

An ecosystem approach for studying the impact of offshore wind farms: a French case study

Jean-Philippe Pezy^{1*}, Aurore Raoux¹, and Jean-Claude Dauvin¹

¹Normandie Univ UNICAEN, UNIROUEN, Laboratoire Morphodynamique Continentale et Côtière, CNRS, UMR 6143M2C, 24 rue des Tilleuls, 14000 Caen, France

*Corresponding author: tel: +33(0)231565722; e-mail: jean-philippe.pezy@unicaen.fr.

Pezy, J.-P., Raoux, A., and Dauvin, J.-C. An ecosystem approach for studying the impact of offshore wind farms: a French case study. – ICES Journal of Marine Science, doi:10.1093/icesjms/fsy125.

Received 27 March 2018; revised 12 August 2018; accepted 17 August 2018.

The French government is planning the construction of offshore wind farms (OWF) in the next decade (around 2900 MW). Following the European Environmental Impact Assessment Directive 85/337/EEC, several studies have been undertaken to identify the environmental conditions and ecosystem functioning at selected sites prior to OWF construction. However, these studies are generally focused on the conservation of some species and there is no holistic approach for analysing the effects arising from OWF construction and operation. The objective of this article is to promote a sampling strategy to collect data on the different ecosystem compartments of the future Dieppe-Le Tréport (DLT) wind farm site, adopting an ecosystem approach, which could be applied to other OWFs for the implementation of a trophic network analysis. For that purpose, an Ecopath model is used here to derive indices from Ecological Network Analysis (ENA) to investigate the ecosystem structure and functioning. The results show that the ecosystem is most likely detritus-based, associated with a biomass dominated by bivalves, which could act as a dead end for a classic trophic food web since their consumption by top predators is low in comparison to their biomass. The systemic approach developed for DLT OWF site should be applied for other French and European installations of Offshore Wind Farm.

Keywords: Ecological Network Analysis, ecopath model, marine renewable energy, offshore wind farms, sampling strategy, trophic web.

Introduction

The worldwide demand for renewable energy development is increasing rapidly, motivated by the challenge to reduce fossil fuel emissions in accordance with political imperatives to combat global climate change (Raoux *et al.*, 2017). For instance, the European Union (EU) has set a target of 20% of energy consumption to be derived from renewable energy sources by 2020 (Directive 2009/28/EC). In accordance with these political ambitions, many Marine Renewable Energies (MRE) (i.e. wind farms, tidal energy farms) are being developed to facilitate the energy transition. At present, wind represents one of the most cost-effective offshore sources of exploitable renewable energy and is by far the most technically advanced of all MRE sources (Leung and Yang, 2012).

The first OWFs (called Horn Rev 1 and 2) were built in 2002–2003 in Denmark, followed by the Netherlands in 2007, the

United Kingdom and Belgium in 2008, and Germany in 2010 (Petersen and Malm, 2006; Leonhard *et al.*, 2011; Wilding *et al.*, 2017). For more than 15 years, Offshore Wind Farms (OWF) have been built in European waters (Raoux, 2017). This development of MRE raises many technical and social issues. Moreover, concerns have been expressed about the potential environmental impacts of these new structures on marine ecosystems, and their potential impacts need to be carefully assessed (Lindeboom *et al.*, 2011; Bailey *et al.*, 2014). The site exploration, construction, operation, and decommissioning of OWFs could indeed lead to temporary and/or permanent effects on marine ecosystems such as local damage of the seabed, or the disturbance of fish and marine mammal populations (OSPAR, 2008; Mueller-Blenkle *et al.*, 2010; Shields and Payne, 2014). All OWFs that have been built in Europe are subject to environmental monitoring programmes (which is a regulatory requirement of several authorities) to

investigate the impacts of these new structures on the surrounding marine ecosystems (Wilding *et al.*, 2017). All these previous studies provide a large amount of data on environmental effects at the species level. However, one of the main issues linked to these environmental monitoring programmes is that they are focused on certain ecosystem components such as: marine mammals, birds, fish, and benthos. Additionally, a particular emphasis was placed on iconic or flagship species not only due to their endangered status (Boehlert and Gill, 2010) but also their highly popular image among the public (Berger *et al.*, 2014). Thus, even if the monitoring of top predators is accepted, other biological compartments, particularly within the benthic community, have not yet been taken into account. However, several studies have stressed the need to include the benthos within ecosystem monitoring of MREs (Villnäs and Norkko, 2011; Wilding *et al.*, 2017). Although the benthos is a core ecosystem component, Wilding *et al.* (2017) highlighted our poor understanding of its interaction with MRE technology.

Moreover, environmental monitoring programmes have so far only attempted to consider the sensitivity to potential disruptions of a number of ecological compartments (plankton, benthos, suprabenthos, fish, marine mammals, and birds), but in a disparate manner without taking into account the trophic links between the compartments (Raoux, 2017; Raoux *et al.*, 2017). Thus, the environmental impacts of OWF construction and operation remain unclear at the ecosystem scale, particularly as regards the trophic web structure and functioning (Bailey *et al.*, 2014). As highlighted by Raoux *et al.* (2017), there is a need to adopt a holistic approach to the impact of OWFs on ecosystem functioning through the use of trophic web modelling tools. In fact, trophic web models can be used for this purpose since they describe the interactions between species at different trophic levels (from prokaryote to top predators) and are based on the quantification of flows of energy and matter in ecosystems.

For instance, the Ecopath with Ecosim trophic web model (Polovina, 1984; Christensen and Walters, 2004) considers all the biotic components of a system simultaneously, and is useful for gaining a better understanding of the system structure and functioning, as well as in predicting ecosystem changes in response to the construction and operation of MRE projects. Thus, the Ecopath with Ecosim modelling approach contributes to the estimation of anthropogenic effects on ecosystems (Raoux *et al.*, 2017). However, a common bias in Ecopath with Ecosim applications arises from the use of non-representative dietary data (i.e. diet composition data for a different time period without taking into account differences in the relative prey species abundances between two time periods) (Plaganyi and Butterworth, 2004). Such models based on poorly representative data will compromise the results of Ecopath mass balance calculations (Plaganyi and Butterworth, 2004). Thus, there is a need to obtain robust estimates of consumption through stomacal content analysis taking into account the chosen time and space-scales of the model under construction (Plaganyi and Butterworth, 2004; Pezy, 2017; Raoux, 2017).

Ecosystem management requires a clear understanding of marine ecosystem structure and functioning. Thus, the objective of the present study is to promote a sampling strategy to collect local data for future OWF environmental monitoring programmes, and to build an Ecopath model based on robust estimates of energy transfer. This holistic view of the impact of OWFs on the ecosystem through trophic web modelling could be replicated on

the other site in the English Channel (EC) and could be useful to analyse the long-term reef and reserve effects in the context of climate change. Indeed, using quantitative modelling to assess the impacts of OWF on the whole ecosystem would allow new knowledge to be brought to the attention of policy makers. It would also facilitate a better integration of ecological considerations in management decisions, and for planning maritime space.

Material and methods

In France, no OWFs have yet been constructed. However, three successive calls for tenders related to OWF development have been issued and seven sites have been selected for future OWF construction. Among these OWF projects, three are planned in the Eastern part of the EC: Courseulles-sur-Mer (50 km², 75 wind turbines), Fécamp (65 km², 83 wind turbines), and Dieppe-Le Tréport (DLT) (67 km², 62 wind turbines) (Raoux *et al.*, 2017). Beyond the fact that the EC is the current hotspot for OWF development in France (Raoux *et al.*, 2017, 2018; Figure 1), this maritime space is also subject to a large panel of anthropogenic disturbances including pollution, transport, fishing, aquaculture, aggregate extraction, or sediment dredging and deposition (Halpern *et al.*, 2008; Dauvin, 2012).

Dieppe-Le Tréport (DLT) offshore wind farm project

The prime contractor of the project is “Eoliennes en mer Dieppe-Le Tréport”, a subsidiary of Engie (formerly named GDF Suez). The proposed wind farm will be located at distances of 15.5 and 17 km offshore from the coast of Le Tréport and Dieppe, respectively. The water depth ranges from 12 to 25 m. The wind farm will cover a total area of approximately 92 km², and will comprise 62 turbines with a capacity of 8 MW each giving a combined nameplate capacity of 496 MW. The wind farm turbines will be connected via an interarray network consisting of 33 kV AC cables which will link up to an offshore transformer substation located within the wind farm. From this station, power will be exported via two 225 kV AC marine cables. The foundations are composed of jacket structures.

Trophic modelling approach

The *Ecopath with Ecosim* (EwE) software (Polovina, 1984; Christensen and Walters, 2004; Christensen *et al.*, 2008) is used here to model the food web flows at the DLT OWF site. This approach uses functional groups or species biomasses which are connected together through a predator–prey diet matrix (DC). This approach allows us to take into account a large number of species in ecological functional groups. Ecopath is designed to build a snapshot of the ecosystem functioning while Ecosim provides a simulation of its dynamic evolution through time. Ecopath is a mass balance (i.e. ignoring year-on-year changes in biomass compared with flows), single-solution model (i.e. yielding only one value per flow) for estimating fluxes between a set of established trophic compartments. Each compartment corresponds to a single species or a group of species similar in terms of predators, prey, and metabolic rates (i.e. trophic group). It is parameterized with biomass (B , gC m⁻²), production to biomass ratio (P/B , year⁻¹), consumption to biomass ratio (Q/B , year⁻¹) and a DC which represents the interactions between predators and prey in the ecosystem.

The parameterization of an Ecopath model is based on satisfying two equations. The first equation [Equation (1)] describes the

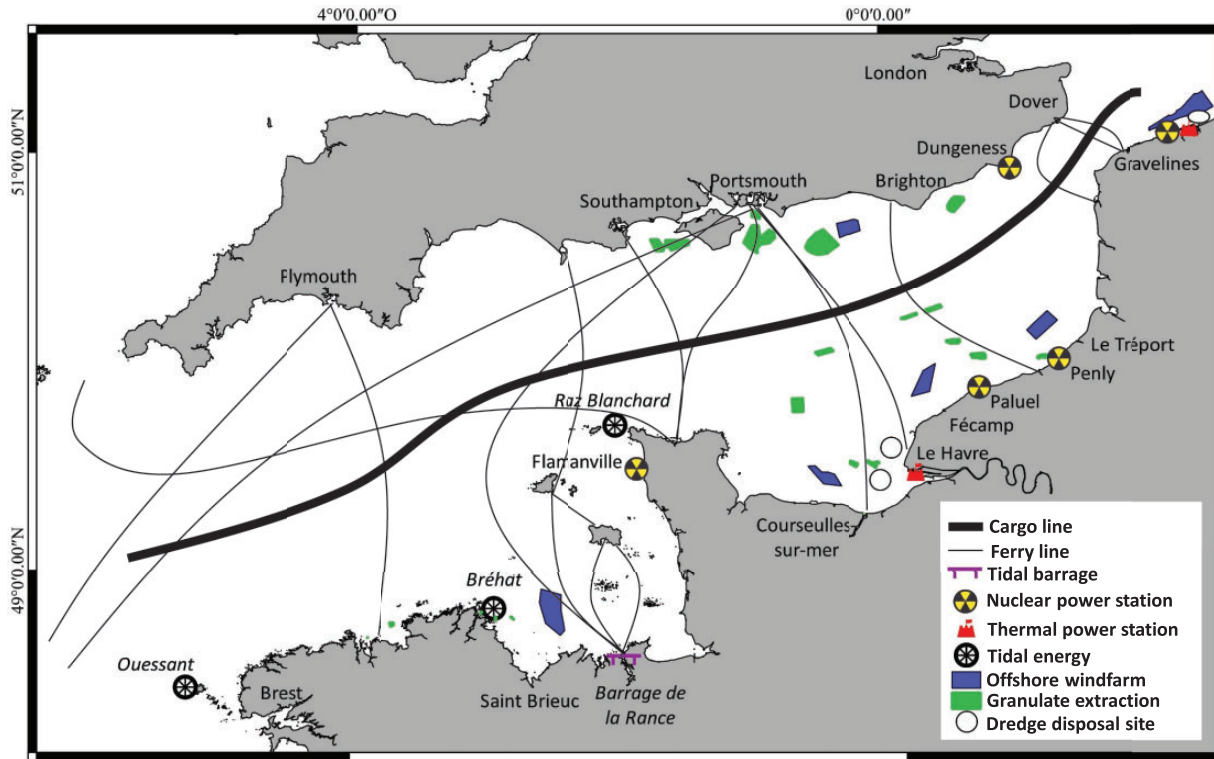


Figure 1. Human activities (without fishing) along the EC. For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.

production of each compartment in the system as a function of the consumption ratio (Q/B) of its predators (j), the fishing mortality (Y_i , gC m^{-2}), the net migration (E_i : emigration – immigration, year^{-1}), the biomass accumulation (BA_i , year^{-1}), and its natural mortality ($1 - EE_i$). EE corresponds to the Ecotrophic Efficiency or the proportion of biomass consumed in the system for each compartment in the system.

$$B\left(\frac{P}{B}\right)_i = \sum_j B_j \left(\frac{Q}{B}\right)_j DC_{ij} + Y_i + E_i + BA_i + B_i \left(\frac{P}{B}\right)_i (1 - EE_i) \quad (1)$$

The second equation [Equation (2)] describes the energy balance within a compartment.

$$Q_i = P_i + R_i + U_i \quad (2)$$

The energy balance of each group in Equation (2) is maintained by assuming that consumption of the i th group (Q_i) is equal to the sum of its production (P_i), respiration (R_i , gC m^{-2}), and excretion of unassimilated food (U_i).

Towards an ecosystem approach concerning offshore wind farms

Sampling and analytical procedures

From 2014 to 2016, sampling was carried out during four surveys. The benthic invertebrates, suprabenthos (or hyperbenthos), demersal fishes, meiofauna, and zooplankton were sampled at two seasons: in March during the winter period and in September during the summer. The different compartments were sampled

during the same week, to limit temporal variability. This sampling strategy was aimed at assessing the initial state of the ecosystem before the installation of the OWF and the status of the different biological compartments taking into account seasonal changes (Figure 2).

Benthic macrofauna

The sampling plan was carried out on 25 stations, 20 located inside and 5 located outside the future OWF, to characterize the benthic communities in its proximity. Five replicates were collected at each station for benthic fauna analysis. Benthic invertebrates were sampled with a 0.1 m^2 Van Veen grab. Sieving was performed on board using a circular 1 mm mesh. The retained material was preserved in 10% formalin until the final treatments in laboratory. Rose Bengal solution was used to facilitate the sorting of organisms from the sediments. The samples were then sorted, and the organisms were identified at the species taxonomic level when feasible. The biomass of each species was determined and expressed in terms of g of Ash-Free Dry Weight (AFDW) per m^2 (loss of weight of dry organisms after 5 h at 500°C).

Demersal fish species and cephalopods

Abundance and biomass data for the demersal fish were collected by sampling with a 3-m beam trawl (utilized by IFREMER). A total of 10 hauls were operated (5 inside and 5 outside the wind farm area) while sailing against the current for a sampling period of 15 min at a speed of approximately 3–3.5 knots (according to the current speed). At each sampling station, large-size invertebrates (mega invertebrates), cephalopods, and fish were

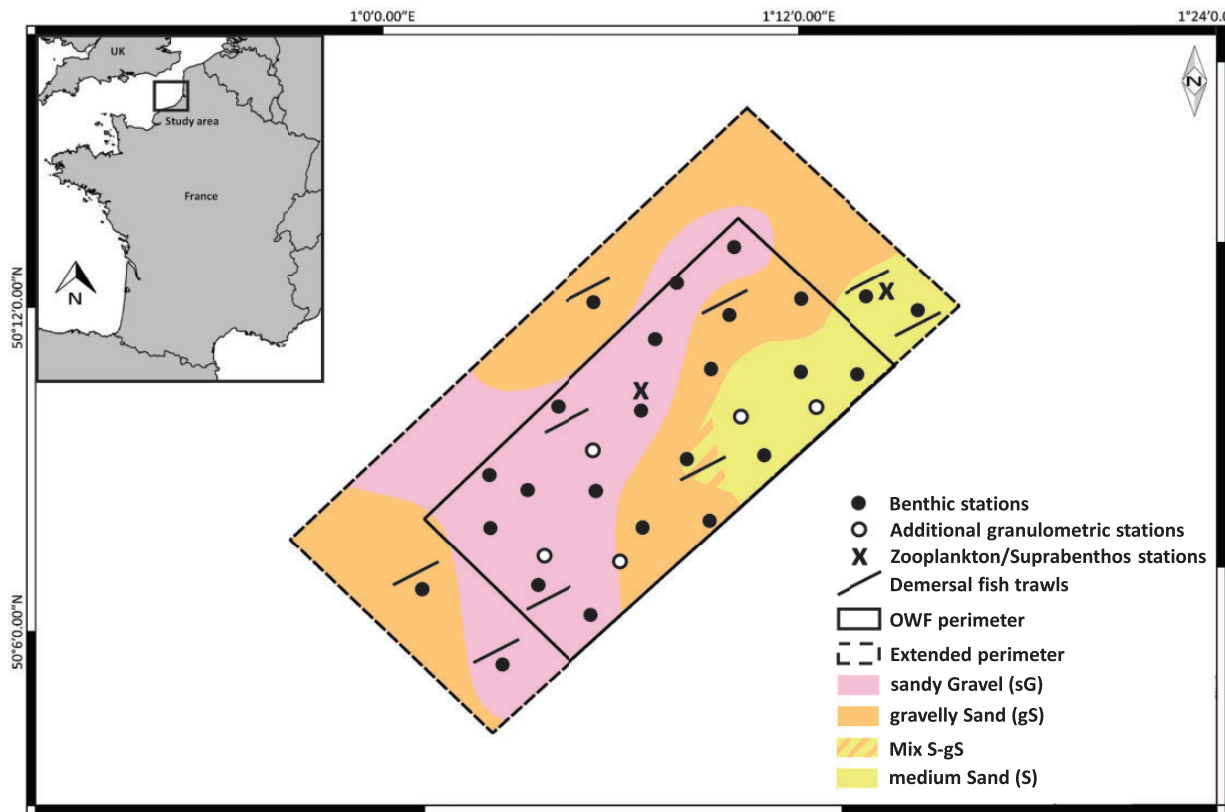


Figure 2. Sampling strategy on the future DLT OWF located in the eastern basin of the EC. For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.

identified, measured, weighed, and fixed with 10% formaldehyde solution for future analysis. The analysis of fish stomach contents is the only way to provide accurate information about the composition of prey species. This analysis allows prey determination at the species level, using hard parts that resist digestion such as crustacean exoskeletons, fish otoliths, and bones (Jackson *et al.*, 2007). Thus, fish stomach content analyses were made at the DLT OWF site to quantify the contribution of benthic prey to the diet of demersal fish species and thus identify the benthic species playing a key role in the trophic web. The biomass of demersal fish individuals was determined based on g of AFDW (loss of weight of dry organisms after 5 h at 500°C). The biomass is given for the area covered by the trawl, which corresponds to the haul length multiplied by the trawl width (3 m).

Meiofauna

The sampling plan was carried out on 25 stations corresponding to the same stations used for the macrofauna. The meiofauna was sampled with a 0.1 m² Van Veen grab and a 3.6-cm diameter sediment core via a hatch on the top of the grab. Sieving was performed using a 1-mm mesh and then a 38- μ m mesh. The samples were preserved in 4% formalin prepared with boiling water and then stored until final treatments in the laboratory. The biomass of each sample corresponds to all the meiobenthic organisms comprised between 1 mm and 38 μ m. Owing to the absence of mud, these organisms could be separated from the sediment by elutriation. The biomass was determined based on g of AFDW per m² (loss of weight of dry organisms after 5 h at 500°C).

Suprabenthos

The suprabenthos is defined as organisms living in the water layer immediately above the seabed, which perform daily vertical migrations and/or seasonal movements at varying distances from the bottom (Brunel *et al.*, 1978). In this study, we consider the suprabenthos *sensu stricto*: peracarids (amphipods, cumaceans, mysids, tanaids, isopods, pycnogonids, leptostracea) and decapods (Dauvin and Vallet, 2006). Here, the suprabenthos is divided into two groups: (i) holosuprabenthos, which corresponds to peracarids, present in the water column adjacent to the seabed at both seasons (summer and winter); and (ii) the merosuprabenthos, which corresponds to individuals present in the column water adjacent to the seabed during a given season (such as decapods larvae, etc.).

Sampling was carried out inside and outside the future wind farm area according to two different sedimentary types. For each sedimentary type, samples were taken during the night and day to study nycthemeral migration. The suprabenthos was sampled using a Macer-GIROQ sledge (Dauvin and Lorgeré, 1989), which consists of four 0.18 m² boxes (0.6 × 0.3 m), designed to filter the water column in four layers above the sea bottom: 0.10–0.40 m (box 1), 0.45–0.75 m (box 2), 0.80–1.10 m (box 3), and 1.15–1.45 m (box 4) (Dauvin *et al.*, 2000). Each box was equipped with a WP2 zooplankton net (500 μ m), including a Tsurimi-Seiki-Kosakusho (TSK) flow meter at its centre to measure the volume of water filtered. The sampling period (i.e. the period during which the sledge was in contact with the seabed) was 15 min at a sledge speed of approximately 1.5 knots.

All sampled organisms were washed, fixed with 10% neutralized formaldehyde, and then transferred to a 70% ethanol

solution. All specimens were sorted, identified and counted under a dissecting microscope to the species level. The species richness corresponds to the total number of taxa found in the four nets, while the abundance and biomass values (AFDW) are standardized to a mean volume of 100 m³ per haul or surface area (m²) corresponding to sledge length multiplied by the sledge width (0.5 m) and the total area of the four nets.

Holozooplankton

The sampling plan was carried out inside and outside the future wind farm at the same stations used for the suprabenthos. The holozooplankton was sampled using a WP2 net (200 µm) with a flow meter at its centre to measure the volume of water filtered. The holozooplankton corresponds to pelagic organisms present during both seasons (winter and summer) in the water column. At each station, four diagonal lines were operated for sampling at a speed of approximately 1 m s⁻¹, two during the day and two during the night. All sampled organisms were washed and sieved on a 100-µm mesh. Then, organisms were identified and sorted into two permanent groups: chaetognaths and copepods. Biomass of each group and sample were determined based on g of AFDW per m².

Defining the model compartments

The functional groups used in this study were defined according to the biological and ecological characteristics of the species, such as their food preference, size, and commercial importance, while also considering data availability. On this basis, we selected 28 groups, 5 of which comprise fish, 15 invertebrates, 1 holozooplankton, 1 merosuprabenthos, 1 primary producer, 1 bacteria, and 1 detritus group. Out of 27 alive functional groups (including consumed and non-consumed categories), 21 were obtained from sampling operations.

Demersal and pelagic fish compartments

Fish were grouped into five functional groups: (i) whiting (*Merlangius merlangus*), which is considered as a piscivore, (ii) the Ammodytidae (e.g. greater sand eel, *Hyperoplus lanceolatus*), (iii) fish benthos feeders, (iv) demersal flatfish, and (v) pelagic planktivorous fish. AFDW biomass was converted to carbon contents using a conversion factor of 0.4 (Elliott and Hemingway, 2002). The consumption to biomass ratio (*Q/B*) and the production to biomass ratio (*P/B*) ratios were taken from Mackinson and Daskalov (2007). The DC was constructed using stomach content analyses.

Invertebrate compartments

Cephalopods

Cuttlefish were grouped into a single functional group: cephalopods. Abundance data (in t km⁻²) for cephalopods were derived from another study focused on the EC (Carpentier *et al.*, 2009). Conversion factors of 0.192 and 0.402 were used to convert wet weights into dry weights and then into carbon content, respectively (Brey, 2001).

Benthic invertebrates

Species were grouped into 15 functional groups, with a special subdivision for 6 of these groups (consumed and non-

consumed): *Branchiostoma lanceolatum*, predators (C & NC), scavengers (C & NC), filter feeders (C & NC), selective deposit feeders (C & NC), non-selective deposit feeders (C & NC), grazers (C & NC), and meiofauna. AFDW biomass was converted to carbon content using a conversion factor of 0.518 (Salonen *et al.*, 1976 in Brey, 2001). *P/B*, *Q/B*, and dietary data were derived from another study focused also on the EC (Garcia, 2010).

Suprabenthos

Species were grouped into two groups: holosuprabenthos and merosuprabenthos. AFDW biomass was converted to carbon contents using a conversion factor of 0.518 (Salonen *et al.*, 1976 in Brey, 2001). *P/B* and *Q/B* and the diet were obtained from another study focused also on the EC (Garcia, 2010).

Zooplankton

Only the holozooplankton were taken into account here, so the merozooplankton corresponds to the merosuprabenthos. AFDW biomass was converted to carbon contents using a conversion factor of 0.4 (Feller and Warwick, 1988). *P/B* ratios were obtained from another study also focused on the EC (Garcia, 2010).

Primary producers, bacteria, and detritus

Data on primary production, bacteria, and detritus were obtained from another study focused also on the EC (Garcia, 2010).

Trophic structure and Ecological Network Analysis

The ENA results presented here were used to characterize the ecosystem state and functioning. The Total System Throughput (T.) measures the size of the ecosystem (Latham, 2006), while Finn's cycling index (FCI) corresponds to the ratio between flows generated by cycling divided by the total system throughput (Finn, 1976). The System Omnivory Index (SOI) measures how the interactions are distributed among trophic levels (Libralato, 2008). The Ascendency (A) integrates system activity (Total System Throughput) with its degree of organization (Average Mutual Information, AMI) (Ulanowicz and Abarca-Arenas, 1997; Ortiz and Wolff, 2002). These above indices are calculated using the network analysis plug-in included in *EwE* (Christensen and Walters, 2004).

Results

The calculated Pedigree index for the model is 0.73. The initial model is not balanced, since there are some ecotrophic efficiencies greater than 1. For instance, biomass and production estimates for the Ammodytidae group are insufficient to support consumption by the whiting. Thus, the biomass of the Ammodytidae is estimated by the model after setting a value of 0.97 for the Ecotrophic Efficiency (biomass proportion consumed in the system for each compartment in the system). The estimated biomass is higher than the input data first entered during model construction. This can be partly explained by the fact that Ammodytidae biomass data were acquired during the day and not during the night.

Biomass and trophic level

The functional groups dominating the biomass are the benthic invertebrates, filter feeders NC (mostly composed of *Glycymeris glycymeris* and the clam *Polititapes rhomboides*), which represent

Table 1. Biomass values and trophic levels (TL) for different compartments at the DLT OWF site.

Compartments	Biomass, gC m ⁻²	TL
Cetaceans	0.0016	4.45
Seals	0.0004	4.33
Cephalopods	0.0161	4.07
Whiting, piscivorous	0.0313	4.16
Fish, planktivorous	0.5570	3.16
Greater sand eel <i>Hyperoplus lanceolatus</i>	0.0511	3.16
Fish, benthos feeders	0.0091	3.79
Fish, flat fish	0.0212	3.34
Benthic inv., predator consumed	0.0789	3.16
Benthic inv., scavenger consumed	0.2369	3.41
Benthic inv., filter feeders consumed	0.2794	2.25
Benthic inv., sDF consumed	0.4249	2.19
Benthic inv., ssDF consumed	0.0761	2.19
Benthic inv., grazer consumed	0.0011	2.00
<i>Amphioxus</i>	0.8340	2.18
Benthic inv., predator not consumed	0.9337	3.10
Benthic inv., scavenger not consumed	0.2899	3.37
Benthic inv., filter feeders not consumed	31.9298	2.23
Benthic inv., sDF not consumed	1.8946	2.19
Benthic inv., ssDF not consumed	2.3412	2.19
Benthic inv., grazer not consumed	0.0129	2.00
Meiofauna	1.3720	2.22
Merosuprabenthos	0.4120	2.51
Holosuprabenthos	0.1131	2.51
Holozooplankton	1.1954	2.15
Bacteria	0.8244	2.00
Phytoplankton	3.1000	

Benthic inv., benthic invertebrate; sDF, surface deposit feeders; ssDF, sub-surface deposit feeders.

70% of the total living biomass of the ecosystem (Table 1). The other major groups of the system are phytoplankton and benthic invertebrates, non-selective deposit feeders NC (mostly composed of the sea urchin *Echinocardium cordatum* and the polychaete *Polygordius lacteus*).

The Trophic Level of functional groups ranged from TL = 1 for primary producers and detritus to a maximum of 4.45 represented by cetaceans (Figure 3; Table 1). The other marine mammals (seals) were ranked just below as top predators in trophic webs with a TL of 4.33. Cephalopods and whiting rank just below with a trophic level of 4.1 (Figure 3; Table 1).

ENA indices

The system is estimated to process 951.3 gC m⁻² year⁻¹ (T.), with 10.8% of the total throughput being recycled (FCI) (Table 2). In addition, the EE (proportion of biomass consumed for each compartment within the system) of detritus is estimated to be 0.6, indicating that more or less all the energy entering this compartment is re-used in the system. The SOI which is a proxy of the food web complexity, yields a value close to 0.20. Finally, the DLT site ecosystem has an Asendency (A) of 1005.8 gC m⁻² an⁻¹.

Discussion

To assess changes in ecosystem structure and functioning in both space and time, this study was undertaken to improve our understanding on the biological compartments prior to construction at the DLT OWF site in order to set up a sampling framework based

on Before After Control Impact (BACI) (Underwood, 1991, 1994; Magurran et al., 2010).

Importance of site-associated data

From a methodological point of view, the model is based on high-quality source data as shown by the high value of the pedigree index compared with the distribution of indices obtained from previous models (Morissette, 2007). In fact, the pedigree index (0.7) is situated at the maximum of the range (0.16 to 0.7) reported in Morissette (2007). This result can be explained by the fact that the biomass data for 21 groups out of 27 were obtained from local, highly replicated and detailed sampling. In addition, the diet compositions of the model fish species are derived from stomach content studies on fish caught at the DLT site. In most cases, Ecopath models are built with biomass data not collected from the study site (same sediment type, depth, and season), using literature data that can induce a bias in the model. In addition, the DC used to build Ecopath models is not always based on the stomach contents of the species of the study site in question, which can also induce a bias in the model and so compromise the Ecopath mass balance results (Plaganyi and Butterworth, 2004). In fact, the diet of marine organisms can vary significantly between individuals of a given species in different areas (Kopp et al., 2015). Such models based on poor data cannot be used for management purposes. Thus, the main strength of the present study is that it is based on the development of an EMR ecosystem approach: (1) using local biomass data from the following compartments: zooplankton, meiofauna, benthos, and demersal fish and 2) taking into account the link between demersal fish and the benthos through stomach content studies. Moreover, stomach content studies allow us to identify the benthos species that are either consumed or not consumed. The results show that the group with the highest biomass is the “benthic invertebrates, Non-Consumed filter feeders”, suggesting that this group could act as a trophic dead end (cul-de-sac) for the fish but participate in the recycling of energy flow as suspension feeders. In fact, there is a dual problem of accessibility and bivalve size in comparison with the sizes of fish living on the study site. Nevertheless, certain bivalves can be consumed by predators such as *Asterias rubens*. In addition, the Non-Consumed filter feeders play a role in the trophic web through their consumption of phytoplankton.

ENA explanation

The values of FCI and the EE (the percentage of production consumed by a predator) for detritus suggests a detritus-based trophic web with detritus acting as a source of food for the bivalves. Concerning the SOI, which is an indicator of the food web complexity (Libralato, 2008), the result obtained can be considered as an intermediate value when compared with the distribution of indices for pre-existing models of Northern Europe (0.14–0.36) (Mackinson and Daskalov, 2007).

The need to adopt an ecosystem approach for MRE projects

Environmental impact assessments for the future OWF in France, which consider the sensitivity of each ecological compartments to potential pressures, are still under development. In addition, these studies are conducted compartment by compartment, which does not allow taking into account the ecosystems complexity and dynamics (Raoux et al., 2017, 2018). Thus, OWF

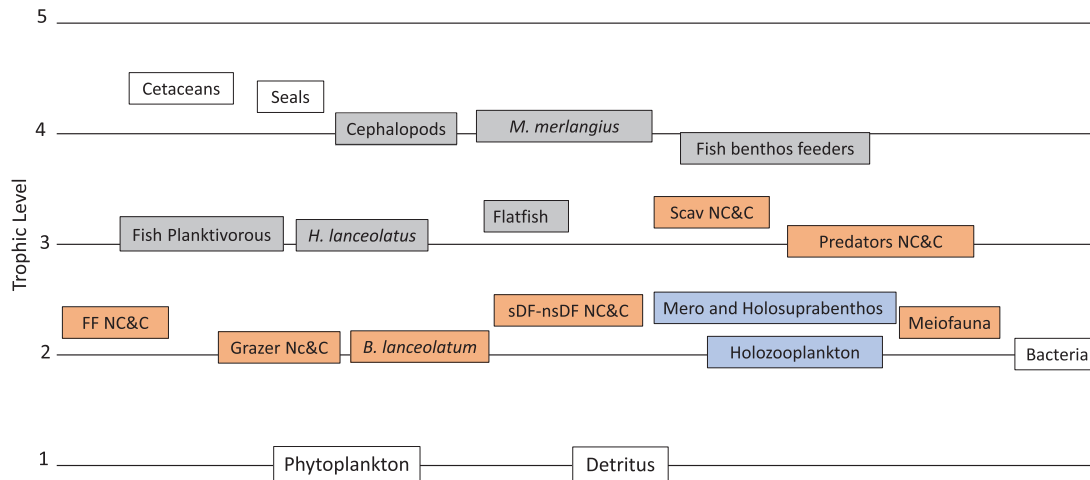


Figure 3. Functional groups of the DLT OWF ecosystem, with trophic level indicated on the y-axis and benthic/pelagic partitioning on the x-axis. White rectangles represent the biomass compartments from the literature and the coloured rectangles represent the biomass compartments from this study. Blue rectangles represent the pelagic invertebrate compartments, orange rectangles represent the benthic invertebrate compartments and the grey rectangles represent the cephalopod and fish compartments. (NC, non-consumed; C, consumed).

Table 2. ENA indices for the DLT OWF model.

ENA	DLT OWF site
T.	951.3
A	1005.8
FCI	10.3
SOI	0.2

The Total System Throughput (T., $gC\ m^{-2}\ year^{-1}$) is calculated as the sum of all flows in the food web. FCI gives the percentage of all flows generated by cycling. The Ascendency (A) is a measure of the system activity (Total System Throughput) linked to its degree of organization (AMI) and is expressed in $gC\ m^{-2}\ year^{-1}$. The SOI is a proxy of the trophic web complexity.

construction effects on the ecosystem structure and function remain unclear (Raoux *et al.*, 2017). In addition, OWFs will integrate into ecosystems already subject to a growing number of natural and anthropogenic disturbances such as granulate extraction, and dumping of spoil sediments (Dauvin, 2012). These can cause changes in the ecosystems functioning and resilience, making them susceptible to changes from one state to another if the cumulative pressures become too frequent and abundant (Knowlton, 1992). Understanding the behaviour of these complex systems is essential in order to anticipate potential changes of states (Hughes *et al.*, 2005) and facilitate the implementation of conservation actions with sustainable development scopes. According to Rosenberg and McLeod (2005), only an ecosystem approach would enable efficient management of the ecosystem. In this context, and as a complementary approach to the traditional impact assessments, the objective of our study was to develop an integrated ecosystem approach using trophic web modelling tools that consider the ecosystem as a whole. Indeed, holistic approaches such as trophic web models are needed as they allow considering, at the same time, the full range of biota size classes, from prokaryote to large top predators. The quantification of the energetic flows between all living organisms in the ecosystem can allow the calculation of numerical indices necessary for the characterization of a system’s functional properties (ENA indices). In fact, as it was illustrated by our case study, ENA indices enable the characterization, among others, of the

recycling, the Omnivory, and the Ascendency (Latham, 2006). In addition, some of these indices have been related to ecological theories about stability, maturity, and stress (Saint-Béat *et al.* 2015) and have been proposed as ecosystem health indicators for describing the food web functioning in different contexts, including the implementation of the Marine Strategy Framework Directive in Europe (Niquil *et al.*, 2014). Our modelling approach can be applied in other OWF implementations and others human activities in European waters. It underlines also the need to have detailed biomass data on a maximum of biological compartment as well as on fish stomach content useful to the modelling of food webs. To do that, it is necessary to persuade the wind farm developers to ensure a long-term monitoring of the new infrastructures impact on the coastal ecosystem taking into account a maximum of biological compartments and estimation of their biomasses, from microbial to top predators and to promote a homogenous ecosystem approach for MRE developments.

The long-term exploitation of wind energy will require long-term monitoring for the different OWFs present along the EC coast. With five OWFs (one in the western basin and four in the eastern basin of the EC), this new activity could provide an observatory at the regional scale that can detect the potential global changes or introduction and/or geographic dispersal of marine species. The development of OWFs corresponds to a new human activity along the French coast, for a period of operation of 30 years. Thus, it provides an occasion to promote a holistic approach to MREs.

The foundation of the wind turbine, as well as the presence of scour protection and unburied cables, will favour the colonization of hard substrates by many species (Wilhelmsson *et al.*, 2006). This reef effect will create a habitat heterogeneity with the creation of a hard substrate on soft substrates, which may lead to species competition for space and resources (Wilhelmsson *et al.*, 2006, 2010; Wilhelmsson and Malm, 2008). OWF development may locally threaten sessile species with a small geographic range, low turnover or recolonization capacity (OSPAR, 2008), as well as engineering species (Di Carlo and Kenworthy, 2008). Thus, the baseline of the DLT OWF site will allow us to monitor the evolution of the ecosystem (functioning, structure, and resilience) after

construction of the OWF. In fact, it is necessary to maintain this ecosystem approach during operational phases to improve our understanding of the behaviour of a given ecosystem. This would allow us to anticipate potential changes of ecosystem states, and implement conservation actions in a sustainable manner.

Conclusion

In the context of OWF development in France, an Ecopath model of the future DLT OWF site was built to characterize the structure and functioning of this ecosystem. The main ecosystem characteristics reflected by ENA indices show that the trophic web is most likely detritus based and that the ecosystem biomass is dominated by “Non-Consumed benthic invertebrates”, which could act as a trophic dead end or cul-de-sac for fish due to the size of these filter feeders against size of sampling fish in the DLT area. Our study highlights the importance of adopting an ecosystem approach for MRE based on local data taking into account the link between demersal fish and the benthos through fish stomach content studies. Moreover, an ecosystem approach needs to be maintained throughout the operational phases of the OWF. Such an approach should be adopted for all future wind farms along the French coast.

Acknowledgments

This study forms part of the doctoral research work of J.P. Pezy, funded by the ANRT (Eoliennes en mer Dieppe-Le Tréport and the French State). We also acknowledge, for their help in the data sampling, the captain and the crew of the Oceanographic Vessel “Celtic Warrior”. The authors are also grateful to M. Carpenter for revising the English style and grammar.

References

- Bailey, H., Brookes, K. L., and Thompson, P. M. 2014. Assessing environmental impacts of offshore wind farms: lessons learned and recommendations for the future. *Aquatic Biosystems*, 10: 8.
- Boehlert, G. W., and Gill, A. B. 2010. Environmental and ecological effects of ocean renewable energy development: a current synthesis. *Oceanography*, 23: 68–81.
- Borger, T., Hattam, C., Burdon, D., Atkins, J. P., and Austen, M. C. 2014. Valuing conservation benefits of an offshore marine protected area. *Ecological Economics*, 108: 229–241.
- Brey, T. 2001. Population Dynamics in Benthic Invertebrates. A Virtual Handbook. <http://thomas-brey.de/science/virtualhandbook>.
- Brunel, P., Besner, M., Messier, D., Poirier, L., Granger, D., and Weinstein, M. 1978. Le traîneau suprabenthique MACER-GIROQ: appareil amélioré pour l'échantillonnage quantitatif étagé de la petite faune nageuse au voisinage du fond. *International Review of Hydrobiology*, 63: 815–829.
- Carpentier, A., Martin, C. C., and Vaz, S. 2009. Channel Habitat Atlas for marine Resource Management, final report/Atlas des habitats des ressources marines de la Manche orientale, rapport final (CHARM phase II). Ifremer, Boulogne-sur-Mer. 626 pp.
- Christensen, V., and Walters, C. J. 2004. Ecopath with Ecosim: methods, capabilities and limitations. *Ecological Modelling*, 172: 109–139.
- Christensen, V., Walters, C. J., Pauly, D., and Forrest, R. 2008. Ecopath with Ecosim Version 6: User Guide. Fisheries Centre, University of British Columbia, Vancouver, BC, Canada.
- Dauvin, J. C. 2012. Are the eastern and western basins of the English Channel two separate ecosystems? *Marine Pollution Bulletin*, 64: 463–471.
- Dauvin, J. C., and Lorgeré, J. C. 1989. Modification du traîneau MACER-GIROQ pour l'amélioration de l'échantillonnage quantitatif étagé de la faune suprabenthique. *Journal de Recherche Oceanographique*, 14: 65–67.
- Dauvin, J. C., and Vallet, C. 2006. The near bottom layer as an ecological boundary in marine ecosystems: diversity, taxonomic composition and community definitions. *Hydrobiologia*, 555: 49–58.
- Dauvin, J. C., Vallet, C., Mouny, P., and Zouhri, S. 2000. Main characteristics of the boundary layer macrofauna in the English Channel. *Hydrobiologia*, 426: 139–156.
- Di Carlo, G., and Kenworthy, W. J. 2008. Evaluation of aboveground and belowground biomass recovery in physically disturbed seagrass beds. *Oecologia*, 158: 285–298.
- Feller, R. J., and Warwick, R. M. 1988. Energetics. *In* Introduction to the Study of Meiofauna, pp. 181–196. Ed. by R.P. Higgings, and H. Thiel. Smithsonian Institution Press, Washington. 488 pp.
- Elliott, M., and Hemingway, K. L. 2002. Fishes in estuaries. Blackwell Science, London. 627 p.
- Finn, J. T. 1976. Measures of ecosystem structure and function derived from analysis of flows. *Journal of Theoretical Biology*, 56: 363–380.
- Garcia, C. 2010. Approche fonctionnelle des communautés benthiques du bassin oriental de la Manche et du sud de mer du Nord. Thèse de Doctorat de l'Université de Lille Nord de France, 396 pp.
- Halpern, B. S., McLeod, K. L., Rosenberg, A. A., and Crowder, L. B. 2008. Managing for cumulative impacts in ecosystem-based management through ocean zoning. *Ocean & Coastal Management*, 51: 203–211.
- Hughes, T. P., Bellwood, D. R., Folke, C., Steneck, R. S., and Wilson, J. 2005. New paradigms for supporting the resilience of marine ecosystems. *Trends in Ecology and Evolution*, 20: 380–386.
- Jackson, G. D., Bustamante, P., Cherel, Y., Fulton, E. A., Grist, E. P. M., Jackson, C. H., Nichols, P. D. *et al.* 2007. Applying new tools to cephalopod trophic dynamics and ecology: perspectives from the Southern Ocean Cephalopod Workshop. *Reviews in Fish Biology and Fisheries*, 17: 79–99.
- Knowlton, N. 1992. Thresholds and multiple stable states in coral reef community dynamics. *American Zoologist*, 32: 674–682.
- Kopp, D., Lefebvre, S., Cachera, M., Villanueva, M. C., and Ernande, B. 2015. Reorganization of a marine trophic network along an inshore–offshore gradient due to stronger pelagic–benthic coupling in coastal areas. *Progress in Oceanography*, 130: 157–171.
- Latham, L. G. 2006. Network flow analysis algorithms. *Ecological Modelling*, 192: 586–600.
- Leonhard, S., Stenberg, C., and Støttrup, J. 2011. Effect of the Horns Rev 1 offshore wind farm on fish communities follow-up seven years after construction. DTU Aqua Rep. 99 pp.
- Leung, D., and Yang, Y. 2012. Wind energy development and its environmental impact: A review. *Renewable and Sustainable Energy Reviews*, 16. doi: 10.1016/j.rser.2011.09.024.
- Libralato, S. 2008. System omnivory index. *Ecological Indicators*, 4: 3472–3477.
- Lindeboom, H. J., Kouwenhoven, H. J., Bergman, M. J. N., Bouma, S., Brasseur, S., Daan, R., Fijn, R. C. *et al.* 2011. Short-term ecological effects of an offshore wind farm in the Dutch coastal zone; a compilation. *Environmental Research Letters*, 6: 035101.
- Mackinson, S., and Daskalov, G. 2007. An ecosystem model of the North Sea to support an ecosystem approach to fisheries management: description and parameterisation. Science Series Technical Report, CEFAS, 142 pp.
- Magurran, A. E., Baillie, S. R., Buckland, S. T., Dick, J. M., Elston, D. A., Scott, E. M., Smith, R. I. *et al.* 2010. Long-term datasets in biodiversity research and monitoring: assessing change in ecological communities through time. *Trends in Ecology & Evolution*, 25: 574–582.
- Morissette, L. 2007. Complexity, cost and quality of ecosystem models and their impact on resilience: a comparative analysis, with emphasis on marine mammals and the Gulf of St. Lawrence

- Zoology. PhD thesis, University of British Columbia, Vancouver, BC, Canada. 278 pp.
- Mueller-Blenkle, C., McGregor, P. K., Gill, A. B., Andersson, M. H., Metcalfe, J., Bendall, V., Sigra, P., Wood, D. T., and Thomsen, F. 2010. Effects of pile-driving noise on the behaviour of marine fish. COWRIE Ref: Fish 06-08.
- Niquil, N., Le Loc'h, F., Tecchio, S., Chaalali, A., Vouriot, P., Mialet, B., Fizzala, X., Féral, J. P., Lamare, S., Dauvin, J. C., and Safi, G. 2014. Ongoing research on ecosystem health indicators for food webs in the MSFD context. Trans-Channel forum proceedings "Science and Governance of the Channel Marine Ecosystem", Caen, France, 14-15.
- Ortiz, M., and Wolff, M. 2002. Trophic models of four benthic communities in Tongoy Bay (Chile): comparative analysis and preliminary assessment of management strategies. *Journal of Experimental Marine Biology and Ecology*, 268: 205–235.
- OSPAR, 2008. Background document on potential problems associated with power cables other than those for oil and gas activities. OSPAR Commission. 50 pp.
- Petersen, J. K., and Malm, T. 2006. Offshore windmill farms: threats to or possibilities for the marine environment. *Ambio*, 35: 75–80.
- Pezy, J. P. 2017. Approche écosystémique d'un futur parc éolien en Manche orientale: exemple du site de Dieppe-Le Tréport. Thèse de Doctorat, Université Caen Normandie. 374 pp.
- Plaganyi, E. E., and Butterworth, D. S. 2004. A critical look at the potential of Ecosim with Ecosim to assist in practical fisheries management. *African Journal of Marine Science*, 26: 261–287.
- Polovina, J. J. 1984. Model of a coral reef ecosystem. I. The ECOPATH model and its application to French Frigate Shoals. *Coral Reefs*, 3: 1–11.
- Raoux, A. 2017. Approche écosystémique des Energies Marines Renouvelables: étude de l'impact sur le réseau trophique de la construction du parc éolien au large de Courseulles-sur-mer et du cumul d'impacts. Thèse de Doctorat, Université Caen Normandie. 251 pp.
- Raoux, A., Dambacher, J. M., Pezy, J. P., Mazé, C., Dauvin, J. C., and Niquil, N. 2018. Assessing cumulative socio-ecological impacts of offshore wind farm development in the Bay of Seine (English Channel). *Marine Policy*, 89: 11–20.
- Raoux, A., Tecchio, S., Pezy, J. P., Degraer, S., Wilhelmsson, D., Cachera, M., Ernande, B. *et al.* 2017. Benthic and fish aggregation inside an offshore wind farm: which effects on the trophic web functioning? *Ecological Indicators*, 72: 33–46.
- Rosenberg, A., and McLeod, K. 2005. Implementing ecosystem-based approaches to management for the conservation of ecosystem services. *Marine Ecological Progress Series*, 300: 270–274.
- Saint-Béat, B., Baird, D., Asmus, H., Asmus, R., Bacher, C., Pacella, S. R., Johnson, G. A. *et al.* 2015. Trophic networks: how do theories link ecosystem structure and functioning to stability properties? A review. *Ecological Indicators*, 52: 458–471.
- Shields, M. A., and Payne, A. I. L., 2014. *Marine Renewable Energy Technology and Environmental Interactions, Humanity and the Sea*. Springer Sciences, Netherlands. 179 pp.
- Ulanowicz, R. E., and Abarca-Arenas, L. G. 1997. An informational synthesis of ecosystem structure and function. *Ecological Modelling*, 95: 1–10.
- Underwood, A. J. 1991. Beyond BACI – experimental designs for detecting human environmental impacts on temporal variations in natural populations. *Australian Journal of Marine and Freshwater Research*, 42: 569–587.
- Underwood, A. J. 1994. On beyond BACI: sampling designs that might reliably detect environmental disturbances. *Ecological Applications*, 4: 3–15.
- Villnäs, A., and Norkko, A. 2011. Benthic diversity gradients and shifting baselines: implications for assessing environmental status. *Ecological Applications*, 21: 2172–2186.
- Wilding, T. A., Gill, A. B., Boon, A., Sheehan, E., Dauvin, J. C., Pezy, J. P., O'Beirn, F. *et al.* 2017. Turning off the DRIP ('Data-rich, information-poor') – rationalizing monitoring with a focus on marine renewable energy developments and the benthos. *Renewables and Sustainable Energy Reviews*, 74: 848–859.
- Wilhelmsson, D., and Malm, T. 2008. Fouling assemblages on offshore wind power plants and adjacent substrata. *Estuarine, Coastal and Shelf Science*, 79: 459–466.
- Wilhelmsson, D., Malm, T., and Öhman, M. C. 2006. The influence of offshore wind power on demersal fish. *ICES Journal of Marine Science*, 63: 775–784.
- Wilhelmsson, D., Malm, T., and Thompson, T. 2010. *Greening Blue Energy: Identifying and Managing the Biodiversity Risks and Opportunities of Off Shore Renewable Energy*. IUCN, Gland, Switzerland. 90 pp.

Handling editor: Steven Degraer