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**Isotopic analyses, a good tool to validate models in the context of Marine
Renewable Energy development and cumulative impacts**

Aurore Raoux^{1,2*}, Jean-Philippe Pezy², Bruno Ernande³, Nathalie Niquil¹, Jean-Claude
Dauvin², Karine Grangeré¹

¹ Unité Biologie des Organismes et Ecosystèmes Aquatiques (BOREA), MNHN, CNRS, IRD,
Sorbonne Université, Université de Caen Normandie, Université des Antilles, CS 14032,
14000 CAEN, France

² Normandie Univ, UNICAEN, UNIROUEN, Laboratoire Morphodynamique Continentale et
Côtière, CNRS UMR 6143 M2C, 24 rue des Tilleuls, 14000 Caen, France

³ IFREMER, Laboratoire Ressources Halieutiques, 150 quai Gambetta, BP 699, 62321
Boulogne sur Mer, France

** Corresponding author: aurore.raoux@unicaen.fr*

ABSTRACT

As part of the energy transition, the French government is planning Offshore Wind Farm (OWF) constructions in the next decades. An integrated ecosystem approach of two future OWF sites of the Eastern English Channel (Courseulles-sur-mer and Dieppe-Le Tréport) was developed to model the marine ecosystems before the OWF implementation. Such ecosystem models allow simulating the possible reef and reserve effects associated to the presence of the farm, and to character the overall changes in the food-web functioning. This holistic view of OWF effects could be replicated on other sites and form the basis of an ecosystem based management of marine renewable energies. However, to use these models for management purpose, they need to be validated. In order to do so, stable isotope ratios of nitrogen were used for determining the accuracy of the effective trophic levels computed in these two models. Results showed that trophic levels estimated by the two models were consistent with the trophic levels estimated by the independent isotopic data. In the context of OWF development and cumulative impacts analysis, this step of validation of the models is essential for developing their use by management actors and policy makers.

KEYWORDS: Offshore Wind Farm, Ecopath with Ecosim, Trophic Level, Isotopic Nitrogen analysis, English Channel

1 Introduction

Clean renewable energy from Marine Renewable Energy (MRE) has increased steadily throughout the Northern Europe as a goal of reducing future carbon emissions. Of all MRE, Offshore Wind Farms (OWF) has seen consistent growth in capacity and it is by far the most technically advanced of all MRE (Leung and Yang, 2012; Willstead et al., 2018). The French government are currently planning the construction of three OWFs in the eastern basin of the English Channel (Fécamp, Dieppe-Le Tréport and Courseulles-sur-mer). Environmental impact studies following the BACI protocol are performed before the implementation, during the exploitation, and during the dismantlement, to investigate the impacts of these OWF structures on the surrounding marine ecosystems (Wilding et al., 2017). All these studies provide a large amount of data on environmental effects at the species level. However, as Raoux et al. (2017, 2019) pointed out, while these studies endeavour to consider the sensitivity of some ecological compartments (benthos, fish and marine mammals), they fail to do so if not taking into account the trophic links between the compartments. Thus, at the beginning of the OWF development in France, a research gap has been identified in the uncertainties around the potential impact of OWF installation on the trophic web structure and ecosystem functioning (Bailey et al., 2014). In this context, and as a complementary approach to the traditional impact assessments, there is a need to adopt an integrated ecosystem approach through modelling tools that consider the ecosystem as a whole (Raoux et al., 2017, 2019; Pezy et al., 2018).

Quantitative 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. Among these different existing approaches, the Ecopath with Ecosim (EwE) model has been intensively used and developed over the last three decades (Colléter et al., 2013, 2015) and was recognised by the US National Oceanographic and Atmospheric Administration (NOAA) as one of the ten biggest scientific breakthroughs in its 200 year history (Heymans et al., 2016). The EwE model has been used to analyse the impact of a wide range of anthropogenic perturbations on marine ecosystems through the world such as fishing activities (Heymans and Tomczak, 2016), invasion of species (Langseth et al., 2012), dumping operations (Pezy et al., 2017, 2018a), and infrastructure development (Tecchio et al., 2015). With the planning of

the construction of three OWF, the English Channel has become a hotspot for future OWF development. Moreover, this area is one of the most human impacted area, with some of the highest cumulative impacts in the worldwide ocean (Halpern et al., 2008). Recently, Raoux et al. (2017, 2019) used the EwE approach in the context of OWF construction and cumulative impacts in the eastern part of the English Channel. In their study, Raoux et al (2017, 2019) built a food web model at the site of Courseulles-sur-mer OWF construction to describe the situation “before” the construction of the wind farm and then performed simulation to analyse the potential impact of the reef and reserve effect (spatial restriction of fisheries for safety measures) on the ecosystem structure and functioning.

This holistic view of the OWF effect on the ecosystem through trophic web modelling could be replicated on other potential installation sites, and used to analyse the long-term reef and reserve effects in the context of global change. Using quantitative modelling to assess cumulative impacts during OWF development would allow bringing new knowledge to policy makers. Results could also be used by the project owner and the “Design Office”, which have the responsibility to provide the Environmental Impact Assessment. However, the main drawback that could limit the utilisation of EwE models for management purposes are that model validations are rare, which may lead to a problem of “quality assurance” (Heymans et al., 2016). In addition, this problem of “quality assurance” is compounded by the fact that the data used (biomass and stomach content) to build Ecopath models are often not collected from the study site (same sediment type, depth, and season) but from literature which can induce a bias in the model and so compromise the quality of the results of the model (Plagányi and Butterworth, 2004).

Thus, in the context of OWF development and cumulative impacts there is an urgent need to validate models in order to test the predictions. For that, the validation should be an integral part of the modelling process. However, it is necessary to have an independent data set with information not use in the model inputs.

Concerning EwE models, special attention has been given to the validation of functional compartments trophic levels calculated by Ecopath using Stable Isotope Analyses (SIA) (Kline et Pauly, 1998; Pauly et al., 1998; Polunin et Pinnegar, 2000; Dame and Christian, 2008;

Nilsen et al., 2008; Milessi et al., 2010; Navarro et al., 2011; Lassalle et al., 2014; Deehr et al., 2014). Compared to modelling, SIA constitutes a completely different approach to analyse trophic interactions within ecosystems (Fry et al., 1987) and the $\delta^{15}\text{N}$ value can be used to describe the trophic level of consumers (Hobson et Welch 1992; Le Loc'h et al., 2008). This approach is based on the fact that there is a relationship between the isotopic signatures of consumer and their prey (Peterson and Fry, 1987). When a predator consumes its prey, there is an energy transfer and the isotopic nitrogen values show a predictable increase in the isotopic ratio between 0.5 and 5‰ across trophic levels (Minagawa and Wada, 1984; Post, 2002). Despite the wide use of both EwE and SIA, few attempts have been made to compare these methods (Dame and Christian, 2008, Lasalle et al., 2014).

In this study, EwE models from two locations were assessed: the site of the future Courseulles-sur-mer Offshore Wind Farm (CSM OWF) (Raoux et al., 2017), and the future Dieppe-Le-Tréport (DLT OWF) (Pezy et al., 2018 b; Pezy et al., in press). The trophic levels estimated from these two models were compared using trophic levels estimated from independent nitrogen isotope data.

2 Materials and methods

2.1 Study area

The eastern English Channel (eEC), where the CSM OWF and DLT OWF will be built in the next years, is a shallow epi-continental area located between France and England. It is about 35 000 km² and it is delimited by the Cotentin peninsula in the west and the Dover Strait to the east. The maximum water depth never exceeds 70 m in the trench running through the centre of the English Channel (Carpentier et al., 2009). The tidal range is greater than 5 m on the French coast but is closer to 2 m on the English side (Dauvin and Lozachmeur, 2006). These currents play an essential role in the sediment (Larsonneur et al., 1982) and benthic communities (Cabioch and Gentil, 1975; Dauvin, 2015) distributions. The eEC is also the focus of many human activities such as transportation, fishing, sediment deposit, and sediment extraction (Dauvin and Lozachmeur, 2006; Dauvin, 2012) and is considered by Halpern et al. (2008) as one of the most anthropized sea of the world.

2.1.1 The Courseulles-sur-mer Offshore Wind Farm (CSM OWF) project

The CSM OWF will be located 10 to 16 km offshore from the Calvados coast between 31 and 34 m of depth. It will be located on the coarse sand and gravels benthic communities of the Bay of Seine (Fig. 1). The OWF will represent an area of 50 km². A total of 75 turbines (each 6 MW) capable of producing 450 MW will be installed by Eoliennes Offshore du Calvados” (EOC) in the next years. The monopile foundation will have a diameter of 7 m and will be driven into the seabed. 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.

2.1.2 Dieppe-Le Tréport Offshore Wind Farm (DLT OWF) project

The prime contractor of the project is “Eoliennes en mer Dieppe-Le Tréport”, a subsidiary of Engie (formerly named GDF Suez). The proposed OWF will be located at distances of 15.5 km and 17 km offshore from the coast off Le Tréport and Dieppe, respectively (Figure 1). The water depth ranges from 12 to 25 m. The OWF 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 foundations are composed of jacket structures. As for the CSM OWF, 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 to the continent.

<Figure 1>

2.2 Data collection

2.2.1 Trophic modelling approach

2.2.1.1 The pre-existing Ecopath models

In order to gain further knowledge on the ecosystem structure and functioning before the construction of the CSM OWF and the DLT OWF, two Ecopath models (Polovina, 1984; Christensen and Walters, 2004; Christensen et al., 2008) describing the initial state of the both sites were built (Raoux et al., 2017; Pezy et al., 2018. Ecopath is a mass-balance (i.e. neglecting year-to year changes in biomass compared to flow values) single-solution model

(i.e. returning only one value per flow) in which the ecosystem is represented by functional groups, which are composed of species or group of species with ecological or biological similarities. Each functional group 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 matrix diet which represents the interactions between predators and prey in the ecosystem.

The parameterization of an Ecopath model is based on two master equations. The first one is the balance equation, which describe the production term:

$$\text{Production} = \text{Catch} + \text{Predation} + \text{Biomass accumulation} + \text{Net migration} + \text{Other mortality}$$

The second equation ensures energy balance for each trophic group:

$$\text{Consumption} = \text{Production} + \text{Respiration} + \text{Unassimilated food}$$

The CSM Ecopath model consisted of 37 compartments ranging from primary producers (phytoplankton) to top predators (sea birds and marine mammals). In the model, fish were grouped into 18 groups; 6 functional groups (sharks and rays, gurnards, piscivorous, planktivorous, benthos feeders, and other flatfish) and 12 single species compartments (mackerel (*Scomber scombrus*), European seabass (*Dicentrarchus labrax*), Atlantic cod (*Gadus morhua*), whiting (*Merlangius merlangus*), Atlantic horse mackerel (*Trachurus trachurus*), pouting (*Trisopterus luscus*), poor cod (*Trisopterus minutus*), European pilchard (*Sardina pilchardus*), European sprat (*Sprattus sprattus*) sea bream (*Spondyliosoma cantharus*), common sole (*Solea solea*), and European plaice (*Pleuronectes platessa*)) (Raoux et al., 2017). Benthic invertebrates were grouped into four functional groups: predators, filter feeders, bivalves and deposit feeders (Raoux et al., 2017). The benthic cephalopods, benthopelagic cephalopod, suprabenthos, zooplankton, bacteria, primary producer, and detritus were represented by one group respectively. The seabirds were grouped into two groups and marine mammals into four groups. 26 of the organism groups were obtained from sampling studies.

The DLT Ecopath model was composed of 28 compartments from primary producers to marine mammals. Fish were grouped into five groups (whiting, Ammodytidae, fish benthos feeders, demersal flatfish, and pelagic planktivorous fish). Benthic invertebrates were

divided into 13 groups with a special subdivision for 6 of these groups (consumed (C) and non consumed (NC)): *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). The consumed organisms were those where the species were found in the fish stomach contents. The cetaceans, seals, cephalopods, meiofauna Merosuprabenthos, holosuprabenthos, holozooplankton, bacteria, primary producer and detritus were represented by one group respectively. 21 of the functional groups were obtained from sampling operations (following methods described by Pezy et al., 2019).

The calculated Pedigree indices for these two models were 0.52 and 0.73 for the CSM Ecopath Model and the DLT Ecopath model respectively. These values were calculated from the EwE pedigree routine which quantifies the input parameter uncertainties (Christensen and Walters, 2004).

A full description and sources of information of the input and output parameters of the CSM and the DLT Ecopath models can be found in Raoux et al. (2017) and Pezy et al. (2018) respectively.

2.2.1.2 Trophic Level in Ecopath

The Trophic Level (TL) defines the trophic position of an organism within the food web (Lindeman, 1942). Ecopath allows calculating a TL for each functional group in the model.

The formula corresponds to the weighted average of the trophic levels of the prey of a functional group, with primary producers and non-living material set at a level of 1:

$$TL_j = 1 + \sum_{i=1}^N DC_{ji} TL_i \text{ (Eq. 1)}$$

where DC_{ji} is the fraction of the prey i in the diet of the predator j .

2.2.2 *Stable isotopes analyses*

2.2.2.1 *Sampling and samples preparation*

Cephalopods were sampled from the GOV (“Grande ouverture Verticale”= high opening) bottom trawl survey in the Bay of Seine during the CGFS (Channel Ground Fish Survey) survey conducted in October 2009 by IFREMER. Other samples of benthic invertebrates were sampled at the site of implantation of the future CSM OWF in October 2015 using a Rallier du Baty dredge.

Fish and benthic invertebrates were collected at the site of the implantation of the future DLT OWF in February 2016 using a 3 m beam trawl and with a 0.1 m² Van Veen grab respectively.

130 samples corresponding to nine species were analysed for nitrogen stable isotope to study the trophic level of each species at the CSM future OWF and 102 samples corresponding to 16 species at the DLT future OWF (Table 1 and Table 2). All samples were kept frozen until processed in laboratory. The preparation of the tissue for stable isotopic analyses varied according to taxon. However, we chose to collect the muscle because the nitrogen isotopic signature of the latter is less variable over time than tissues from other organs such as the digestive gland or heart tissue (Pinnegar and Polunin, 1999). For bivalves, a sample was taken for the abductor muscle, for cephalopods, a mantle tissue was taken, for crabs a mantle tissue was taken from chelipeds, for ophiurid a sample was taken from the arms, for sea urchins, the muscle of the lantern of Aristotle was used and for fish a sample of white dorsal muscle was taken.

<Table 1>

<Table 2>

All samples collected were oven dried (60°C for 48 h) and then each dried sample was ground into a homogeneous powder using a mixer mill. Approximately 1 mg of powder was weighed into small tin cups. Isotopic analyses were performed with elemental analyser EA 3000 (EuroVector) coupled with a mass spectrometer (Pinnegar and Polunin, 1999) at the

Plateau d'Analyse d'Isotopie de Normandie (PLATin, SFR ICORE) at the University of Caen Normandie (France) and at the LEMAR laboratory (Brest University, France).

Results are expressed in standard δ notation based on international standards [N_2 for $\delta^{15}N$ (Peterson and Fry, 1987)] according to the following equation:

$$\delta X (\text{‰}) = [(R_{\text{sample}} / R_{\text{standard}}) - 1] \times 1000 \text{ (Eq. 2)}$$

where $X = ^{15}N$, $R_{\text{sample}} = ^{15}N/^{14}N$ in sample and $R_{\text{standard}} = ^{15}N/^{14}N$ in international standards.

2.2.2.2 Trophic level calculation from the isotopic analyses

The trophic level of each species was calculated according to the following equation (Minagawa et Wada, 1984):

$$TL = \lambda + (\delta^{15}N_{\text{organism}} - \delta^{15}N_{\text{base}}) / \Delta N \text{ (Eq. 3)}$$

where $\delta^{15}N_{\text{organism}}$ is the isotopic signatures of the organism, ΔN is the average $\delta^{15}N$ enrichment from prey to predator assumed to be 3.4‰ (Minagawa et Wada, 1984) and the $\delta^{15}N_{\text{base}}$ is the mean value of a species close to the trophic web base chosen as trophic baseline and λ its trophic level. In our study, the benthic filter feeders, *Aequipecten opercularis* (Linnaeus, 1758), was used to establish isotopic baseline as this species integrates the short term spatial and temporal variability displayed by primary producers (Jennings and Warr, 2003). Thus, $\lambda = 2.18$.

It is worth to note that published isotopic analyses (Trophic Levels) of 23 taxa (zooplankton, benthic invertebrates and fish) sampled in October 2009 in the Bay of Seine were also used (Kopp et al., 2015) for the comparison of the TLs calculated from the CSM Ecopath with those calculated from SIA. Isotopic results are detailed for all species sampled in the Bay of Seine in Kopp et al. (2015).

2.3 Comparison between Ecopath mass balanced model and isotope results

The correlation between TL derived from SIA and those derived from the two Ecopath models were tested using the Spearman-rank correlation coefficient test (Zar, 1984). In the case of multi-species functional groups, the TL of the functional group was calculated as the mean TL of the species within the functional group weighted by their biomass proportions. The same method was applied to calculate the TL of multi-species functional groups from SIA.

3 Results

Results showed that the TLs estimated by the both Ecopath mass balanced models (CSM and DLT) were highly and positively correlated with the SIA (CSM: $r^2_{\text{Spearman}} = 0,79$; $p < 0.0001$ (Figure 2; Table 1); DLT: $r^2_{\text{Spearman}} = 0,95$; $p < 0.0004$ (Figure 3 ; Table 2).

<Figure 2>

<Table 1>

In fact, the Figure 2 revealed that half of the points are located either above or below the 1:1 line of perfect agreement (or first bisector) in the range of TLs being studied. This result suggested that the EwE approach tended to slightly underestimate the trophic positions of the functional groups located above the first bisector and to slightly over estimate the trophic positions of the functional groups located under the first bisector. For instance, according to the CSM Ecopath model, the zooplankton presented a TL of 2 whereas the zooplankton TL derived from SIA revealed a TL of 2.56. On the same line, the EwE approach also underestimated the TLs of the following groups: Benthopelagic cephalopods, mackerel, European sea bass, Atlantic horse mackerel, gurnard, European pilchard, European sprat, sea bream and Benthic inv, predators. However, the EwE approach overestimated the TL of the following functional groups: Benthic cephalopods, sharks and rays, Atlantic cod (*Gadus morhua*), whiting (*Merlangius merlangus*), pouting (*Trisopterus luscus*), poor cod (*Trisopterus minutus*), fish benthos feeders, other flat fish, King Scallops (*Pecten maximus*) and Benthic inv, deposit feeders. In addition, perfect agreements were found for two compartments, one mono-species functional group corresponding to the plaice (*Pleuronectes platessa*) (17), and a multi-species functional group corresponding to the bivalves (20) (Figure 2; Table 1). Finally, the cephalopods showed the biggest difference

between the TLs estimated by both methods. This difference did not exceed 20% for any of the studied trophic compartment.

<Figure 3>

<Table 2>

Concerning the comparison of the TLs derived from the DLT Ecopath model and those from SIA, the results showed that most of the points were located under the first bisector suggested that the EwE approach tended to slightly overestimate the trophic positions of the functional groups located under the first bisector (Figure 3; Table 2). For instance, the EwE approach overestimated the TLs of the following functional groups: fish benthos feeders, flat fish, benthic invertebrates not consumed and the benthic invertebrates filter feeders not consumed. However, the EwE approach underestimated the TL of the benthic invertebrates scavengers not consumed. In fact, according to the DLT Ecopath model, this functional group presented a TL of 3.36 whereas its TL derived from SIA revealed a TL of 3.2 (Figure 3, Table 2). However, it is worth noting that these results were based on a low number of trophic groups and so they should be taken with precaution.

4 Discussion

4.1 Importance of local data

The objective of the present study was to evaluate the CSM OWF model and the DLT OWF model through the comparison of the TLs computed by these two models with TLs estimated from independent nitrogen isotope data. Results showed that TLs estimated by the Ecopath mass balanced models were consistent with the TLs estimated by the SIA as an independent test of validity. These results suggested that diet data use in the CSM Ecopath model and in DLT Ecopath were of good quality and highlighted the importance to build model based on high-quality source data. In addition, these results are consistent by the high value of the pedigree index of these two Ecopath models. In fact the pedigree index of the CSM Ecopath model (0.5) and of the DLT Ecopath model (0.7) are situated at the maximum of the range (0.16 to 0.7) reported in Morissette (2007). These pedigree high values can be explained by the fact that most of functional group biomass data, in the two models, were obtained from local, highly replicated and detailed sampling (Raoux et al., 2017; Pezy et al., 2018). In addition, the fish diet compositions of the models were derived from local stomach content

studies (Raoux et al., 2017; Pezy et al., 2018). It is worth to note that in most cases, Ecopath models are built with biomass data not collected from the study site but using literature data that can induce a bias in the model. This is mainly due to the lack of data in many ecosystem studies. In addition, the diet matrix used to build Ecopath models is not always based on local stomach contents which can also induce a bias in the model and so can compromise the Ecopath mass balance results (Plagányi and Butterworth, 2004). In fact, as pointed out by Kopp et al. (2015), marine organism diet can vary significantly between individuals of a given species in different areas. In additions, there can be significant temporal and spatial variability in the diet of marine organisms or during ontogenetic shift (Grangeré et al., 2012, Gaudron et al., 2016). Thus, highlight the importance of site-associated data in the construction of the EwE model.

4.2 Comparison of the TLs derived from the EwE approach and SIA

The popular utilisation of both the Ecopath mass balanced model and SIA make their comparison and cross validation highly relevant. However, to date very few attempts have been made to validate the Ecopath model results with results from SIA. Among these few attempts, Kline and Pauly (1998), Nilsen et al. (2008), Milessi et al. (2010), Lasalle et al. (2014), Dheer et al. (2014) also found a positive correlation between the TLs estimated by the EwE and the SIA approach. However, Dheer et al. (2014) highlighted that the EwE approach tended to estimate incorrectly the TLs of the detritivorous species. This can be explained by the fact that in Ecopath, a TL of 1 is automatically associated to detritus leading to an underestimation of TLs compared to SIA (Nilsen et al., 2008; Navarro et al., 2011).

Other potential factors could partly explain the deviation observed between the TLs derived from the two Ecopath models and those calculated from SIA. First, the temporal scale associated with the methods. In Ecopath, TLs are commonly calculated based on data from stomach content analyses, which only records what a predator has eaten recently, whereas the SIA provide information on diet integrated over a period of several weeks or months (Stowasser 2006). In this perspective, collecting muscle tissue is important since nitrogen isotopic signature has less temporal variability than tissue from other organs such as the digestive gland (Pinnegar and Polunin, 1999). This is what we did in this study. A second potential source of discrepancy between the two approaches could be the use of a standard

isotopic fractionation values (in the calculation of TLs with the SIA approach). The value of 3.4‰ used in this study is considered as a robust average for $\delta^{15}\text{N}$ isotopic fractionation (Minagawa and Wada, 1984) but it can range from 0.5 to 5.5‰ (Post, 2002). The fractionation and isotopic signature can also vary among and within species, and at different stages of development according to their growth rate (DeNiro and Epstein 1978, 1981; Weidel et al., 2011), and as a function of the sea temperature (Barnes et al., 2007). Moreover, feeding differences and fish migration between locations can also cause variation in $\delta^{15}\text{N}$ values (Deegan and Garritt, 1997). Finally, the aggregation of species within compartments in the Ecopath models, and the resulting composition of the compartments can influence the TL calculation, and could partly explain the deviation between the two approaches. In this study, aggregation and definition of the functional groups were based on the biological and ecological characteristics of the species, such as their food preference, size, and commercial importance, as well as on data availability (Raoux et al., 2017; Pezy et al., 2018). However, the difference observed in the zooplankton TL could be explained by the fact that the CSM Ecopath model has only one compartment of zooplankton including both prey (mesozooplankton) and predators (copepods), whereas SIA were made only on copepods, which have a higher trophic level than mesozooplankton. Moreover, the SIA were done on copepods that had not undergone fasting, which could introduce a bias in isotopic analysis. Finally, the differences observed in the cephalopods TLs could be explained by the fact that the prey species with a low trophic level are often digested more rapidly than prey species of higher trophic levels, and would therefore be underestimated during stomach analyses. Finally, it is worth noting that the results from the DLT

5 Conclusion

In the context of the energy transition, the French government is planning the construction of OWFs in the next decade. Several studies have been done to document the environmental conditions and ecosystem functioning in selected sites before the OWF construction. However, these studies endeavor to consider the sensitivity of some ecological compartments, but they fail in taking into account the trophic links between the compartments. Thus, there is lacking holistic studies on the effects of the OWF constructions and operation. Trophic web models appear to be a good complementary approach to the traditional OWF impact assessments. They allow to consider the ecosystem as a whole, and

to simulate the effects of the OWF construction on the ecosystem structure and functioning (Raoux et al., 2017, 2019; Pezy et al., 2018b). This holistic view of the OWF effects on the ecosystem could with advantage be replicated on other site in the English Channel, and a potentially useful to analyses the long-term OWF effects in the context of climate change (Pezy et al., 2018b). In order to use these models for management purposes, there is an urgent need to validate these models to better simulate the potential impacts associated with this OWF development. In this study, we used isotope ratios of nitrogen as a validation tool for determining the trophic levels accuracy computed by two Ecopath models that describe the ecosystem before the implantation of two futures OWF (CSM and DLT OWF). Results showed a good correlation between Ecopath-calculated trophic levels of the two ecosystems and the $\delta^{15}\text{N}$ values. These results indicated that the diet data used for the two Ecopath models were of good quality. However, as mentioned below these results must be taken with precaution as they are based on only two cases and the results from DLT case were based on a low number of trophic groups. Nonetheless, the SIA seems to be a good independent tool to validate Ecopath models. SIA could allow to quantify the uncertainty associated with the diets of the different compartments, and could complement the result of the pedigree index already in use in Ecopath. Ideally, isotopic analyses should be done before the model construction, and they could be used to define the model trophic compartments. In the context of OWF development and cumulative impacts, we strongly believe that this step of validation is essential in order to use these models and simulations by policy makers.

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 592

Figure captions

Figure 1: Location of the Courseulles-sur-Mer future offshore wind farm (which corresponds to the CSM Ecopath model) and of the Dieppe-Le Tréport future offshore wind farm (which corresponds to the CSM Ecopath model) (Modified from Larssonneur et al., 1982)

Figure 2: TLs estimated from stable isotopes (TL_{IA}) plotted against their corresponding levels estimated by the Ecopath model ($TL_{Ecopath}$) for the CSM OWF trophic web. Invertebrates are represented by rectangles (black: benthic organisms and benthic cephalopods; white: zooplankton) and fish by circles (black: flatfish; grey: demersal fish; white: pelagic fish). See Table 1 for matching between number and functional groups.

Figure 3: TLs estimated from stable isotopes (TL_{IA}) plotted against their corresponding levels estimated by the Ecopath model ($TL_{Ecopath}$) for the DLT OWF trophic web. See Table 2 for matching between number and functional groups.

Table 1: Comparison between the TLs derived from the CSM ecopath model and the TLs calculated from SIA

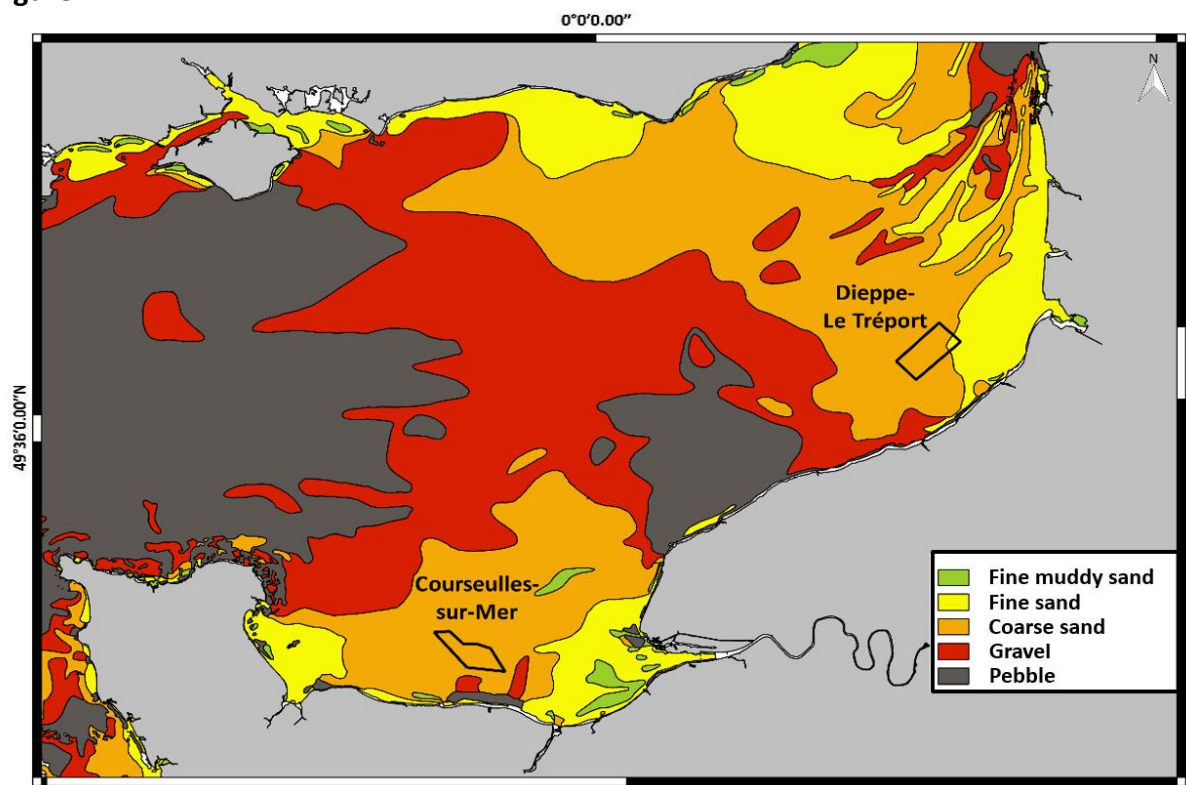
Number	Functional group	TLs Ecopath	TLs IA	Source for IA
1	Benthic cephalopods	3.92	3.22 ± 0.16	This study
2	Benthopelagic cephalopods	4.07	4.22 ± 0.25	Kopp et al., 2015
3	Fish, mackerel	3.14	3.30 ± 0.18	Kopp et al., 2015
4	Fish, European seabass	3.75	3.98 ± 0.25	Kopp et al., 2015
5	Fish, sharks and rays	4.15	3.46 ± 0.2	Kopp et al., 2015
6	Fish, Atlantic cod	4.03	3.66 ± 0.2	Kopp et al., 2015
7	Fish, whiting	4.12	3.98 ± 0.25	Kopp et al., 2015
8	Fish, Atlantic horse mackerel	3.90	4.03 ± 0.26	Kopp et al., 2015
9	Fish, gurnard	3.50	3.90 ± 0.25	Kopp et al., 2015
10	Fish, pouting	3.76	3.61 ± 0.23	Kopp et al., 2015
11	Fish, poor cod	3.72	3.28 ± 0.21	Kopp et al., 2015
12	Fish, European pilchard	2.80	2.95 ± 0.23	Kopp et al., 2015
13	Fish European sprat	3.00	3.12 ± 0.16	Kopp et al., 2015
14	Fish, benthos feeders	3.76	3.48 ± 0.18	Kopp et al., 2015
15	Fish, sea bream	3.20	3.71 ± 0.22	Kopp et al., 2015
16	Fish, sole	3.44	3.32 ± 0.18	Kopp et al., 2015
17	Fish, European plaice	3.37	3.37 ± 0.17	Kopp et al., 2015
18	Fish, other flatfish	3.35	3.21 ± 0.16	Kopp et al., 2015
19	Benthic inv. predators	3.07	3.22 ± 0.16	This study
20	Benthic inv. bivalves	2.10	2.11 ± 0.05	This study
21	King scallop	2.10	2.00 ± 0.16	Kopp et al., 2015
22	Benthic inv. deposit feeders	2.21	2.02 ± 0.024	This study
23	Zooplankton	2.00	2.56 ± 0.17	Kopp et al., 2015

613 **Table 2:** Comparison between the TLs derived from the DLT Ecopath model and the TLs
614 calculated from SIA
615

Number	Functional group	TLs Ecopath	TLs IA	Source
1	Fish benthos feeders	3.51	3.32 ± 0.16	This study
2	Fish flatfish	3.38	3.35 ± 0.13	
3	Benthic inv. predator not consumed	3.09	2.84 ± 0.12	
4	Benthic inv. filter feeders not consumed	2.25	2.10 ± 0.11	
5	Benthic inv. Scavenger not consumed	3.36	3.42 ± 0.18	

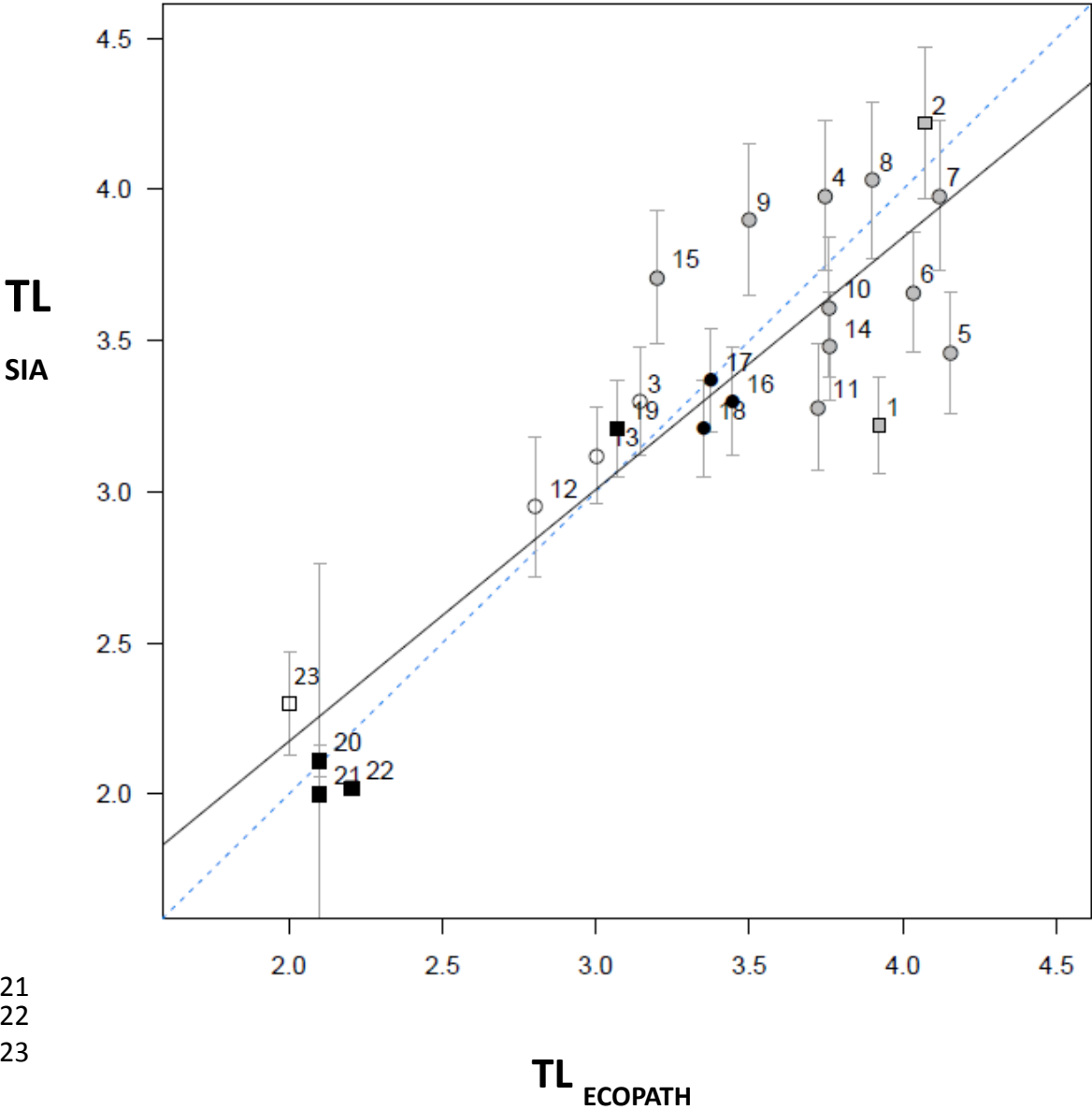
616
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618 **Figure 1**



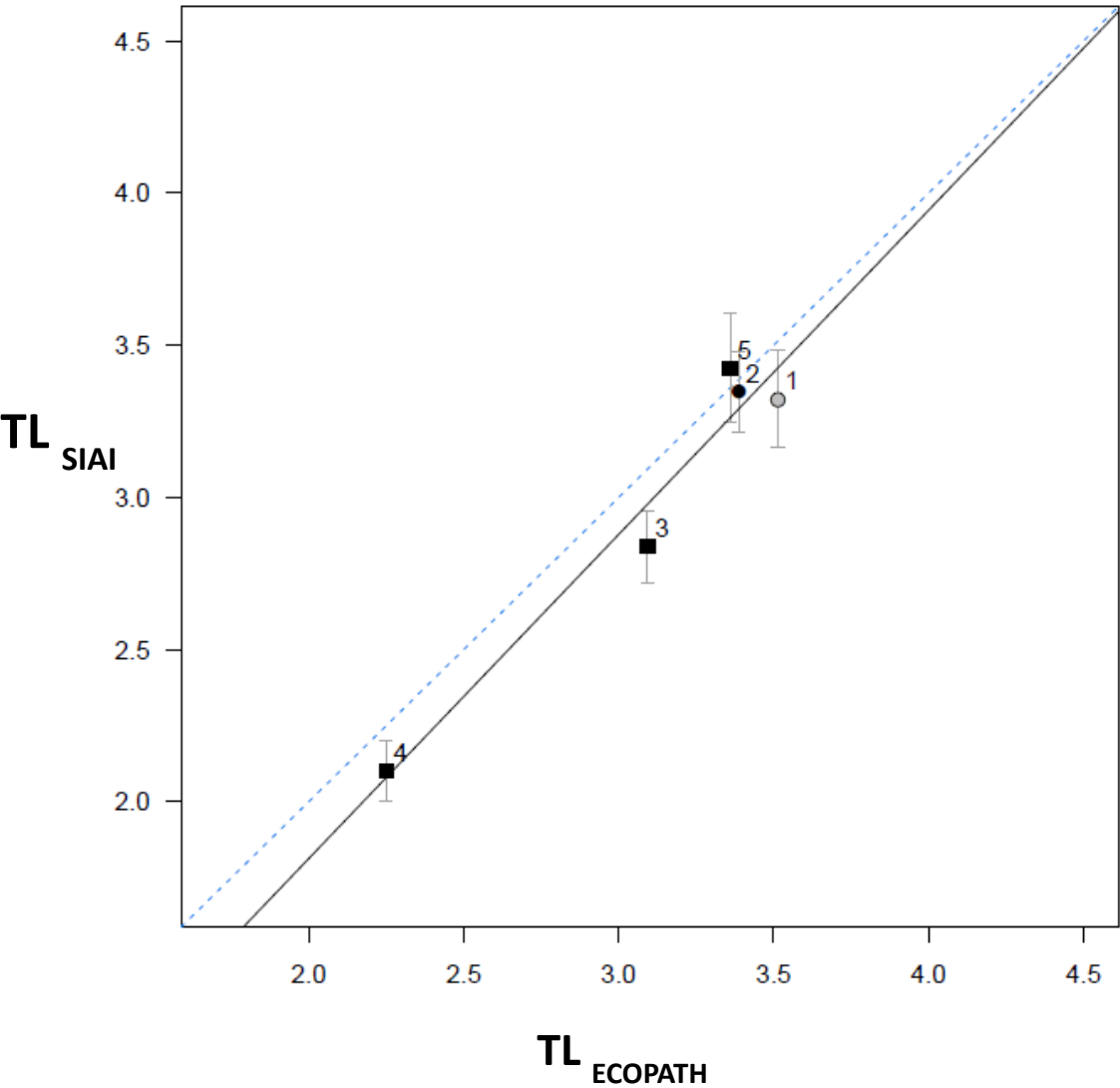
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620 **Figure 2**



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624 **Figure 3**



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