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

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14

15

16 **ABSTRACT**

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

31

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

34

## 35 **1 Introduction**

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

55 Quantitative trophic web models can be used for this purpose since they describe the  
56 interactions between species at different trophic levels (from prokaryote to top predators)  
57 and are based on the quantification of flows of energy and matter in ecosystems. Among  
58 these different existing approaches, the Ecopath with Ecosim (EwE) model has been  
59 intensively used and developed over the last three decades (Colléter et al., 2013, 2015) and  
60 was recognised by the US National Oceanographic and Atmospheric Administration (NOAA)  
61 as one of the ten biggest scientific breakthroughs in its 200 year history (Heymans et al.,  
62 2016). The EwE model has been used to analyse the impact of a wide range of anthropogenic  
63 perturbations on marine ecosystems through the world such as fishing activities (Heymans  
64 and Tomczak, 2016), invasion of species (Langseth et al., 2012), dumping operations (Pezy et  
65 al., 2017, 2018a), and infrastructure development (Tecchio et al., 2015). With the planning of

66 the construction of three OWF, the English Channel has become a hotspot for future OWF  
67 development. Moreover, this area is one of the most human impacted area, with some of  
68 the highest cumulative impacts in the worldwide ocean (Halpern et al., 2008). Recently,  
69 Raoux et al. (2017, 2019) used the EwE approach in the context of OWF construction and  
70 cumulative impacts in the eastern part of the English Channel. In their study, Raoux et al  
71 (2017, 2019) built a food web model at the site of Courseulles-sur-mer OWF construction to  
72 describe the situation “before” the construction of the wind farm and then performed  
73 simulation to analyse the potential impact of the reef and reserve effect (spatial restriction  
74 of fisheries for safety measures) on the ecosystem structure and functioning.

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

88  
89 Thus, in the context of OWF development and cumulative impacts there is an urgent need  
90 to validate models in order to test the predictions. For that, the validation should be an  
91 integral part of the modelling process. However, it is necessary to have an independent data  
92 set with information not use in the model inputs.

93  
94 Concerning EwE models, special attention has been given to the validation of functional  
95 compartments trophic levels calculated by Ecopath using Stable Isotope Analyses (SIA) (Kline  
96 et Pauly, 1998; Pauly et al., 1998; Polunin et Pinnegar, 2000; Dame and Christian, 2008;

97 Nilsen et al., 2008; Milessi et al., 2010; Navarro et al., 2011; Lassalle et al., 2014; Deehr et al.,  
98 2014). Compared to modelling, SIA constitutes a completely different approach to analyse  
99 trophic interactions within ecosystems (Fry et al., 1987) and the  $\delta^{15}\text{N}$  value can be used to  
100 describe the trophic level of consumers (Hobson et Welch 1992; Le Loc'h et al., 2008). This  
101 approach is based on the fact that there is a relationship between the isotopic signatures of  
102 consumer and their prey (Peterson and Fry, 1987). When a predator consumes its prey,  
103 there is an energy transfer and the isotopic nitrogen values show a predictable increase in  
104 the isotopic ratio between 0.5 and 5‰ across trophic levels (Minagawa and Wada, 1984;  
105 Post, 2002). Despite the wide use of both EwE and SIA, few attempts have been made to  
106 compare these methods (Dame and Christian, 2008, Lasalle et al., 2014).

107  
108 In this study, EwE models from two locations were assessed: the site of the future  
109 Courseulles-sur-mer Offshore Wind Farm (CSM OWF) (Raoux et al., 2017), and the future  
110 Dieppe-Le-Tréport (DLT OWF) (Pezy et al., 2018 b; Pezy et al., in press). The trophic levels  
111 estimated from these two models were compared using trophic levels estimated from  
112 independent nitrogen isotope data.

113

## 114 **2 Materials and methods**

### 115 **2.1 Study area**

116 The eastern English Channel (eEC), where the CSM OWF and DLT OWF will be built in the  
117 next years, is a shallow epi-continental area located between France and England. It is about  
118 35 000 km<sup>2</sup> and it is delimited by the Cotentin peninsula in the west and the Dover Strait to  
119 the east. The maximum water depth never exceeds 70 m in the trench running through the  
120 centre of the English Channel (Carpentier et al., 2009). The tidal range is greater than 5 m on  
121 the French coast but is closer to 2 m on the English side (Dauvin and Lozachmeur, 2006).  
122 These currents play an essential role in the sediment (Larsonneur et al., 1982) and benthic  
123 communities (Cabioch and Gentil, 1975; Dauvin, 2015) distributions. The eEC is also the  
124 focus of many human activities such as transportation, fishing, sediment deposit, and  
125 sediment extraction (Dauvin and Lozachmeur, 2006; Dauvin, 2012) and is considered by  
126 Halpern et al. (2008) as one of the most anthropized sea of the world.

127

128 *2.1.1 The Courseulles-sur-mer Offshore Wind Farm (CSM OWF) project*

129 The CSM OWF will be located 10 to 16 km offshore from the Calvados coast between 31 and  
130 34 m of depth. It will be located on the coarse sand and gravels benthic communities of the  
131 Bay of Seine (Fig. 1). The OWF will represent an area of 50 km<sup>2</sup>. A total of 75 turbines (each 6  
132 MW) capable of producing 450 MW will be installed by Eoliennes Offshore du Calvados”  
133 (EOC) in the next years. The monopile foundation will have a diameter of 7 m and will be  
134 driven into the seabed. The wind farm turbines will be connected via an interarray network  
135 consisting of 33 kV AC cables which will link up to an offshore transformer substation located  
136 within the wind farm. From this station, power will be exported via two 225 kV AC marine  
137 cables.

138

139 *2.1.2 Dieppe-Le Tréport Offshore Wind Farm (DLT OWF) project*

140 The prime contractor of the project is “Eoliennes en mer Dieppe-Le Tréport”, a subsidiary of  
141 Engie (formerly named GDF Suez). The proposed OWF will be located at distances of 15.5 km  
142 and 17 km offshore from the coast off Le Tréport and Dieppe, respectively (Figure 1). The  
143 water depth ranges from 12 to 25 m. The OWF will cover a total area of approximately 92  
144 km<sup>2</sup>, and will comprise 62 turbines with a capacity of 8 MW each giving a combined  
145 nameplate capacity of 496 MW. The foundations are composed of jacket structures. As for  
146 the CSM OWF, wind farm turbines will be connected via an interarray network consisting of  
147 33 kV AC cables which will link up to an offshore transformer substation located within the  
148 wind farm. From this station, power will be exported via two 225 kV AC marine cables to the  
149 continent.

150 <Figure 1>

151

152 **2.2 Data collection**

153 *2.2.1 Trophic modelling approach*

154 *2.2.1.1 The pre-existing Ecopath models*

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

160 (i.e. returning only one value per flow) in which the ecosystem is represented by functional  
161 groups, which are composed of species or group of species with ecological or biological  
162 similarities. Each functional group is parameterized with biomass ( $B$ ,  $\text{gC m}^{-2}$ ), production to  
163 biomass ratio ( $P/B$ ,  $\text{year}^{-1}$ ), consumption to biomass ratio ( $Q/B$ ,  $\text{year}^{-1}$ ) and a matrix diet  
164 which represents the interactions between predators and prey in the ecosystem.

165

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

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

169

170 The second equation ensures energy balance for each trophic group:

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

172

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

188

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

192 divided into 13 groups with a special subdivision for 6 of these groups (consumed (C) and  
193 non consumed (NC)): *Branchiostoma lanceolatum*, predators (C & NC), scavengers (C & NC),  
194 filter feeders (C & NC), selective deposit feeders (C & NC), non-selective deposit feeders (C &  
195 NC), grazers (C & NC). The consumed organisms were those where the species were found  
196 in the fish stomach contents. The cetaceans, seals, cephalopods, meiofauna  
197 Merosuprabenthos, holosuprabenthos, holozooplankton, bacteria, primary producer and  
198 detritus were represented by one group respectively. 21 of the functional groups were  
199 obtained from sampling operations (following methods described by Pezy et al., 2019).

200

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

205

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

### 209 2.2.1.2 Trophic Level in Ecopath

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

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

214

$$215 \quad TL_j = 1 + \sum_{i=1}^N DC_{ji} TL_i \text{ (Eq. 1)}$$

216

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

218

219 2.2.2 *Stable isotopes analyses*

220 2.2.2.1 *Sampling and samples preparation*

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

226

227 Fish and benthic invertebrates were collected at the site of the implantation of the future  
228 DLT OWF in February 2016 using a 3 m beam trawl and with a 0.1 m<sup>2</sup> Van Veen grab  
229 respectively.

230 130 samples corresponding to nine species were analysed for nitrogen stable isotope to  
231 study the trophic level of each species at the CSM future OWF and 102 samples  
232 corresponding to 16 species at the DLT future OWF (Table 1 and Table 2). All samples were  
233 kept frozen until processed in laboratory. The preparation of the tissue for stable isotopic  
234 analyses varied according to taxon. However, we chose to collect the muscle because the  
235 nitrogen isotopic signature of the latter is less variable over time than tissues from other  
236 organs such as the digestive gland or heart tissue (Pinnegar and Polunin, 1999). For bivalves,  
237 a sample was taken for the abductor muscle, for cephalopods, a mantle tissue was taken, for  
238 crabs a mantle tissue was taken from chelipeds, for ophiurid a sample was taken from the  
239 arms, for sea urchins, the muscle of the lantern of Aristotle was used and for fish a sample  
240 of white dorsal muscle was taken.

241

242 <Table 1>

243

244 <Table 2>

245

246 All samples collected were oven dried (60°C for 48 h) and then each dried sample was  
247 ground into a homogeneous powder using a mixer mill. Approximately 1 mg of powder was  
248 weighed into small tin cups. Isotopic analyses were performed with elemental analyser EA  
249 3000 (EuroVector) coupled with a mass spectrometer (Pinnegar and Polunin, 1999) at the

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

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

254

$$255 \quad \delta X (\text{‰}) = [ (R_{\text{sample}} / R_{\text{standard}}) - 1 ] \times 1000 \text{ (Eq. 2)}$$

256

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

258

### 259 2.2.2.2 Trophic level calculation from the isotopic analyses

260

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

263

$$264 \quad TL = \lambda + (\delta^{15}N_{\text{organism}} - \delta^{15}N_{\text{base}}) / \Delta N \text{ (Eq. 3)}$$

265

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

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

278

### 279 **2.3 Comparison between Ecopath mass balanced model and isotope results**

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

### 286 **3 Results**

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

290 <Figure 2>

291 <Table 1>

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

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

311 <Figure 3>

312 <Table 2>

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

324

## 325 **4 Discussion**

### 326 **4.1 Importance of local data**

327 The objective of the present study was to evaluate the CSM OWF model and the DLT OWF  
328 model through the comparison of the TLs computed by these two models with TLs estimated  
329 from independent nitrogen isotope data. Results showed that TLs estimated by the Ecopath  
330 mass balanced models were consistent with the TLs estimated by the SIA as an independent  
331 test of validity. These results suggested that diet data use in the CSM Ecopath model and in  
332 DLT Ecopath were of good quality and highlighted the importance to build model based on  
333 high-quality source data. In addition, these results are consistent by the high value of the  
334 pedigree index of these two Ecopath models. In, fact the pedigree index of the CSM Ecopath  
335 model (0.5) and of the DLT Ecopath model (0.7) are situated at the maximum of the range  
336 (0.16 to 0.7) reported in Morissette (2007). These pedigree high values can be explained by  
337 the fact that most of functional group biomass data, in the two models, were obtained from  
338 local, highly replicated and detailed sampling (Raoux et al., 2017; Pezy et al., 2018). In  
339 addition, the fish diet compositions of the models were derived from local stomach content

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

351

#### 352 **4.2 Comparison of the TLs derived from the EwE approach and SIA**

353 The popular utilisation of both the Ecopath mass balanced model and SIA make their  
354 comparison and cross validation highly relevant. However, to date very few attempts have  
355 been made to validate the Ecopath model results with results from SIA. Among these few  
356 attempts, Kline and Pauly (1998), Nilsen et al. (2008), Milessi et al. (2010), Lasalle et al.  
357 (2014), Dheer et al. (2014) also found a positive correlation between the TLs estimated by  
358 the EwE and the SIA approach. However, Dheer et al. (2014) highlighted that the EwE  
359 approach tended to estimate incorrectly the TLs of the detritivorous species. This can be  
360 explained by the fact that in Ecopath, a TL of 1 is automatically associated to detritus leading  
361 to an underestimation of TLs compared to SIA (Nilsen et al., 2008; Navarro et al., 2011).

362

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

372 isotopic fractionation values (in the calculation of TLs with the SIA approach). The value of  
373 3.4‰ used in this study is considered as a robust average for  $\delta^{15}\text{N}$  isotopic fractionation  
374 (Minagawa and Wada, 1984) but it can range from 0.5 to 5.5‰ (Post, 2002). The  
375 fractionation and isotopic signature can also vary among and within species, and at different  
376 stages of development according to their growth rate (DeNiro and Epstein 1978, 1981;  
377 Weidel et al., 2011), and as a function of the sea temperature (Barnes et al., 2007).  
378 Moreover, feeding differences and fish migration between locations can also cause variation  
379 in  $\delta^{15}\text{N}$  values (Deegan and Garritt, 1997). Finally, the aggregation of species within  
380 compartments in the Ecopath models, and the resulting composition of the compartments  
381 can influence the TL calculation, and could partly explain the deviation between the two  
382 approaches. In this study, aggregation and definition of the functional groups were based on  
383 the biological and ecological characteristics of the species, such as their food preference,  
384 size, and commercial importance, as well as on data availability (Raoux et al., 2017; Pezy et  
385 al., 2018). However, the difference observed in the zooplankton TL could be explained by the  
386 fact that the CSM Ecopath model has only one compartment of zooplankton including both  
387 prey (mesozooplankton) and predators (copepods), whereas SIA were made only on  
388 copepods, which have a higher trophic level than mesozooplankton. Moreover, the SIA were  
389 done on copepods that had not undergone fasting, which could introduce a bias in isotopic  
390 analysis. Finally, the differences observed in the cephalopods TLs could be explained by the  
391 fact that the prey species with a low trophic level are often digested more rapidly than prey  
392 species of higher trophic levels, and would therefore be underestimated during stomach  
393 analyses. Finally, it is worth noting that the results from the DLT

394

## 395 **5 Conclusion**

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

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

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592

593 **Figure captions**

594

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

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

603

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

607

608 **Table 1:** Comparison between the TLs derived from the CSM ecopath model and the TLs  
609 calculated from SIA  
610  
611

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

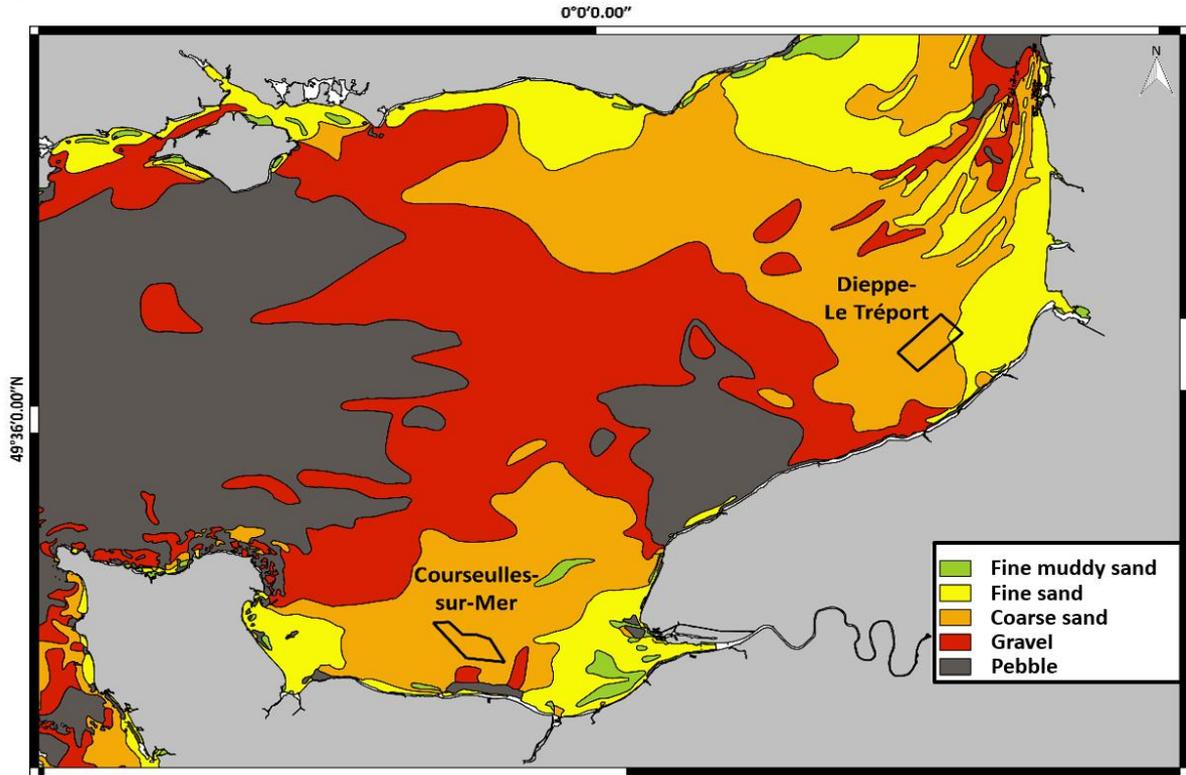
612

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

<b>Number</b>	<b>Functional group</b>	<b>TLs Ecopath</b>	<b>TLs IA</b>	<b>Source</b>
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	

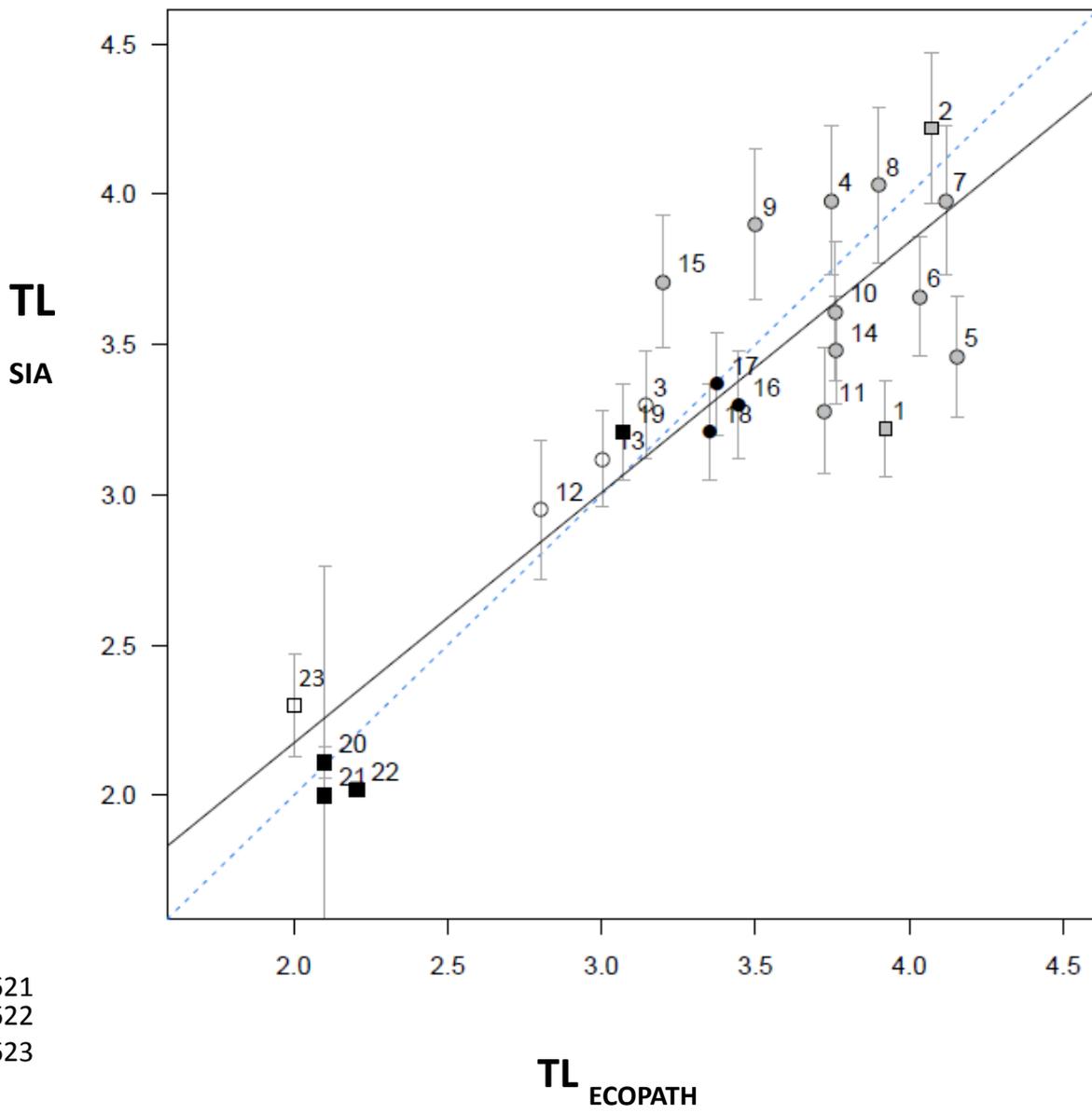
616  
617

618 **Figure 1**



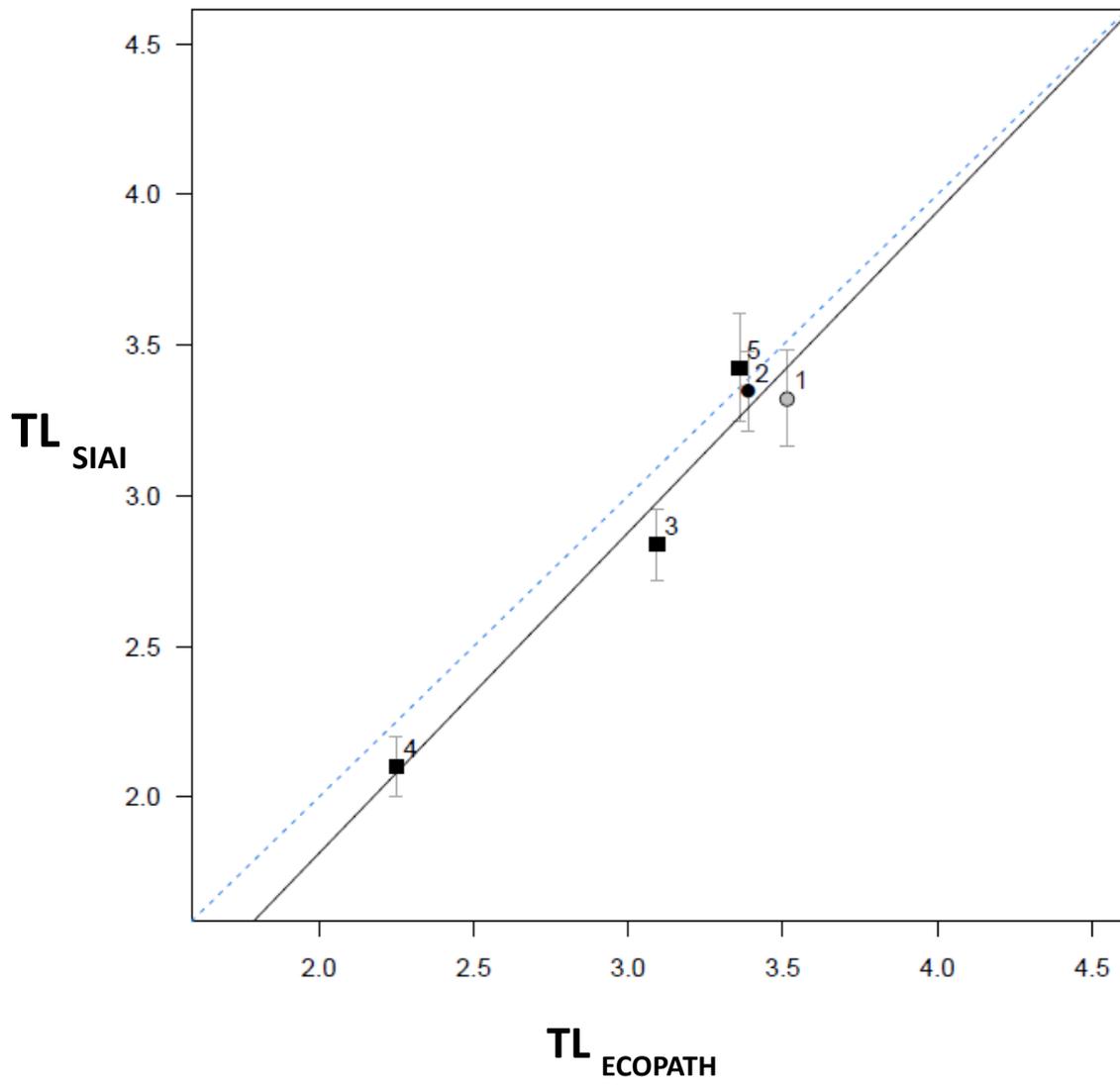
619

620 **Figure 2**



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624 **Figure 3**



625  
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