prolonged impact onto the marine fauna. When standardised to megawatt installed (3 MW turbines on the monopiles and 6 MW turbines on the jacket foundations) both types of piling will score about equally.

Samenvatting

Windmolenparken op zee genereren onderwatergeluid tijdens de bouw, de exploitatie en de afbraak. Er bestaan verschillende types funderingen voor windturbines in zee. In het Belgisch deel van de Noordzee worden er stalen monopiles, jacket-funderingen (die steunen op vier pinpiles) en gravitaire funderingen geplaatst. Zowel de pinpiles voor de jacket-funderingen als de monopiles worden in de zeebodem geheid, hetgeen zeer hoge niveaus van onderwatergeluid veroorzaakt. In deze studie wordt een vergelijking gemaakt van het onderwatergeluid tijdens het heien van de funderingen tijdens de bouw van twee windmolenparken die een verschillend type fundering gebruikten: Belwind (Bligh Bank) installeerde monopiles, C-Power (Thorntonbank) gebruikte jacket-funderingen met vier pinpiles per turbine voor fase twee en drie van hun windmolenpark.

Het onderwatergeluid werd gemeten op verschillende afstanden van de heilocaties vanop een ‘rigid hull inflatable boat’. De analyses focusden op de quantificatie van het discontinue impulsgeluid dat onstond tijdens het heien. Het ‘nul tot piek’ geluidsdrukniveau (Lz-p), het ongewogen geluidsdrukniveau (SEL), het cumulatieve SEL en het 1/3 octaaf spectrum werden berekend om het onderwatergeluid te kunnen quantificeren en vergelijken. Lz-p en SEL werden genormaliseerd tot 750 m afstand van de heilocatie. Het heien van de monopiles genereerde een Lz-p tussen 179 en 194 dB re 1µPa. Er werd verwacht dat het heien van de kleinere pinpiles minder onderwatergeluid zou genereren, maar er werden geen significante verschillen gevonden met het heien van de monopiles (pinpiles Lz-p: 172-189 dB re 1 µPa). Ook voor het SEL, dat varieerde tussen 145 en 168 dB re 1 µPa²s, werd er geen significant verschil gevonden. De analyses van de spectra van het onderwatergeluid toonden aan dat deze bijna identiek waren voor het heien van de beide types funderingen.

De grote variabiliteit in en de grote overlap van de resultaten van de twee types van heien kan mogelijks toegeschreven worden aan de (kleinschalige) verschillen in bodem type op de verschillende

locaties en de gebruikte setup tijdens het heien. De straal waarbij er ongemak ('discomfort') optreedt voor de bruinvis *Phocoena phocoena* werd geschat op 30 km en 19 km respectievelijk voor monopiles en pinpiles. Het heien van pinpiles (vier per fundering) neemt ongeveer 2.5 meer tijd in beslag dan het heien van monopiles en heeft dus een langere impact op de mariene fauna. Wanneer dit gestandaardiseerd wordt naar het aantal geïnstalleerde megawatt (3 MW turbines op de monopiles en 6 MW turbines op de jacket-funderingen), dan scoren beide funderingstypes gelijkaardig.

10.1. Introduction:

The European Marine Strategy Framework Directive obliges every member state to achieve or maintain good environmental status, under which also the introduction of energy including underwater noise is considered a main concern (Tasker *et al.*, 2010). In Belgium, the MSFD indicator for impulsive noise reads as "the level of anthropogenic impulsive sound sources is less than 185 dB re 1 µPa (zero to peak sound pressure level (SPL) – $L_{z\text{-}p}$) @ 750m from the source[1]" (Anonymous, 2012).

One of the major concerns in excessive underwater noise emissions is linked to offshore wind farms, as this industry is relatively new to the marine environment (Huddelston, 2010), is developing fast and is highly diverse in technology used (EWEA 2012). As such, at present major attention is paid to the underwater noise generated during the construction, operation and (future) dismantlement of offshore wind farms (Huddelston, 2010). Here, four different phases should be distinguished in relation to the life cycle of an offshore wind farm: (1) the before implantation phase - reference situation, (2) the construction phase, (3) the operational phase and (4) the dismantlement phase (Nedwell & Howell, 2004).

For the BPNS, the underwater noise emissions were documented for the first three phases, with reference to sound pressure levels (SPL) of about 100 dB re 1 µPa at the Thorntonbank and Bligh Bank (Henriet *et al.*, 2006; Haelters *et al.*, 2009). So far, seven wind farms are planned for the Belgian part of the North Sea (BPNS), of which four have been granted both a domain concession and environmental permit. Two wind farms have actually been constructed. The first six wind turbines (C-Power project, phase 1; Thorntonbank) were built on concrete gravity based foundation (GBF), while in a second and third phase jacket foundations, involving the piling of four pinpiles per jacket, were used. In a second wind farm (Belwind project, Bligh Bank) only monopile foundations were applied. During the operational phase finally Norro *et al.* (2011) measured a 20 dB re 1 µPa increase in mean SPL emitted in case of a steel monopile foundation (totalling 120 dB re 1 µPa at 100 Hz), while hardly any noise emission was observed in case of GBFs.

This paper focuses on the differences in underwater noise emissions by two different types of piling, i.e. piling of large monopiles (further called: monopiling) and the piling of the jacket foundation pinpiles (further called: pinpiling). Next to Lz-p, the best measures for comparing noise from pile driving also include sound exposure level (SEL), as the latter is better related to the energy emitted by the piling. Comparison of both piling activities therefore focused on both $L_{z\text{-}p}$ and SEL. We finally also compared their noise spectra and attenuation functions.

10.2. Material and methods:

Analysis focused on the quantification of the discontinuous impulsive pile driving-generated underwater noise. Underwater noise was measured at various distances from the pile driving location during the installation of steel monopiles and jackets at the Bligh Bank and Thorntonbank site, respectively. Zero to peak sound pressure level ($L_{z\text{-}p}$), unweighted sound exposure level (SEL), cumulative SEL and 1/3 octave spectra were computed in order to quantify the underwater noise emitted during the construction phase.

---

[1] Not applicable for bomb explosion (defence).

10.2.1. Measurement methodology

The same measurement protocol, as used for previous underwater noise measurements in Belgian wind farms was used for the present study (see Haelters *et al.* 2009). In summary, measurements of wind farm construction noise were performed from a drifting rigid hull inflatable boat (RHIB) in the vicinity of the piling site. To avoid interaction with the hydrophone, the engine, radar and echosounder, if present onboard, were turned off. The geographic position and time of measurement were recorded with a handheld GPS GARMIN GPSMap60 at a frequency of one position every 5 seconds. The clock of the recorder was synchronised beforehand with the GPS-time (UTC). At the start and the end of each measurement a reference signal was recorded.

Several recordings of few minutes each (1 to 5 min.) were performed at different locations on 26th September 2009 and 15th January 2010 at the Bligh Bank (Norro *et al.*, 2010) and on the 11th May and the 12th of July 2011 at the Thorntonbank site (table 1). Weather conditions encountered during fieldwork featured wind of 1-3 BF and a sea state of 1 to 2.

Table 1. Metadata of the underwater noise measurements at the Blighbank site: monopile A02 and monopile B10; and at the Thorntonbank site phase 2 & 3: jacket CG3 and jacket CB6.

Position start recording		**Distance (m) from piling location**
Latitude	**Longitude**	
Monopile A02 (26th September 2009)		
51°40.39’	2°50.03’	~3000
51°39.41’	2°50.64’	~4820
51°38.25’	2°51.25’	~6990
Monopile B10 (15th January 2010)		
51°34.59’	2°57.31’	~14150
51°38.52’	2°48.16’	~1580
51°38.50’	2°47.44’	~770
Jacket CG3 (11th May 2011)		
51°33.92’	2°58.94’	~250
51°51.34’	2°58.36’	~500
51°33.96’	2°58.93’	~250
Jacket CB6 (12th July 2011)		
51°33.07’	2°53.94’	~600
51°32.96’	2°52.59’	~1700
51°32.65’	2°53.42’	~750
51°32.22’	2°53.01’	~1600

10.2.2. Acoustic measurement equipment

For every measurement, a Brüel & Kjær hydrophone (type 8104) was deployed at a depth of 10 m. A Brüel & Kjær amplifier (Nexus type 2692-0S4) was connected between the hydrophone and the recorder in order to allow for an amplification and filtration of the signal. A reference signal was used together with the output sensitivity of the Nexus to calibrate the amplitude of the recorded signal. The signal was recorded using an audio MARANTZ Solid State Recorder (type PMD671). It was operated with the highest possible sampling rate of 44100 Hz. The signal was recorded in WAVE format (.wav) on Compact Flash cards of 2 GB (Sandisk Ultra II). Batteries powered all equipment.

10.2.3. Response variables

$L_{z\text{-}p}$ is defined as in Ainslie (2011),

$$L_{z-p} = 10\ log_{10} \frac{p_{z-p}^2}{p_{ref}^2} \quad in\ dB\ re\ 1\ \mu\ Pa$$

For impulsive sound, however, the unweighted SEL better characterises the energy produced by a given stroke, extracted from a complete piling event. SEL is computed as defined by Ainslie (2011).

The SEL is the level of a continuous sound during the integration period and having the same sound energy as the impulse.

$$SEL = 10\log\left(\frac{1}{T}\int_{T1}^{T2}\frac{p(t)^2}{p_0^2}\,dt\right) = 10\log\frac{E}{E_{ref}}\quad in\ dB\ re\ 1\ \mu\ Pa^2\ s$$

, where T is 1 second, T1 and T2 are respectively the start and the end of the integration time window (the complete stroke being included in this window), p(t) is the sound pressure signal and p_0 is the reference sound pressure of 1 µPa. When more than one noise pulse is generated as is the case for pile driving, it is possible to compute a cumulative sound exposure level. For a series of strokes, the cumulative SEL is computed following the definition given by de Jong *et al.* (2011), advising not to rely only on cumulative SEL, but to also include the total number of blows and the frequency of piling. Following Nehls *et al.* (2007) and Müller (2011) measurement made at various distances can be normalized to a reference distance of 750 m using the equation:

$$L_{norm} = L_{measured} + 15\log_{10}(distance/750)$$

This normalization has been used in this study in order to allow for an appropriate comparison of noise characteristics collected at various distances from the source.

The third octave band spectrum of the underwater sound pressure level was computed according to the norm IEC1260. All these computations were made using dedicated routines developed using the MATLAB environment.

A Kruskal-Wallis test, followed by post hoc multiple comparison tests, were used to identify statistically significant differences in the underwater noise emitted by the different foundation types.

10.2.4. Piling activity details

For the piling of the 56 monopile foundations at the Bligh Bank, a hammer IHC hydrohammer S1200, operated from the support vessel Svanen, was used. The hammer features a maximum power of 1200 kJ. The average energy used for each stroke was 706 kJ (Table 2). For the installation of the 49 jacket foundations at the Thortonbank, the piling of 196 pinpiles was required. The hammer used was an IHC hydrohammer S-800 featuring a maximum power of 800 kJ for a nominal power of 720 kJ. Average energy used for each stoke was 412 kJ. The hammer log did not record a time stamp for every blow along with the other information, hampering a direct comparison between the measurements and the hammer log.

Table 2. Summary statistics of the piling activities of monopile A02 and B10 and jacket foundations CB6 and CG3, targeted in this study, as well as the averages and total (where appropriate) for the 56 monopiles installed at the Bligh Bank (source: Belwind) (from Norro *et al.* 2010) and the 49 jacket installed on the Thorntonbank (Source C-Power).

Monopile Piling activities (pile diameter = 5m)					
	Unit	**A02**	**B10**	**Average**	**Total**
Pile length	m	55	63	54	
Mass	t	401	453	375	
Number of strokes required		2114	3848	2982	168550
Average energy per stroke	kJ	642	839	706	
Duration of piling	min	64	163	120	6779
Net piling frequency	# strokes/minute	42	39	40	
Total energy	kJ	1356154	3223808	2084454	118908074
Jacket Piling activities (pinepile diameter = 1.8m)					
	Unit	**G3**	**B6**	**Average**	**Total**
Pile length	m	48	21	37	
Mass	t	96	46	77	
Number of strokes required		13321	4288	9476	464328
Average energy per stroke	kJ	436	321	412	
Duration of piling	min	405	162	319	15646
Net piling frequency	# strokes/minute	about 40	about 40		
Total energy	kJ	5804717	1376243	3908793	191530843

10.3. Results:

10.3.1. Underwater noise sound pressure and exposure levels

The highest normalised $L_{z\text{-}p}$ of 194 dB re 1µPa was observed for the piling of the B10 monopile at the Bligh Bank while a maximum of 189 dB re 1µPa was observed for the piling of the jacket pinpiles (CG3) at the Thorntonbank (Table 3). The lowest $L_{z\text{-}p}$ value of 172 dB re 1µPa was observed for the piling of the jacket CB6, while the lowest $L_{z\text{-}p}$ for monopiles was 179 dB re 1µPa. The piling of the jacket foundation CG3 and the piling of the monopile A02 exert similar normalized $L_{z\text{-}p}$ values of about 186 dB re 1 µPa. Some lower normalized $L_{z\text{-}p}$ (by 15 to 20 dB re 1 µPa) is observed for the piling of the jacket CB6.

Normalized maximum SEL values range between 151 and 178 dB re 1 µPa^{2}s. The maximum observed normalised SEL for jacket foundation piling was 178 dB re1 µPa^{2}s (CG3), while the maximum observed normalized SEL for monopiles (B10) was some 10 dB lower with a maximum of 166 dB re1 µPa^{2}s. Normalized mean SELs show similar behaviour with the highest value of 168 dB re1 µPa^{2}s measured at CG3 and the lowest value for jacket piling of 145 dB re1 µPa^{2}s (CB6). Normalized mean SELs for both steel monopile are in between with 168 dB re1 µPa^{2}s for B10 and 164 dB re 1 µPa^{2}s for A02. Whereas statistically significant differences were detected between the four piling events for normalized maximum SEL (Kruskal-Wallis test: $p = 0.016$) and mean SEL ($p = 0.020$), *post hoc* multiple comparisons revealed differences only between the two jacket piling events ($p = 0.008$ and $p = 0.018$, respectively).

Table 3: Normalized @ 750 m zero to peak sound pressure level ($L_{z\text{-}p}$) in dB re 1µPa. Normalized @750m mean and maximum sound exposure level (SEL) in dB re 1 µPa^{2}s.

	Record	**Norm. $L_{z\text{-}p}$ @ 750m**	**Norm. mean SEL @ 750 m**	**Norm. max. SEL @ 750 m**
Monopile A02	1	186	161	164
	2	189	164	166
	3	180	160	164
Monopile B10	1	194	162	166
	2	190	168	162
	3	179	163	166
Jacket CG3	1	185	168	174
	2	189	168	178
	3	186	168	175
Jacket CB6	1	180	155	159
	2	172	145	151
	3	176	150	152
	4	180	152	157

10.3.2. Underwater noise spectra

For both monopile and jacket piling, the highest underwater noises are emitted between the frequencies of 60 to 2000 Hz. Moreover, while the shape of the spectra are similar in the frequency domain 100 to 500 Hz, the spectra show more isolated peaks for the piling of jacket than for the piling of monopile, for which only one larger peak is found. The decay of the spectra shows a similar slope for both foundation types.

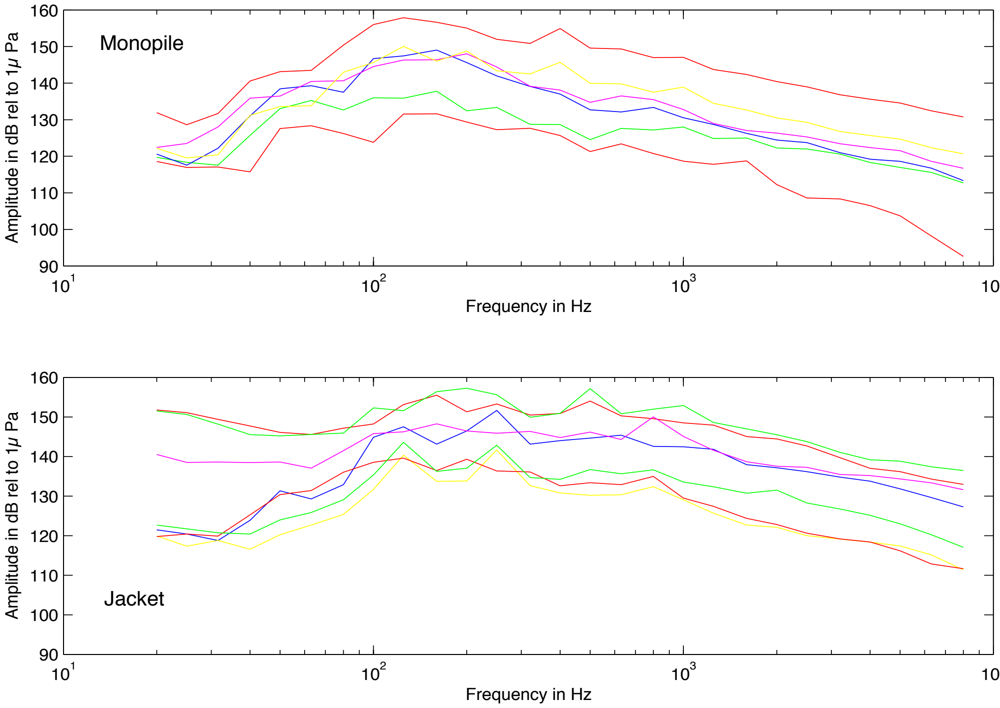


Figure 1. Spectral analysis (1/3 octave) of the emitted underwater noise during the piling of the monopiles at the Bligh Bank wind farm location and the jackets at the C-Power (phases 2 and 3), Thorntonbank.

On average, a jacket foundation requires about three times more blows per foundation (Table 4) than a monopile, but when that parameter is normalized to MW installed, that figure is inversed with 33% less blows/MW installed for jacket foundations than for monopile. Nevertheless, the average piling time required is higher for jacket foundation than for monopile (factor 2.5) and remains somewhat higher when normalized to MW installed (factor 1.3).

Table 4: Characterization of the monopile and jacket piling activities. Computed from data provided by the concessioners Belwind (Bligh Bank) and C-Power (Thorntonbank) or computed from our measurements of underwater noise. Normalized maximum sound exposure level (Norm. Max. SEL @ 750 m).

Foundation type	**Monopile (3 MW)**	**Jacket (6 MW)**
Average n° blow / foundation	3010	9476
Average n° blow / MW installed	1021	1612
Average energy (kJ)/ blow	705	412
Average energy (kJ) / foundation	2123358	3908792
Average energy (kJ) / MW installed	720655	665038
Norm. Max. SEL @750 m (dB re 1 μ Pa²s)	166	178
Average duration of piling (min)/ foundation	120	319
Average duration of piling (min)/ MW installed	41	55
Average piling frequency (blow/min)	25	30

10.3.3. Noise propagation and attenuation

The propagation model developed by Norro *et al.* (2010) for the Belwind site was also applied for the C-Power phase 2&3 site that is only few NM away and characterised by similar bathymetry. Application of the model permits (Figure 2) to compute the distance needed to reach the discomfort level $L_{z\text{-}p}$ for Porpoise of 134 dB re 1 μ Pa (Tougaard *et al.,* 2011) proposes a peak to peak sound pressure level of 140 dB re 1 μ Pa. Peak to peak level ≊ $L_{z\text{-}p}$ + 6 dB). These values are respectively 19 km away from the C-Power pinpiling activities and 30 km away from the Belwind monopiling activities.

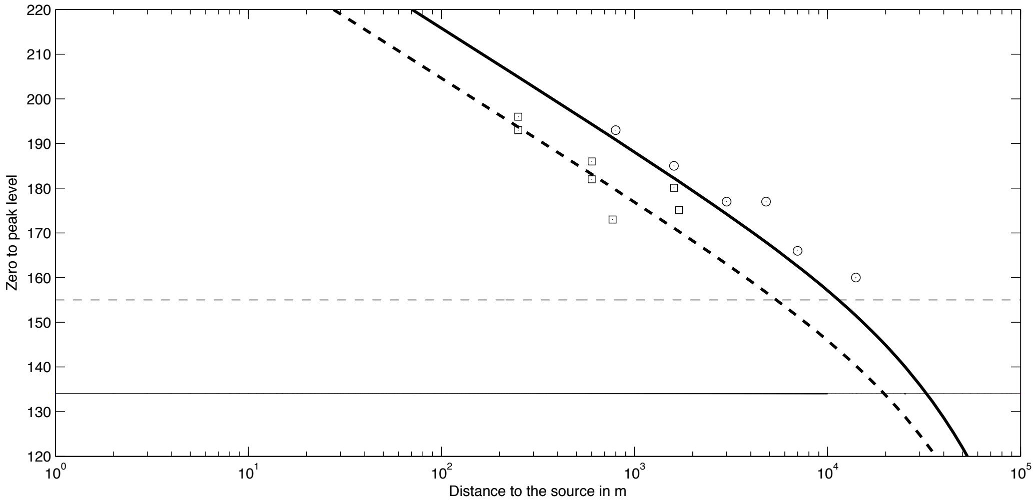


Figure 2. Application of the propagation model to jacket piling (dashed line) and monopile piling (plain line). Squares and circles are the measured $L_{z\text{-}p}$ respectively for jacket and monopile while the horizontal line at 134 dB re 1 μ Pa represents the discomfort level ($L_{z\text{-}p}$) for harbour porpoise (Tougaard et al., 2011) and the dashed line at 155 dB re 1 μ Pa the threshold level for major disturbance Bailey *et al.* (2010).

Bailey *et al.* (2010) propose a threshold for behavioural disturbance for harbour porpoise with minimal disturbance possible at 90 dB re 1 μ Pa (which is even below the background value at the BPNS), and while major disturbance above 155 dB re 1 μ Pa. Application of our model shows major disturbance inside a 6 km diameter circle around C-Power phase 2&3 while the diameter of that major disturbance zone extends to 11 km from the Belwind wind farm when it was under construction.

10.3.4. Cumulative sound exposure level

The mean number of strokes required for the complete piling of one monopile foundation was 3010 strokes (Table 4). As 3010 strokes represent an increase of the normalized @750m mean SEL of 35 dB (10 log10(3010)), the mean cumulative SEL for monopile was estimated at 196 dB re 1 µ Pa^2s. The mean duration of piling for one foundation was 120 min. A mean number of 9476 strokes was required for the installation of one jacket foundation (incl. four pinpiles). This represents an increase of 40 dB, giving a cumulative normalized SEL of 196 dB re 1 µ Pa^2s @750 m. The mean duration of piling for one jacket was 319 min. The same cumulative SEL values were hence observed for both foundation type, but with a longer disturbance time for jacket foundations compared to monopiles. Similar parameter renormalized at 750 m for the Q7wind farm (Netherlands, 4m diameter, De Jong & Ainslie, 2008) was 209 dB re 1 µ Pa^2s. Nehls *et al.* (2007) presents a table featuring peak level for various wind farms. The piling duration is not given and comparison is only possible with Lz-p levels when re-normalized from 500 to 750 m. Lz-p ranges between 185 and 199 dB re 1 µ Pa for pile diameter ranging between 3,3 and 4,7 m. These results are of the same order of magnitude and coherent with what was observed in the Belgian part of the North Sea.

10.4. Discussion:

As expressed earlier, the piling work linked to the installation of the jacket foundation requires the piling of four pinepiles, while the monopile design requires the piling of only one large monopile. Jacket foundations may however accommodate larger turbines than monopoles (EWEA 2012). A less powerful hammer can be used for the installation of the jacket foundations than for the monopile foundations. However, jacket design requires longer piling time than the monopile design (here: mean time of 319 min for jacket against 120 min for monopile), but at lower noise levels with a normalized Lz-p of maximum 194 dB re 1 µ Pa for a monopile against 189 dB re 1 µ Pa for a jacket. Jacket installation hence impacts a smaller zone, but for a longer time.

In terms of energy, the total piling energy needed to achieve the complete construction of the C-Power project, phases 2 and 3 at the Thorntonbank (49 jacket foundations) was just above 0.19 TJ (Table 2), while the same figure for the Belwind wind farm implanted at the Bligh Bank and featuring 56 monopile foundations was 0.12 TJ. The overall message is that more energy was used and therefore transmitted to the environment for the installation of the new Thorntonbank wind farm (288 MW installed capacity) than for the installation of the monopiles at Bligh Bank (165 MW installed capacity). This is further confirmed by the SEL data (Table 3) featuring a maximum value for the normalized SEL of 178 dB re 1 µ Pa2s for the Thorntonbank wind farm against 166 dB re 1 µ Pa2s for the Blighbank one.

When underwater noise is generated by pile driving, the size of the pile, power of the pile driver (hammer) and sedimentological and geological properties are important variables, affecting the effective underwater noise produced. Nehls *et al.* (2007) showed that for similar sediment properties, using a larger pile driver would generate less noise because of a lower impact velocity applied when hammering. It also could be more economical to use a large pile driver operated at 2/3 of its nominal power than a smaller used at its maximum power. The use of a less powerful hammer (800kJ) for pinpiling (versus 1200 kJ for monopiling) in conjunction with the use of smaller pile produced lower Lz-p values than for the monopiling at the Bligh Bank (some 5 dB re 1 µ Pa @ 750m). The higher SEL identified for the piling of jacket CG3 at the Thorntonbank (Table 2) in comparison with the piling of the jacket CB6 is most probably related to the use of the hammer at a higher power, even if we cannot demonstrate that relation due to the unavailability of a timestamp on every blow. However, to conclude on the differences observed between pinpiling and monopiling, a significant difference was found inside the pinpiling group (Table 3). This significant difference can be explained by the fact that the piling of one of the jackets (CB6) required only a third of the mean energy used for the installation of the other jackets (Table 2). This could indeed be related to the small-scale differences in sedimentological and geological properties within the C-Power concession area.

Nevertheless, when renormalizing these data to the installed power expressed in MW, the message is different with a little lower average energy per MW used for the jacket foundation (665038 kJ) than for the monopile foundation (720655 kJ). While jacket foundation pinpiling use less piling energy per MW, the average duration of piling per installed MW remains 26 % higher with 55 minutes for a jacket and only 41 minutes for a monopile. However, an even better normalization would be obtained when standardising to the MW produced instead of the MW installed. Such standardisation would however be premature at this moment, since the wind farms are operational either for a short period of time (Belwind) or not yet operational at all (C-Power, phases 2 and 3).

One has to note that some of the levels observed here for both the monopile or jacket type foundations exceeds the 185 dB re 1µPa permitted by the Belgian MSFD descriptor 11 and as such mitigating measures will need to be undertaken in the future.

10.5. Conclusion:

While jacket foundations involved smaller diameter pinepiles and while the emitted normalized at 750m $L_{z\text{-}p}$ values are lower than for monopiling, therefore impacting a smaller zone, the overall energy needed for the complete piling was 58% higher for the 49 jackets than for the 56 monopiles. The normalized @750 SEL was also higher for jacket than for monopile foundation piling.

When normalized to installed MW the figure is inversed and average energy needed by installed MW is 8% lower for jacket than for monopile.

Finally, for both maximum and mean normalized @ 750m SEL, no statistically significant difference on the emitted underwater noise between pinpiling and monopiling could however be observed.

The radius of discomfort level for harbour porpoises can extend to 30 km in the case of monopiling, while radius for major disturbance was modelled to reach 11 km. These numbers read respectively 19 km and 6 km for pinpiling.

Some measurements are above the proposed Belgian MSFD requirements and this both for monopile as well as for jacket foundations.

10.6. Acknowledgement:

The officers and crew of the RV Belgica are acknowledged for their help provided during fieldwork. We would like to thank the financial support of the companies C-Power, Belwind and Northwind.

10.7. References

Ainslie M.A. (2011). Standard for measurement and monitoring of underwater noise, Part I. Physical quantities and their units. TNO report. TNO-DV 2011 C235. 52pp. and 3 appendixes.

Anonymous, 2012. MSFD-Belgium. Définition du bon état écologique & definition d'objectifs environnementaux. Directive-cadre stratégique pour le milieu marin Art 9&10. 01/04/2012. 28pp.

Bailey, H., Senior, B., Simmons, D., Rusin, J., Picken, G. & Thompson, P., (2010). Assessing underwater noise levels during pile-driving at an offshore windfarm and its potential effects on marine mammals. Marine Pollution Bulletin 60: 888-897.

De Jong C.A.F, Ainslie M.A & Blacquière G. (2010). Standard for measurement and monitoring of underwater noise, Part II: Procedures for measuring underwater noise in connection with offshore wind farm licensing. TNO report. TNO-DV 2011 C251. 57pp.

De Jong C.A.F. & Ainslie M.A., (2008). Underwater radiated noise due to the piling for the Q7 Offshore Wind Park. Acoustics 2008 Paris.

EWEA (2012). Wind energy- The facts- A guide to the technology, economics and future of wind power- European Wind Energy Association. 488pp.

Haelters J, Norro A & Jacques T., (2009). Underwater noise emission during the phase I construction of the C-Power wind farm and baseline for the Belwind wind farm. *In:* Degraer, S. & Brabant, R. (Eds.). Offshore wind farms in the Belgian part of the North Sea. State of the art after two years of environmental monitoring. Chapter 10: 237-266.

Henriet, J.-P., Versteeg, W., Staelens, P., Vercruysse, J. & Van Rooij, D., (2006). Monitoring van het onderwatergeluid op de Thorntonbank: referentietoestand van het jaar nul, eindrapport. Studie in opdracht van het KBIN/BMM, rapport JPH/2005/sec15 , Renard Centre of Marine Geology Ghent University, Belgium July, 2006.

Huddelston, J. (ed) (2010). Understanding the environmental impacts of offshore windfarms COWRIE 138 pp.

Müller A. & Zerbs C., (2011). Offshore wind farms. Measurement instruction for waterborne sound measurements. Report M88 607/5 21pp. Müller-BBM GmbH

Nedwell, J.R. & Howell, D. (2004). A review of offshore windfarm related underwater noise sources. Report N° 544 R 0308 COWRIE 57 pp.

Nehls G, Betke K., Eckelmann S. & Ros M. (2007). Assessment and costs of potential engineering solutions for the mitigation of the impacts of underwater noise arising from the construction of offshore windfarms. COWRIE report ENG-01-2007. 47 pp.

Norro A., Haelters J., Rumes B. & Degraer S. (2010). Underwater noise produced by the piling activities during the construction of the Belwind offshore wind farm (Bligh Bank, Belgian marine waters). *In:* Degraer, S., Brabant, R., Rumes, B. (Eds.). Offshore wind farms in the Belgian part of the North Sea. Early environmental impact assessment and spatio-temporal variability. Chapter 4: 37-51.

Norro A., B.Rumes & S.Degraer (2011). Characterisation of the operational noise, generated by offshore wind farms in the Belgian part of the North Sea. *In:* Degraer, S., Brabant, R., Rumes, B. (Eds.). Offshore wind farms in the Belgian part of the North Sea. Selected findings from the baseline and targeted monitoring. Chapter 3: 17-26.

Southall, B.L., Bowles, A.E., Ellison, W.T., Finneran, J., Gentry, R., Green, C.R., Kastak, C.R., Ketten, D.R., Miller, J.H., Nachtigall, P.E., Richardson, W.J., Thomas, J.A. & Tyack, P.L., (2007). Marine Mammal Noise Exposure Criteria. Aquatic Mammals 33: 411-521.

Tasker M.L., Amundin M.,Andre M., Hawkins A., Lang B., Merck T., Sholik-Scholmer A., Teilmann J., Thomsen F., Werner S., Zakharia M, (2010). Indicator for the good environmental status for underwater noise and other form of energy. The main report of task group 11 for Marine Strategy Framework Directive's descriptor 11 Draft 11:01/2010. 39pp ICES/JRC report.

Tougaard, J., Kyhn, L.A., Amundin, M., Wennerberg, D. & Bordin, C., (2011). Harbour porpoise are deterred by simulated pile driving noise. 25th Conference of the European Cetacean Society, Book of Abstracts, 21–23 March 2011, Cádiz, Spain.

Annexes

Annex 1: Systematic species list of the soft substratum macrobenthos

Phylum	Class	Order	Family	Species
Annelida	Clitellata	/		*Oligochaeta* sp.
	Polychaeta	Capitellida	Capitellidae	*Capitella minima*
				Notomastus latericeus
		Cirratulida	Paraonidae	*Aricidea (Acmira) catherinae*
				Aricidea simonae
				Aricidea sp.
		Opheliida	Opheliidae	*Ophelia limacina*
				Ophelia rathkei
				Travisia forbesii
		Orbiniida	Orbiniidae	*Scoloplos (Scoloplos) armiger*
		Phyllodocida	Glyceridae	*Glycera alba*
				Glycera lapidum
			Hesionidae	*Microphthalmus similis*
			Nephtyidae	*Nephtys cirrosa*
				Nephtys juv.
				Nephtys longosetosa
			Nereididae	Nereidinae sp.
			Pholoidae	*Pholoe minuta*
			Phyllodocidae	*Eteone longa*
				Eumida sanguinea
				Hesionura elongata
				Phyllodoce juv.
				Phyllodoce laminosa
				Phyllodoce mucosa
				Phyllodoce rosea
				Phyllodoce sp.
			Polynoidae	*Gattyana cirrhosa*
				Harmothoe extenuata
				Harmothoe glabra
				Harmothoe sp.
				Polynoinae sp.
		Magelonida	Magelonidae	*Magelona mirabilis*
		Spionida	Spionidae	*Aonides paucibranchiata*
				Boccardiella ligerica
				Spio sp.
				Spiophanes bombyx
				Streblospio juv.
		Terebellida	Cirratulidae	*Aphelochaeta filiformis*
			Pectinariidae	*Lagis koreni*
			Terebellidae	*Lanice conchilega*
Arthropoda	Malacostraca	/	/	Caridean larvae
		Amphipoda	Amphilochidae	*Amphilochus neapolitanus*
			Aoridae	*Aora typica*
			Atylidae	*Atylus swammerdami*
				Nototropis falcatus
			Calliopiidae	*Apherusa ovalipes*
			Caprellidae	*Pariambus typicus*
				Phtisica marina
			Corophiidae	*Monocorophium acherusicum*
			Ischyroceridae	*Jassa herdmani*
			Leucothoidae	*Leucothoe incisa*

				Leucothoe spinicarpa
			Melitidae	*Abludomelita obtusata*
			Oedicerotidae	*Perioculodes longimanus*
				Pontocrates altamarinus
				Pontocrates sp.
				Synchelidium maculatum
			Pontoporeiidae	*Bathyporeia elegans*
				Bathyporeia guilliamsoniana
				Bathyporeia pilosa
				Bathyporeia sp.
			Stenothoidae	*Stenothoe marina*
			Urothoidae	*Urothoe brevicornis*
				Urothoe elegans
		Cumacea	Bodotriidae	*Bodotria arenosa*
			Diastylidae	*Diastylis rathkei*
				Diastylis rugosa
			Pseudocumatidae	*Monopseudocuma gilsoni*
				Pseudocuma (Pseudocuma) longicorne
				Pseudocuma simile
				Pseudocuma sp.
		Decapoda	/	Decapoda juv.
			/	Zoea larvae (Brachyura)
			Crangonidae	*Crangon crangon*
				Crangon juv.
			Polybiidae	*Liocarcinus* sp.
			Paguridae	*Pagurus bernhardus*
				Pagurus sp.
			Thiidae	*Thia scutellata*
		Isopoda	Cirolanidae	*Eurydice spinigera*
		Mysida	Mysidae	*Gastrosaccus spinifer*
		Nebaliacea	Nebaliidae	*Nebalia bipes*
	Maxillopoda	/	/	Copepoda sp.
Chordata	/	/	/	Pisces sp.
	Actinopterygii	Pleuronectiformes	Pleuronectidae	*Pleuronectes* sp.
	Leptocardii	Amphioxiformes	Branchiostomidae	*Branchiostoma lanceolatum*
Cnidaria	Anthozoa	Actiniaria	/	Actiniaria sp.
			Edwardsiidae	*Edwardsia* sp.
	Hydrozoa	/	/	Hydrozoa sp.
Echinodermata	Asteroidea	Forcipulatida	Asteriidae	*Asterias rubens*
				Asteriidae juv.
	Echinoidea	Clypeasteroida	Echinocyamidae	*Echinocyamus* juv.
				Echinocyamus pusillus
		Spatangoida	Loveniidae	*Echinocardium cordatum*
				Echinocardium juv.
	Ophiuroidea	/	/	Ophiuroidea juv.
		Ophiurida	Ophiuridae	*Ophiura* juv.
				Ophiura ophiura
Mollusca	Bivalvia	/	/	Bivalvia juv.
		Euheterodonta	Pharidae	*Ensis* juv.
		Veneroida	Mactridae	*Spisula elliptica*
				Spisula juv.
				Spisula subtruncata
			Montacutidae	*Kurtiella bidentata*
				Tellimya ferruginosa
			Tellinidae	*Angulus fibula*
				Angulus pygmaeus

				Angulus tenuis
				Macoma balthica
	Gastropoda	Littorinimorpha	Naticidae	*Lunatia catena*
				Euspira pulchella
Nematoda	/	/	/	Nematode sp.
Nemertea	/	/	/	Nemertea sp.
	Anopla	/	/	Heteronemertea sp.
/	/	/	/	Egg/Larvae

Nematoda, Pisces and rare species (all species found in maximum three samples, with a maximum of two individuals per sample) were excluded from all analyses (species highlighted in grey).

Annex 2: Results Two-way SIMPER analysis

Two-way SIMPER analysis: Average abundance of Gradient groups (Southwest, Southeast, Northwest, Northeast) across all Distance groups. Average Similarity (Av.Sim), Average Abundance (Av.Abund.), Contribution% to the total similarity.

Southwest (Av.sim.: 46.97)	Av.Abund	Contrib%	**Southeast** (Av.sim.: 64.90)	Av.Abund	Contrib%
Asteriidae juv.	74.53	21.76	*Spio* sp.	48.98	32.15
Spio sp.	21.31	22.36	*Spiophanes bombyx*	30.67	15.99
Nemertea sp.	20.9	7.57	Asteriidae juv.	28.02	10.84
Spiophanes bombyx	20.26	6.93	*Nephtys cirrosa*	12.67	7.06
Echinocardium juv.	13.01	3.55	*Gastrosaccus spinifer*	12.34	5.88
Lanice conchilega	12.88	3.72	*Bathyporeia pilosa*	10.79	3.76
Nephtys cirrosa	11.72	9.64	*Bathyporeia elegans*	9.62	3.93
Gastrosaccus spinifer	7.13	2.99	*Ophelia limacina*	8.24	4.79
Northwest (Av.sim.: 57.34)	Av.Abund	Contrib%	**Northeast** (Av.sim.: 48.21)	Av.Abund	Contrib%
Spio sp.	52.86	35.36	*Spio* sp.	40.1	28.53
Asteriidae juv.	23.82	10.85	*Spiophanes bombyx*	20.49	5.58
Spiophanes bombyx	22.26	6.86	*Nephtys cirrosa*	16.79	17.93
Nephtys cirrosa	14.76	10.81	Asteriidae juv.	16.6	10.07
Gastrosaccus spinifer	12.39	9.44	*Gastrosaccus spinifer*	10.39	6.71
Nemertea sp.	11.55	7.42	*Urothoe brevicornis*	7.54	3.6
Ophelia limacina	10.7	4.41	zoea larvae	7.42	7.5
Urothoe brevicornis	6.9	3.31	*Bathyporeia pilosa*	3.18	3.01

Two-way SIMPER analysis: Average abundance of Distance groups (15, 25, 50, 100 and 200m) across all Gradient groups. Average Similarity (Av.Sim), Average Abundance (Av.Abund.), Contribution% to the total similarity.

15m (Av.sim.: 56.75)	Av.Abund	Contrib%	**25m** (Av.sim.: 44.98)	Av.Abund	Contrib%
Asteriidae juv.	63.37	15.43	Asteriidae juv.	83.56	20.39
Spio sp.	35.36	18.4	*Spio* sp.	34.71	24.92
Spiophanes bombyx	29.73	9.16	*Spiophanes bombyx*	27.29	5.86
Nemertea sp.	19.98	7.43	Nemertea sp.	15.98	5.88
Bathyporeia pilosa	15.07	3.98	*Lanice conchilega*	12.61	2.68
Urothoe brevicornis	13.06	5.82	*Nephtys cirrosa*	12.47	6.42
Lanice conchilega	12.42	3.92	*Urothoe brevicornis*	11.08	4.97
Echinocardium juv.	12.37	3.9	*Gastrosaccus spinifer*	9.53	5.92
Nephtys cirrosa	10.72	4.76	*Ophelia limacina*	8.21	3.26
Ensis juv.	10.32	4.59	*Spisula* juv.	5.24	2.96
Bathyporeia elegans	9.6	5.33			
50m (Av.sim.: 57.91)	Av.Abund	Contrib%	**100m** (Av.sim.: 60.55)	Av.Abund	Contrib%
Asteriidae juv.	51	17.61	*Spio* sp.	50.84	39.78
Spio sp.	47	29.1	*Spiophanes bombyx*	16.52	6.29
Spiophanes bombyx	33.83	15.13	Asteriidae juv.	13.7	10.45
Nephtys cirrosa	15.62	8.67	*Nephtys cirrosa*	13.66	10.66
Gastrosaccus spinifer	12.04	5.16	*Gastrosaccus spinifer*	12.23	8.31
Nemertea sp.	12.04	4.35	*Ophelia limacina*	10.38	7.65
Ophelia limacina	9.97	2.95	Nemertea sp.	8.37	3.77
Bathyporeia pilosa	8.75	3.4	*Bathyporeia pilosa*	4.49	3.61
Urothoe brevicornis	8.12	1.95			
Bathyporeia elegans	7.14	3.06			
200m (Av.sim.: 53.98)	Av.Abund	Contrib%			
Spio sp.	35.99	31			
Spiophanes bombyx	18.67	11.45			
Nephtys cirrosa	15.38	16.18			
Asteriidae juv.	10.8	8.59			
Gastrosaccus spinifer	9.65	7.02			
Nemertea sp.	7.04	4.8			
Ophelia limacina	6.46	3.14			
Bathyporeia elegans	4.13	2.54			