RECOMMENDATIONS FOR THE QUANTITATIVE ASSESSMENT OF METAL INPUTS IN THE MARINE ENVIRONMENT FROM THE GALVANIC ANODES OF OFFSHORE RENEWABLE ENERGY STRUCTURES

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RECOMMENDATIONS FOR THE QUANTITATIVE ASSESSMENT OF METAL INPUTS IN THE MARINE ENVIRONMENT FROM THE GALVANIC ANODES OF OFFSHORE RENEWABLE ENERGY STRUCTURES

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Recommandations for the quantative assessment of metal inputs in the marine environment
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Preface

The fast-growing offshore wind industry represents a potential new anthropogenic pressure for the marine environment. Despite the fact that it is considered as a green/clean energy source, the large-scale deployment of these devices could induce some environmental impacts that need to be studied. Understanding these impacts is in line with the Marine Strategy Framework Directive (MSFD) (European Commission, 2008) of the European Union which targets good environmental status for the marine environment.

The development of offshore renewable energies (ORE), in particular floating and fixed offshore wind structures in France, gives rise to new and more precise questioning from the French public authorities and civil society. In particular, a lot of questions have emerged in relation to the use of Galvanic Anode Cathodic Protection (GACP) systems and about the release of heavy metals associated with their functioning. These anodes are composed of different metals which are more reducing than the metal to be protected. The anodes are generally made with magnesium, aluminium or zinc. In offshore industries such as offshore wind turbines or oil & gas platforms, the anodes used are mainly composed of an aluminium alloy which contains zinc and other trace metals.

Their installation in fishing zones is a source of major concern and this affects several offshore wind farm projects (such as Île d'Yeu and Noirmoutier). This growing concern from public authorities and civil society has even led to galvanic anodes being abandoned in certain projects in order to overcome this problem. For example, the chosen solution for the jacket foundations of the Dieppe Le Tréport project is to implement cathodic protection by impressed current (ICCP) with titanium anodes). GACP has been replaced by ICCP in several projects although no scientific study has been conducted to prove the environmental impact of GACP in the ORE context, and no scientific study has proven that ICCP has no environmental impact.

In this context, a chemical risk assessment process has been conducted through a 1-year collaborative R&D project called ANODE in order to determine whether the chemicals released from GACP may represent a risk for the marine environment. Such chemical risk assessments are based on the comparison of the predicted or measured concentration (PEC - Predicted Environmental Concentration) for a contaminant with its hazard data (PNEC - Predicted No Effect Concentration) for all environmental compartments (water column, sediment and biota for the marine environment). This is an iterative process the first step of which is based on a worst-case approach in order to ensure a high protective level for the marine environment. Through the ANODE project, the risk assessment was conducted on the water column only. A hydrodynamic model was developed as a first approach for the prediction of the dispersion of these metals in the water column. From this numerical model, a predicted concentration PEC was determined to be compared with existing $PNEC_{seawater}$. Three sites were studied representing different hydrodynamic conditions directly influencing the metal dispersion and different technologies (i.e. fixed and floating offshore wind turbines). The project objectives were as follows:

- To perform a literature review of the different cathodic protection systems used in the offshore industry with a focus on GACP and current knowledge of its potential impact on the marine environment.

- To draw up an overview of the data required for the numerical simulation and the availability of these data in the study areas.

- To model the hydrodynamics of the selected areas, i.e. Courseulles-sur-Mer in the English Channel, Leucate in the Mediterranean and Groix & Belle-Île off the Atlantic coast, and simulate the metal releases and the evolution over time of their concentrations in the various case studies.

- To conduct a risk assessment by comparing the results obtained using the model with existing toxicity thresholds (in the water column).

Following these objectives, this report summarises the main results of the project and outlines a number of recommendations. This report is constructed as follows:

• The first section provides a summary of the principal results of the project;

• The second part presents the methodological recommendations for the environmental risk assessment;

• The third part lists methodological recommendations for future ORE installations and their associated impact studies.

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1 - Summary of work performed

1.1 Literature review

During this project, the first step was to perform a literature review of current knowledge of cathodic protection systems and how to estimate their potential impact on the marine environment. These systems are actually designed to protect the offshore structures against the electrochemical corrosion. The steel of which the structures are composed loses electrons to return to its most stable form (Fe_{2+} ion) which is the first and principal step in this corrosion process.

In order to prevent this loss, GACP or ICCP systems provide electrons to the structure (see Figure 1). In the case of GACP, this is achieved through the degradation of galvanic anodes, while ICCP systems directly inject a current to the structure (hence its name). However, the use of these devices will induce the release of different elements. In the offshore industries, the main composition of galvanic anodes is an aluminium alloy which of course contains a large share of aluminium (about 95%), as well as zinc (about 5%) and other trace metals (< 1%): copper, iron, indium or cadmium. The composition of these anodes is normed by DNVGL-RP-B401 (2017). The degradation of the anodes induces the release of heavy metals in the seawater in different forms which strongly depend on the pH level. In natural seawater, Millero *et al.* (2009) reported that the pH level is close to 8.1 and is favourable to the presence of $Al(OH)_{4-}$ in dissolved form which represents almost 67% of the solution while aluminium hydroxide $Al(OH)_3$ in particulate form represents 32.18%. In this project, only the dissolved form was considered as it is the most readily assimilated by marine organisms. Hence, our hypothesis, which states that 100% of each metal dissolves into the seawater, constitutes the worst-case scenario to be tested.

To estimate the release rate of this dissolved form, the recommended practice DNVGL-RP-B401 gives a method depending on the total anode mass installed on the devices and the estimated life-time. This method will be used in the modelling work (which is summarised in the following sections) to implement the source points of aluminium and other metals in the model with a realistic release rate. On the Courseulles-sur-Mer study site for example, a total of 15 tonnes of anodes are positioned on each wind turbine for a

life-time of 25 years. Based on the recommended practice, it was considered that 85% of the anodes were degraded after this duration. Therefore, the release rate of each metal was estimated as a constant by the ratio between the total amount of anode mass released after 25 years and the offshore wind farm life-time.

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Figure 1. Example of galvanic anode cathodic protection

The ICCP system also induces the release of some elements. However, its functioning is not based on the degradation of an anode but on the generation and injection of an electrical current. It induces the chlorination of the seawater which releases some dichlorine. This element has a half-life time of 10 minutes in seawater. Therefore, it is considered that ICCP systems release elements from the degradation of dichlorine into the seawater such as bromamines, monochloramine, bromoform or chloroform. Based on the amount of injected current, the dichlorine release rate is determined and applied to all other elements. Therefore, we considered that all the dichlorine was transformed either into bromamines, monochloramine, bromoform or chloroform, which once again constitutes the worst-case scenario. This release rate is determined as a constant and was used for the numerical modelling.

The risk assessment was performed in accordance with the REACH (Registration, Evaluation, Authorisation, and Restriction of Chemicals) technical guidance described by ECHA (European Chemicals Agency, 2008; 2016). In the risk characterisation, exposure levels (PEC) are compared to hazard information (PNEC). For the hazard assessment, a search for available PNECs in the literature cited in the bibliography was carried out (see section 1.4 Risk assessment of galvanic anodes in the marine environment).

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1.2 Selection of the study sites

Of all the future commercial and pilot offshore wind farms planned in France, three were selected for the ANODE project. These farms will be deployed in different areas which are exposed to diverse hydrodynamic conditions. Moreover, the wind turbines are to be installed on either fixed or floating foundations (see Figure 2). In order to take all these elements into account, one fixed foundation farm was selected on the English Channel coast and two with floating structures were selected on the Atlantic coast and in the Mediterranean Sea (see Table 1).

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Figure 2. The three future offshore wind farms: Courseulles-sur-Mer, Leucate and Groix & Belle-Île (from the left to the right).

French coasts and main specificities		Atlantic / English Channel *Tidal-driven currents*	Mediterranean Sea *Wind-driven currents*
Model per site with associated structure technology and cathodic protection type	Fixed offshore wind farm (monopile)	Model 1: Courseulles-sur-Mer GACP Scenario	-
	Floating offshore wind farm	Model 3: Groix & Belle-île GACP Scenario	Model 2: Leucate GACP Scenario & ICCP Scenario

Table 1. Scenarios planned to be implemented for the selected sites.

The selection was performed taking into account the available data on each site and based on the data required to run the numerical model, namely:

• Acoustic Doppler Current Profiler (ADCP) measurements on the study site for the validation of the numerical model;
• The exact location of each future device to implement the metal/chlorine-based element source points;
• The location of the anodes on the structure to position them in the water column;
• The exact composition and quantity of galvanic anodes to estimate the release rate of each element;
• A potential release rate estimated by the developers for validation;
• Concentration measurements of each element on the study site as an initial state.

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1.3 Three-dimensional numerical modelling of the dispersal of metallic/chlorine -based elements

Following the conclusion of the two previously described studies, the numerical modelling was then performed on the three selected sites of Courseulles-sur-Mer, Leucate and Groix & Belle-Île (see Figure 3). The dispersion of each element was performed over a 5-year period for the sites dominated by the tidal flow (Courseulles-sur-Mer and Groix & Belle-Île) and a 1-year period for the Leucate site where the hydrodynamic conditions are driven by the atmospheric forcing.

Figure 3. Numerical domain of the (a) Courseulles-sur-Mer, (b) Leucate and (c) Groix & Belle-Île study sites. Courseulles-sur-Mer is the only site considered as a future commercial farm (75 wind turbines). The two others are pilot farms (3 to 4 wind turbines). The colour scale represents water depth in meters.
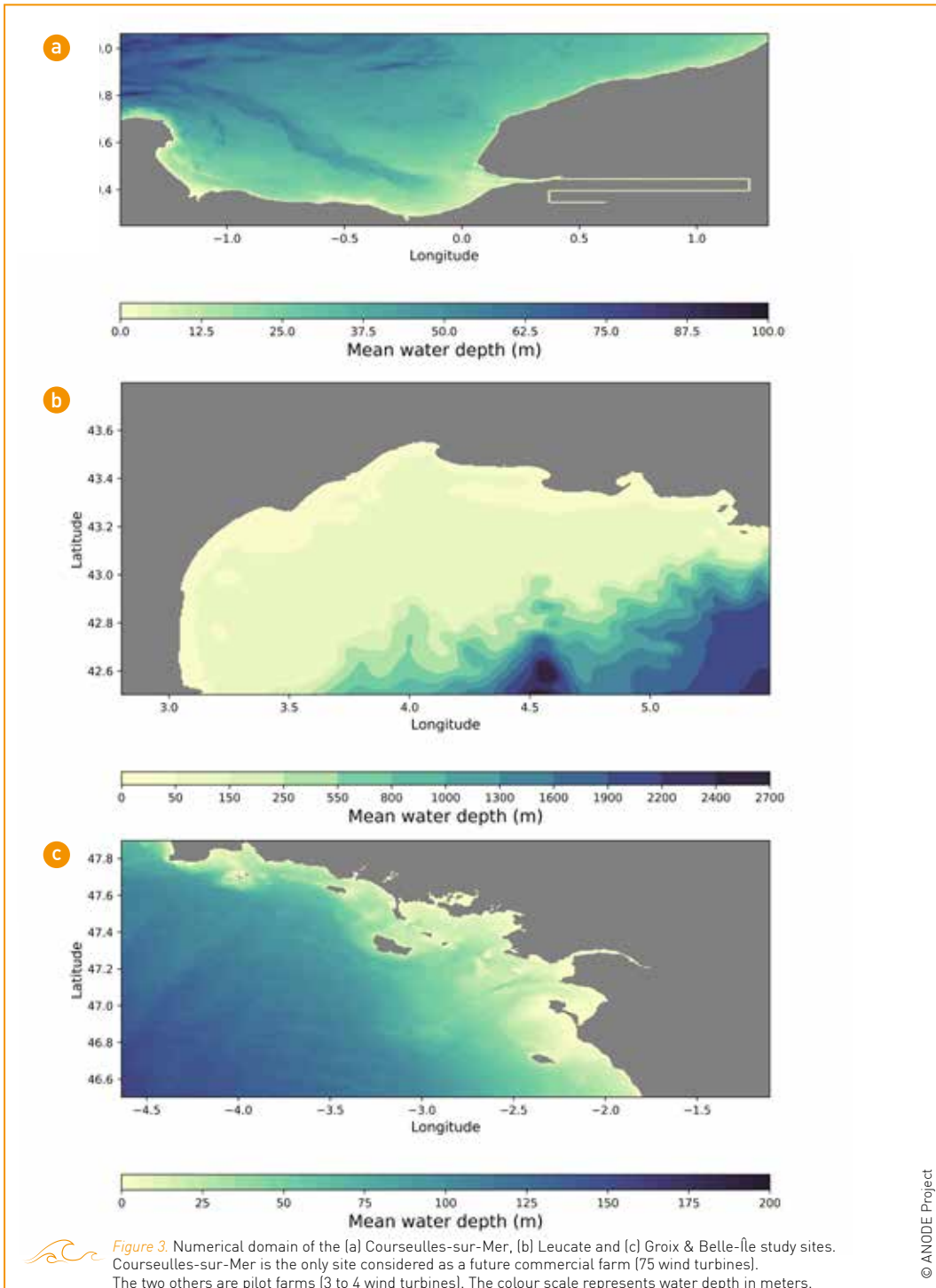
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The hydrodynamic conditions of each site were simulated as a three-dimensional configuration using the Coastal and Regional Ocean COmmunity model (CROCO). The tidal forcing is implemented in the simulations performed on the Courseulles-sur-Mer and Groix & Belle-Île sites. All scenarios were configured to integrate the atmospheric forcing using Météo France ARPEGE/AROME model results and river runoff which plays an important role in the definition of the density gradient on all sites.

Once the hydrodynamic conditions were validated, the GACP/ICCP releases were implemented in the three domains as source points (located at each wind turbine foundation positions) with the release of a certain mass of each element at every time step of the computation (30s). In the model, each metallic element was emitted into the water column in its dissolved form and considered as unchanged thereafter. Results were then produced for each study site. It was decided to present in this document only the results obtained for the Courseulles-sur-Mer site because it is a commercial farm with a large number of turbines.

At the Courseulles-sur-Mer study site, the dispersion of the metals composing the galvanic anodes into the water column is mainly driven by the tidal currents. In the long term, the dissolved metals follow the southern coastal residual currents exiting through the western part of the bay and then are advected out of the domain in accordance with the study by Salomon & Breton (1993). In this domain, aluminium is the only element to exceed the toxicity threshold, while the maximum concentration estimated for the other elements is less than 5% of their respective PNEC$_{marine water}$. Therefore, for the other metals (zinc, iron, indium, copper and cadmium) the results indicate that in principle there is no risk.

The maximum aluminium point concentration reached during the 5 years of the simulation in the domain is 1 µg.L^{-1} recorded in the middle of the farm (see Figure 4.). However, this maximum concentration registered was only reached for a period of 24h (1 record). To study the evolution of the aluminium concentration over time, four different locations were selected. The point P2 is located within the commercial farm, P1 and P3 are located in the immediate vicinity and P4 is in the entrance to the Bay des Veys (see Figure 4). Regarding the evolution of the concentration at these locations, exposure characteristics were described and compared to the available data regarding the toxicity of the aluminium.

The concentration reached during a short time period of two consecutive days at these locations is naturally higher in the vicinity of the farm. At the P1 and P3 locations, its value remains around 0.75 µg.L^{-1} during a period of 2 consecutive days, while it is around 0.41 µg.L^{-1} for a period of 2 consecutive weeks. At the P2 location, the concentrations remains higher than at the other locations, as it is inside the farm with a value of around 0.96 µg.L^{-1} for two consecutive days and around 0.67 µg.L^{-1} for 2 consecutive weeks. At the P4 location, variation is very low and the concentration reaches 0.44 µg.L^{-1} for two days and stays close to 0.33 µg.L^{-1} for 2 weeks.

Figure 4. Maximum concentration of aluminium reached during the 5-year simulation period. The black line outlines the area where the PNEC = 0.005 µg L^{-1} is exceeded.

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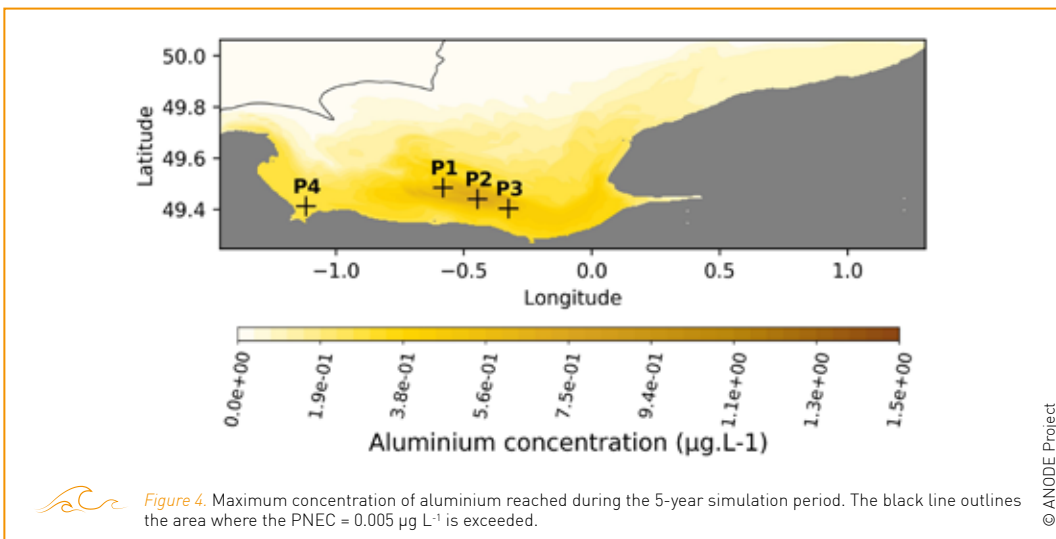
However, notwithstanding the discussion on the relevance of the aluminium PNEC in seawater (see section 1.4 Risk assessment of galvanic anodes in the marine environment), these concentrations would not represent the majority of the aluminium concentration in the bay if we considered aluminium releases by the Seine. In order to estimate the difference between the anodes and the natural contributions from the Seine, two other simulations were performed over the same time period considering only the aluminium re-

leased naturally by the Seine river which were set to either 25 or 70 µg.L^{-1} based on the measurements performed by Gabelle *et al.* (2012).

The results of these simulations (see Figure 5) show that throughout the farm, where the anodes' contribution is at its highest, the natural Seine contribution could reach almost 10 µg L^{-1}. The release induced by the Seine is thus almost 10 times higher than the maximum concentration registered from the anodes inside the future farm area.

Figure 5. Maximum concentration values reached by the Seine river contribution during the 5-year simulation period using (a) the minimum measured concentration of 25 µg.L^{-1} and (b) the maximum of 70 µg.L^{-1}.

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Over the 4 locations described previously, the maximum and mean concentrations of aluminium were compared between the anodes and the natural Seine contributions and are reported in Table 2. The mean concentrations indicate that the anode contribution for the entire farm (at locations P1, P2 and P3 where it is the highest) represents between 10 and 50% of the Seine contri-bution while in the Bay des Veys (location P4) it represents between 10 and 30% of the Seine contribution. The maximum contribution values from the Seine estuary show an even greater difference, with the anode contribution representing between 4 and 15% of the Seine release over the area of the farm while in the Bay des Veys it varies between 10 and 30%.

Release source	Location P1 concentrations (Mean / Max)	Location P2 concentrations (Mean / Max)	Location P3 concentrations (Mean / Max)	Location P4 concentrations (Mean / Max)
Anodes	0.12 / 0.54	0.24 / 0.65	0.17 / 0.51	0.1 / 0.3
Seine (25 µg.L^{-1})	0.23 / 3.6	0.46 / 4.6	0.66 / 4.7	0.33 / 1.03
Seine (70 µg.L^{-1})	0.64 / 10 .1	1.29 / 12.9	1.84 / 13.16	0.92 / 2.88

Table 2 : Mean and maximum concentrations (in µg.L^{-1}) at P1, P2, P3 and P4 locations in the three different configurations.

1.4 Risk assessment of galvanic anodes in the marine environment

The risk assessment was performed following the REACH (Registration, Evaluation, Authorisation, and Restriction of Chemicals) technical guidance described by ECHA (European Chemicals Agency, 2016). The chemical risk assessment for the environment is a compartmental approach conducted for each substance of interest in several matrices: water column, sediment, and biota. However, for the ANODE project, the risk assessment is conducted only on the seawater column, first because exposure modelling can only be conducted on this compartment and second, because this compartment is considered to be the main diffusion vector of the bioavailable forms of the elements released by the anodes. In a first intention, the risk is deliberately maximised to ensure optimum protection of the compartment. Other iterations need to be carried out in case of risk in order to refine the assessment.

1st iteration of the chemical risk assessment

Step 1 - Inventory of substances.
Each metallic element composing the galvanic anodes is considered: Al, Zn, Si, In, Fe, Cu, Cd.

Step 2 – Hazard assessment.
Existing PNECs were identified for each metallic element (see Table 3). The search for existing PNECs was conducted on three online databases: European Chemicals Agency (ECHA), ETOX (Information System Ecotoxicology and Environmental Quality Targets) and the INERIS (French National Institute for Industrial Environment and Risks) chemical substances portal.

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Compound	Priority substance list	Metal fraction	Seawater PNEC ($\mu g.L^{-1}$)	Derivation method	Source
Aluminium (Al)	–	Inorganic monomeric Al	0.005	Transposal of a freshwater PNEC with an added factor of 10. Derived from an EC_{30} (Observed Effect on 30% of the tested population) on growth inhibition of the algae *Chlorella pyrenoidosa* at pH = 6. Dataset composed of 27 chronic endpoints from 8 freshwater species covering 5 taxa (algae, fish, molluscs, crustacean, flatworms). Assessment Factor (AF) total = 50*10 = 500.	(Crane, *et al.*, 2007)
	–	Total dissolved Al	24	Derived from ecotoxicity tests on marine species with an SSD approach. Dataset of marine data: 10 EC_{10} and 1 NOEC (No Observed Effect Concentration) between 18 and 72h, on 11 species (some of them are endemic to the Australian coastal reef) covering 5 taxa (micro and macro algae, mollusc, cnidarian, echinoderm). No information on the AF used.	(Golding, *et al.*, 2015)
Zinc (Zn)	River Basin Specific Pollutant (RBSP) for freshwater	Total dissolved Zn	3.0	Derived from ecotoxicity tests on marine species with an SSD approach. Dataset composed of chronic results for 27 marine species. AF = 2.	(Nendza, 2014)
Silicon (Si)	–	–	No data	–	–
Indium (In)	–	Total dissolved In	40.6	Derived from ecotoxicity tests on freshwater and marine species with an SSD approach. Dataset composed of 23 ecotoxicity endpoints for 16 species from 3 trophic levels. Short-term and long-term data pooled together, a majority of freshwater data with only 2 marine species. AF = 3.	(ECHA, 2018)
Iron (Fe)	–	Total dissolved Fe	1.6	Transposal of a freshwater PNEC with an added factor of 10. Derived from a NOEC on reproduction for *Daphnia magna*. Dataset: NOECs covering 5 taxa. AF method, AF = 10*10 = 100.	(Johnson, *et al.*, 2007)
Copper (Cu)	River Basin Specific Pollutant (RBSP) for freshwater	Total dissolved Cu	2.64	Derived from ecotoxicity tests on marine species with an SSD approach. Dataset composed of 58 NOECs for 29 marine species covering 8 taxa. AF = 1.	(Maycock, *et al.*, 2011)
Cadmium (Cd)	Priority Hazardous Substance (PHS) & Carcinogenic, Mutagenic, Reprotoxic (CMR)	Total dissolved Cd	0.21	Dataset: 16 NOECs covering 7 taxa. SSD method used with AF = 2.	(Common Implementation Strategy for the Water Framework Directive, 2005)

Table 3 : Summary of the seawater PNECs of interest for the risk assessment.

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No PNEC$_{seawater}$ was available for silicon. The lack of hazard information on this metal could be explained by the fact that this is the most common element on earth, and it is essential for the development of many living organisms (skeleton, shell). The dissolved form of silicon in the marine environment is silicic acid. Its concentration in the water column generally does not exceed a few hundred µg of silicon per litre (Laruelle et al., 2002). It is an essential nutrient for organisms performing its biomineralisation, such as diatoms. There is no known toxicity of silicic acid to marine organisms at realistic environmental concentrations. It could also play a role in aluminium homeostasis by controlling its bioavailability (Exley et al., 2002).

Two PNEC$_{seawater}$ were available for aluminium, but both judged not fully satisfactory. The lowest concentration (0.005 µg.L^{-1}), proposed by UK TAG (Crane et al., 2007), was derived for inorganic monomeric aluminium based on ecotoxicity data deemed poorly reliable when assessed against Klimisch criteria (Klimisch et al., 1997), obtained on fresh-water species (transposal of PNEC$_{freshwater}$ with a supplementary assessment factor, AF = 10). Inorganic monomeric aluminium is the most bioavailable form of this metal in seawater. The highest concentration (24 µg.L^{-1}), proposed by Golding et al. (2015), was derived for dissolved aluminium upon marine organisms but covering only few taxa and using a derivation method that does not correspond to European standards (Whitehouse et al., 2018). For want of other options, we used the most protective option with the lower PNEC$_{seawater}$ of 0.005 µg.L^{-1} proposed by Crane et al. (2007) for the risk assessment.

Step 3 – Exposure assessment.

Predictions of exposure concentrations reached during the 5-year simulation were modelled with CROCO for the Bay of Seine. Over this domain, the results extracted for the 4 locations P1, P2, P3 and P4 were considered for the risk assessment as they describe the exposure inside the farm, in the vicinity of it and in an area of interest. The maximum inputs found at these locations are summarised in Table 4. To estimate protective predicted environmental concentrations (PECs) in the seawater column, the maximal inputs of the anodes and the initial environmental concentrations of each metallic element in the vicinity of the wind farm must be added together.

Data on metal concentrations assessed directly in the marine water column near the wind farm were not available. In absence of closer data, we used values found in literature obtained at the estuary of the main river feeding into the bay: the Seine, put forward by Chiffoleau et al. (1997) and Gabelle et al. (2012). This approach is part of a worst-case scenario where the initial concentrations are supposedly overestimated (assuming that the Seine is the main contributor of metallic elements to the bay and not taking into account the dispersion and dilution effect of elements through the bay) to maximise the protection of the marine water column.

Compound	Fraction	Max. input of the anodes (µg.L^{-1})	Input of the Seine river at the estuary (µg.L^{-1})	PEC$_{seawater}$ (µg.L^{-1}) (Anodes + Seine inputs)
Aluminium	Dissolved	0.65	25 to 70	70.65
Zinc	Dissolved	6.82x10^{-2}	2.78 (± 1.45)	2.8482
Silicon	Dissolved	1.23x10^{-3}	No data	Insufficient data
Indium	Dissolved	4.84x10^{-4}	No data	Insufficient data
Iron	Dissolved	1.05x10^{-3}	0.138 (± 0.10)	0.1391
Copper	Dissolved	3.73x10^{-5}	1.454 (± 0.69)	1.4540
Cadmium	Dissolved	2.50x10^{-5}	0.053 (± 0.017)	0.0530

Table 4 : Maximal input in seawater of the metals released during galvanic anode dissolution and summary of the metallic input of the Seine to the bay. The SEINE-AVAL results are presented as: concentrations (µg.L^{-1}) ± standard deviation.

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Step 4 - Risk characterisation.
With the PEC and the PNEC, it is possible to carry out a risk characterisation for the seawater column for almost all the metallic elements which compose the anodes. To this end, the Risk Characterisation Ratio (RCR) is estimated using the following equation:

$$RCR = \frac{PEC_{seawater}}{PNEC_{seawater}} \; ; Risk \; if \; RCR > 1$$

The risk ratios for silicon and indium could not be calculated. For silicon, we do not have enough data to characterise either the level of exposure (lack of information on the initial environmental concentration), or the hazard information on the substance (no existing $PNEC_{seawater}$ for this metal). For indium, we do not know the initial environmental concentration for this compartment near the Bay of Seine, so the $PEC_{seawater}$ for indium could not be calculated.

Based on the results of these ratios, for our 5-year simulation on the Courseulles-sur-Mer study case, we note that only one RCR is above 1. Indeed, a risk was characterised for seawater for one metal: aluminium, while for zinc, iron, copper, and cadmium no potential risk for the seawater compartment was determined with this first approach. However, in the case of zinc, the risk characterisation ratio was particularly close to 1, and to dismiss any concerns for this metal a second iteration of the risk assessment could be performed in the future. For aluminium, a second iteration of the chemical risk assessment was judged necessary to refine the risk assessment based on aluminium-specific data.

Compound	$PNEC_{seawater}$ (μg.L^{-1})	$PEC_{seawater}$ (μg.L^{-1})	$PEC_{seawater}$ / $PNEC_{seawater}$
Aluminium	0.005	70.65	$1.41 \cdot 10^{4}$
Zinc	3.0	2.85	0.95
Silicon	No data	Insufficient data	Insufficient data
Indium	40.6	Insufficient data	Insufficient data
Iron	1.6	0.14	$3.4 \cdot 10^{-3}$
Copper	2.64	1.45	0.55
Cadmium	0.21	0.05	0.25

Table 5 : 1st iteration of the chemical risk assessment for the components of galvanic anodes ($PNEC_{seawater}$, $PEC_{seawater}$ and $PEC_{seawater}$ / $PNEC_{seawater}$).

2nd iteration of the chemical risk assessment

For the second iteration of the risk assessment, steps 2 to 4 were thoroughly looked at for aluminium.

Step 2 – Hazard assessment, refinement of the aluminium $PNEC_{seawater}$.
The ecotoxicity datasets of the PNECs derived by Crane *et al.* (2007) and Golding *et al.* (2015) were closely studied, in particular according to the Klimisch criteria, to initiate the refinement of the $PNEC_{seawater}$ for aluminium. These criteria are used to assess the quality of experimental toxicological and ecotoxicological data. The classification is composed of 4 categories :
1 – Reliable without restrictions;
2 – Reliable with restrictions;
3 – Not reliable;
4 – Not assignable.

Criteria are awarded according to the data quality based on the sound and detailed methodology in accordance with the Organisation for Economic Co-operation and Development (OECD) or other internationally accepted testing guidelines, preferably Good Laboratory Practice (GLP) (Whitehouse *et al.*, 2018).

However, from these two datasets, too few marine ecotoxicity data were reliable without restrictions according Klimisch criteria. Thus, a search for ecotoxicity data considered reliable according to Klimisch criteria was initiated based on the literature (ECOTOX database by the US EPA) in an attempt to complete the existing dataset. Nevertheless, the search yielded only a few new ecotoxicity data classified as reliable with restrictions, meaning that a new $PNEC_{seawater}$ could not be derived at this point.

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Step 3 – Exposure assessment, refinement of the aluminium PEC$_{seawater}$.

The PEC$_{seawater}$ for aluminium was refined by simulating the dispersal of the metal from the Seine estuary through the bay again with the CROCO model. The simulation takes into account a constant release from the Seine estuary, and gives an initial value at the farm location of 12.59 µg.L^{-1}. Therefore it allowed us to estimate the initial environmental concentration of aluminium near the wind farm in a less protective way.

Step 4 - Risk characterisation, refinement of the risk characterisation ratios.

The refined PEC$_{seawater}$ was lower than that defined in the first assessment. However the risk ratio remains above 1 for aluminium with the PNEC$_{seawater}$ of Crane *et al.* (2007), while no risk is identified with the one of Golding *et al.* (2015). It is important to note that the PNEC$_{seawater}$ proposed by Crane *et al.* was derived from inorganic monomeric aluminium, while the PEC$_{seawater}$ was obtained from the dissolved fraction of aluminium. The risk ratio calculated during this risk assessment is thus derived from different fractions of aluminium, which is not ideal. Ideally, to assess the potential chemical risk for pelagic organisms, both the PEC$_{seawater}$ and the PNEC$_{seawater}$ should be derived from the most bioavailable fraction of the metal to be as precise and protective as possible. So, supplementary iterations of the risk characterisation should be conducted in relation to the recommendations made in the remainder of this document.

Compound	PNEC$_{seawater}$ (µg.L^{-1})	Refined PEC$_{seawater}$ (µg.L^{-1})	Refined PEC$_{seawater}$ / PNEC$_{seawater}$
Aluminium	0.005 24	13.59	$2.718 \cdot 10^3$ 0.566

Table 6 : 2nd iteration of the chemical risk assessment for aluminium (PNEC$_{seawater}$, Refined PEC$_{seawater}$ and Refined PEC$_{seawater}$ / PNEC$_{seawater}$)

Summary of work performed

1

Recommandations for the quantative assessment of metal inputs in the marine environment from the galvanic anodes of offshore renewable energy structures

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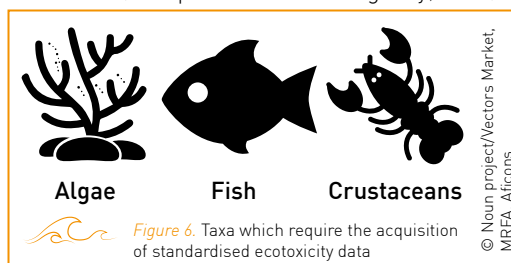## 2 - Methodological recommendations for the environmental risk assessment of anode dissolution

The risk assessment (in the water column) was carried out for galvanic anodes according to the REACH guidance document (European Chemicals Agency, 2016), following an iterative approach and applying a protective scenario, on the Courseulles-sur-Mer site (see part 1.4 Risk assessment of galvanic anodes in the marine environment). During its first iteration, no risk was identified for zinc, iron, copper or cadmium, while a risk was identified for aluminium. The risk ratios for silicon and indium could not be calculated due to the lack of information on the environmental concentrations near the wind farm and/or hazard information ($PNEC_{seawater}$).

Refinement of the risk characterisation of aluminium in seawater was initiated in a second iteration of the assessment, however in view of the lack of data we propose, in this document, recommendations for future investigations on this metal.

Hazard assessment

For the **hazard part** of this assessment, we used the $PNEC_{seawater}$ determined on the most bioavailable form of aluminium in seawater: inorganic monomeric aluminium (using the soluble form $Al(OH)_4^-$) by Crane et al. (2007). However, during the second iteration of the risk assessment, we further examined the methodology and the ecotoxicity dataset used to derive this $PNEC_{seawater}$ as well as the $PNEC_{seawater}$ derived from dissolved aluminium by Golding et al. (2015). Both of these $PNEC_{seawater}$ were judged unsatisfactory, as their ecotoxicity datasets were judged incomplete due to too few data classified as reliable according to the Klimisch criteria. Taxa covered by these datasets were also limited for seawater organisms, and the methodology of determination was not always in accordance with the REACH guidance document (European Chemicals Agency, 2008). The search for recent ecotoxicity data on seawater and freshwater initiated to complete the existent dataset yielded few data classified as reliable with restrictions according to Klimisch criteria. We thus recommend the acquisition of standardised ecotoxicity data for aluminium on marine organisms, especially on the following taxa: algae, fish, and crustaceans (see Figure 6). Tests should be conducted in accordance with OECD or other internationally accepted testing

guidelines (preferably GLP), to ensure the reliability of the data according to Klimisch criteria. The methodology applied to derive the future $PNEC_{seawater}$ should also follow the REACH guidance document (European Chemicals Agency, 2008).

Algae   Fish   Crustaceans

Figure 6. Taxa which require the acquisition of standardised ecotoxicity data

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Exposure assessment

For the **exposure part** of this assessment, the lack of data on the initial concentrations of aluminium in seawater in the vicinity of the wind farm led us to use an approximation of the aluminium concentrations in the Bay of Seine. For the second iteration, a modelled estimation was obtained by simulating the dispersal of dissolved aluminium from the Seine estuary (see Figure 7) into the bay, taking the highest concentration measured at the estuary by Gabelle et al. (2012) as the constant input. Using this approach, and considering the Seine as the main contributor/source of aluminium in the bay, we obtained an upper bound estimation of the initial environmental concentration of dissolved aluminium in the seawater column. Then, we added this estimation to the predicted inputs of aluminium released by the anodes to obtain a PEC. However, the predicted input of aluminium from anode releases was modelled considering that all the aluminium released was stable over time and in a bioavailable form. Hence, to further refine the exposure estimation and overcome the knowledge gap we recommend the acquisition of in situ concentrations of aluminium in the seawater column near the wind farm.

Figure 7. Estuary of the Seine

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Risk assessment

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Recommandations for the quantative assessment of metal inputs in the marine environment from the galvanic anodes of offshore renewable energy structures

Two methods can be used, both are complementary:

- Measurement by Diffusive Gradient in Thin films (DGT) to characterise the initial concentration at the site and to regularly monitor the concentration once the anode is installed on the wind farm. This will allow an assessment of the evolution of the concentration in relation to the use of the anode. The protocol is presented in part 3.1 Initial state characterisation - Measurement protocol.

- In addition to DGT, seawater samples could be analysed in parallel in order to measure the total dissolved fraction of aluminium or zinc in the filtered water (0.45 µm).

- These measurements should be performed once the maximum concentration is thought to have been reached.

Risk assessment

In order to assess the risk, it is important to consider the fraction measured *in situ* and that defined for the PNEC. A PNEC defined in the bioavailable fraction is ideally what we need in order to better assess the potential toxicity for organisms. But in the case of aluminium, it has been shown that PNEC derivation is based on very few data and derived from ecotoxicity data taken from fresh water only. Furthermore, no laboratory is identified that can measure isomeric monomeric aluminium in seawater. Thus, in order to compare a PEC to a PNEC, it is necessary for the fractions considered both in the PEC and PNEC to be consistent and compatible. Thus, to compare to a $PNEC_{seawater}$ derived from dissolved aluminium, *in situ* measurements should be taken on filtered seawater. To compare this to a $PNEC_{seawater}$ derived from isomeric monomeric aluminium (most bioavailable form of aluminium), *in situ* measurements can be performed using DGT, or filtered seawater as these fractions are considered as at least upper than the isomeric monomeric aluminium, thus it ensures a protective approach. The predicted concentrations of aluminium released by anodes should also be supported by laboratory experimentations to validate the model, for example by monitoring aluminium releases using DGT (measurements of the labile fraction) and spot water sample analysis (measurements of the dissolved fraction).

We also recommend continuing the work begun by assessing the risk for environmental compartments not considered within the ANODE project: sediment (for the protection of benthic organisms) and biota (for the protection of top predators).

2

Risk assessment

Recommandations for the quantative assessment of metal inputs in the marine environment from the galvanic anodes of offshore renewable energy structures

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3 - Recommendations for new implementation areas

From this project, it appears that the different impact studies do not fully answer the question of anode releases. Indeed, most of these studies do not include measurements of the initial concentrations of aluminium or other released elements in the area. Moreover, the numerical modelling work performed in this project is not systematically performed in all impact studies. The influence of hydrodynamics on metal dispersal is therefore not known.

To complete the impact study for the future park, this report suggests implementing the above elements and the following sections describe a proposed methodology.

3.1 Initial state characterisation – Measurement protocol

To characterise the initial state of the concentration of metal, analysis should be performed both on spot water samples and by a passive sampling technique. The methodology for sampling seawater for metal analysis is described in the AQUAREF document (Amouroux & Claisse, 2015). The passive sampling technique method is described here for the sampling of some metals present in galvanic anodes. No method is available for the elements released from ICCP systems. Concentration measurements of Metallic Trace Elements (MTEs) in dissolved form in the water column will be taken using the DGT technique which is used to obtain temporal integrated concentrations and gives a good evaluation of the bioavailability of the measured MTE (see Figure 8). This approach was developed by Ifremer and has already been applied to contribute (in controlled conditions in an experimental device) to the impact assessment of most elements released from galvanic anode dissolution on the chemical contamination of the marine environment (Deborde *et al.*, 2015).

The aim of the DGT measurements will be to monitor the evolution of the chemical quality of the seawater around the area of a farm protected by galvanic anodes composed of aluminium, zinc and other trace metals.
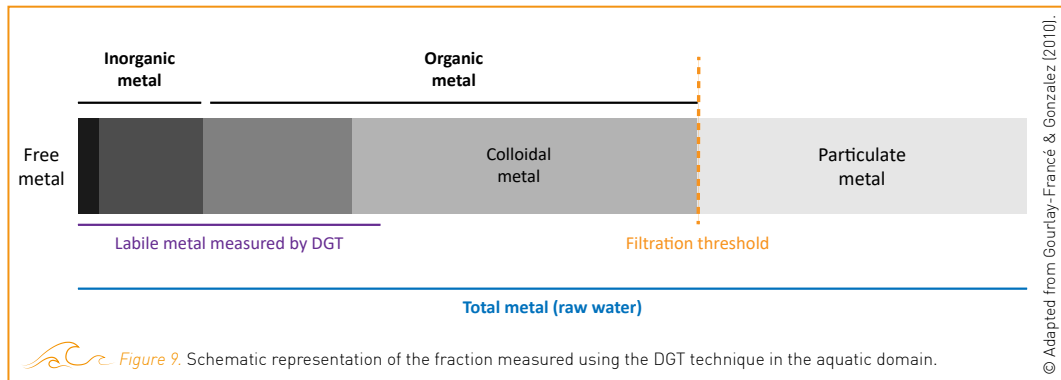

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Figure 8. Illustration of water spot sample and DGT retrieval

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New implantation areas

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Description of the method

Currently, the DGT technique can be used to measure the concentrations of many metallic trace elements (Al, As, Cd, Co, Cr, Cu, Hg, Ni, Pb, Sn, Zn...) in the marine environment at low levels (see Figure 9). For a given MTE, the measured concentrations are representative of one part of the "dissolved" fraction which is the most bioavailable for phytoplankton (the so-called "labile" fraction: hydrated ions, mineral complexes, "small" organic complexes).This technique was described by Davison & Zhang (1994) and by Zhang &

Davison (1995). Vidéos tutorials are also available at Ifremer website (Gonzalez *et al.*, 2020). The most "labile" dissolved metallic cations are accumulated on the resin and depend on the metal concentration in water and on the immersion time of the DGT (a few hours to several days). DGT passive samplers could be immersed anywhere in the water column but the device has to stay immersed (tides shall be taken into account) and the DGTs must not scrape the sediment.

Figure 9. Schematic representation of the fraction measured using the DGT technique in the aquatic domain.

© Adapted from Gourlay-Francé & Gonzalez (2010).

Back in the lab, after the recovery (1 to 2 weeks), DGTs are processed under "ultra clean" conditions. The resin is collected and eluted (HNO$_3$ 1M). The contaminant mass accumulated into the resin is measured by Inductively Coupled Plasma coupled with Mass Spectrometry (ICP-MS). With the immersion time and water temperature, it is then possible to calculate the MTE concentration in the seawater ("labile" species) (Davison & Zhang, 1994). The processing of the DGTs (resin recovery, elution) and analysis are to be performed by Ifremer's Biogeochemistry and Ecotoxicology research unit and especially the Biogeochemistry of Metal Contaminants laboratory.

Description of the measurement protocol

Measurement of the time-integrated concentration of the MTE under "labile" form by the DGT technique: aluminium, zinc and other trace metals which compose the anodes installed on the future wind turbine foundations (see Figure 10).

- DGT immersed in triplicate (size of the device: about 10 x 20 cm; weight of less than 30 g).

- Duration of the immersion period: 1 to 2 weeks. Ideally at the same time of year. Plan 2 measurements during each campaign year: one associated with low river runoff and another with strong flow. Additional measurement associated with extreme weather events.

- Time of the campaign:

  o Control 5 years after commissioning

  o A campaign prior to installation to determine the initial state of the area

  o A campaign 1 year after commissioning, and another one year after if an effect was noted

- Mooring: one DGT device close to the surface and another close to the seabed linked to a non-metallic weight and maintained below the sea surface with a surface or a subsurface float. The device could also be fixed to an immersed point (e.g. a foundation).

- Measurement locations: a point located close to a wind turbine foundation at the centre of the farm and another location outside of the potential impact area as a reference.

Recommandations for the quantative assessment of metal inputs in the marine environment from the galvanic anodes of offshore renewable energy structures

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REPLICATION

- DGT immersed **in triplicate**
- Size of the device: about 10 x 20 cm
- Weight of the device: ‹ 30 g

DURATION

- Immersion period:
  1 to 2 weeks
- Same periods of the year

SCHEDULE

- **Before the installation:**
  1 campaign to determine the initial state

- **After the commissioning:**
  o 1 year after, 2 years after if an effect was noted the previous year, 5 years after to control
  o 2 measurements during each campaign year (1 associated with low river runoff and 1 with strong flow) + 1 additional measurement associated to extreme weather events if necessary

WATER DEPTH

- 1device close to **the surface**
- 1 device close to **the seabed** linked to a non-metallic weight and maintained under the sea surface by a surface or a subsurface float. The device could also be fixed to an immersed point (e.g. a foundation).

LOCATION

- 1 point located close to a wind turbine foundation **at the centre of the farm**
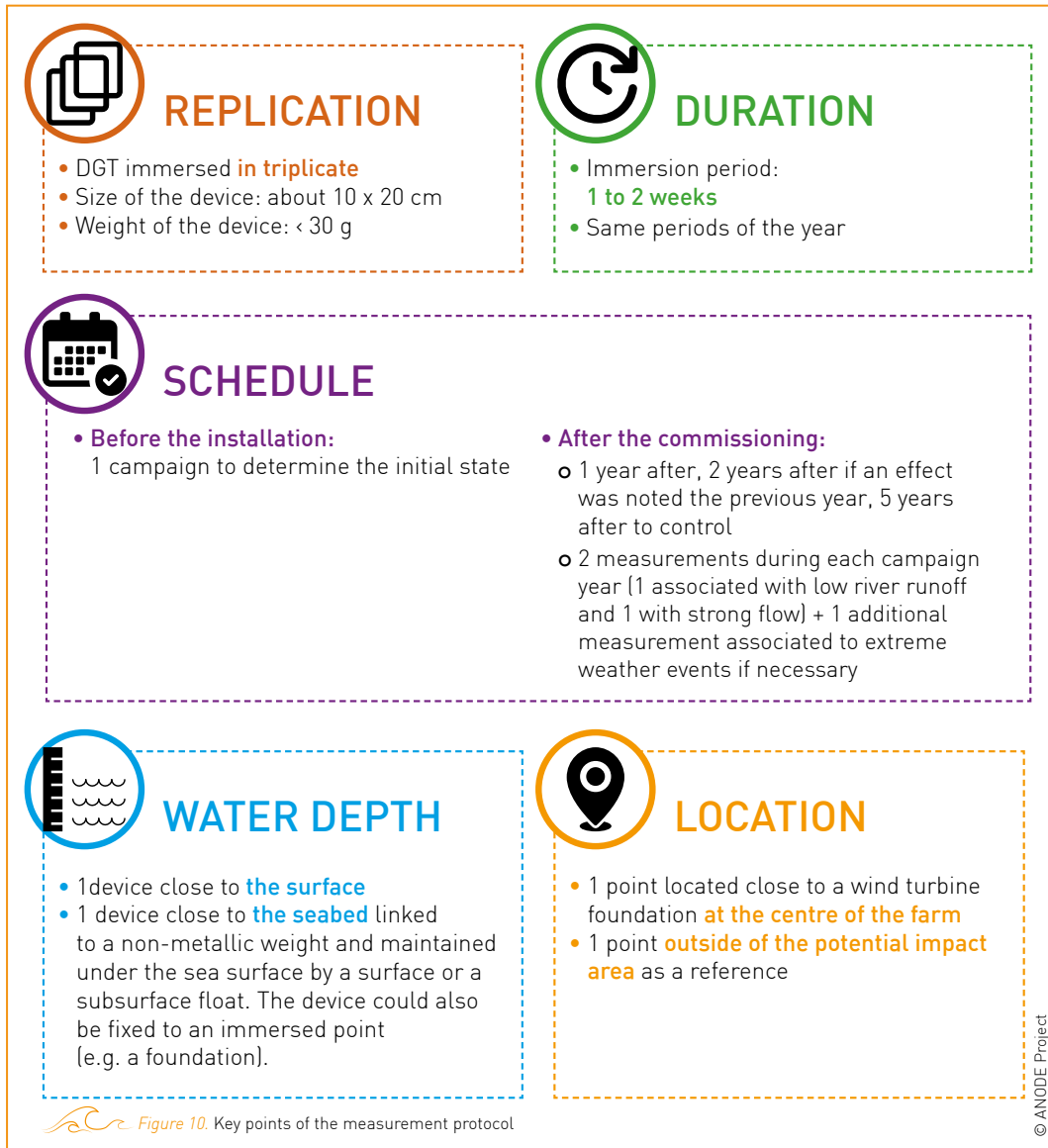- 1 point **outside of the potential impact area** as a reference

Figure 10. Key points of the measurement protocol

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New implantation areas

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Recommandations for the quantative assessment of metal inputs in the marine environment from the galvanic anodes of offshore renewable energy structures

3.2 Study of the dispersal of the chemical component by numerical modelling

After this physical protocol, the authors also recommend performing numerical modelling of the dispersal of metallic elements released by galvanic anodes. In other words, it is recommended to perform a study similar to the ANODE project for all future farms. Indeed, the hydrodynamics of the farm area play a major role in the dispersal of these elements and could vary between locations especially along the coasts of France. Moreover, the previously outlined protocol also recommends performing a measurement at a location outside of the potential impact area. Therefore, a numerical investigation would be necessary in order to, at least, determine the size of this area. In this report the main steps of the numerical methodology followed are described to provide an example for future developers (see Figure 11).
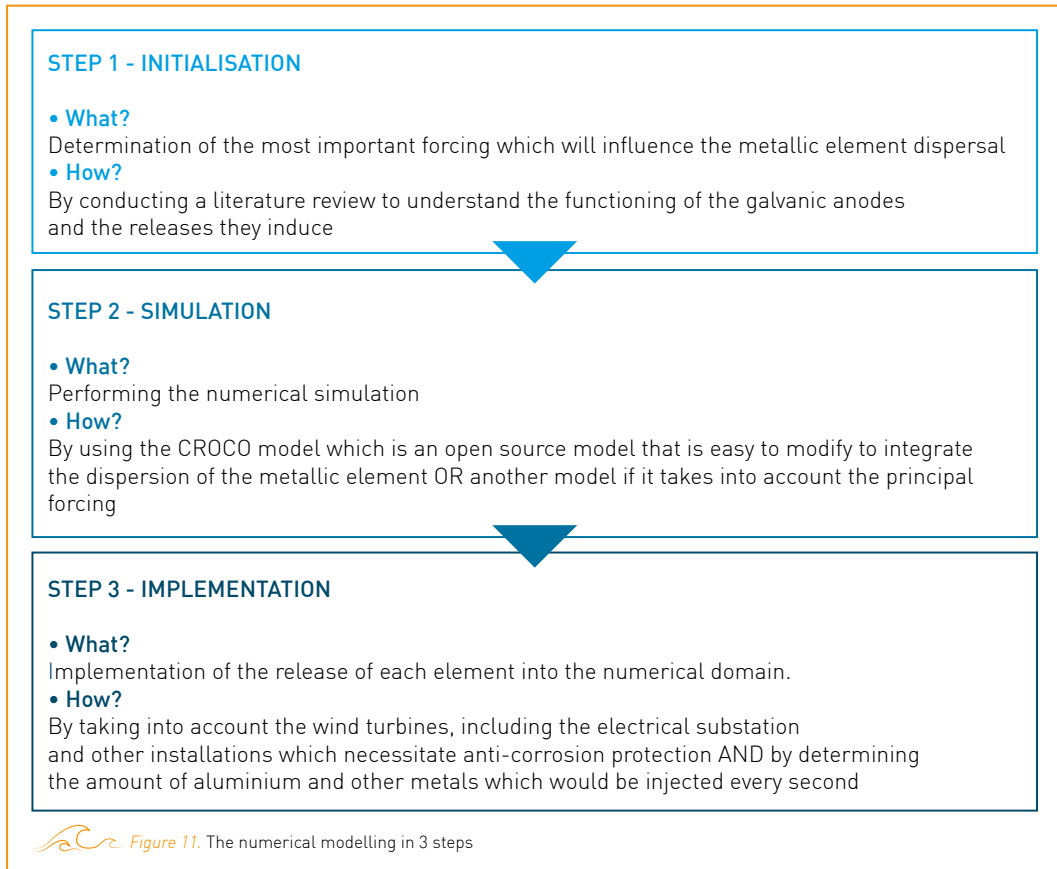
The first step is to determine the most important forcing which will influence the metallic element dispersal. For this a simple literature review is suggested in order to understand the functioning of the galvanic anodes and the releases they induce. To perform the numerical study, the CROCO model was selected for the numerical simulation as it is an open source model that is easy to modify to integrate the dispersion of the metallic element. However, another model could be used if it takes into account the principal forcing. Here the most important forcing was identified as the tides only for the Atlantic English Channel facades, and as the wind, river runoff and solar flux for all French facades. Wave action could also be included as it is expected to have an important role in vertical mixing.

The final step is obviously to implement the release of each element into the numerical domain. In this report, the study performed on the Courseulles-sur-Mer site takes into account the 75 wind turbine structures initially planned, although this number has since been reduced to 64 devices. However, to get closer to reality, it would be interesting to also include the electrical substation (if there is one) and other installations which require anti-corrosion protection. Therefore, we could say that the results presented here are more conservative than the reality. To implement the releases from the galvanic anodes installed on each structure, the amount of aluminium and other metal which would be injected every second needs to be determined. The method to determine this mass per second is based on the NF EN 12496 (AFNOR, 2013) and on DNVGL-RP-B401 (2017).

In the model, each element is taken into account as a passive tracer, which means that the hydrodynamics of the area would have an influence on dispersion but the concentration of these elements would not be affected by the current magnitude, direction or other parameters linked to the hydrodynamics. It also means that all released metallic elements would be considered as dissolved into the seawater. Readers should note that it is an approximation to consider the worst-case scenario which is sufficient to determine whether an ecotoxicological risk is present or not.

The mass of aluminium and other metals is defined to be injected every three-dimensional time step (20 or 30 seconds depending on the configuration in this project). Therefore, at every time step, a certain mass of metal (corresponding to the degradation of the anodes installed on one wind turbine foundation) is diluted at each wind turbine location by dividing this value by the cell volume. Therefore, for a model which divides the water column following a sigma coordinate system, this volume would vary every time step as a function of the sea surface elevation.

Recommandations for the quantative assessment of metal inputs in the marine environment from the galvanic anodes of offshore renewable energy structures

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STEP 1 - INITIALISATION

• What?
Determination of the most important forcing which will influence the metallic element dispersal
• How?
By conducting a literature review to understand the functioning of the galvanic anodes
and the releases they induce

STEP 2 - SIMULATION

• What?
Performing the numerical simulation
• How?
By using the CROCO model which is an open source model that is easy to modify to integrate
the dispersion of the metallic element OR another model if it takes into account the principal
forcing

STEP 3 - IMPLEMENTATION

• What?
Implementation of the release of each element into the numerical domain.
• How?
By taking into account the wind turbines, including the electrical substation
and other installations which necessitate anti-corrosion protection AND by determining
the amount of aluminium and other metals which would be injected every second

Figure 11. The numerical modelling in 3 steps

3

New implantation areas

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Recommandations for the quantative assessment of metal inputs in the marine environment from the galvanic anodes of offshore renewable energy structures

4 - General conclusion

The ANODE project developed a methodology for assessing the chemical risk of wind farms by combining a hydrodynamic model with hazard data. The modelling was performed using the CROCO model for the study sites of Courseulles-sur-Mer (English Channel), Groix & Belle-Île (Atlantic Ocean) and Leucate (Mediterranean Sea). The model was modified in order to include the heavy metal releases from galvanic anode degradation (due to their functioning). The aim of the simulations in each site configuration was to estimate the concentration of each metal component of the anodes released into the environment. To reach this objective, the first step of this work was to research the main features and forcing driving the local hydrodynamics and the residence time of the water masses in the domain. Considering the main hydrodynamic processes on the study site and the residence time, the optimal duration of the simulation could then be estimated. In the case of Courseulles-sur-Mer, presented in this document, the hydrodynamics are mainly driven by the tides and also influenced by the wind and Seine runoff. A 5-year simulation was then considered sufficient to estimate the different dispersion scenarios. Several validations of the hydrodynamic model results were performed on the free-surface elevation, the current direction and amplitude, as well as over the density gradient through comparison with different data sources, namely *in situ* measurements, satellite observations and other numerical models. The heavy metal releases were included in the numerical code and the concentration level that could be reached was estimated. Globally the results show that an upper bound value of the concentration of each metal emerges in each point of the domain but strong variations around a median value were observed over short time periods.

The study performed in this project is considered sufficient to obtain a first estimation of exposure associated with the metals released by the galvanic anodes. A similar approach with this model is recommended for the future farm. However, it is necessary to improve this approach with *in situ* chemical measurements. Several measurements require to be performed in order to determine the initial condition of the heavy metal concentrations at the study site location and estimate the contribution of the galvanic anode to the environment. The recommendation specifies that these measurements should be performed before the installation, one and five years after it and at different location. However, it is also recommended to perform measurements during different hydrodynamic scenarios such as low tidal currents or strong wind which could be identified by the hydrodynamic model. These measurements would then be useful to estimate the initial state and refine the PEC for the risk assessment, as well as to validate the heavy metal dispersion model.

Based on available data, the chemical risk assessment performed identifies a risk for aluminium for the species living in the water column. However, in the same way as for the PEC, the PNEC needs to be refined. To refine the $PNEC_{seawater}$, it is recommended to conduct standardised experimental studies in laboratory conditions for the acquisition of ecotoxicity data for aluminium on marine organisms, especially on algae, fish and crustaceans. In addition, a field study could be planned in order to measure the metal concentration associated with anode dissolution in the seawater, sediment and biota (exposed caged organisms) and to measure the bioaccumulation of aluminium and biomarkers/bioassays of effects. There are few bioassays or biomarkers specific to the metals except methalothionein but a list of some bioassays and biomarkers can be selected from the ICES and OSPAR (Davies & Vethaak, 2012) and/or OECD literature. The final risk assessment for aluminium (3rd iteration) can only be achieved once these detailed data are acquired.

Finally, the work begun should be pursued by assessing the risk for environmental compartments not considered within the ANODE project: sediment (for the protection of benthic organisms) and biota (for the protection of top predators). The potential cumulative impacts in terms of chemical contamination of wind farms on marine wildlife health deserve to be estimated and monitored over the long term.

4

General conclusion

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4

General conclusion

5 - Acronyms

ADCP = Acoustic Doppler Current Profiler
AF = Assessment Factor
CMR = Carcinogenic, Mutagenic, Reprotoxic
CROCO = Coastal and Regional Ocean COmmunity model
DGT = Diffusive Gradient in Thin films
EC = Effect Concentration
ECHA = European CHemicals Agency
ETOX = Information System Ecotoxicology and Environmental Quality Targets
GACP = Galvanic Anode Cathodic Protection
GLP = Good Laboratory Practice
ICCP = Impressed Current Cathodic Protection
ICES = International Council for the Exploration of the Sea
ICPMS = Inductively Coupled Plasma-Mass Spectrometry
INERIS = Institut National de l'Environnement industriel et des RISques
LBCM = Laboratoire de Biogéochimie des Contaminants Métalliques
MSFD = Marine Strategy Framework Directive
MTE = Metallic Trace Element
NOEC = No Observed Effect Concentration
OECD = Organisation for Economic Co-operation and Development
ORE = Offshore Renewable Energy
OSPAR = Convention for the Protection of the Marine Environment of the North-East Atlantic
OWF = Offshore Wind Farm
PEC = Predicted Environmental Concentration
PHS = Priority Hazardous Substance
PNEC = Predicted No-Effect Concentration
RBSP = River Basin Specific Pollutant
RCR = Risk Characterisation Ratio
REACH = Registration, Evaluation, Authorisation and Restriction of Chemicals
SSD = Species Sensitivity Distribution
UK TAG = United Kingdom Technical Advisory Group
US EPA = United States Environmental Protection Agency

5

Acronyms

Recommandations for the quantative assessment of metal inputs in the marine environment from the galvanic anodes of offshore renewable energy structures

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Bibliography

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In the context of the development of offshore wind turbine projects in France, the potential environmental impact of cathodic protections, and particularly of Galvanic Anode Cathodic Protection (GACP) systems, raises a real concern for public authorities and civil society. In order to effectively address this issue, a combined study comprising numerical modelling and an ecotoxicological study was performed on three different study sites. In order to cover different hydrodynamic conditions (*i.e.* Atlantic/English-Channel vs Mediterranean conditions), technologies (*i.e.* floating vs fixed offshore wind farms) which could also influence the dispersal of these elements and anticorrosion protection system, the site of Courseulles-sur-Mer, Groix & Belle-Île and Leucate sites were considered. The dispersion of the metals released by the galvanic anodes or of the chlorine-based elements from the use of Impressed Current Cathodic Protection (ICCP) systems was then estimated by a fully three-dimensional numerical approach using the Coastal and Regional Ocean Community model (CROCO). The detailed results for the Courseulles-sur-Mer site were considered in an ecotoxicological study which, following the REACH (Registration, Evaluation, Authorisation and Restriction of Chemicals) technical guidance, consisted in a risk assessment of galvanic anodes in the environment using the available data on the toxicity thresholds of each element composing the anodes. While these results allow the potential impact of the galvanic anodes to be estimated, this combined study also led to several recommendations for future farms in order to improve the toxicity threshold determination, the initial state of concentration and the dispersion estimated by the numerical model.

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