Review

Emissions from corrosion protection systems of offshore wind farms: Evaluation of the potential impact on the marine environment

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ABSTRACT

Offshore wind energy is a fast growing sector of renewable energies worldwide. This will change the marine environment and thus, a wide range of environmental impacts of offshore wind farms are subject of current research. Here we present an overview about chemical emissions from corrosion protection systems, discuss their relevance and potential impact to the marine environment, and suggest strategies to reduce their emissions. Corrosion is a general problem for offshore infrastructures and corrosion protection systems are necessary to maintain the structural integrity. These systems are often in direct contact with seawater and have different potentials for emissions, e.g. galvanic anodes emitting substantial amounts of metals. Organic coatings may release organic substances due to weathering and/or leaching. Current assumptions suggesting a low environmental impact, but monitoring data is not sufficient to assess the environmental impact of this new source.

1. Introduction

1.1. Political background and current situation of offshore wind energy in Europe and Germany

The goal of the Paris Agreement to limit the temperature increase well below 2°C was negotiated at the meeting of the United Nations Framework Convention on Climate Change (COP21, Paris 2015). In order to achieve this goal, the global greenhouse gas emissions have to be decreased significantly in the next decades. The reduction of greenhouse gas emissions by the increase of the use of renewable energy is one of the key strategies to mitigate global warming. The European Union renewable energy directive has the target to reach a contribution of at least 20% of renewable energy to the total energy consumption in 2020 and 27% in 2030 (European Commission, 2016a). This directive was transformed in national specific energy targets which range from 10% of renewable energy for Malta to 49% for Sweden. The climate action plan of the Federal Government of Germany aims to reduce the emissions of greenhouse gases by 40% until 2020 and by 80% to 95% in 2050, compared to emissions in 1990 (BMUB, 2016; 2014). The increase of renewable energy production and energy efficiency along with the reduction of energy consumption are the key elements used to reach this ambiguous goal. In 2017, the contribution of renewable energy to the gross electricity consumption in Germany was 36.2%. Offshore wind energy contributed only 8.2% to the total produced renewable energy, due to the fact that most of the offshore wind farms (OWF) were still under construction or not connected. Onshore wind energy is currently the leading renewable energy source in Germany with a contribution of 40.7% to total renewable energy production, followed by solar energy (18.3%) and biogas/biomethane (14.3%) (AGEE-Stat, 2018). For 2025, Germany aims to reach a contribution of renewable energy of around 40–45% to the total energy production and offshore wind energy is one important technology to reach this goal. Germany plans to achieve an offshore wind energy capacity of 15 GW by 2030 (Offshore Wind Energy Act, 2017).

On a European scale in total 15.8 GW capacity are installed offshore in 2017, which corresponds to 4149 grid-connected offshore wind turbines (OWT) (WindEurope, 2018) and an increase is expected for the next decades (Fig. 1). Based on the number of grid-connected wind turbines, United Kingdom (1753 turbines) is currently in the lead, followed by Germany, Denmark and the Netherlands. The major part of the OWF is located in the North Sea (72%). If compared to the current status of the offshore gas and oil industry (1350 facilities), a significantly higher number of offshore constructions in relation with wind
energy production can be expected in the future and will significantly increase the total number of offshore facilities in the OSPAR (Convention for the Protection of the Marine Environment of the North-East Atlantic) region and worldwide (OSPAR Commission, 2015). This demonstrates the dimension of offshore wind energy development, especially in the North Sea region (see Fig. 2). In 2017, 7.5 bn € (18.2 bn € in 2016) were invested for offshore projects in Europe (WindEurope, 2018). Also offshore wind energy is globally increasing. China reached already 2.8 GW in 2017 when the total global offshore wind energy capacity was 18.8 GW, demonstrating that Europe is currently leading in offshore wind energy (GWEC, 2018). This might change in the future, because several other countries are planning offshore wind projects (e.g. only the USA offshore wind project development pipeline includes 24.1 GW of potential installed capacity (Musial et al., 2016); and China wants to reach 30 GW by 2020 (Hong and Möller, 2012), which will increase the number of offshore wind farms rapidly (Poudineh et al., 2017).

In 2010, the first German offshore wind farm “Alpha Ventus” (54° 0’ 30” N, 6° 35’ 54” E) started its operation. Currently (as of 6/2018) already 17 wind farms are installed in the German North and Baltic Seas; six are currently under construction. The majority of these wind farms are situated beyond the German territorial waters in Germany’s exclusive economic zone (EEZ) in the North Sea. In total, 1138 OWT are already installed or under construction in the North Sea and 232 OWT in the Baltic Sea representing a total power of 4695 MW and 692 MW of the installed OWT, respectively. To achieve an effective energy conversion and transmission to the mainland via subsea-cables, further offshore facilities are consequently constructed. Thus, 23 offshore substations (OSS) and 8 converter platforms (HVDC stations) are installed or under construction in German marine waters.

1.2. Offshore wind energy

The major advantage of offshore wind energy compared to onshore wind energy is the increased availability of the resource “wind”. Although offshore wind is cost intensive in planning, design, construction, operation and maintenance, compared to onshore, it is economically very attractive. Higher speed and continuous wind allows a high percentage of full-load hours. Furthermore less wind turbulences in the offshore areas may increase the lifetime of wind turbines (Bilgili et al., 2011; Esteban et al., 2011). The offshore and onshore wind turbine capacity is still comparable, but future developments will allow...
for higher turbine capacity offshore. Currently high-capacity 8 MW turbines are planned to be installed in ongoing projects, but 9–12 MW turbines are already announced by different producers. Furthermore, if compared to the onshore wind farms, OWFs are constructed far away from populated areas which reduce noise emissions and visual impacts for the population. The majority of the European OWFs are constructed in shallow waters in depths between 10 and 50 m (average 27.5 m, 2017) and in a distance of up to 100 km (average 41 km, 2017) from the coast (WindEurope, 2018). The currently used foundation designs are tripods, jacket foundation, gravity foundation or monopiles (Hammar et al., 2010), whereby the latter one is with around 87% (2017) the most frequently used technique in the German EEZ and in Europe (WindEurope, 2018). Several new developments try to improve these foundations designs for application also in greater depth. An additional approach is floating platforms which are in focus of current research (Orth and Becker, 2016). First pilot projects have already been started (e.g. Norway Hywind Demo, Onstad et al., 2016) and further projects are planned in e.g. Scotland (Hywind Scotland, Kincardine, Dourey Tri), Portugal (WindFloat Atlantic, Pacheco et al., 2017) or France (Leucate zone, Faraman area). This will allow the construction of offshore wind farms in depths of > 50 m and will further increase the potential of offshore wind energy worldwide (Negro et al., 2014). However, that means more complex structures (subsea-cables and offshore facilities like OSS, and HVDC platforms) for the energy transfer to land.

1.3. Environmental issues

Offshore wind energy is a new and fast growing industry and a new anthropogenic pressure for the marine environment. Long-term effects are currently unknown; especially in small and shallow seas where the area consumption, like the North Sea and Baltic Sea, is high. To date about 850 km² are used by OWFs or OWFs under construction only in Germany. Although it is considered a “clean/green” energy source, offshore wind energy has an environmental impact. Understanding this environmental impact is important for the Marine Strategy Framework Directive (MSFD) (European Commission, 2008) of the European Union which includes the target of a good environmental status for the marine environment. To assess the environmental impact of OWF environmental impact assessments are mandatory during windfarm approvals and national frameworks Guidance or guidelines are established or under development (e.g. the German StUK, Standard for Environmental Impact Assessment (BSH, 2013); UK: PD 6900:2015 Environmental impact assessment for offshore renewable energy projects – Guide (British Standards Institution, 2015)). In addition, OSPAR published recommendations for an environmental impact assessment for offshore wind farms (OSPAR on Environmental Considerations for Offshore Wind Farm Development). Furthermore, several research projects accompanying wind farms. In Germany, the impact of OWFs on the marine environment was investigated in the pilot offshore wind farm “alpha ventus” (Dursteiwitz and Lange, 2017; Fischer et al., 2015; Wiesen et al., 2013) and discussed in several reviews and specific studies (Bailey et al., 2014; Kaldellis et al., 2016; Lindeboom et al., 2015, Lindeboom et al., 2011; OSPAR Commission, 2008a, 2008b; Garstensen et al., 2006; Riefolo et al., 2016; Saidur et al., 2011; Willsteed et al., 2017; Wilson et al., 2010), but is still not completely known. Most of this research is focused on the impact on benthos and pelagic species (Hammar et al., 2014; Krone et al., 2017; Scheidat et al., 2011), influences on the food web (Raoux et al., 2017), birds and their migration routes (Busch et al., 2013; Christel et al., 2013; Cleasby et al., 2015; Cook and Robinson, 2017; Garthe et al., 2017; Masden et al., 2012; Vanermen et al., 2015), noise emissions during construction and operation (Bailey et al., 2010; Dolman and Jasny, 2015), impact on ecosystem services (Hooper et al., 2017; Klain et al., 2018), and the overall carbon footprint (Kaldellis and Apostolou, 2017). Besides the environmental impact, first studies evaluate the possibility to use OWFs for aquaculture (Buck et al., 2017; Buck et al., 2010; Van Den Burg et al., 2017; Di Tullio et al., 2017; Jansen et al., 2016), biofouling harvesting as source for nutritional material (Weiss and Wittke, 2012), and as resettlement areas of former North Sea species like the lobster (Krone et al., 2013) and the European oyster (Gerken and Schmidt, 2014).

Compared to the previously mentioned studies and reviews about the different ecological aspects and pressures on the marine environment, less is known about the chemical emissions from OWF during construction, operation and decommission and their impacts which might be also important for the environmental status of the sea and/or sea food quality. Potential chemical emissions from the offshore wind industry may originate from: the increased traffic and an increased risk of accidental spills from this traffic, the re-mobilization of contaminated sediment due to seabed disturbance by subsea cable and foundation construction, discharges from waste water treatment plants and cooling water from platforms, artificial scour protection materials, atmospheric emissions from diesel generators, direct chemical emissions and spills during accidents (e.g. fire on platform and the use of firefighting foams or accidental spills of oil, lubricants or coolants). However, the risk of emissions or accidently release of these chemicals can be substantially reduced by using constructive preventive measures such as backup systems, secondary containments, closed loop systems, and recovery tanks. Recently, Tornero and Hanke (2016) (Tornero and Hanke, 2016) presented a generic list of potential chemicals released from offshore wind energy facilities, which includes aluminium, copper, zinc, iron, diuran, irgarol, hydrocarbons (BTEX, PAHs), silicon fluids, mineral oils, (bio-) diesel, vegetable oils, synthetic esters, ethylene glycol, propylene glycol, and sulfuric acid. There is no data available about the total amount of chemical emissions from OWFs, but they are supposed to be very low compared to chemical emissions from fossil energy related offshore platforms (e.g. UK oil & gas industry reported 102,500 t of chemical emissions in 2015 (Oil&Gas UK, 2016)). This review focuses on an additional source of chemical emissions by offshore constructions to the marine environment: corrosion protection systems and their emissions. These systems are essential to protect the steel construction against corrosion in the harsh marine environment during their service life time and are an important factor for the long-term stability of the offshore structure. We discuss the following corrosion protection systems in detail concerning their potential for chemical emissions: a) cathodic protection systems using galvanic anodes or impressed current cathodic protection systems and b) corrosion allowances (“thicker steel”) and c) coatings (e.g. epoxy resins and polyurethane). This review gives insight into the technical aspects of the protection systems and the advantages and disadvantages of each system. Furthermore, it will critically discuss these emissions, their distribution, and potential impact on the marine environment. This review gives an overview about potential substances, but cannot provide a detailed environmental impact assessment for all compounds. We identify knowledge gaps, address different needs for research and monitoring activities to increase the awareness of the environmental impact of corrosion protection systems, and suggest strategies to reduce their emissions.

2. Corrosion and corrosion protection

2.1. The problem of corrosion in the marine environment

The exposure to the harsh marine environment is one major challenge for the construction and duration of offshore structures, which are mainly made of steel. OWT, OSS and converter platforms are designed for at least 25 years of lifetime. The chemical properties of the immersion medium, i.e. sea or brackish water, have a strong influence on the corrosion processes of metals. Seawater is very corrosive compared to drinking water, as corrosion is increasing with the salinity of seawater, but also parameters such as oxygen concentration, pH (seawater 7.8–8.3), and temperature affect corrosion processes (Adedipe et al.,
2.2. Technical and environmental regulations

Technical requirements for the corrosion protection systems are provided by national and international standards (e.g. DNVGL-RP-0416 (2016), DNVGL-RP-B401 (2017), NORSOK (e.g. M-501, 2012), VGB/BAW-Standards Part 1–4 (VGB-S-021-01-2018-04-EN, 2018; VGB-S-021-02-2018-04-EN, 2018; VGB-S-021-2018-03-04-EN, 2018; VGB-S-021-04-2018-07-DE, 2018)) and national regulations for offshore infrastructures to assure the use of the best available techniques. However, environmental issues are not in the focus of these technical standards and are regulated by the national authorities. In Germany, environmental requirements for OWT and their corrosion protection systems are generally regulated in e.g. the Standard Design – Minimum requirements concerning the constructive design of offshore structures within the Exclusive Economic Zone (BSH, 2015). This includes a “zero emissions/discharge” policy and some basic environmental requirements: corrosion protection systems should have the lowest possible level of pollutants and that the use of tributyltin is prohibited. OSPAR stated in their guidance that all chemicals, paints, coverings etc. used in the construction should be approved for use in the marine environment and their ecotoxicological properties should be known (OSPAR Commission, 2008a, 2008b). In addition, the VGB/BAW standard for offshore wind turbines is regulating the requirements for e.g. coatings processes. The coatings have to be applied onshore to avoid direct emissions by dripping or other losses of material. This means also, that coatings are cured and hardened, when installed offshore. However, offshore repair and maintenance work may result in small direct emissions of coating material.

2.3. Corrosion protection systems

2.3.1. Cathodic protection systems

Cathodic protection systems are the most common used techniques for all kinds of steel-constructions like ships and infrastructures in marine and harbor environments. Hereby the electrical potential of the metal structures will be reduced to slower down oxidation processes. This can be realized by galvanic “sacrificial” anodes (GACP) or inert anodes (ICCP) which provide the protection current for the polarization of the metal surface by either the galvanic reaction of a less noble material compared to the steel structure (GACP) or the active current (ICCP).

For offshore structures Al-based galvanic anodes are mainly used, due to its high electrochemical capacity (2000 Ah kg\(^{-1}\)) in seawater and low weight, compared to zinc (780 Ah kg\(^{-1}\)) or magnesium anodes (DNVGL-RP-B401, 2017; HTG, 2009). The advantages are the easy installation even for complex foundations like jackets and the low maintenance efforts. The disadvantages are that a high amount of anode material is necessary and replacement in case of malfunction under marine conditions is difficult. However, to protect the steel structures of OWT, and substations- or converter platforms from corrosion, several tons of anode material often is necessary. The amount of material depends on the foundation type (different surface areas), the desired life time (e.g. at least 25 years), the seawater conditions, and whether the GACP is combined with other techniques. The annually necessary material and thus related emissions from galvanic anodes are in the range of several kilos (for e.g. monopile foundations) to tons per year (for huge jacket structures of platforms) (see calculation in the next section). Depending on its specific alloy, the producer, and applied standard, Al anodes also contain Zn (2.5–5.75%) and In (0.015–0.04%). Indium is necessary for activation and to avoid passive oxide films on the surface, while the other trace metals remain as impurities from manufacturing (Smith and Goosley, 1996). These impurities are e.g. Cd, Mn, Fe, Si, Pb, Cu, and other trace metals (DNVGL-RP-B401, 2017) and were reported from laboratory analyses of the anodes material (Voigt et al., 2017).

Impressed current cathodic protection is an emerging technique in

offshore wind farm construction that actively protects the submerged zones from corrosion and is already used for ships and waterway infrastructures. In contrast to galvanic anodes, where the protection current is provided through the difference in the electrochemical reduction potential of the materials (i.e., galvanic anodes are “less noble”), ICCP systems deliver direct current (DC) to the steel surface by using chemically inert anodes. The advantages of these systems are the provision of high amounts of protection current per anode and the possibility to be remotely monitored and actively regulated. Moreover, they do not have any significant emissions. However, there are still concerns about higher maintenance and inspection costs and risk of damages, compared to GACP. Currently used anode materials are titanium, iridium or iridium/ruthenium-MOX coatings, magnetite or platinum with titanium, niobium, and tantalum coatings. ICCP can be used in combination with coatings. It is important to note that not all coatings are compatible with ICCP systems, as detachments problems can occur for some of the coatings (Fischer et al., 1991; Hillebrand and Schwenk, 1982; Mahdavi et al., 2017).

2.3.2. Corrosion allowance
The so called “corrosion allowance” as corrosion protection means using thicker steel as necessary for the construction. This can be seen as a backup solution, if conventional protection systems are not working, or to cover damages in coatings. Furthermore, corrosion allowance is necessary to cover time periods until the installation of the ICCP and the polarization of the steel. This can increase the lifetime of the OWFs if there are technical problems with ICCP or galvanic anodes (e.g., passivation). The thickness is calculated on the estimated corrosion rate of the steel alloy in the marine environment. The recommended corrosion allowance is between 0.2 and 1.2 cm (DIN 50929-3, 2018). The steel (according to DIN EN 10225-2009) is made mainly out of iron, but also contains traces between 0.01 and 1.65% of C, Si, Mn, P, S, Cr, Co, Ni, Mo, V, Al, and Cu.

2.3.3. Epoxide resin and polyurethane coatings
Epoxied resins (EP) and polyurethane (PUR) based coatings are a state of the art technique for corrosion protection in a wide range of marine applications (shipping, harbors and offshore infrastructures) and are an artificial barrier to separate the steel from the corrosive environment (Lyon et al., 2017; Price and Figueira, 2017). These coatings can be applied in different parts of the OWTs, where the areas with direct contact with water (SWZ, TWZ, and UWZ) are the most relevant for potential chemical emissions to the seawater. They are mandatory for the tidal and the wave affected zones and normally in this section they also contain the yellow color (RAL 1023) necessary for marine traffic security reasons in e.g. Germany. In the submerged zone coatings can be used in addition to galvanic anodes or ICCP systems. The combination reduces the necessary anode material, but not all coatings are compatible with ICCP systems as mentioned before (Fischer et al., 1991; Hillebrand and Schwenk, 1982; Mahdavi et al., 2017). Coatings with anti-fouling agents are normally not used for offshore wind turbines. There are several new developments, like self-healing coatings or nano-coatings, but they are currently not applied in the offshore sector due to the harsh environmental conditions. Most of the OWT currently are not coated in the UWZ but if coated, epoxy resins are used. In the SWZ and TWZ (for monopile the transition peace) often combination of different coatings are used. It is important that the different coating layers are compatible. Mombert and Marquardt (2018) providing an overview of different coatings applied in German Offshore Wind farms. They may consist of a priming layer (e.g. EP-Zn or 1C/2C PUR–Zn, EP or EP) and 2–4 layers of intermediate or top coatings (EP or 1C/2C PUR). The number of layers and dry film thickness (up to 700 µm or more) of the coating depends on the applied type of coatings systems and which technical standard is referred to. Top coatings in the SWZ and TWZ are normally PUR, because their higher UV-stability compared to EP. In the UWZ, EP is mainly used as topcoat. The evaluation whether coatings can be applied for OWT is based mainly on their technical properties (e.g., mechanical resistance, weathering, and long-term exposure). However, it should be mentioned that in current certification processes for coatings the test methods are designed for a life time of only 15 years (DIN EN ISO 12944-1, 2017). The application of coatings for offshore infrastructures is regulated in different national and international standards: e.g., DIN EN ISO 12944-1 (2017), VGB-BAW-Standard (VGB-S-021-03-2018-04-EN, 2018), GfKorr (2016), NORSOK M-501 (2012) and ISO 12944-9 (2018). Recently, the use of a metallic (Al, 350 µm) coating applied by thermal spray aluminium (TSA) in combination with organic coatings was announced for a wind farm in the Baltic Sea as an alternative technique to avoid the use of anodes and thus reduce metal emissions.

3. Chemical emissions and environmental impact

3.1. Metals
The necessary anode material depends on the protection current which is needed for the steel structures. The calculation depends on several factors and technical requirements, like the foundation design, size, and whether the foundation is coated or at least partially coated, sea-water or sediment conditions, and anode distribution. Therefore, the anodes material varies between the OWF, as different solutions are necessary for the individual OWT, OSS and HVDC stations. The chemical emissions from GACP are similar to the amount of anode material, if we assume that the majority of the material will be consumed over the 25 years life time and enter into the marine environment. Therefore, the potential emissions by GACP can be roughly estimated by using e.g. the DNV-GL-RP-B401 (2017) calculations, one of the established standards. Table 1 summarizes these estimates for the different foundation designs and an OSS. These estimated amounts of Al-anode mass for non-coated founding designs ranging between 13,000 kg (monopiles) and 32,000 kg (tripod). An additional coating would reduce the total amount of Al-anode mass. In this case, only between 6000 kg (monopiles) and 10,700 kg (tripod) of Al-anode mass is necessary.

It is important to mention that these estimates do not include any additional factors, like the distribution of anodes over the entire structure which has to be considered for each individual part of the foundation design and may change the total amount of anodes. If we consider an OWF with 80 OWT with monopile foundations and an OSS and use the values given in Table 1, an OWF will release annually 45 tons of Al and 2 tons of Zn (if we assume 5% Zn). The use of Zn anodes would increase the total annual emissions by a factor of around 2.5 (118 tons), due to the lower current capacity and in consequence

<table>
<thead>
<tr>
<th>Surface in water (m²)</th>
<th>Monopile</th>
<th>Tripod</th>
<th>Jacket</th>
<th>OSS jacket</th>
</tr>
</thead>
<tbody>
<tr>
<td>850</td>
<td>2500</td>
<td>1800</td>
<td>12,000</td>
<td></td>
</tr>
<tr>
<td>850</td>
<td>300</td>
<td>180</td>
<td>1200</td>
<td></td>
</tr>
<tr>
<td>Total amount of anode material</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>a) Blank Steel (kg)</td>
<td>13,000</td>
<td>32,000</td>
<td>22,500</td>
<td>150,000</td>
</tr>
<tr>
<td>b) Coated in the water column (kg)</td>
<td>8000</td>
<td>16,000</td>
<td>11,000</td>
<td>80,000</td>
</tr>
<tr>
<td>c) Coated in the water column (kg)</td>
<td>6000</td>
<td>10,700</td>
<td>7500</td>
<td>50,000</td>
</tr>
</tbody>
</table>

a Assuming that coatings are completely lost after 15 years, as standards for coatings only assume a life time of 15 years.

b Assuming a total coating damage of 32% after 25 years (breakdown factor calculated with DNV GL standard values).
more material is necessary. In the case of Zn-anodes, 99% of the emissions are Zn. Although technically suitable, Zn-anodes are currently not used in offshore wind farms in the German EEZ, because the necessary higher number of anodes has constructive disadvantages. The use of coatings would significantly reduce the necessary anode material, and the emissions of an OWF with Al anodes can be reduced to 19–25 tons of Al per year, but currently the majority of the OWT are not coated in the submerged zone. It should be noted that these values are average emissions over the 25 years of life time, as they are generally higher in initiation phase to polarize the entire foundation. In addition, wind farms often have an OSS with larger foundations as those of OWTs and need significantly more anode material (roughly estimated emissions: 0.5–1 t yr⁻¹). The largest infrastructures for offshore wind farms are HVDC converter stations, where just the jacket steel structure has a total mass of several thousands of tons, hence a high amount of anode material is necessary (roughly estimated emissions: 5–15 t yr⁻¹). This will increase the total emissions of metals from a wind farm.

Some OWTs are using galvanic anodes in the inner part of monopiles or tripods whereby the suitability is still under discussion due to several concerns (Black et al., 2015). Monopiles and other foundation designs have only a limited or no exchange with the surrounding water. In consequence, the environmental conditions may change inside, including e.g. oxygen concentrations (change from aerobic to anaerobic conditions), reduction of pH values, and formation of hydrogen gas within the pile. These changes may influence corrosion processes and functionality of galvanic anodes inside the piles, e.g. decreased pH causing accelerating consumption of the anodes (Black et al., 2015; Delwiche et al., 2017). In addition hydrogen generation is causing the risk of explosions if piles are not ventilated. Furthermore, the sediment and water inside the piles will be enriched with Al, Zn, and other trace metals released from the galvanic anodes in the 25 years. The introduction of a high amount of low pH and highly metal-enriched water should be avoided, as it may have a negative impact on the marine environment. This should be considered at least during decommissioning of OWT (Grabe, 2017; Topham and McMillan, 2017).

Summarizing, the use of galvanic anodes outside and inside the offshore infrastructures will have a significant local input of metals to the marine environment over the lifetime of the OWFs. Here we will discuss the three major metals (Al, Zn, In) expected to be emitted from galvanic anodes. However it should be mentioned that galvanic anodes also contain other heavy metals like Pb or Cd in traces, which will also be emitted during the consumption.

3.1.1. Aluminium

Aluminium is the third most abundant element in the earth crust, because of its presents in clay minerals. In addition, Al concentrations in sediment in the German Bight are in the range of 6.2–7.7% (< 20 μm fraction, superficial sediment, German Bight, 2011, (BSH, 2016). Dissolved Al concentrations in the seawater are generally low with concentrations in a range from 0.008 to 0.68 μg L⁻¹ (open ocean), 0.5 to 0.68 μg L⁻¹ (coastal waters) and 0.6 to 0.9 μg L⁻¹ (North Sea) (Angel et al., 2015). Due to the natural pH conditions of seawater (pH = 8.1), the dominant Al species are the hydroxides Al(OH)₃⁻ and Al(OH)₆⁰⁰⁻ (Benezeth et al., 1997), but Al can also bond to Fluor, dissolved organic matter or are organically chelated. Environmental toxicity is known for fresh water algae and fish under more acidic conditions (Gensemer and Playle, 1999). Al concentrations are regulated for drinking water with e.g. 200 μg L⁻¹ in Germany (TrinkwV, 2001). To the best of our knowledge, there are no national or international guidelines or environmental quality assessment values for Al concentrations in seawater. Golding et al. (2015) proposed a no effect concentration (NOEC) of 24 μg L⁻¹ of total Al for an Australian marine water guideline, based on the results of toxicity test of 11 marine species from different trophic levels and demonstrated that dissolved and/or particulate Al can contribute to the toxicity depending on the investigated species. The most sensitive marine species in that study was one diatom species. The NOEC of the second most sensitive species (oyster) was already 4 times higher with 100 μg L⁻¹ (Golding et al., 2015). These NOEC values are significantly higher as measured concentrations of dissolved Al in the marine environment.

More specifically, a few studies investigated the effects of Al emissions from galvanic anodes, mainly in harbor environments or laboratory experiments (Caplat et al., 2010; Pinaeu et al., 2014; Gabelle et al., 2012; Mao et al., 2011). Gabelle et al. (2012) demonstrated a significant increase of Al concentration in the sediment in the vicinity of the anodes in a harbor basin (Le Havre, France), whereas water concentrations of Al did not show any increase due to dilution effects. Experiments with tanks filled with seawater without sediment by Deborde et al. (2015) showed high concentrations in the settled particles and in suspended particulate matter (SPM). They assumed that this enrichment of Al in the SPM might be relevant for filter-feeding organisms. For fresh water fish species, higher dissolved organic carbon and matter reduce the Al toxicity (Gensemer and Playle, 1999). Deborde et al. (2015) also suspected that the dissolved fraction of Al in seawater is not relevant under environmental conditions, due to the dilution effects. Mao et al. (2011) investigated Al accumulation in Mytilus edulis in a harbor environment and demonstrated that the digestive glands are short and medium-term storage sites for Al. Caplat et al. (2010) investigated the effects of dissolved Al and Zn from galvanic anodes and their corresponding sulfate salt with sea urchin bioassays and suggest lesser effects from Al and Zn originating from galvanic anodes. The impact of dissolved and precipitated Al species on different diatoms was investigated by Gillmore et al. (2016). They demonstrated that the toxicity can be caused either by dissolved or precipitated Al depending on the diatom species.

As demonstrated in the previous section, high amount of Al will be emitted by galvanic anodes. Whether these emissions will increase the dissolved Al concentration in seawater is unclear, due to the high dilution in the open sea and the potential emission as non-soluble species or amorphous aluminium hydroxides. Galvanic anodes may increase Al concentrations in SPM as demonstrated in the tank experiments (Deborde et al., 2015). This SPM bounded Al and the non-soluble fraction may be buried in the sediment in the near surrounding of OWF. However, it is currently unknown if these emissions will have an effect on sediment concentrations and on benthos organisms. Detailed emission studies including the investigation of metal species and metal complex generation on the anode surface and the effects on the local or even regional scales are unknown, because published data is not available. The total Al content of marine sediments is already high, because it originates from clay minerals and it might be difficult to differentiate natural Al background and the impact of the galvanic anodes on Al concentrations in sediment. However, this differentiation is important to evaluate the potential eco-toxicological effects of the Al hydroxide species emitted by the galvanic anodes. Analyses of Al-species in the sediment would be necessary in order to better understand the fate of Al emissions from galvanic anodes.

3.1.2. Zinc

Zinc is an essential element and ubiquitous distributed in the earth system, but shows also dose depending toxicological effects. Zn is one of the marine micronutrient and is ubiquitously distributed in the marine environment (Conway and John, 2014). Zn species and fractionation between sediment, suspended matter and dissolved in seawater strongly depends on the physicochemical properties of the environment. This distribution in the marine environment is affected by salinity, pH, redox conditions and organic content. Zn concentrations from the North Sea were reported with median concentrations 1 μg L⁻¹ and 0.3 ng L⁻¹ in coastal and open sea waters, respectively (BSH, 2016). These North Sea concentrations are in the range of measured coastal water concentrations in different regions from 0.3–300 μg L⁻¹ in Mottin et al. (2012). Sediment concentrations in the German Bight, are currently ranging between 158 and 864 mg kg⁻¹ dw (< 20 μm fraction, superficial
sediment, (BSH, 2016). Zn is included in the OSPAR environmental monitoring program for riverine input and the annual fluxes of direct discharges. The riverine inputs to the North Sea, as one major source, were 2689 tons in 2013 (OSPAR, 2013). Compared to this input, emissions of OWFs might be low. OSPAR convention eco-toxicological assessment criteria (EAC) are between 0.5 and 5 μg L\(^{-1}\) in water and 50–500 mg kg\(^{-1}\) dw in sediments. These values have no legal character and are only for a preliminary assessment to identify potential concern. The WHO recommends limits of 5 mg L\(^{-1}\) for drinking water based on taste conditions, only.

The fate and impact on sediment and seawater of Zn anodes was already investigated in various studies, as Zn anodes are frequently used in ships, in ballast water tanks, and harbor sheet piling (Bird et al., 1996; Caplat et al., 2012; Mottin et al., 2012; Rousseau et al., 2009; Wagner et al., 1996). Gomiero et al. (2011, 2015) analyzed mussels (Mytilus galloprovincialis) from offshore gas platforms in the Adriatic Sea and discussed galvanic anodes as potential source for metal accumulation (Zn, Cd, Ni) in the mussels resulting an overall weak signal of biological disturbance. In Al anodes, Zn is the second most abundant metal with a contribution of 2.5–5.75% to the total anode mass. Tank experiments with Zn anodes demonstrated an increase of Zn\(^{2+}\) ions in the water and precipitation as hydroxide, but also in form of complexes or attached to suspended matter (Rousseau et al., 2009). Deborde et al. (2015) analyzed Al, Zn and Fe emitted by Al anodes in a seawater tank. They demonstrated an increase of the dissolved Zn fraction only in the beginning of the experiment and a decrease later due to sorption to SPM and dilution effects during water renewal. They observed a highly hydrous amorphous white layer on the surface of galvanic anodes, with high Al concentration in the solid part and high Zn concentrations in the liquid part of this white layer. Caplat et al. (2010) studied the toxicity of Zn emissions from galvanic anodes for sea urchin embryos and sperm, resulting in low or no damage compared to the salts of Zn and Al. Mottin et al. (2012) investigated exposed oysters to emissions from Zn anodes and observed an impact on the immune system activities of C. gigas, with a sensitivity to acute Zn toxicity but only moderate affects by mild Zn concentration.

Although studies suggesting a rather low toxicity of Zn from galvanic anodes and the emissions from offshore windfarms will be significantly lower (approximately 2 t a\(^{-1}\), following the previous example of an offshore wind farm with Al-Zn-In anodes) compared to Al, investigations are still necessary to understand the fate and potential effects of Zn emissions from this new source.

### 3.1.3. Indium

Indium contributes only 0.01–0.04% of the anodes material. Compared to its low environmental occurrence with only 0.05 ppm in the earth crust, galvanic anodes might be a significant new source of indium to the marine environment. Indium is generally used as indium tin oxide for electric displays, photovoltaic cells and LEDs. Its industrial production increased in the past 20 years, and is the major source of indium in the environment (White and Hemond, 2012). In most of the previously mentioned emissions studies and tank experiments, indium was not investigated in detail due to its low concentration and analytical issues. Only Deborde et al. (2015) analyzed the precipitated sediment fraction of a tank experiment and showed the occurrence of indium in the anode exposed sediments. Data of indium in the marine environment are rare and mainly discussed in relation to geochemical processes. The toxicological effects of indium species were mainly investigated due to the exposure of workers in the industry, suggesting inhalation related toxicological effects (Tanaka et al., 2010). As a result of the increased demand and application of indium in the industry and thus the emission through wastewater, first studies investigated the toxicological effects on fresh water aquatic species, with median lethal concentrations for indium (III) between 6.9 and 21.5 mg L\(^{-1}\) (LC\(_{50}\)) (Yang, 2014). To the best of our knowledge, there is currently no data for indium in the European marine environment available. Concentrations in the Mediterranean Sea and the Atlantic Oceans were below 0.6–1.6 ppm kg\(^{-1}\) (Albo et al., 1999) and concentrations in the Sea of Japan were even lower (Obata et al., 2007).

### 3.2. Organic compounds

Organic compounds released from organic coatings (EP and PUR) in contact with sea-water may originate through leaching processes, weathering processes or material losses. Applied EP and PUR coatings contain a wide range of chemicals like binding agents, pigments, fillers, organic modifier, solvents and different additives. In addition, the resins are a reaction product of different components and the final products or artefacts are not always known, but might be leachable. The leaching or releases through other processes during the lifespan of an organic coating was investigated for different applications and additives (e.g. biocides from facade coatings, renders and paints; (Bollmann et al., 2017, Bollmann et al., 2016; Burkhardt et al., 2011), but not well studied for marine coatings. Epoxy resins were investigated mainly for drinking water or food related applications (Bruchet et al., 2014; Rajasirikâlâ et al., 2016), or the transfer to the aquatic environment from EP coatings for onshore civil engineering infrastructures (Vermeiren et al., 2017). In these leaching experiments also eco-toxicological effects were investigated under laboratory conditions and demonstrated the occurrence of some xenoestrogens such as BPA. Results and quantity of released organic substances depended strongly on the specific coating product (Burkhardt et al., 2015). A first report of leaching of different organic substances from one component PUR (1C-PUR) corrosion protection systems was recently published (Laft et al., 2017). A dependence of hardening time and released substances was observed in that study, but as already mentioned before; coatings have to be applied and harden onshore, which reduce emissions and avoid the release of not fully cured coating material directly into the water. However, it is unknown how frequently coatings are maintained offshore, which might be an additional source of emissions (especially from the application process), but it's difficult to quantify.

#### 3.2.1. Bisphenol A

Bisphenol A (or) F diglycidyl ethers are common starting products of epoxy resins and after application the coating might contain leachable bisphenol A or F in traces. Bisphenols can also be used as an ingredient compound, thus as free bisphenol A in epoxy resins, but it is rarely used for coatings in the offshore industry. For epoxy resins without any free BPA as ingredient compound, assessment studies assumed only residual BPA levels of 10 ppm which might be leachable and enter the environment (Fischer et al., 2014). This estimate does not include releases through weathering, damages, or any other residual or alternative bisphenols, like BPF. In addition the leaching of e.g. BPA and other chemicals from coatings depends on several factors: the concentration of BPA in the resin which differs between products, the used amount, how the resin was applied, salt water conditions, exposure to sunlight, and stress corrosion cracking of the coating influence the emissions. The available data is too insufficient to estimate the total emissions and further research is needed.

Bisphenol A is already ubiquitous in the environment, although there are no natural sources (Corrales et al., 2015; Cousins et al., 2002; Fischer et al., 2014; Flint et al., 2012; Fromme et al., 2002; van Soestbergen et al., 2014). It is produced in high volumes because it is 30% of the total bisphenol A consumption in 2008 was used for the production of epoxy resins (Fischer et al., 2014), with an estimated annual usage of epoxy resins for marine coatings of 51.000 t in Europe (Epoxy Resin Committee, 2015). The reactions between bisphenol A (or bisphenol F as a substitute) and epichlorohydrin are the key products for epoxy resins which are widely used coatings for marine applications and for offshore wind turbines.

Only a few studies demonstrated that BPA and also BPF can be
leached from epoxy coatings for different applications in laboratory experiments or drinking water applications (Bruchet et al., 2014; Burkhardt et al., 2015; Rajasäärkä et al., 2016). Most leaching studies focus on BPA in consumer products (like water bottles) as it is a direct uptake source for humans. In the marine environment BPA is often discussed as a result of leaching from marine litter, microplastics or the riverine input (Heemken et al., 2001; Koelmans et al., 2014; Staniszewska et al., 2016; Zhang et al., 2014).

BPA is under discussion as substances of environmental concern (European Commission, 2010; U.S. EPA, 2011). In addition, different reaction products for epoxy resins (e.g. 4,4′-Isopropylidenediphenol (bisphenol A), oligomeric reaction products with 1-chloro-2,3-epoxypropane; formaldehyde, oligomeric reaction products with 1-chloro-2,3-epoxypropane and phenol; 2,3-epoxypropyl neodecanate; 2,3-epoxypropyl o-tolyl ether); are listed as currently under evaluation as they are suspected as potential endocrine disrupters (ECHA, Community rolling action plan (CoRAP), evaluated by Denmark). It is also listed on the OSPAR list of substances of potential concern (Group B, OSPAR). Provisional predicted no effect concentration (PNEC), as one environmental quality criteria for pollutants, is 0.15 μg l\(^{-1}\) for BPA in marine waters (European Commission, 2010; Ghekiere et al., 2013). First studies demonstrated also eco-toxicological effects of the substitutes, like BPF (Moreman et al., 2017). Reported environmental concentrations of BPA in the open North and Baltic Sea are generally low and below the PNEC value. Stachel et al. (2003) and Heemken et al. (2001) detected BPA only in river water samples, but not in the open North Sea (detection limit 50 ng l\(^{-1}\)). In the Gulf of Gdansk in the Baltic Sea average BPA concentrations were 59.2 ng l\(^{-1}\) in surface water and 79.2 ng l\(^{-1}\) in bottom water (Staniszewska et al., 2014). Belfroid et al. (2002) reported concentration from <10–330 ng L\(^{-1}\) from estuarine waters of the Netherlands. Although the extend of emissions from coatings is currently unknown, the high number of OWFs may be an new point source of organic chemicals like BPA in the North and Baltic Sea as current background concentrations are generally low.

3.2.2. Other organic compounds

Screening of ingredient information from security data sheets of various coating products show a wide range of phenolic compounds like p-tert-butylphenol, octylphenol (OP), nonylphenol, or dodecylphenol (and their isomers) and several other organic compounds (e.g. xylol and its isomers, ethylbenzene, methyl isobutyl ketone, trimethylbenzenes, ethyltoluene, propylbenzene, butyl acetate, ethyl 3-ethoxypropionate, 1-methoxy-2-propanol and its acetate, benzyl alcohol), are often used in coating materials as solvents, viscosity regulators or catalysts of hardening (especially 2,4,6-tris(dimethylaminomethyl)phenol) in EP. The EP curing agents for the epoxy oligomers are mostly diamines, poly(aminoamide)s and imidazol derivates. The following compounds in that study belong to chemical groups: derivatives of N-(tosyl)carbamate, p-toluenesulfonamide, 4,4′-methyleneidiphenyl disiocyanate, toluenediisocyanate, and p-toluenesulfonic acid. Apart from the leaching related emissions from intact coatings, deterioration and damages may increase also from the underlying coating layers, but are also a source for particle emissions. As mentioned before, coatings have an expected life time of 15–20 years and failures rates of at least 10–20% of the coated surface in the first 15 years can be normally expected. This loss of coating materials was demonstrated by a performance assessment of corrosion protection systems of offshore wind power transmission platforms by Mombert (2016). He documented coating damages and suggested that the majority of these damages are due to the constructive design and the mechanical load. Also an enrichment of particles from e.g. anti-fouling paints was observed in the near surrounding of boat repair facilities, abandoned structures and grounded ships (Turner, 2010).

4. Moving forward

The ambitious targets of the various countries to increase the share of renewable energy will significantly increase the number of OWFs in Europe and worldwide. Long-term environmental effects of OWFs are currently unknown and are subject of current research. Chemical emissions from OWFs appear to be low, especially if compared to other offshore activities, but may become more relevant for the marine environment with increasing numbers of OWFs. Corrosion protection systems are a direct source of chemical emissions from all types of offshore structures in windfarms: OWT, OSS and HVDC platforms. Currently there is no clear evidence of a negative impact of corrosion protection systems on the marine environment, but it is necessary to increase our knowledge to understand the potential (long-term) effects of corrosion protection systems. We identified the following needs:

- **scientific investigations of the OWF's corrosion protection systems and their environmental impact**
- **develop environmental monitoring strategies for chemical emissions in OWFs to assess the long-term impact**
- **consider corrosion protecting systems and their chemical emissions in the obligatory EIA**
- **OWF and platform planner should include environmental aspects when choosing corrosion protection systems (besides technical aspects)**

The discussed emissions suggest currently a low environmental risk of the different systems, also in comparison to other sources of chemicals into the marine environment (e.g. oil and gas industry, riverine input, ship traffic, atmospheric deposition). However, the available data from OWFs is scarce and makes it difficult to assess the impact of these emissions on the marine environment. Scientific investigations should address this issue and should develop suitable analytical methods (if not available) and provide detailed analyses of seawater, sediment and biota in OWFs and the surrounding. In addition, bio-marker approaches as applied for gas platforms (Gomiero et al., 2011, 2015) or passive sampling strategies could be used as additional tools for an environmental monitoring. This would help to better understand the fate and impact of these emissions, also in comparison to the other natural and anthropogenic sources of these substances in the marine environment. In addition, (standardized) test systems should be developed to investigate leachable substances and/or eco-toxicological...
effects of corrosion protection systems, because first studies demonstrated the release of substances from these systems (e.g. Luft et al., 2017; Vermeirssen et al., 2017). Leaching experiments followed by target and non-target screening methods might be one suitable tool. If leachable substances were identified, field experiments are necessary to confirm their occurrence and fate in the environment. These environmental analyses in combination with laboratory test systems may help to deduce monitoring strategies to generate long-term data and to address potential substances of concern. This information is also important to evaluate the “good environmental status” of the marine environment, as e.g. requested by the European Marine Strategy Framework Directive. Furthermore it should be noted that chemical emissions from corrosion protection systems are only one aspect of a suite of potential chemical and particulate (e.g. erosion of coatings, blades etc.) emissions from OWFs and environmental monitoring should also include substances from other potential emission sources. If scientific findings of these studies demonstrate the importance of chemical emissions of OWFs, these emissions should be more taken into account in the environmental impact assessment and also national guidelines and standards should address this issue.

More knowledge about the impact of corrosion protection systems will raise the awareness of chemical emissions of OWF by planners and national authorities. It will help the national authorities and OWF planner to apply the environmentally best solution for future projects. This will increase the environmental performance of OWFs. With ICCP, a technical solution is already available, which has significant lower emissions than Zn or Al galvanic anodes (Table 2). Although the environmental impact of galvanic anodes with their high emissions is currently unknown, national authorities should encourage the planners to use ICCP systems if technically possible. Galvanic anodes contain environmentally critical elements (e.g. Pb, Hg, Cu, Cd) as impurities. These impurities should be reduced to a minimum to lower environmental risk. Coatings should be used also in the underwater zone to reduce the necessary galvanic anode material. However, currently emissions from epoxy resins or polyurethane coatings are difficult to evaluate because composition and ingredients depend on the product and detailed information about leaching and the fate into the marine environment is not available. If this data is available, coatings with the best environmental performance should be used. Furthermore, offshore wind energy is a young and fast growing market and the demand of suitable corrosion protection systems is increasing. The growing awareness of the environmental impact of corrosion protection systems may provoke developments of new products and techniques which will reduce these emissions. Searching for the environmentally best solution which also matches the technical requirements will further reduce the environmental footprint of offshore wind farms.

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