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Potential impacts of submarine power cables from marine renewable energy projects on benthic communities

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Rapporteurs avant soutenance :

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Composition du Jury :

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À mon père,

“そいつはただひとりぼっちだっただけさ。
夢の中で生きているような、そんな男だった。”
カウボーイビバップ 天国の扉 2001

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La FRIME

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List of publications presented

Bastien Taormina ; Juan Bald ; Andrew Want ; Gérard Thouzeau ; Morgane Lejart ; Nicolas Desroy ; Antoine Carlier. "**A review of potential impacts of submarine power cables on the marine environment: knowledge gaps, recommendations and future directions**". Published in *Renewable and Sustainable Energy Reviews* Volume 96, November 2018, 380–391. https://doi.org/10.1016/j.rser.2018.07.026

Bastien Taormina ; Martin Marzloff ; Nicolas Desroy ; Xavier Caisey ; Olivier Dugornay ; Emmanuelle Metral Thiesse ; Aurélien Tancray ; Antoine Carlier. "**Optimised underwater imagery method to investigate epibenthic communities**". Published in *ICES Journal of marine science* Volume 77, Issue 2, March 2020, 835–845. https://doi.org/10.1093/icesjms/fsz249

Bastien Taormina ; Arthur Percheron ; Martin P. Marzloff ; Nolwenn Quillien ; Morgane Lejart ; Xavier Caisey ; Nicolas Desroy ; Olivier Dugornay ; Antoine Carlier. "**Succession of epibenthic communities on artificial reefs associated with offshore renewable energy facilities within a tide-swept environment**". Accepted in *ICES Journal of marine science*. https://doi.org/10.1093/icesjms/fsa129

Bastien Taormina ; Martial Laurans ; Martin Marzloff ; Noémie Dufournaud ; Morgane Lejart ; Nicolas Desroy ; Didier Leroy ; Stéphane Martin ; Antoine Carlier. "**Renewable energy homes for marine life: habitat potential of a tidal energy project for benthic megafauna**". Under review in *Marine Environmental Research*.

Bastien Taormina ; Carole Di Poi ; Ann-Lisbeth Agnalt ; Antoine Carlier ; Nicolas Desroy ; Rosa Helena Escobar-Lux ; Jean-François D'eu ; Florian Freytet ; Caroline M.F. Durif. "**Impact of magnetic fields generated by AC/DC submarine power cables on the behavior of juvenile European lobster (*Homarus gammarus*)**". Published in *Aquatic Toxicology* Volume 220, March 2020. https://doi.org/10.1016/j.aquatox.2019.105401

Bastien Taormina ; Jessica Laurent ; Jean-Dominique Gaffet ; Morgane Lejart ; Antoine Carlier ; Nicolas Desroy. "**The power to protect: Do submarine cables generate unintended reserves?**". In preparation.

List of communications

"*Caractérisation morphologique, sédimentaire et biologique d'un espace naturel remarquable : le plateau rocheux de la Méloine*". Poster. CARHAMB'AR 2017 ; Brest (France). Laurent Lévêque ; Axel Ehrhold ; **Bastien Taormina** ; Caroline Broudin ; Jean-Charles Leclerc ; Line Le Gall ; Eric Houlgatte ; Eric Thiébaut.

"*Optimisation of an imagery analysis method to characterise the epibenthic communities of submarine power cables*". Talk. EIMR 2018 ; Kirkwall (United-Kingdom). **Bastien Taormina** ; Morgane Lejart ; Emmanuelle Thiesse ; Nicolas Desroy ; Antoine Carlier.

"*France Energies Marines : Environmental Impact & Resource*". Talk. FOWT 2018 ; Marseille (France). **Bastien Taormina** ; Nolwenn Quillien ; Morgane Lejart ; Guillaume Damblans.

"*Optimisation of an imagery analysis method to characterise the reef effect*". Talk. Atelier Imagerie IFREMER 2018 ; Brest (France). **Bastien Taormina** ; Morgane Lejart ; Emmanuelle Thiesse ; Nicolas Desroy ; Antoine Carlier.

"*Utilisation of underwater imagery to characterise the reef effect*". Talk. ICOE 2018 ; Cherbourg (France). **Bastien Taormina** ; Morgane Lejart ; Emmanuelle Thiesse ; Martial Laurans ; Nicolas Desroy ; Antoine Carlier.

"*Exploring Submarine power cables from offshore wind farms Environmental Impacts*". Talk. CIGRE 2019 ; Aalborg (Denmark). Lisa Garnier ; Laurent Chauvaud ; Antoine Carlier ; **Bastien Taormina** ; Morgane Lejart ; Aurélie Jolivet ; Sylvain Chauvaud.

"*Are juvenile European lobsters (Homarus gammarus) on a highway to hell due to AC/DC magnetic fields generated by submarine power cables?*". Poster. Benthic Ecology Meeting 2019; Saint-Johns (Canada). **Bastien Taormina**; Nicolas Desroy; Antoine Carlier; Florian Freytet; Caroline Durif.

"*Are juvenile European lobsters (Homarus gammarus) on a highway to hell due to AC/DC magnetic fields generated by submarine power cables?*". Poster. SEANERGY 2019; Dunkerque (France). **Bastien Taormina** ; Nicolas Desroy ; Antoine Carlier ; Florian Freytet; Caroline Durif.

"*Why and how characterise biofouling for FOWT?*". Talk. FOWT 2019 ; Montpellier (France). Nolwenn Quillien; G. Damblans; K. Boukerma; J.-F. Briand; C. Bressy; A. Carlier; C. Compère; C. Dreanno; D. Jacob; V. Leblanc; A. Maison; M. Reynaud; F. Schoefs; **B. Taormina**; M. Lejart

"*Succession of epibenthic communities on artificial structures*". Talk. EIMR online conference 2020. **Bastien Taormina** ; Arthur Percheron ; Martin P. Marzloff ; Nolwenn Quillien ; Morgane Lejart ; Xavier Caisey ; Nicolas Desroy ; Olivier Dugornay ; Antoine Carlier.

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List of abbreviations

AC Alternating Current

ANOSIM ANalyses Of SIMilarities

ANOVA ANalyses Of VAriance

CATAMI Collaborative and Automated Tools for Analysis of Marine Imagery

CL Carapace Length

CV Coefficient of Variation

DC Direct Current

EA Exclusion Area

EMF ElectroMagnetic Field

FEve Functional Evenness

FOri Functional Originality

FRic Functional Richness

FSpe Functional Specialisation

GIS Geographic Information System

HMF1 High Magnetic Field 1

HMF2 High Magnetic Field 2

HVAC High Voltage Alternating Current

HVDC High-Voltage Direct Current

LMF3 Low Magnetic Field 3

LMF4 Low Magnetic Field 4

MF Magnetic Fields

MRE Marine Renewable Energy

nMDS non-metric Multi-Dimensional Scaling

PCA Principal Component Analysis

PERMANOVA PERMutational multivariate ANalysis Of VAriance

RDA Redundancy Analysis

SI Supplementary Information

SPC Submarine Power Cable

SPL Sound Pressure Level

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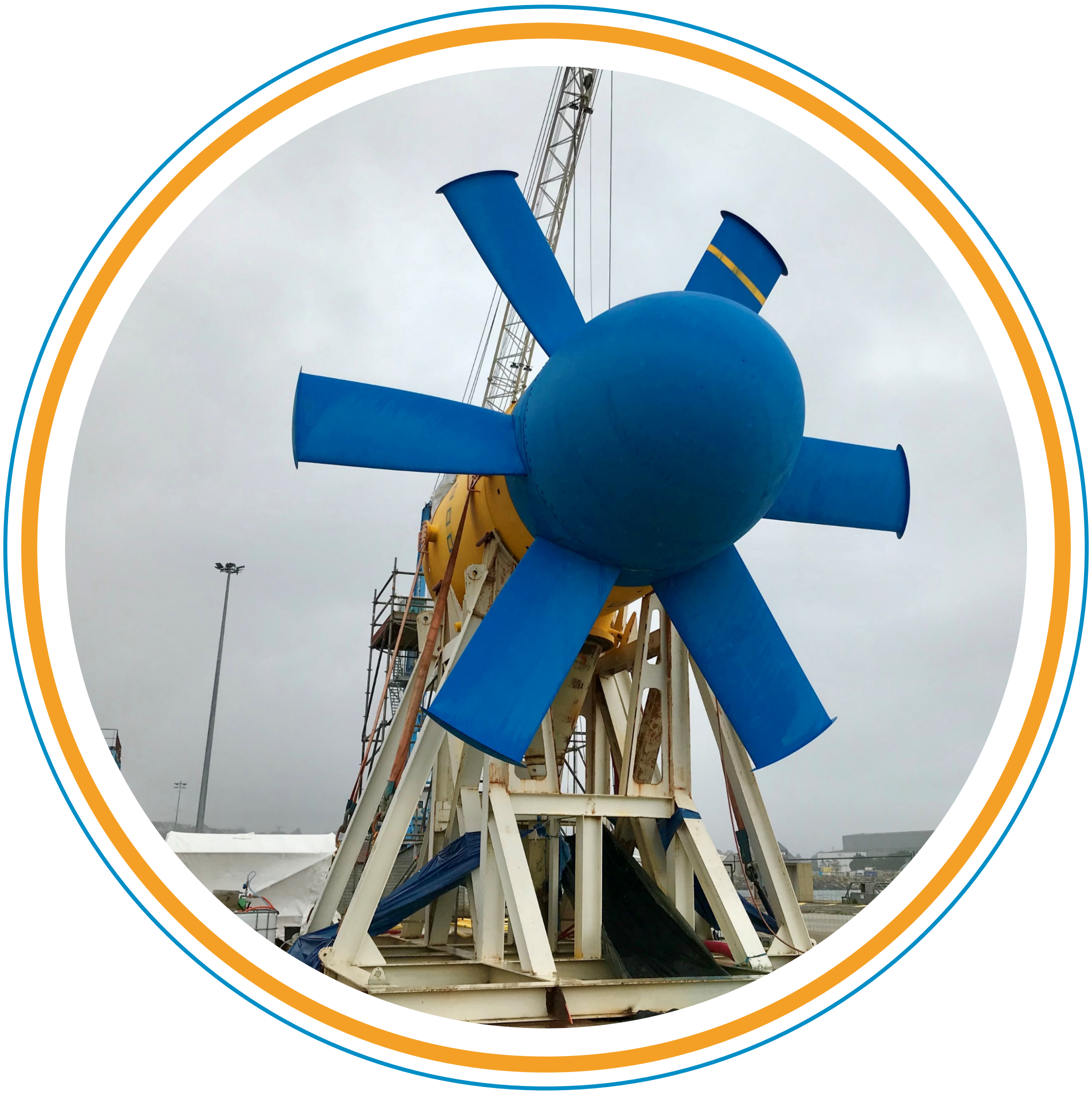
General Introduction



1. Marine renewable energy to combat global change

In 2017, the power industry (*i.e.* power and heat generation plants) was the main carbon dioxide emitting sector with around 40% of worldwide emissions (Muntean et al., 2018). In current attempts to combat climate change, the replacement of fossil fuel energy by renewable energy constitutes one major priority. To tackle this issue, the European Commission, Parliament and Council set up renewable energy targets in 2008 and renewed them in 2018. The aim is to achieve at least a 32% share of renewable energy in final energy consumption by 2030 (STATEMENT/18/4155). For nations with coastal and ocean territory, the development of the exploitation of Marine Renewable Energies (MRE) provides an attractive potential contribution to the targeted renewable energy mix. MRE can be summarised as the technologies that generate energy from the ocean, through winds, tides, waves and temperature differentials in seawater (Figure 1).

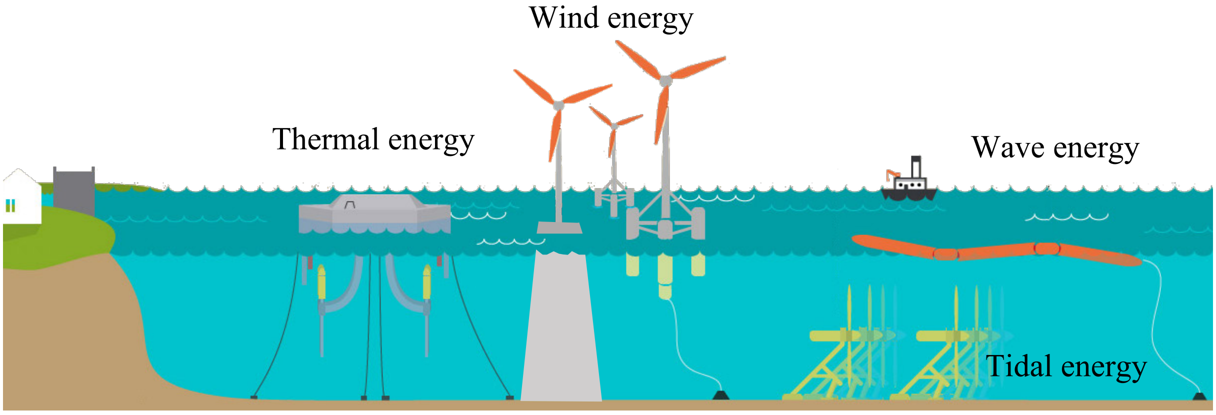


Figure 1: Diagram of the main types of Marine Renewable Energy (MRE) technologies with, from left to right: ocean thermal energy conversion plants (thermal energy), offshore fixed-foundation and floating wind turbines (wind energy), tidal turbines (tidal energy) and wave energy converters (wave energy). Courtesy of France TV.

Since the installation of the world's first offshore windfarm in 1991 in Denmark, MRE has become a fast growing industry which now extends across large coastal areas. So far, more than 4500 fixed-foundation offshore wind turbines (representing more than 100 farms and a total cumulated capacity of 18.5 GW) have been installed and grid connected in the European seas (Wind Europe, 2019), and other types of MRE (*e.g.* floating wind turbines, tidal and wave energies *etc.*) are under development. But although the introduction of MRE devices constitutes a green measure for sustainable development, it also represents a new source of potential anthropogenic disturbances on coastal environments which are already under high human pressure.

2. Coastal ecosystems under high pressures

While coastal waters represent only 7% of the surface area of the world's oceans, their importance is major both from socio-economic and ecological points of view (Costanza *et al.*, 1997; Harley *et al.*, 2006; Snelgrove *et al.*, 2014). They account for at least 25 % of global primary productivity, 90 % of the world's marine fish catch and 17 % of open ocean CO_2 uptake (Agardy *et al.*, 2005; Cai, 2011). The benthos (*i.e.* the assemblages of organisms living in, on or close to the seabed) constitutes a paramount compartment for the global functioning of coastal ecosystems. Benthic organisms represent a significant share of coastal biodiversity and support a wide panel of ecosystem processes involved in biogeochemistry cycles and the provision of food resources (Dannheim *et al.*, 2019). Additionally, numerous benthic species provide a wide variety of biogenic habitats which constitute important sanctuaries for biodiversity such as mangrove forests, salt marshes, coral reefs, seagrass meadows and kelp forests (Agardy *et al.*, 2005; Kovalenko *et al.*, 2012).

Due to the large number and variety of ecosystem services that coastal environments offer to mankind, a disproportionate share of the global human population (39 % in 2005; Agardy et al., 2005) lives within 100 km of a coastal area, leading to high anthropogenic pressures. Because of their position at the interface between land and sea, coastal environments end up between Scylla and Charybdis, suffering from both terrestrial and marine anthropogenic pressures (Halpern et al., 2008; Figure 2). Land-based activities can cause the removal, alteration or destruction of natural habitats through urbanisation (Agardy *et al.*, 2005) and affect the runoff of nutrients and chemical/organic contaminants (Islam and Tanaka, 2004). Coastal

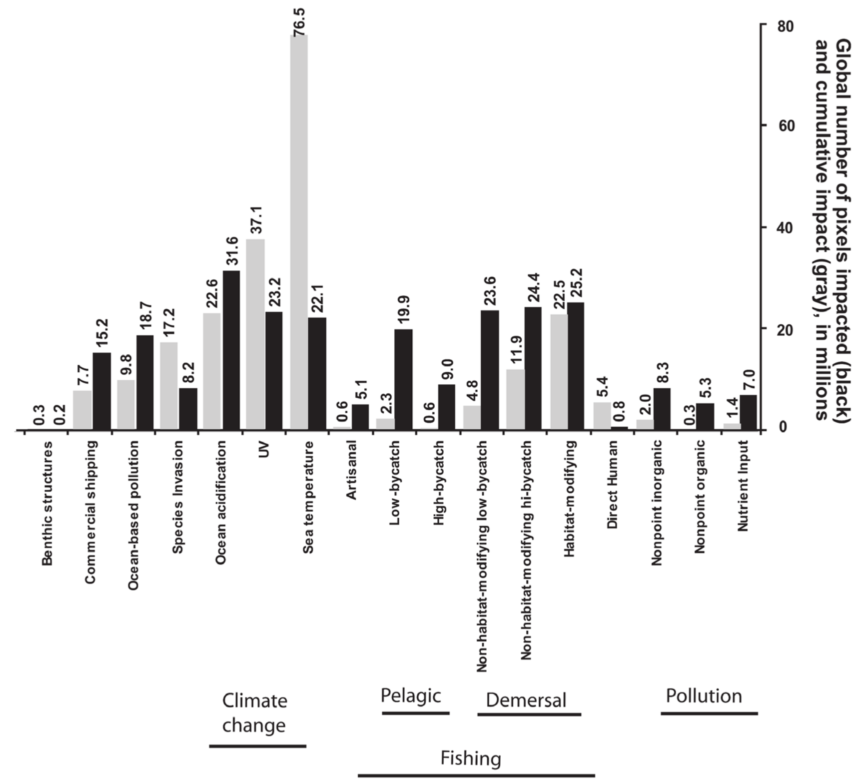


Figure 2: Total area affected (square kilometres, grey bars) worldwide and summed threat scores (rescaled units, black bars) for each anthropogenic driver for all coastal regions <200 m in depth. Values for each bar are reported in millions (Modified from Halpern *et al.* 2008)

ecosystems are also highly impacted by overfishing, which lead to global depletion of stocks of finfish, crustaceans and molluscs (Agardy *et al.*, 2005; Lotze *et al.*, 2006b). Fishing activities like trawling also cause direct physical impact considerably damaging benthic habitats (Eigaard *et al.*, 2017). In addition, coastal waters are the zones which are most heavily impacted by introduced and invasive species which use maritime transport and aquaculture as their main introduction vectors (Grosholz, 2002). In addition to all the above-mentioned pressures, coastal ecosystems are affected by global warming, leading to *i*) an increase in the water temperature, *ii*) an acidification of the water and *iii*) a rise in sea level (Harley *et al.*, 2006). In response to this cocktail of anthropogenic pressures, coastal ecosystems exhibit the most rapid environmental change: for example 35% of mangrove forests (Valiela *et al.*, 2001) and 19% of coral reefs (Wilkinson, 2008) have disappeared during the last few decades.

Most anthropogenic disturbances cumulate in the coastal zone, leading to unpredictable changes in ecosystem functioning when a “tipping point” (*i.e.* a critical threshold at which a tiny perturbation can qualitatively alter the state or development of a system) is exceeded (Lenton *et al.*, 2008). In this context, the addition of a new type of anthropogenic disturbance through the development of MRE, even of low intensity, is closely scrutinised.

3. MRE: a new source of pressure

Several publications give a synthesis of the major environmental concerns linked to MRE development (Boehlert and Gill, 2010; Lindeboom *et al.*, 2011; Copping *et al.*, 2016; Dannheim *et al.*, 2019). Looking closely at the benthic compartment, impacts of MRE devices can occur during the construction, operational, or decommissioning phases (Dannheim *et al.*, 2019). The construction/decommissioning phases can lead to *i*) mechanical disturbance to the seafloor and associated macrobenthos (Coates *et al.*, 2015) and *ii*) to the emission of noise from pilling activities resulting in relocation of the distribution of certain fish species (Neo *et al.*, 2014). Once installed, the different devices act as artificial reefs and are subject to rapid and extensive colonisation by sessile organisms (Sheehan *et al.*, 2018), large decapods (Langhamer and Wilhelmsson, 2009) and pelagic and demersal fishes (Reubens *et al.*, 2011). Indirect impacts also exist such as those caused by the exclusion of fishing activity within MRE parks. This exclusion can result in the restoration of marine communities (Lindeboom *et al.*, 2011) and an abundance increase of commercially important species (*e.g.* the European lobster *Homarus gammarus*; Roach et al., 2018).

Due to a lack of field studies, a high degree of uncertainty is associated with some of the environmental concerns about MRE development (Lindeboom *et al.*, 2015; Copping *et al.*, 2016). Uncertainties tend to heighten the perceptions of risk and contribute to slow siting and consenting of MRE development worldwide (Copping *et al.*, 2019). In front of the urgent global situation, such information is greatly needed to support energy policy developments and planning decisions (Hooper *et al.*, 2017; Dannheim *et al.*, 2019).

4. Outline and objectives of the thesis

The aim of this thesis work is to bring new information on the impacts of a specific component of ocean energy development that is common to all types of MRE projects: the submarine power cable. In this context, power cables are used to convey *i*) power generated from each device to a substation and *ii*) from this substation to the mainland. However, submarine power cables are also used for a variety of other applications such as connecting autonomous grids or supplying power to islands, marine platforms or subsea observatories. Nevertheless, associated environmental concerns have been the subject of very few studies, and considering the current increase in the number of connections, it is now vital to clearly identify the associated impact.

In this context, the first step was to conduct an exhaustive literature review of the potential impacts of submarine power cables on the marine environment (**Chapter 1**). Afterwards, considering submarine power cables as specific artificial reefs, their 'reef' effect was studied *in situ* on sessile epibenthic and megafauna compartments of the benthic ecosystem. This work began with the development and optimisation of a methodological approach based on underwater image analysis in order to describe epibenthic communities (**Chapter 2**). Then, applying this method, the colonisation dynamics of sessile epibenthic communities on different habitats associated with an unburied subsea power cable was studied in **Chapter 3**. We also studied the habitat potential for mobile benthic megafauna created by protection structures associated with cable (**Chapter 4**). After studying the artificial reef effect, attention was paid to the impacts of magnetic fields generated by power cables on benthic organisms. **Chapter 5** thus presents an experimental study conducted on the impact of magnetic fields on juvenile European lobsters (*Homarus gammarus*). Finally, **Chapter 6** reports the potential benefits of an anthropogenic activity exclusion area caused by the presence of power cables on benthic macrofauna.

Chapter 1

A review of potential impacts of submarine power cables on the marine environment: knowledge gaps, recommendations and future directions

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Abstract

Submarine power cables (SPC) have been in use since the mid-19th century, but environmental concerns about them are much more recent. With the development of marine renewable energy, it is vital to understand their potential impacts. The commissioning of SPC may temporarily or permanently impact the marine environment through habitat damage or loss, noise, chemical pollution, heat and electromagnetic field emissions, risk of entanglement, introduction of artificial substrates, or reserve effect. While growing numbers of scientific publications focus on impacts of the marine energy harnessing devices, data on impacts of associated power connections such as SPC is scarce and knowledge gaps persist. The present study *i*) examines the different categories of potential ecological effects of SPC during the commissioning, operation and decommissioning phases and hierarchizes these types of interactions according to their ecological relevance and existing scientific knowledge, *ii*) identifies the main knowledge gaps and needs for research, and *iii*) sets recommendations for a better monitoring and mitigation of the most significant impacts. Overall, ecological impacts associated with SPC can be considered weak or moderate, although many uncertainties remain, particularly concerning electromagnetic effects.

Keywords

Submarine power cables; marine renewable energy; environmental impacts; ecosystem functioning; benthic habitats

1. Introduction

In 1811, a powered cable was laid down across the Isar River in Germany. This is considered to be the first underwater power cable in the world. More than a century later, the first commercial High Voltage Direct Current (HVDC) cable, installed in 1954 in the Baltic Sea, was set up to link Sweden and Gotland Island. Since then, submarine power cables (SPC), using direct current (DC) or alternating current (AC), have continued to spread across the globe. Technologies have constantly improved with respect to materials, cable length and width, but also installation techniques. Applications of SPC are numerous: they can be used to connect autonomous grids, to supply power to islands, marine platforms or subsea observatories, and to convey power generated by marine renewable energy (MRE) installations to electric sub-stations. While most SPC are on top of or buried within the seafloor, some (known as dynamic cables) are deployed through the water column between the surface and the seafloor. This last category of cables is used for offshore oil platforms and, recently, to export energy produced by floating MRE devices (like wind turbines), a technology still under development. In 2015, almost 8000 km of HVDC were present on the seabed worldwide, 70% of which were in European waters. In comparison, the total length of all submarine cables deployed (including AC and DC power cables and telecommunication cables) is of the order of 10^6 km (Ardelean and Minnebo, 2015).

SPC, like any other man-made installation or human activity at sea, may cause disturbances to marine life and habitats. First, when talking about anthropogenic disturbances, ‘effects’ must be distinguished from ‘impacts’, according to the framework proposed by Boehlert and Gill (2010). Effects are modifications of environmental parameters (or “stressors”), such as the nature of the substratum, hydrodynamics, water temperature, noise, or electromagnetic fields beyond the range of natural variability. Impacts correspond to changes observed at “receptor” level, *i.e.*, the different ecosystem compartments (biotopes, biocenosis), or levels (community, populations) or some ecological processes within marine ecosystems (trophic interactions). Impacts may be positive or negative, although this distinction remains subjective.

Scientific interest in interactions between marine life and submarine cables started with the first records of cable damage caused by whale entanglements (16 events between 1877 and 1955; Wood and Carter, 2008) or by fish and shark bites (at least 39 events from 1907 to 2006; International Cable Protection Committee, 2016). Although such events have decreased significantly with technological improvements (cable burial and advances in design or protection; Carter *et al.*, 2009), they generate ecological concern about submarine cables. Nowadays, ecological issues refer not only to direct physical interactions between large animals and cables but also to less obvious impacts of cables on marine communities and habitats.

Numbers of SPC will increase drastically in coming decades with increasing grid connections of islands and archipelagos and the development of MRE projects (offshore wind farms, tidal and wave turbines). Several inter-governmental organisations have set objectives for the next decades. For example, in 2014, the European Council set 27% as a target for the minimum proportion of total electricity consumption produced by renewable energies in the EU by 2030 (EUCO 169/14). In 2008, the global electric energy supply produced by all grid-connected renewable energy installations taken together was estimated at 12.9%, and several predictions estimate an increase to 17% by 2030 and 27% by 2050 (Edenhofer *et al.*, 2011).

Despite more than 10 years of scientific work on potential environmental impacts of MRE projects (Lindeboom *et al.*, 2015; Copping *et al.*, 2016), SPC have received much less attention than MRE devices themselves. Indeed, only nine published papers focusing on *in situ* effects or impacts of SPC were found during the literature research. These studies addressed the impacts of SPC on benthic communities, considering both installation or operation phases (Andrulewicz *et al.*, 2003; Kogan *et al.*, 2006; Bacci *et al.*, 2013; Dunham *et al.*, 2015; Love *et* *al.*, 2017a), examined communities colonising unburied structures (Sherwood *et al.*, 2016; Love *et al.*, 2017a), and/or reported species-specific changes of behaviour (Westerberg and Lagenfelt, 2008; Love *et al.*, 2015, 2017b). Considering the current exponential increase in SPC worldwide, a robust and accurate assessment of their potential environmental impacts has become a priority.

In this context, the aims of the present study are *i*) to review the existing knowledge concerning potential ecological impacts from SPC during installation, operation and decommissioning phases, *ii*) to attempt to hierarchize these impacts according to their significance and *iii*) to point out knowledge gaps and recommendations for monitoring and mitigation of these impacts.

2. Methods

A literature search was conducted using online databases and internet search tools (Web of Science, Science Direct, Google Scholar, ResearchGate) to create a bibliographic database including peer-reviewed scientific publications, books, theses and non-peer-reviewed consultancy and technical reports. Owing to the lack of published studies, a large proportion of current knowledge comes from industrial or governmental reports and environmental impact assessments that may have associated confidentiality issues. The literature search first focused on general publications about SPC generalities and their global environmental impacts before aiming at specific literature for each of the different identified impacts. Documents focussing on anthropogenic disturbances other than SPC, but potentially inducing comparable impacts (*e.g.*, artificial reefs or sediment reworking for example) were also considered. Based on the main conclusions of the reviewed literature, the relative importance of the different potential impacts and the associated scientific uncertainty was compiled.

3. Features of submarine power cables

3.1 Technical characteristics

Table 1 Description of five generic submarine power cable types (Photos: 1 = General Cable; 2, 3, 4 = Ningbo Orient Wires and Cables Co. Ltd; 5 = ABB Sweden), XLPE: Cross-Linked Polyethylene; EPR: Ethylene Propylene Rubber (reproduced from ; Worzyk, 2009).

					
Type	1	2	3	4	5
Rated voltage	33 kV AC	150 kV AC	420 kV AC	320 kV DC	450 kV DC
Insulation	XLPE, EPR	XLPE	Oil/paper or XLPE	Extruded	Mass-impregnated
Typical application	Supplying small islands, connection of offshore wind turbines	Connecting islands with large populations, offshore wind parks export cables	Crossing rivers/straights with large transmission capacity	Long distance connections of offshore platforms or wind farms	Long distance connection of autonomous power grids
Maximum length	20─30 km	70─150 km	<50 km	>500 km	>500 km
Typical rating	30 MW	180 MW	700 MW/three cables	1000 MW/cable pair	600 MW/cable

SPC are specifically designed to relay electric currents either as Alternating Current (AC) or Direct Current (DC), the transmission type being determined by the capacity and length of the transmission line, as well as commercial issues. For example, a DC line can transmit more power than an AC line of the same size, but is more expensive. AC transmission presents some limitations since the reactive power flow due to the large cable capacitance limits the maximum transmission distance (<100 km) due to power loss. DC is therefore the only viable technical option for long distance cable links. AC is more frequently used within grids of marine renewable energy devices (Copping *et al.*, 2016). Cables in use today include monopolar, bipolar and three-phase systems. SPC diameters are between 5 and 30cm and weigh between

15 and 120 kg m^{-1} (including stabilization devices such as articulated steel shell). Different methods exist to insulate electric cables in order to contain the emitted electric fields. Specific designs have been addressed for dynamic cables, with specific armouring layers and internal components. Indeed, their high position in the water column makes them more susceptible to fatiguing pressure and twist caused by hydrodynamics (particularly swell). Table 1 describes most types of recently installed SPC.

3.2 Cable installation

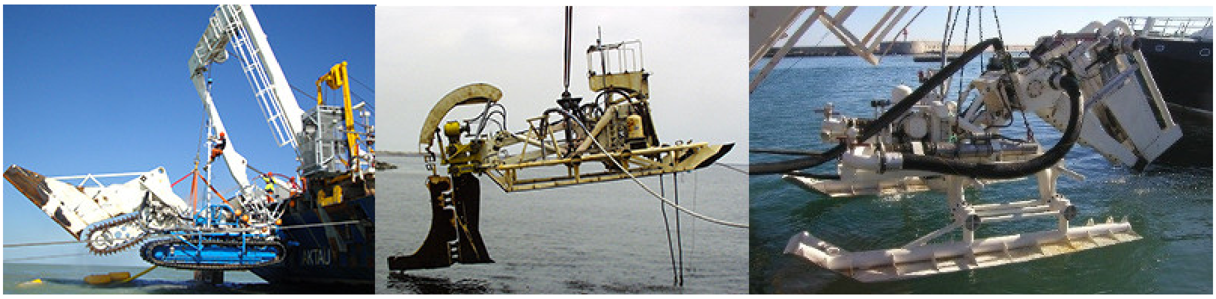


Figure 1: Wheel cutter (left); Plough (centre) and Towed Jetting Vehicle (right) (courtesy: www.ldtravocean.com).

Before any deployment, the cable route must be chosen, depending on the bathymetry, seabed characteristics and economic activities of an area. The route must first be prepared, sometimes with adjustment of the slope and depth, or removal of obstacles before the passage of the cable-laying device. An example of an established method is the pre-lay grapnel run, consisting of dragging a hooking device at low speed along the planned route to remove any material, such as abandoned ropes or fishing nets.

Cable deployment is a complex process requiring highly specialised equipment. The cables are usually buried within the seafloor by different techniques including trenching with a cutting wheel in rocky sediments and ploughing or water jetting in soft sediments (Figure 1; Worzyk, 2009). Ploughing generally allows trenching, laying the cable and burying it with the extracted sediment to be done in a single operation. Special backfill materials for burial can be required when burial is technically complicated. In the case of hard or deep bottoms, the cable can simply be laid on the seafloor and stabilised with suitable cover. The duration of the cable installation process determines the magnitude of some environmental effects, such as increased turbidity or anthropogenic noise. The duration of installation can be highly variable according to methods and seafloor characteristics, as cable laying is much more difficult for a route with obstacles such as boulders, rocks or outcrops, compared with a featureless seafloor (Worzyk, 2009). The rate of cable-laying may vary from 1.85 $km \cdot h^{-1}$ for a cable that is simply laid down to 0.13─0.21 $km \cdot h^{-1}$ for a cable buried using water jetting (OSPAR Commission, 2008). For cable burial in the upper intertidal zone, the trench is often dug with more common devices such as mechanical excavators, and directional drilling is sometimes employed.

3.3 Cable protection

Depending on anthropogenic and natural perturbations in the route area, the cables may need to be protected from damage caused by fishing gears or anchors (OSPAR Commission, 2008), strong hydrodynamic forces or storms. When trenching is not possible, other methods exist for unburied cables, such as rock-mattress covering, cable anchoring, ducting, cast-iron shells, concrete slabs, steel plates or dumped rocks (OSPAR Commission, 2008). On uneven seafloors, the cable may form "free spans" along its route where it will hang without touching the seafloor. This may result in vibration, chafing, fatigue and, ultimately, cable failure (Worzyk, 2009). One solution is to fill the empty space between the cable and the seafloor with rock dumping or concrete bags. As an example of protection methods employed, the cable

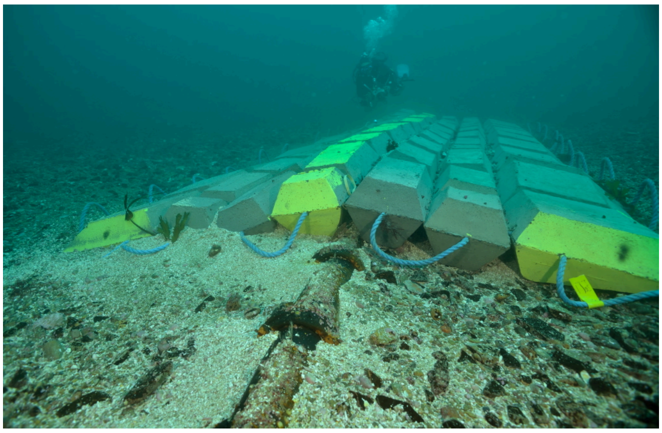


Figure 2: Photograph of iron shells and concrete mattresses used to protect an unburied cable at the tidal Paimpol-Bréhat turbine test site, France (courtesy: Olivier Dugornay, 2013).

connecting the French tidal turbine test site of Paimpol-Bréhat to the land was installed on a highly hydrodynamic and hard seafloor (rock and pebbles). The cable is unburied over a large portion of its route but it is protected with cast-iron shells and concrete mattresses (Figure 2); the free spans are filled with concrete bags. Most of the time, combined to these different protection methods, authorities create a protected area encompassing the cable route, with prohibition of other human activities (fishing, anchoring, dredging, *etc.*) in order to protect the cable from any damage.

4. Environmental effects and impacts

Potential environmental effects associated with SPC are summarised in Figure 3. During installation, maintenance and decommissioning phases, these effects may include physical habitat disturbances, sediment resuspension, chemical pollution and underwater noise emission. More long-term effects may occur during the operational phase, with changes of electromagnetic fields, heat emission, risk of entanglement, chemical pollution, creation of artificial reef and reserve effects.

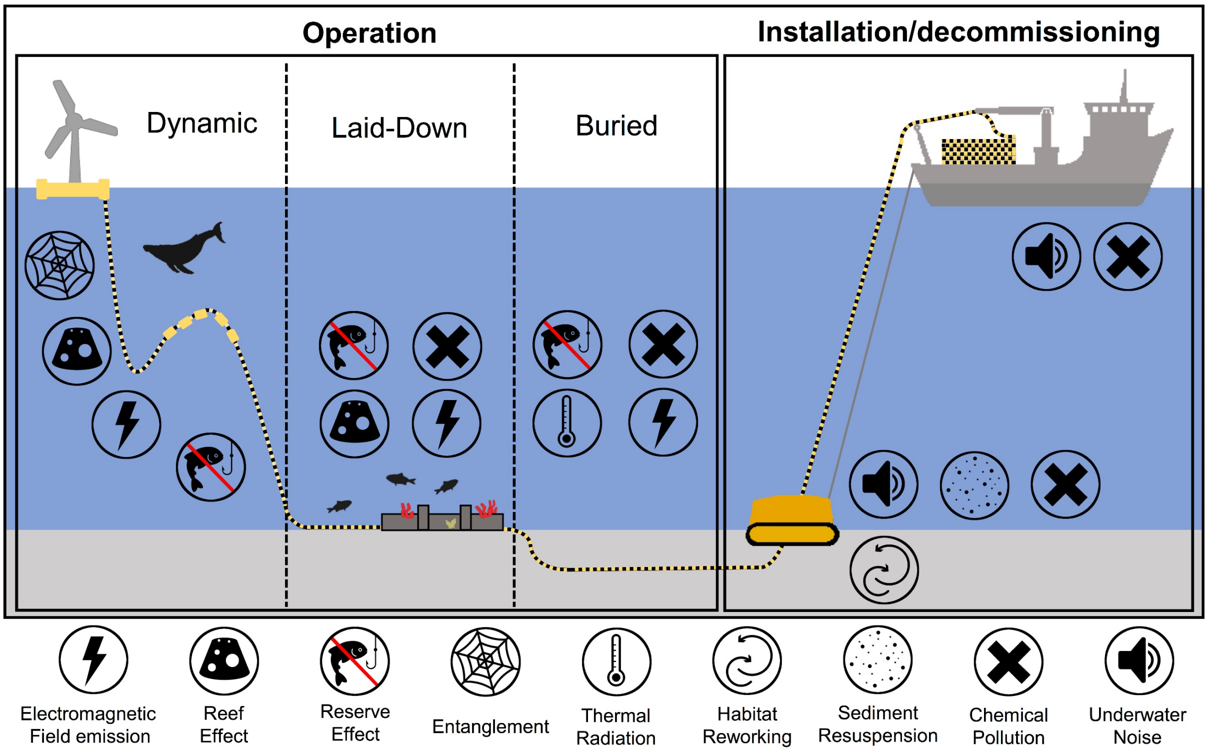


Figure 3: Diagram of the potential impacts caused by different types of SPC immersion (Dynamic, Laid-Down and Buried) during their operation and installation/decommissioning phases.

4.1 Habitat reworking

4.1.1 Physical changes

Figure 4: . Installation works of the 2000 FLAG Atlantic 1 in the intertidal area, Brittany, France (courtesy: www.ldtravocean.fr).

Substratum alterations are mainly created by equipment used for cable route preparation (grapnels such as in the aforementioned Pre-Lay Grapnel Run) and installation of the cable (ploughing, jetting and cutting-wheels). The surface of disturbed area can be enlarged when installation techniques require large ships with several anchoring stabilizers (Worzyk, 2009).

These methods of reworking the seabed may lead to direct destruction of benthic habitats, flora and fauna. However, such effects are usually restricted to a limited area, the width and intensity of disturbance, depending on the installation method. For example, a trenching plough's footprint may vary from 2 to 8 m depending on device size (Carter *et al.*, 2009). According to Vize *et al.* (2008), ploughing methods seem to cause less seabed disturbance than other methods. These disturbances are usually limited in time, as installation works only require a few hours or days per km of cable (Rees *et al.*, 2006). Ploughing and jetting methods favour a quicker recovery of bottom topography, as the trench is filled with displaced and re-suspended material immediately after digging and cable laying. In intertidal areas, physical impacts on the substrate usually occur over a larger surface area, of the order of tens of metres, due to the utilisation of vehicles such as mechanical excavators (Figure 4). Alternatively, underground horizontal directional drilling may be used in intertidal areas up to distances of 700-1000m (10 m below the sediment surface), and occasionally up to 1800 m (Worzyk, 2009). This installation technique only disturbs the substrate and biota locally over a few m² at the land and sea entrance points.

Unburied cables may also cause habitat loss, but to a lesser extent than buried cables. Disturbance is limited to the cable width itself, or to the dimensions of the materials used to stabilise and protect it (Wilhelmsson *et al.*, 2010). In shallow areas, some sections of unstabilised unburied cables may act as dragging elements that disturb the sediments due to their strumming movement induced by the swell during the operation phase (Bald *et al.*, 2010). Wave action may shift the cable, and direct interaction with the hard seafloor can result in surficial scraping and incisions in rock outcrops (Kogan *et al.*, 2006). Maintenance (to a lesser extent) and/or decommissioning phases may generate similar effects to those of installation, but their magnitude will depend on the duration and scale (repairs *vs.* inspections) of the works.

With respect to other human activities at sea, physical disturbance to the seabed caused by cables is spatially very limited. For example, the footprint of submarine cables in the UK coastal area is about 0.3 km^2, representing less than 0.01% of the coastal seabed (Foden *et al.*, 2011), whilst in the Basque Country coastal zone (Northern Spain), the footprint of cables and pipelines is about 2.3 km^2, or 0.02% of the area between the coastline and the exclusive economic zone (Borja *et al.*, 2011).

4.1.2 Biological changes

These substratum alterations may affect related benthic communities by direct impacts such as displacement, damage or crushing of organisms. Andrulewicz *et al.* (2003) examined the environmental impact of the installation of a buried submarine power cable on soft bottoms of the Baltic Sea. They concluded that there were no significant changes in benthic diversity, abundance or biomass on the cable route or in its close proximity one year after the installation.

Magnitude and significance of biological changes depend on several factors linked to the sensitivity and resilience capability of the species or communities directly affected. Habitat or community resilience defines capacity to get back to its initial ecological state after a perturbation (cabling in this case), and then the duration of the impact. The weaker the resilience is, the more sensitive the habitat or the community. Thus resilience depends on several factors, such as: nature and stability of the substratum (Newell *et al.*, 1998; Kaiser *et al.*, 2006; Foden *et al.*, 2010), habitat depth (Foden *et al.*, 2011; Clark *et al.*, 2016) and life cycle of disturbed species (for example, seagrass meadows, which grow very slowly, may take several years to recolonise a disturbed area (Erftemeijer and Robin Lewis III, 2006)).

The magnitude of biological changes is also dependent on the composition of the community itself, *i.e.*, the relative occurrence of benthic species (abundance and biomass) and assemblages (richness) along the cable route, compared with their occurrence at the regional scale. Due to the small spatial footprint of cabling, the overall impact on benthic communities is negligible if its spatial distribution is significantly homogenous.

Benthic community resilience after commissioning of submarine cables remains poorly understood owing to the lack of long-term studies (*i.e.* several years). Despite a relatively small spatial footprint, future studies should focus on the resilience of habitats and communities of particular ecological or economic interest (*e.g.* sea grass, maerl beds and nursery areas).

4.2 Sediment resuspension

Depending on the nature of the seafloor, sediment reworking by installation, maintenance or decommissioning can lead to turbid plumes that can reach several tens of hectares, with suspended particulate matter concentrations that can reach several dozen mg l^{-1} (Fissel and Jiang, 2011). Apart from sediment type, the extent and properties of plumes will depend on factors such as installation technique, hydrodynamic conditions and the scale of cable-laying. For instance, in the Nysted offshore wind farm (Denmark) where the substrate is dominated by medium sand sediment, cable installation in water depths between 6 and 9.5m, generated mean particle concentrations of 14 mg l^{-1} (up to 75 mg l^{-1}) at 200 m from the operation site during trenching with a backhoe dredger, and 2 mg l^{-1} (up to 18 mg l^{-1}) during jetting (Seacon, 2005 in Vize *et al.*, 2008) . Turbidity can persist for several days depending on the duration of the whole cable-laying process. For instance, one month was necessary to excavate 17,000m^3 of sediment for a 10.3-km long, 1.3-m wide and 1.3-m deep cable trench for Nysted Offshore Wind Farm (Dong Energy, 2006). However, at any given location on a cable route, disturbance can persist from a few hours to a few days.

Decrease in water transparency and deposition of the resuspended material may limit light for primary producers and impact feeding ability of fish that detect their preys visually (Utne-Palm, 2002). The efficiency of invertebrate filter-feeding could also be temporarily modified (Last *et al.*, 2011; Szostek *et al.*, 2013). In the case of species that lay eggs on the bottom, resuspension/deposition processes through the plume may bury the eggs. The presence of mineral particles in the water column may also lead to gill damage in young fish larvae (Au *et al.*, 2004; Wong *et al.*, 2013). For example, early survival of cod recruits (whose eggs are pelagic) may be affected by the sediment plume created by cable trenching (Hammar *et al.*, 2014).

Nevertheless, turbidity increases resulting from cable installation and decommissioning constitute localised and short-term effects. Although no study has focused on the impact of particle resuspension induced by cable installation and decommissioning on marine communities, it should generally have negligible impacts on marine ecosystems.

4.3 Chemical pollution

The main chemical risk is the potential release of sediment-buried pollutants (*e.g.*, heavy metals and hydrocarbons) during sediment re-suspension caused by cable burial, decommissioning or repair works. The highest contaminant concentrations are generally found in coastal areas due to human activities. To reduce the release of contaminants, a preliminary analysis to assess the level of sediment toxicity should be performed in potentially polluted areas to select a cable route which avoids the remobilisation and dispersion of pollutants (Merck and Wasserthal, 2009).

Pollution can also occur during the operation phase, especially for monopolar DC cables using sea electrodes for the return current path (which represent around 30% of HVDC in service use ; Sutton *et al.*, 2017). Indeed, the cathode and the anode of sea electrodes release toxic electrolysis products like chlorine and bromine which can impact the close water quality (Andrulewicz *et al.*, 2003; Sutton *et al.*, 2017). To a lesser extent, some older cables have hydrocarbon fluid insulation and may leak contaminants into the marine environment when damaged. The amount of fluid released will vary according to the time needed to detect and repair the leakage, its location and the extent of the damage, but several tens of litres per hour can be released in the worst cases (Schreiber *et al.* 2004, in Meißner *et al.*, 2006). It should be noted that installation of oil-insulated cables ceased in the 1990s (Carter *et al.*, 2013). Furthermore, ships and hydraulic equipment pose a higher potential risk of accidental oil leakage during operations (Bald *et al.*, 2010; Polagye *et al.*, 2011). Cables also include copper, lead and other heavy metals that are potential sources of metal contamination. For example, a cable consisting of a 3.5-mm lead sheath contains 12 kg lead.m^{-1} (Schreiber *et al.*, 2004 in Meißner *et al.*, 2006). Heavy metals can potentially dissolve and spread into the sediment from damaged and abandoned cables, but the quantities released are considered insufficient to have significant impacts. Furthermore, such pollution is rare as cables are usually removed when no longer in operation. Although no studies focus specifically on the SPC-related contaminants, this source of disturbance is considered to be rare, spatially localised and unlikely to have significant impacts on benthic communities.

4.4 Underwater noises

Anthropogenic noises can be produced during route clearance, trenching and backfilling, cable and cable protection introduction, and by the vessels and tools used during these operations. Intensity and propagation of underwater noise will vary according to bathymetry, seafloor characteristics (*e.g.*, sediment type and topography), vessels and machines used, and water column properties. *In-situ* data on such noise is scarce, and modelling approaches have been used to estimate the sound pressure levels (SPL) expected during installation. Nedwell and Howell (2004) examined the noise produced by plough trenching in a sandy gravel area for the installation of an electric cable within a Welsh offshore wind farm. Results showed a maximal noise emission of 178 dB re 1µPa (on a frequency range from 0.7 to 50 kHz) at 1 m from the trenching area. A similar study by Bald *et al.* (2015) focused on noises from trenching and cable installation of a wind-farm platform in a sandy area of the Bay of Biscay. During the installation phase, average sound level was 188.5 dB re 1µPa (at 11 kHz) at 1m from the source. Modelling using these in situ data estimated that the underwater noise would remain above 120 dB re 1µPa in an area of 400 km² around the source.

Another lesser noise emission caused by submarine cables comes from vibrations during operation of several kinds of HVAC (High Voltage Alternating Current) cables because of the Coulomb force occurring between conductors (Zabar *et al.*, 1992). For example, a 138 kV transmission cable situated in Canada emits a SPL, for the 120 Hz tonal vibration, of approximately 100 dB re 1 µPa at 1 m (JASCO Research Ltd., 2006). Compared to cable installation, such SPL is low, but continuous because it occurs during the whole operation phase.

There is no clear evidence that underwater noises emitted during cable installation affect marine mammals or any other marine animal, although it is accepted that many marine animals (notably mammals and fishes) detect and emit sounds for different purposes such as communication, orientation or feeding. Marine mammals have high frequency functional hearing ranges from 10 Hz to 200 kHz (Richardson *et al.*, 2013), while fish typically hear at much lower frequencies, often from 15 Hz to 1 kHz (Gotz *et al.*, 2009). For other taxa, organisms such as sea turtles (O'Hara and Wilcox, 1990; Bartol *et al.*, 1999) and many invertebrates like decapods (Popper *et al.*, 2001), cephalopods (Packard *et al.*, 1990; André *et al.*, 2011) or Cnidaria (Solé *et al.*, 2016) have also been shown to be sound-sensitive. Many studies highlight the reaction of cetaceans to anthropogenic sounds of different intensities (Gordon *et al.*, 2003; Bailey *et al.*, 2010). For fish, sounds generated by ship activity can impact the behaviour of different species (Sarà *et al.*, 2007; Popper and Hastings, 2009). Anthropogenic underwater noise can affect marine life in different ways, by inducing species to avoid areas, disrupting feeding, breeding or migratory behaviour, masking communication and even sometimes causing animal death (Rossington *et al.*, 2013). So far, characterisation of acoustic thresholds causing temporary or permanent physical damage are much better described for marine mammals (Southall *et al.*, 2007; National Marine Fisheries Service, 2016), than for fish (Slabbekoorn *et al.*, 2010), and remain unknown for marine invertebrates and sea turtles (Popper *et al.*, 2014).

Compared with other anthropogenic sources of noise, such as sonar, piling or explosions, underwater noises linked to undersea cables remain low. Cable installation is a spatially localised temporary event, so the impact of noise on marine communities is expected to be minor and brief. HVAC cable vibration, although significantly lower than potential SPL during the installation phase, requires special attention though because its long-term impacts are unknown.

4.5 Reef effect

Like other immersed objects (*e.g.* shipwrecks, oil/gas platforms, and MRE devices) unburied submarine cables and associated protection/stabilisation represent permanent artificial reefs, and induce the so-called 'reef' effect (Langhamer, 2012). Artificial reefs have been commonly used for centuries to enhance fisheries, and more recently for habitat rehabilitation or coastal protection (Jensen *et al.*, 2000a). These structures are colonised by hard-substrate benthic species including epifauna and mobile macrofauna, and may also attract mobile megafauna, such as decapods or fishes.

The extent of reef effect depends on the size and nature of the cable protection structure, but also the characteristics of the surrounding area and native populations (Langhamer, 2012). Such artificial structures are expected to have limited reef effect when located within a naturally hard substratum environment. For example, Sherwood *et al.* (2016), looking at the effects of laying and operating the BassLink HVDC cable, found that, 3.5-year after the cable installation, the benthic sessile community present on the half-shell cover was similar to the one present on the surrounding basalt reef area (Figure 5.B). Other similar investigations showed no significant differences between communities on powered cables and hard bottom control areas (Dunham *et al.*, 2015; Kuhnz *et al.*, 2015; Love *et al.*, 2017a). By contrast, on soft sediments, unburied cables generate a stronger reef effect and host a new community, as illustrated by the unburied sections of the ATOC/Pioneer cable (Half Moon Bay, California) colonised by actinarians (Kogan *et al.*, 2006). In this case, sea anemones became more abundant on the cable than on the surrounding soft bottom 8 years after cable installation (Figure 5.A) and fish species were more abundant close to the cable, probably in response to increased habitat complexity compared with the surrounding environment.

'Reef effect' is often considered as a positive effect, as artificial reefs generally have higher densities and biomass of fish and decapod crustaceans than surrounding soft bottoms.

Also, when associated with a fisheries exclusion area (as described in section 4.6), artificial reefs may function as refuges for these populations, with potential spill-over benefits for adjacent stocks and fisheries (Wilhelmsson and Langhamer, 2014). This is particularly true for commercial species, like the European lobster (*Homarus gammarus* ; Figure 5.C) or edible crab (*Cancer pagurus*) observed on offshore wind-farm foundations (Hooper and Austen, 2014;

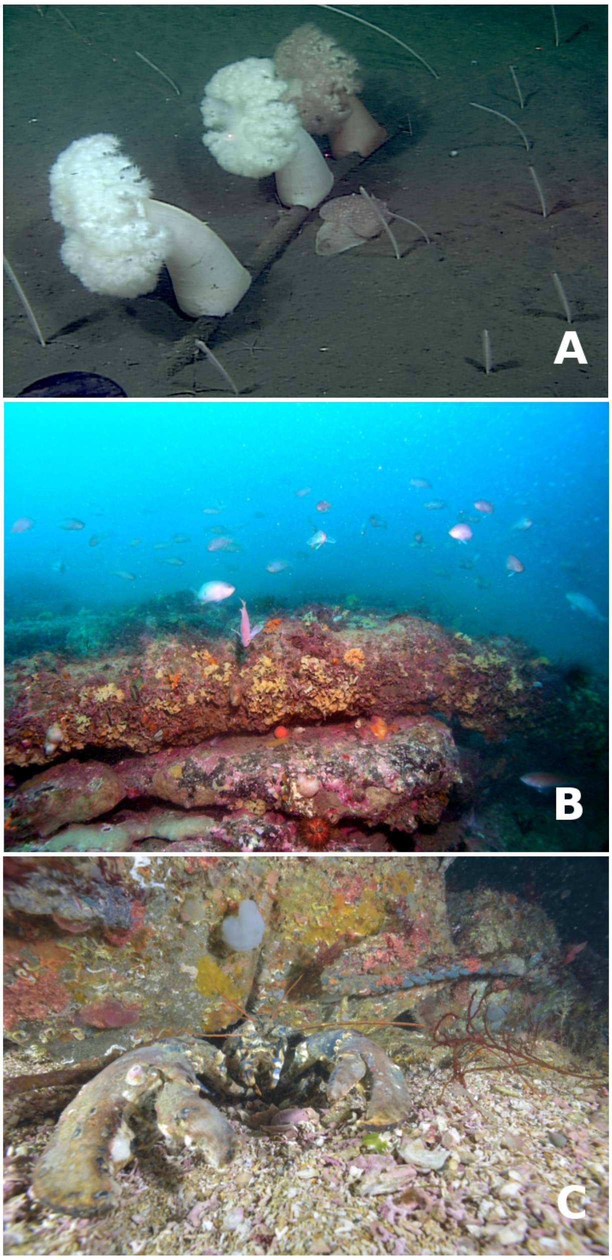


Figure 5: Photographs of laid-down cables: (A) the ATOC/Pioneer Seamount cable (California, USA) in an unconsolidated sandy silt area showing three *Metridium farcimen* settled on the cable (courtesy: Kogan *et al.,* 2006); (B) the BassLink cable (Tasmania, Australia), protected by a cast-iron half-shell, showing a heavy encrustation of algal and invertebrate species as on the underlying basalt reef (courtesy: Sherwood *et al*., 2016); and (C) the rock mattresses used to stabilize the cable connecting the Paimpol-Bréhat tidal turbine test site, France, to the land, show heavy colonisation by megafauna species like the European lobster (*Homarus gammarus*) (courtesy: Olivier Dugornay – IFREMER).

Krone *et al.*, 2017). In some cases, the cable reef effect is considered a compensatory measure for habitat destroyed during cable installation (Langhamer, 2012). Concerning dynamic cables used to connect offshore floating MRE projects, in addition to the processes of colonisation and concentration, biofouling can significantly increase cable weight and wear at least on the first tens of metres, creating technical problems (Yang *et al.*, 2017).

On the contrary, reef effect may potentially result in long-term negative effects if the structures facilitate the introduction of non-indigenous sessile species. Indeed, the number of non-native species present on new hard artificial substrate can be 2.5 times higher than on natural substratum (Glasby *et al.*, 2007). Thus, the presence of a new hard substratum, such as a cable or its protection structures, on soft sediment can potentially open a corridor to a new area for some hard-bottom sessile species. Such processes can potentially lead to the spread of new introduced species by a stepping stone process across biogeographical boundaries (Adams *et al.*, 2014). Although cable routes are narrow and often buried in areas of soft sediment, and no spread of invasive species caused by SPC has been documented, this question needs to be considered in light of the exponential growth of offshore wind farms.

4.6 Reserve effect

The potential reserve effect of SPC is linked to the limitation/interdiction by local authorities of environmentally damaging human activities (trawl fishing, anchoring, dredging, *etc.*) around the cable route during the operation phase and is considered as a positive effect for ecosystems. The size of the protected zone and the level of restriction depend on the cable installation method (buried or not), the number of cables present in the area and the size of the electric connections. For example, the Cook Strait cables have an extensive protected area to prevent damage to three submarine HVDC cables and one fibre-optic cable which link the North and South Islands of New Zealand over 40 km. An area seven kilometres wide around these cables, where anchoring and fishing of any type are prohibited, was created by New Zealand authorities, corresponding to a marine protected area of approximately 236 km² (Figure 6; TRANSPOWER, 2011).

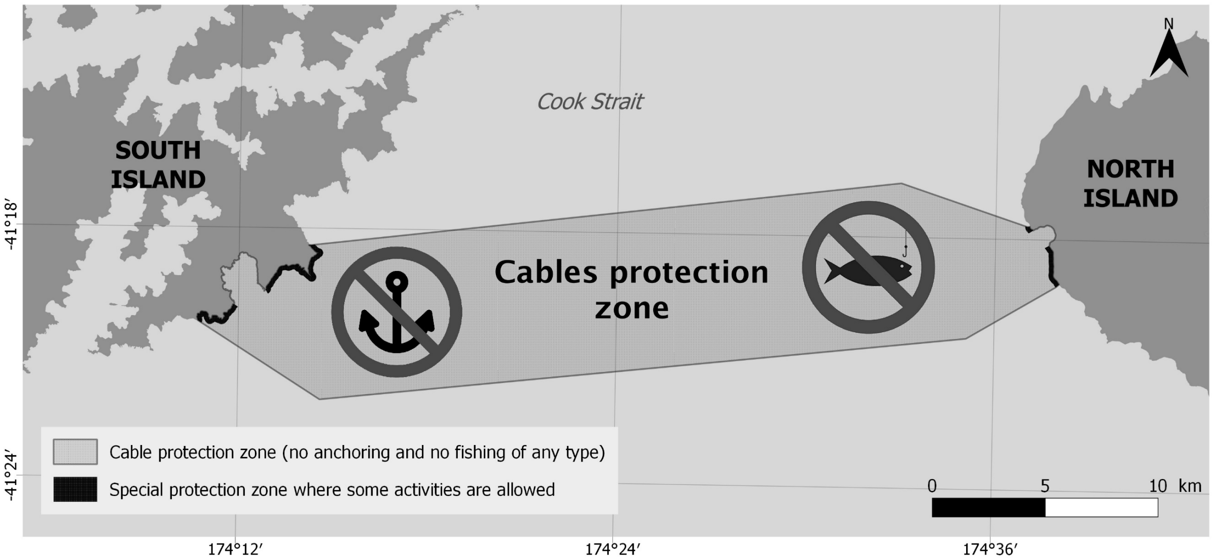


Figure 6: Protection zone of three SPC and one fibre-optic cable situated across Cook Strait, New Zealand. The total protected area covers approximately 236 km² (reproduced from TRANSPOWER, 2011).

With fishing access restricted, economically exploited sedentary species (such as scallops or clams) will be protected throughout their lives, but protection of mobile species (such as fish) will only be effective during the time they live in/pass through the cable area. The use of passive fishing equipment (nets, lines, and traps) is sometimes permitted, reducing the protection of targeted species. A study focusing on fish found no significant differences in species richness inside and outside a protection zone (Shears and Usmar, 2006). The reserve effect has been clearly demonstrated for some commercial offshore wind farms, including their associated electric cable grids. Within the Dutch Offshore Wind farm Egmond aan Zee, where all nautical activities are prohibited, the habitat heterogeneity (Lindeboom *et al.*, 2011), benthic biodiversity and possibly the use of the area by the benthos, fishes, marine mammals and some bird species have increased (although counterbalanced by a decreasing use of several other bird species). These changes occurred during the first two years of wind-farm operation, in response to the establishment of the marine protected area but also other factors, such as the reef effect of the wind turbine foundations, rockfill and cables. Nenadovic (2009) studied a protected area associated with a fibre-optic cable route on the coast of the Gulf of Maine (USA) and showed a significant difference in epifaunal community structure between protected and unprotected areas. In particular, engineer species were more frequent near the cable route. The maintenance of such species with a complex biological structure highlights the structuring effect of marine protected areas.

4.7 Electromagnetic fields

The potential impact of electromagnetic fields (EMF) is one of the environmental issues for which there is the most concern. EMF are generated by current flow passing through power cables during operation and can be divided into electric fields (called E-fields, measured in volts per metre, V/m) and magnetic fields (called B-fields, measured in µT). Electric fields increase in strength as voltage increases and may reach 1000 µV per m for an electric cable (Gill and Taylor, 2001), but are generally efficiently confined inside cables by armouring. EMF characteristics depend on the type of cable (distance between conductors, load balance between the three phases in the cable, *etc.*), power and type of current (direct *vs.* alternating current – AC generates an AC magnetic field which creates a weak induced electric field of a few µV/m, called an iE-field, near the cable), and whether it is buried or not (Ohman *et al.*, 2007; Copping *et al.*, 2016). When the cable is buried, the sediment layer does not entirely eliminate the EMF, but reduces exposure to the strongest EMF existing in direct contact with the cable (CMACS, 2003). The strength of both magnetic and induced electric fields increases with current flow and rapidly declines with distance from the cable (Normandeau Associates Inc. *et al.*, 2011).

Electric currents with intensities of 1,600 A are common in submarine cables. In response, magnetic fields of approximately 3,200 µT are generated, decreasing to 320 µT at 1 m distance, 110 µT at 4 m and values similar to the terrestrial magnetic field (50 µT) beyond 6 m (Bochert and Zettler, 2006). By contrast, according to AWATEA (2008), a standard submarine cable carrying 132 kV AC (350 A) generates a magnetic field of 1.6 µT on the "skin" of the cable (*i.e.*, within millimetres), while cables carrying 10-15 kV DC do not generate a significant magnetic field beyond a few centimetres from the cable surface. The magnetic field varies greatly as a function of the cable type, and modelling of the magnetic field induced by either DC (Figure 7.A) or AC cables (Figure 7.B) reveals this heterogeneity (1 to 160 µT at the cable surface; Normandeau Associates Inc. *et al.*, 2011). Particular attention must be paid to monopolar DC cables using sea electrodes for the return current path, the design of which leads to higher magnetic and electric fields (Normandeau Associates Inc. *et al.*, 2011; Sutton *et al.*, 2017). Although modelling presents serious limitations in the understanding of ecosystem-scale responses to such disturbances, the rare *in-situ* EMF studies available for review yielded values of measured EMF comparable to those calculated by modelling (Andrulewicz *et al.*, 2003; Sherwood *et al.*, 2016).

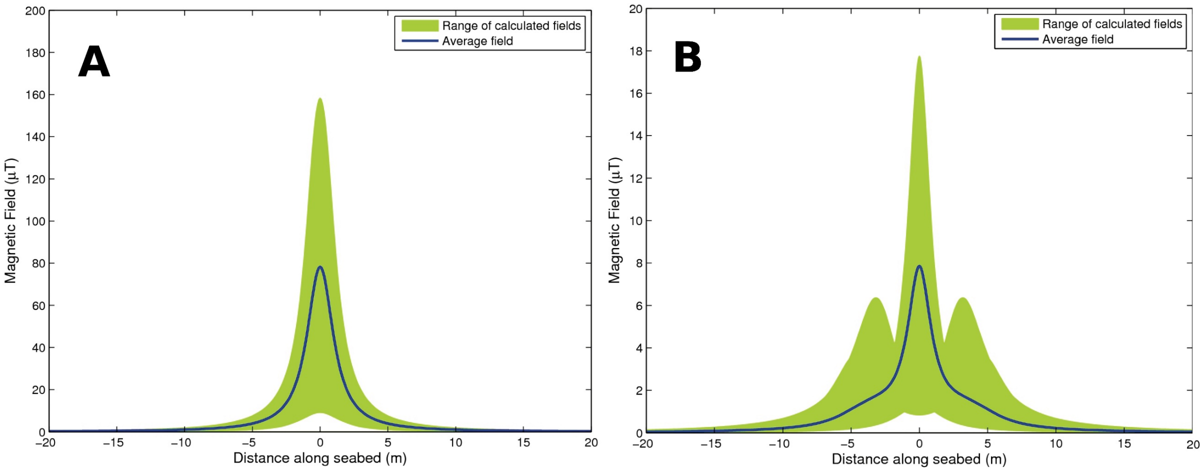


Figure 7: Modelled magnetic fields at the sediment-water interface originating from different types of buried and in operation submarine cables; (A) Calculated data based on 9 cables for DC ; (B) Calculated data based on 10 cables for AC (courtesy: Normandeau Associates Inc. *et al.*, 2011).

Many marine species around the world are known to be sensitive to electromagnetic fields, including elasmobranchs (rays and sharks), fishes, mammals, turtles, molluscs and crustaceans. Indeed, the majority of these taxa detect and utilize Earth's geomagnetic field for orientation and migration (Kirschvink, 1997; Willows, 1999; Walker *et al.*, 2002; Lohmann *et al.*, 2008; Lohmann and Ernst, 2014). Some are electrosensitive, like elasmobranchs, which are able to detect E-fields and iE-fields through specific organs called ampullae of Lorenzini (Peters *et al.*, 2007; Gill *et al.*, 2014). This electrosense can be used to detect electric fields emitted by preys, conspecifics or potential predators, as well as for orientation (Gill *et al.*, 2014). A few incidents of bites observed on unburied SPC may also be linked to the electric field emitted by cables.

Thus, SPC can possibly interact in a negative way with sensitive marine species, especially benthic and demersal organisms through:

- effects on predator/prey interactions,
- avoidance/attraction and other behavioural effects,
- effects on species navigation/orientation capabilities,
- and physiological and developmental effects.

Elasmobranchs can detect very low electric (starting from 0.005 µV cm^{-1} ; Normandeau Associates Inc. *et al.*, 2011), and magnetic (20─75 µT ; Walker *et al.*, 2002; Bochert and Zettler, 2006) fields. Power cables inducing a strong electric field can repel many elasmobranch species, preventing some movement between important areas (such as feeding, mating and nursery areas). As part of the COWRIE (Collaborative Offshore Wind energy Research Into the Environment) project, Gill *et al.* (2005) reported that Elasmobranchs are attracted by electric fields generated by DC between 0.005 and 1 µV cm^{-1}, and repelled by electric fields of approximately 10 µV cm^{-1} and higher. Mesocosm studies on impacts of EMF emitted by submarine cables on several elasmobranch species showed that the response was not predictable and seemed to be species specific, maybe even specific to individuals (Gill *et al.*, 2009). Teleosts, especially diadromous fish, also use natural EMF to migrate. Westerberg and Lagenfelt (Westerberg and Lagenfelt, 2008) showed that the swimming velocity of European eel (*Anguilla anguilla*) slightly decreased when crossing the electromagnetic field of a non-buried 130 kV cable, but did not report evidence of population-scale impact. Furthermore, no substantial impacts have been shown on physiology or survival of these taxa (Gill *et al.*, 2012; Woodruff *et al.*, 2012).

Concerning invertebrates, data are scarce except for a few studies relating to minor or non-significant impacts of anthropogenic electromagnetic fields on benthic invertebrates (Bochert and Zettler, 2004; Woodruff *et al.*, 2012, 2013, Love *et al.*, 2015, 2017b). However, a recent experimental study performed by Hutchison *et al.* (2018), highlights a subtle change in the behavioural activity of the American lobster (*Homarus americanus*) when exposed to the EMF of a HVDC cable.

Another noteworthy issue is that the groups with significant data gaps include many pelagic species (like pelagic shark, marine mammals or fishes) that interact with dynamic cables. However, it is almost impossible to evaluate such impacts at a population scale, which explains the substantial data gap.

4.8 Heat emission

SPC can also emit heat. When electric energy is transported, a certain amount is lost as heat by the Joule effect, leading to an increase in temperature at the cable surface and a subsequent warming of the immediate surrounding environment (OSPAR Commission, 2012). The constant water flow around a laid-down or a dynamic cable likely dissipates the thermal energy in close proximity and confines it at the cable surface (Worzyk, 2009). However, for buried cables, thermal radiation can significantly warm the surrounding sediment in direct contact with the cable, even at several tens of centimetres away from it, especially in the case of cohesive sediments (Emeana *et al.*, 2016). Heat emission is higher in AC than DC cables at equal transmission rates. Heat emission can be modulated by physical characteristics and electrical tension of the cable, burial depth, bottom type (thermal conductivity, thermal resistance, *etc.*) and physical characteristics of the environment (OSPAR Commission, 2008, 2012; Emeana *et al.*, 2016).

Despite the evidence for thermal radiation from subsea cables, very few studies exist on the subject and most consist of numerical modelling (Worzyk, 2009; Hughes *et al.*, 2015). One of the rare field measurement studies concerned the offshore wind array of Nysted (maximal production capacity of about 166 MW), in the proximity of two AC cables of 33 and 132 kV buried in a medium sand area, approximately 1-m deep. Results showed a maximal temperature increase of about 2.5 ºC at 50 cm under the seafloor vertical with the cable (Meißner *et al.*, 2006). Transposition of these results to other locations is difficult, considering the large number of factors impacting thermal radiation, and other field studies are necessary to gain a better understanding of thermal radiation effects.

Temperature increases near the cable can modify chemical and physical properties of the substratum, such as oxygen concentration profile (redox interface depth) and, indirectly, the development of microorganism communities and/or bacterial activity. Physiological changes in benthic organisms living at the water-sediment interface and in the top sediment layers can also potentially occur (Rhoads and Boyer, 1982; OSPAR Commission, 2008). Temperature radiation can potentially cause small spatial changes in benthic community structures by way of migratory behaviour modification, the cryophile species being excluded from the cable route in favour of other, more tolerant species.

To our knowledge, the impacts of local temperature increase caused by electric cables on benthic communities (macrofauna diversity or microbial structure and functioning) have rarely been examined, and *in-situ* investigations are lacking. Furthermore, studies using controlled temperature increases are often unrealistic about the extent of suspected warming. This considerable knowledge gap prevents drawing conclusions about ecological impacts of long-lasting thermal radiation on ecosystems, but considering the narrowness of the corridor and the expected weakness of thermal radiations, impacts are considered as non-significant. Nevertheless, new field measurements and experiments are required to fully understand this phenomenon under operational conditions and to assess its impacts on potentially exposed biological compartments.

4.9 Entanglement risks

Before the 1960s, entanglement of mobile megafauna with cables occurred during the operation phase leading, in the worst cases, to lacerations, infections, starvations and drowning of the trapped marine mammals (Benjamin *et al.*, 2014). Technical improvements made since the 1960s for installation of laid-down cables have reduced this risk (Wood and Carter, 2008). Currently, entanglement risks only concern dynamic SPC. Although this risk is considered to be non-significant, concerning a single dynamic SPC (pilot scale projects still under development), it may require more attention in the future in the case of commercial farms of floating devices and associated webs of dynamic SPC and mooring lines hanging in the water column. According to Kropp (2013), arrays of dozens of dynamic cables and mooring lines per km^2 can potentially affect large marine animals such as whales.

According to existing reports, entanglements caused by dynamic SPC will remain a low risk (Kropp, 2013; Harnois *et al.*, 2015). The large diameters of SPC (>5 cm) should make them relatively inflexible (Federal Energy Regulatory Commission, 2010), and mooring lines and dynamic SPC would be tight enough to reduce entanglement (Kropp, 2013). However, indirect entanglement resulting from discarded fishing gears wrapped around dynamic SPC (Benjamin *et al.*, 2014) may significantly impact a larger set of species, including marine mammals, sharks or fishes. Quantifying such risks will only be possible when floating MRE installations are operational. Consequently, entanglement risk remains highly speculative at this stage of knowledge, but will probably need to be considered in coming years.

5. Recommendations

5.1 Mitigation and compensation measures

Potential environmental impacts of cables must be foreseen prior to the installation phase by applying avoidance and reduction measures. In order to mitigate potential environmental disturbances caused by cabling activity, measures exist and should be applied, including the choice of an appropriate cable route and installation technique, answering the following:

- Planning the cable route to avoid impacts on habitats and benthic species that are most sensitive to disturbance or of special ecological interest (with special attention to slow-growing long-lived species). Particularly important and sensitive habitats in the North Atlantic include biogenic reefs comprising *Modiolus modiolus* (Horse mussel beds), *Sabellaria spinulosa* (honeycomb worm), maerl beds and *Zostera* seagrass meadows.
- Selecting landing zones and cable routes in order to prevent the re-mobilisation of contaminants present in sediments and contamination of the trophic food web.
- Using cable technology suitable for reducing the emission of magnetic fields, such as three-phase AC cables and bipolar HVDC transmission systems (Merck and Wasserthal, 2009), and minimising the emission of directly generated electric fields through adequate shielding (Nedwell and Howell, 2004).
- Avoiding the use of monopolar DC cables using sea electrodes, which produce toxic compounds, generate higher EMF and accelerate corrosion of manmade structures, in favour of cable systems with other return path options causing less disturbance (Sutton *et al.*, 2017).
- Deploying dynamic SPC with the lowest risks of entanglement for marine megafauna where relevant. Appropriate configurations, as for mooring lines (Harnois *et al.*, 2015), and appropriate cable type, with diameters and colours allowing visual tracking by affected species (Kropp, 2013).

- Managing installations to respect life cycles of mobile species (winter dormancy, migration, mating and/or spawning, *etc.*) to avoid disturbance of sensitive species (*e.g.*, fish, crustaceans, marine mammals, marine turtles or resting/feeding birds).
- Prioritizing burial depth appropriate to the substratum type. To reduce exposure of sensitive species to electromagnetic fields and heat emission, the physical distance between animals and the cable can be adjusted. According to models proposed by Normandeau *et al.* (2011 ; Figure 7), the EMF level at the water-sediment interface with a 2m burial depth would be approximately 25% of its initial value- versus 60% for a 1m burial depth.
- Prioritizing the laid-down option rather than burying in the presence of unavoidable fragile benthic soft bottom habitats (*e.g.*, seagrass beds; Bacci *et al.*, 2013).
- Installing devices with a strategy to reduce electrical connections and limiting the number of export cables (*i.e.*, when several MRE projects are present in close proximity).

To complement reduction and avoidance strategies, compensation measures should be considered if residual impacts persist. When possible, and only after having addressed avoidance and reduction options, compensation measures may be applied directly to the implantation site, or in close proximity. Discussions between stakeholders are recommended to establish parameters for scale and responsibilities for compensation measures.

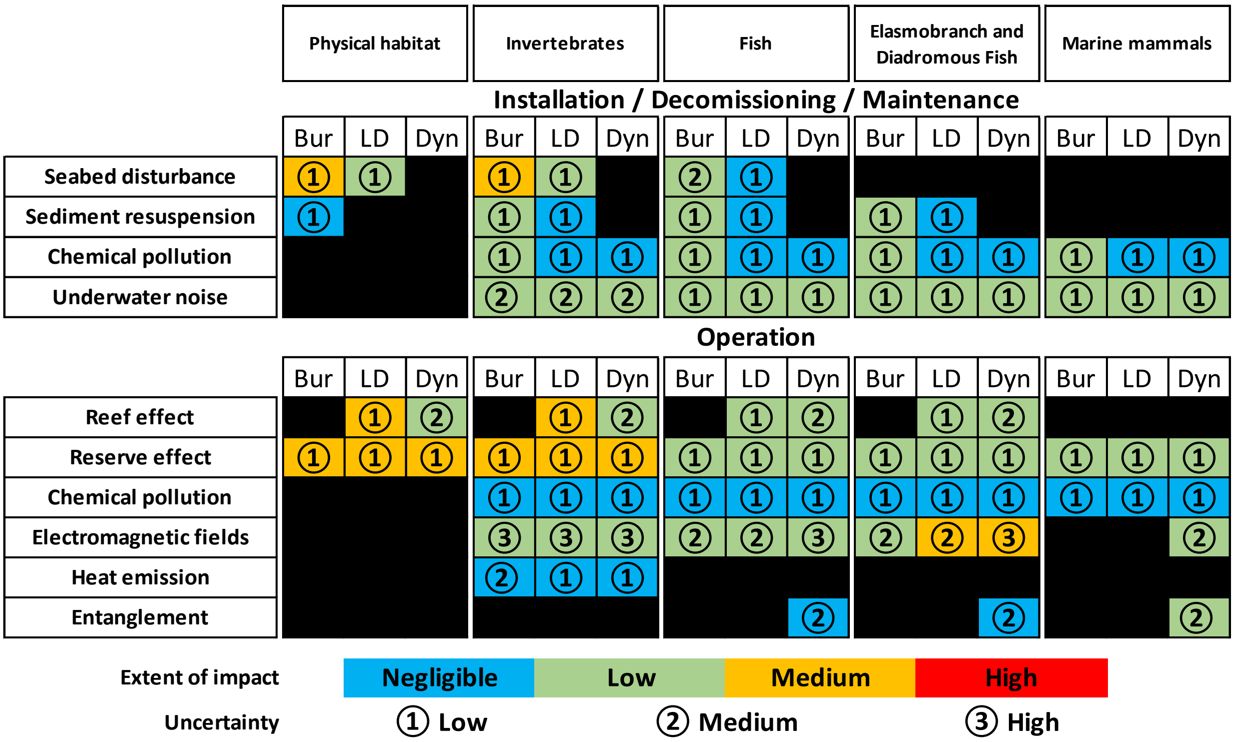
A possible form of compensation measures can consist in performing experimental work to improve knowledge about ecosystems functioning and resilience after a disturbance, in order to select appropriate ecological engineering strategies for future projects. For example, on the Paimpol-Bréhat French tidal turbine test site, the route of the cable connecting turbines to the land crossed important seagrass meadows containing *Zostera noltei* and *Z. marina*. In response, the prime contractor (EDF, Electricité De France) developed an experimental protocol aiming to transplant some seagrass plants located on the route area to another barren place before cable burial. Such measures aimed to test transplantation techniques and acquire knowledge about the mechanism of recolonisation by seagrass after installation of a cable (Barillier *et al.*, 2013). Similar transplantation experiments are currently being tested in the context of SPC installation (*e.g.*, ongoing project by Red Eléctrica de España in Majorca and Ibiza).

Environmental monitoring strategies performed in parallel with a cable installation project must be appropriate. Environmental monitoring should: *i*) verify the impact predictions made in the environmental impact study and detect unforeseen alterations, *ii*) ensure the fulfilment of mitigating measures proposed, and *iii*) provide data to improve future environmental impact assessments and installation plans (Moura *et al.*, 2010).

5.2 Future research priorities

A hierarchical model of potential impacts based on the expected levels of ecological impact and the associated levels of scientific knowledge (or uncertainty) is presented in table 2. This synthetic output corresponds to a concerted expert judgement of the authors, and takes into account the main conclusions of the literature cited in this paper. The main priorities concern potential impacts of electromagnetic fields, reef and reserve effects and benthic habitat disturbance. A substantial data gap remains concerning the impacts of EMF because data on sensitivity thresholds or tolerance are available only for a small number of taxa. Major uncertainties therefore remain for several large groups (cetaceans, pinnipeds, fishes, crustaceans, and many pelagic species ; Normandeau Associates Inc. *et al.*, 2011). Better knowledge of the different sensitivity thresholds is needed to fill these data gaps, especially for several key species at different stages of their development. Additionally, environmental issues may arise following industrial-scale deployment of MRE devices using multiple submarine electric cables installed in close proximity and creating a network impacting a large area. The cumulative effects of more than one activity or perturbation factor, which may act in synergy, must be considered (Crain *et al.*, 2008). For example, recovery of benthic communities after

Table 2: Synthesis of the importance of potential impacts caused by Submarine Power Cables (SPC) on different marine compartments during installation, operation, maintenance and decommissioning, based on the author's interpretation of the reviewed literature. For each interaction, the extent of the impact and associated uncertainty are each quantified as 'Negligible', 'Low', 'Medium' or 'High'. Bur = Buried SPC; LD = Laid-Down SPC; Dyn = Dynamic SPC. Black fill = no impact.



Installation / Decomissioning / Maintenance

	Physical habitat			Invertebrates			Fish			Elasmobranch and Diadromous Fish			Marine mammals		
	Bur	LD	Dyn	Bur	LD	Dyn	Bur	LD	Dyn	Bur	LD	Dyn	Bur	LD	Dyn
Seabed disturbance	🟧 ①	🟩 ①	⬛	🟧 ①	🟩 ①	⬛	🟩 ②	🟦 ①	⬛	⬛	⬛	⬛	⬛	⬛	⬛
Sediment resuspension	🟦 ①	⬛	⬛	🟩 ①	🟦 ①	⬛	🟩 ①	🟦 ①	⬛	🟩 ①	🟦 ①	⬛	⬛	⬛	⬛
Chemical pollution	⬛	⬛	⬛	🟩 ①	🟦 ①	🟦 ①	🟩 ①	🟦 ①	🟦 ①	🟩 ①	🟦 ①	🟦 ①	🟩 ①	🟦 ①	🟦 ①
Underwater noise	⬛	⬛	⬛	🟩 ②	🟩 ②	🟩 ②	🟩 ①	🟩 ①	🟩 ①	🟩 ①	🟩 ①	🟩 ①	🟩 ①	🟩 ①	🟩 ①

Operation

	Physical habitat			Invertebrates			Fish			Elasmobranch and Diadromous Fish			Marine mammals		
	Bur	LD	Dyn	Bur	LD	Dyn	Bur	LD	Dyn	Bur	LD	Dyn	Bur	LD	Dyn
Reef effect	⬛	🟧 ①	🟩 ②	⬛	🟧 ①	🟩 ②	⬛	🟩 ①	🟩 ②	⬛	🟩 ①	🟩 ②	⬛	⬛	⬛
Reserve effect	🟧 ①	🟧 ①	🟧 ①	🟧 ①	🟧 ①	🟧 ①	🟩 ①	🟩 ①	🟩 ①	🟩 ①	🟩 ①	🟩 ①	🟩 ①	🟩 ①	🟩 ①
Chemical pollution	⬛	⬛	⬛	🟦 ①	🟦 ①	🟦 ①	🟦 ①	🟦 ①	🟦 ①	🟦 ①	🟦 ①	🟦 ①	🟦 ①	🟦 ①	🟦 ①
Electromagnetic fields	⬛	⬛	⬛	🟩 ③	🟩 ③	🟩 ③	🟩 ②	🟩 ②	🟩 ③	🟩 ②	🟧 ②	🟧 ③	⬛	⬛	🟩 ②
Heat emission	⬛	⬛	⬛	🟦 ②	🟦 ①	🟦 ①	⬛	⬛	⬛	⬛	⬛	⬛	⬛	⬛	⬛
Entanglement	⬛	⬛	⬛	⬛	⬛	⬛	⬛	⬛	🟦 ②	⬛	⬛	🟦 ②	⬛	⬛	🟩 ②

Extent of impact	🟦 Negligible	🟩 Low	🟧 Medium	🟥 High
Uncertainty	① Low	② Medium	③ High	

cable installation may be slower and less efficient if the benthic ecosystem is already threatened by other anthropogenic disturbances such as chemical pollution, eutrophication, or invasive species (especially in enclosed and shallow areas). The assessment of impacts due to interactions between different kinds of disturbances remains highly speculative, partly since environmental impacts of single cables are still poorly understood.

6. Conclusions

Although SPC have been used since the mid-19th century, environmental concerns associated with their installation and operation are much more recent. This is due to the rapid expansion of MRE and the growing demand for electric interconnections between countries that have adopted a common energy strategy. Thus, even though they are usually considered low impact, a better knowledge of potential ecological impacts is becoming essential.

The main potential environmental impacts associated with SPC during their operational phase are those related to the production of electromagnetic fields, the creation of artificial reefs and “reserve effects” caused by the interdiction of certain human activities. Cable installation, maintenance and decommissioning also impact the environment, causing direct benthic habitat modification, which can be problematic in the case of sensitive bioconstructed habitats. These phases of SPC may also induce significant particle and pollutant resuspension events in very confined and modified shallow coastal areas. Mitigation measures are possible before, during or after projects to limit the ecological impacts of SPC and associated maritime operations.

Although environmental effects generated by SPC are recognised, their amplitude is generally considered to be non-significant. Most of the time, these disturbances likely create minor and short-term impacts on ecosystem structure and functioning. Nevertheless, the nature and amplitude remain uncertain for some categories of specific impacts, particularly for EMF impacts on elasmobranchs, diadromous fishes and invertebrates, as well as for cumulative impacts. Despite these drawbacks, the present review provides a quantification and ordering of the different impacts of SPC on marine environments and offers updated practical recommendations for developers.

Acknowledgements

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Chapter 2

Optimised underwater imagery method to investigate epibenthic communities

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Abstract

Underwater imagery is increasingly used as an effective and repeatable method to monitor benthic ecosystems. Nevertheless, extracting relevant ecologically information from a large amount of raw images remains a time-consuming and somehow laborious challenge. Thus, underwater imagery processing needs to strike a compromise between time-efficient image annotation and accuracy in quantifying benthic community composition. Designing and implementing robust image sampling and image annotation protocols is therefore critical to rationally address these trade-offs between ecological accuracy and processing time.. The aim of this study was to develop and to optimise a reliable image scoring strategy based on the point count method using imagery data acquired on tide swept encrusting benthic communities. Using a stepwise approach, we define an underwater imagery processing protocol that is effective in terms of *i*) time allocated to overall image analysis (~45 minutes per picture and 6.75 hours for all replicates of a site), *ii*) reaching a satisfactory accuracy to estimate the occurrence of dominant benthic taxa (with a mean percentage cover ≥ 5%) and *iii*) adopting a sufficient taxonomic resolution (*i.e.* the so-called 'CATAMI' classification that is accessible to non-experts) to describe changes in community composition. We believe that our method is well adapted to investigate the composition of epibenthic communities on artificial reefs, and can be useful in surveying colonisation of other human structures (wind turbine foundations, pipelines *etc.*) in coastal areas. Our strategy meets the increasing demand for inexpensive and time-effective tools for monitoring changes in benthic communities in a context of increasing coastal anthropogenic pressures.

Keywords

Underwater magery; sampling design; benthic monitoring; fouling community; taxonomic resolution

1. Introduction

Coastal benthic ecosystems are increasingly impacted by a cocktail of anthropogenic pressures, including sea bottom fishing (trawling/dredging in particular), harbour development, tourism, industry, energy production, urban coastal development, *etc* (Halpern *et al.*, 2008b). As a direct consequence, both quality and extent of vulnerable coastal habitats have declined worldwide (Jackson *et al.*, 2001; Lotze and Milewski, 2004; Lotze *et al.*, 2006a; Le Pape *et al.*, 2007). In this context, there is an increasing demand for a regular cost-effective monitoring of the ecological quality of ecosystems. Underwater imagery has for several reasons been increasingly used as an effective and repeatable method to monitor benthic ecosystems. Firstly, the collection of large amounts of high-resolution information on benthic biodiversity is rapid; secondly, the method is non-invasive, which is key for long-term monitoring of selected sites (no or limited perturbation of ecological communities); and thirdly, cameras operated by scuba divers or underwater vehicles provide access to remote sites (for instance due to depth or seafloor topography) that are difficult to sample with classic methods. Consequently, underwater imagery is widely used to describe a diverse range of coastal benthic habitats such as tropical coral reefs (Brown *et al.*, 2004; Lam *et al.*, 2006; Dumas *et al.*, 2009; Molloy *et al.*, 2013), algal assemblages (Preskitt *et al.*, 2004; Vroom and Timmers, 2009; Deter *et al.*, 2012; Berov *et al.*, 2016), rocky substrates (Macedo *et al.*, 2006; Van Rein *et al.*, 2011), artificial reefs (Page *et al.*, 2006; Walker *et al.*, 2007; Jerabek *et al.*, 2016; Jimenez *et al.*, 2017), highly hydrodynamic sites (Foveau *et al.*, 2017; O'Carroll *et al.*, 2017a) and mesophotic or deep-sea ecosystems (Sen *et al.*, 2016; Domke *et al.*, 2017; Marzloff *et al.*, 2018).

While underwater imagery produces large amounts of raw data of seafloor communities, the extraction of ecologically relevant information through taxonomic identification to species level is often challenging, sometimes impossible without collected specimens, expert knowledge or extensive taxonomy literature (Althaus *et al.*, 2015). So, benthic ecologists have developed classification methods adapted to assess benthic biodiversity solely from imagery. Such classifications are often region-specific and inconsistent as they may use different terminologies to label a given category of organism (Schlacher *et al.*, 2010; Harrison and Smith, 2012; Oh *et al.*, 2015). In response to these inconsistencies across worldwide image-based benthic surveys, Althaus *et al.* (2015) developed a standardised classification for identifying benthic categories from underwater imagery called CATAMI (Collaborative and Automated Tools for Analysis of Marine Imagery), which aims to facilitate image annotation, data management and data sharing.

However, even with the appropriate classification, the extraction of relevant information (taxon occurrence, count of individuals or colonies, size or cover estimation *etc.*) from the entire raw images relies on laborious and time consuming analysis (Pech *et al.*, 2004; Preskitt *et al.*, 2004; Nakajima *et al.*, 2010). For instance, concerning benthic sessile communities on hard substrates, the challenge lies in quantifying the occurrence or percentage cover of each taxon on each image to describe the community composition. This can be achieved by labelling all organisms visible on the picture or exhaustively delineating their shape (for percentage cover). However, this method is not applicable to a large set of images or to diverse encrusting communities as it is highly time consuming. The 'point count' approach provides a reliable time-effective alternative to this comprehensive image analysis (Pielou, 1974). It consists in distributing a certain number of points on an image, and then visually labelling the benthic category (taxa or substratum type) lying under each point. Then, the community composition can be assessed by calculating the percentage cover of each category as the ratio between the number of points attributed to a target category and the total number of points, on a given sampled surface. This method was facilitated by the development of dedicated software, such as CPCe (Coral Point Count estimation, Kohler and Gill, 2006), PhotoQuad (Trygonis and Sini, 2012) or more recently BIIGLE (Langenkämper *et al.*, 2017). However, the accuracy of the percentage covers estimated with this method increases with the density of points scored and depends also on the method used to project points on the image. So, the optimal point density strikes a compromise between the desired accuracy level and the time needed for image processing. It also depends on the seafloor area sampled per image, as well as the size, relative occurrence and distribution patterns of the targeted taxa (Pante and Dustan, 2012; Perkins *et al.*, 2016). Except for a limited number of methodological studies (Dumas *et al.*, 2009; Deter *et al.*, 2012; Pante and Dustan, 2012; Berov *et al.*, 2016; Perkins *et al.*, 2016), the chosen density of points scored per image is rarely justified. Furthermore, all of these mentioned studies focus on benthic organisms of sizes superior to 10 cm (*i.e.* megafauna/flora). Thus, to our knowledge, no information is as yet available concerning optimal point count method when targeting macroepibenthic communities.

The aim of this study was to develop and optimise a protocol of underwater image analysis suitable for describing macroepibenthic communities colonising natural and artificial hard substrates. Using a stepwise approach, we defined a reliable image scoring strategy using imagery data acquired on subtidal tide swept encrusting benthic communities by optimising: (*i*) density of points, (ii) way of point projection, (*iii*) total sampling area and (*iv*) taxonomic resolution (by testing the CATAMI classification). Finally, we discuss our results in the broader context of possible applications of the point count method to score underwater imagery of benthic ecosystems.

2. Methods

2.1 Context of the study

We developed an optimised protocol of underwater image analysis by studying macroepibenthic community settling on marine renewable energy installations. Environmental studies on these installations frequently require method as image-based monitoring, to characterise the environmental impacts associated with deployment of offshore generators (*e.g.*

wind farms, tidal turbine *etc.*). Marine renewable energy's devices and their associated infrastructures (maintenance platforms, submarine power cables and associated protection and stabilising structures *etc.*) constitute permanent artificial reefs (Wilson and Elliott, 2009; Langhamer, 2012) colonised by hard-substrate benthic species, including epifauna.

2.2 Study site

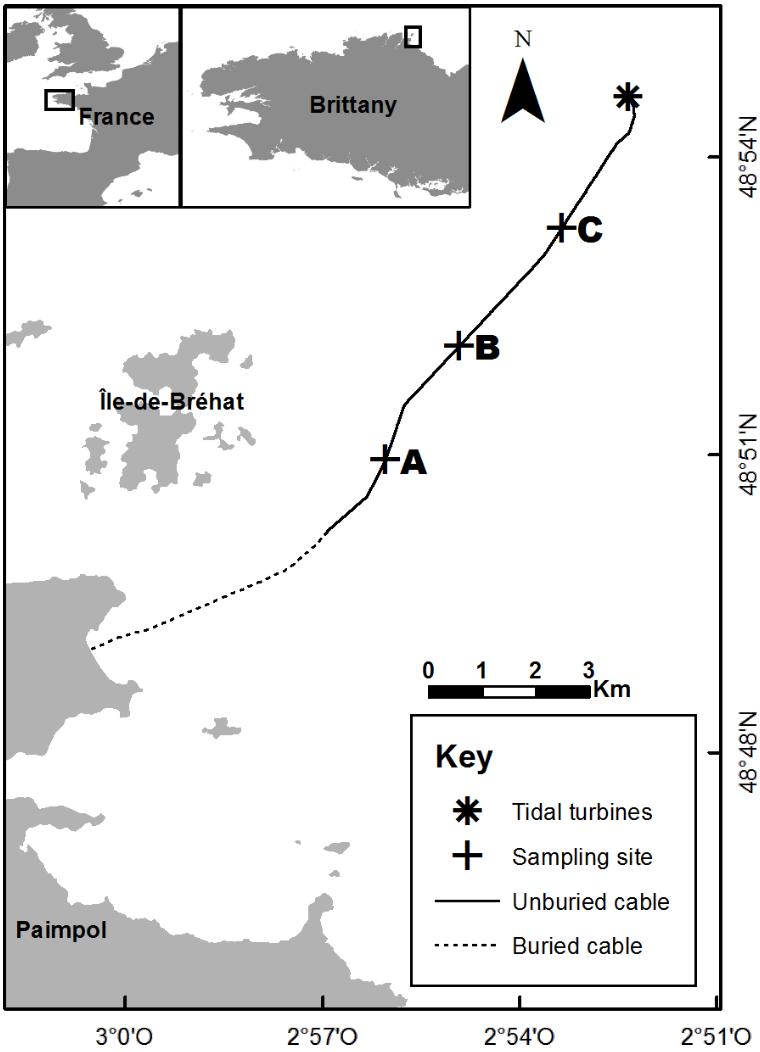

Figure 1: Map of the study area off the north coast of Brittany in western France (top-left and top-centre panels), which shows the location of the Paimpol-Bréhat tidal turbine test site where A, B and C indicate the three study sites surveyed along the cable route (bottom).

The study site is a 15 km-long submarine power cable (8 MVA - 10 kVDC) set up in 2012 to connect the tidal test site of Paimpol-Bréhat (Brittany, France; Figure 1) developed by Electricité de France – Energies Nouvelles (EDF-EN). Because of the seafloor characteristics (dominance of pebbles and rocks), 11 km of cable are unburied but fully protected with nested iron half-shells (50 cm long, 15 cm diameter). The cable is also stabilised by 120 concrete mattresses (6 m long, 3 m wide) installed in 2013 (Figure 2), which prevent its displacement due to high hydrodynamic site conditions (current speed up to 5 knots during Spring tides). Due to several setbacks in the commissioning progress of the project, no electric current has transited through the cable so far and associated protection structures have actually acted as a simple artificial reef.

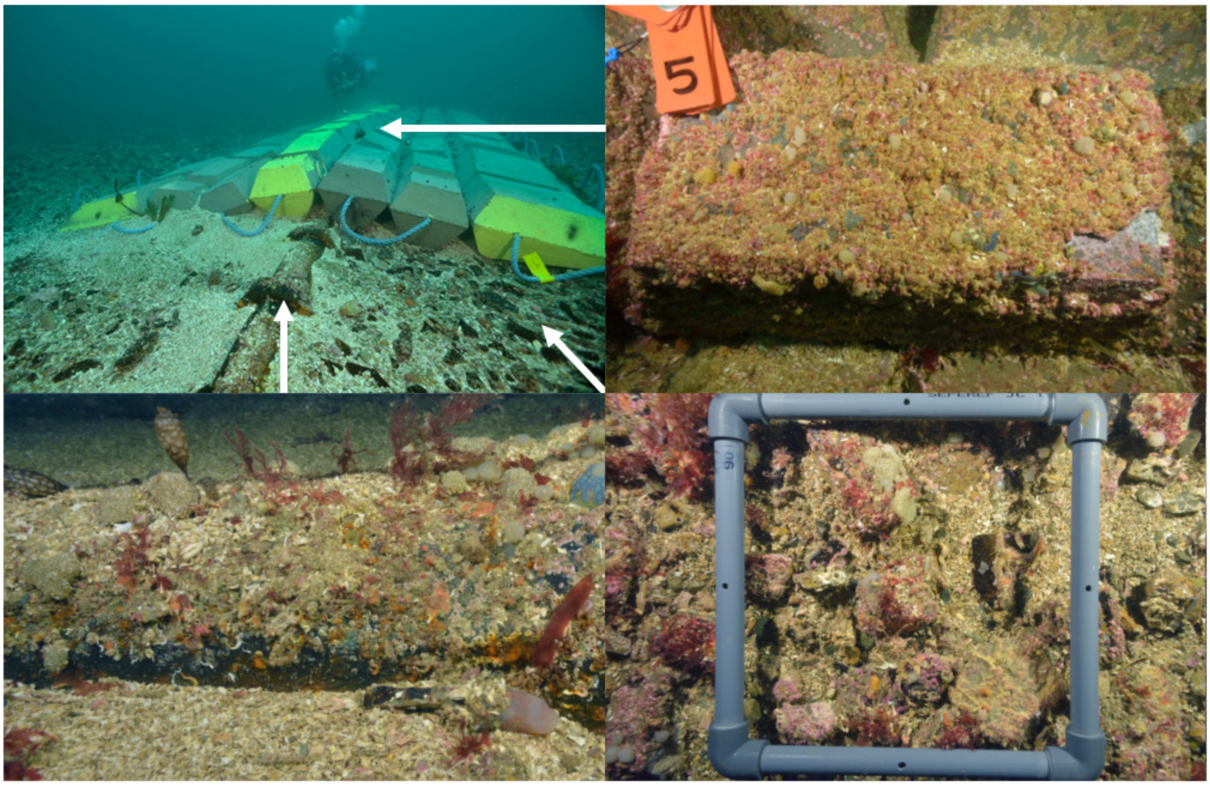

Figure 2: Overall view of one of the survey sites including cast-iron half-shells, a concrete mattress (freshly installed) and natural habitat (top-left); Close-up views of one of the mattresses concrete units (top-right), one cast-iron half- shell (bottom-left), and one of the quadrats placed on the natural habitat (bottom-right) (courtesy: Olivier Dugornay).

2.3 Image acquisitions

A yearly underwater imagery benthic survey undertaken by divers was started in September 2014, at three sites along the cable route: A, B and C (Figure 1). The three sites present similar depths (between 18 and 20m). At each site, high-definition photographs of benthic communities were taken by divers both on natural bottom and artificial habitats that protect the cable (iron half-shells for sites A, B and C and concrete mattresses for sites B and C) with the following strategy:

i) each side of each 50 cm long iron half-shell on a 10 m transect;

ii) 16 regularly spaced concrete units (whether 47x38 cm or 47x20 cm) of the mattress;

iii) quadrat of 25*25 cm randomly placed on the natural habitat 10 m apart from the cable route (Figure 3A).

Photographs were taken at a resolution of 37 million pixels per image with a Nikon D810 inside a Ikelite underwater housing, with a 20 mm lens and 2 Keldan LED lights (105W, 9000 lumens). All images were calibrated with a scale bar. A total of six campaigns carried out over four years (September 2014, March and September 2015, September 2016, September 2017 and March 2018) produced more than 1,500 pictures.

2.4 Point count strategy at the image level

Briefly, we followed a 3-step approach (detailed in the following sections) to define the optimal image scoring strategy, in terms of number of points and point projection method, by:

i) describing exhaustively the benthic biodiversity on 9 ‘reference’ images (3 for each type of habitat);

ii) using these 9 ‘reference’ images, assessing how the point sampling designs (point density combined with projection method) impact the estimation of benthic biodiversity;

iii) based on the obtained relationships, identifying the optimal density of point and projection method.

Exhaustive analysis of ‘reference’ images

We selected one image representative of the complexity of the benthic community (in terms of diversity and spatial heterogeneity) for each habitat (half-shell, mattress and natural bottom) and from three different surveys (2014, 2015, 2017). On these nine 'reference' pictures, an area equivalent to 625 cm² was cropped for analysis. Using ArcGIS, all benthic categories (being either taxa or substrates) visible in this area were manually cut out and annotated after visual identification (at the lowest possible taxonomic level for biological categories). The comprehensive scoring of each reference image took between 14 and 21 hours. This first step resulted in nine raster files that provided a comprehensive description of benthic biodiversity, and for which each pixel was assigned to a benthic category (Figure 3B).

Point count simulations

Then, we tested how a range of point count image-scoring strategies effectively reflects the true benthic community composition. These point sampling strategies were generated by combining 100 different point densities (from 5 to 500 points per 625 cm² image area, by increments of 5 points) and two different projection methods (random and stratified-random; Figure 3C). For each of the nine 'reference' images, 1,000 random simulations were performed for each combination, giving a total of 200,000 simulations. For each simulation, we computed the percentage cover of each benthic category. All the simulations were performed with RStudio (v 1.0.0143) using the SpCosa package to implement stratified-random sampling (Walvoort *et al.*, 2010).

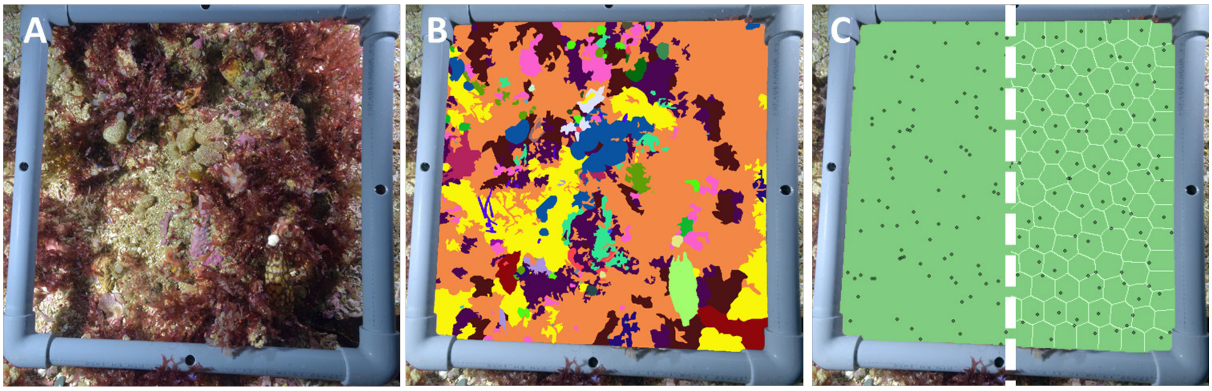

Figure 3: Illustration of image processing. (A) An example of 25*25 cm quadrat image of the natural bottom (Site B September 2017 – Courtesy: Olivier Dugornay); (B) Result of the exhaustive picture taxonomic analysis performed with ArcGIS, each colour corresponding to a different benthic category (*i.e.* substratum type or taxon); (C) Example of point count simulation with 200points (*i.e.* 0.32pt.cm-2), using the random (left) or stratified-random (right) projection methods.

Selection of the optimal method

Our aim was to achieve an optimal scoring method that would enable us to estimate the occurrence of benthic categories with a percentage cover superior to 5% and an accuracy corresponding to a CV of the estimated occurrence ≤ 0.25. This threshold was chosen because it has been shown that the point count method is generally not suitable to accurately characterise benthic categories with a percentage cover inferior to 5% (Dumas *et al.*, 2009; Deter *et al.*, 2012; Perkins *et al.*, 2016).

To assess the accuracy of alternative point sampling strategies, we computed the Coefficient of Variation (CV, see Eq. 1) of the estimation of percentage cover computed for each category across 1,000 random simulations. The CV constitutes a good proxy of the accuracy and the repeatability of a measure (the higher the CV, the lower the accuracy).

$$\text{(Eq. 1) } \mathbf{CV}(\mathrm{i,n,m}) = \frac{\sigma\,(\mathrm{i,n,m})}{\bar{\mathrm{X}}(\mathrm{i,n,m})}$$

with i, the i^{th} benthic category; n, the number of points scored ($5 \leq \mathrm{n} \leq 500$ by interval of 5); m, the projection method (random or stratified-random); $\bar{\mathrm{X}}$(i,n,m), the mean percentage cover of category i across 1,000 simulations under a given method; σ(i,n,m), the standard deviation of the percentage cover of category i across 1,000 simulations under a given method.

We used a nonlinear model (function *nls* of the R package *stats*) using Rstudio (RStudio Team, 2015; v 1.0.0143) to characterise the number of points required to reach a CV of 0.25 for taxa that exhibit a range of percentage cover (represented in bold white line on Figure 4). The black dotted line highlights the specific case of benthic categories associated with a 5% cover. For each habitat (natural bottom, iron half-shell, concrete mattress) and projection method, we identified the minimum number of points required to achieve a CV ≤ 0.25 for benthic categories with a 5% cover (which corresponds to our accuracy threshold). Based on these CV estimates, we identified an optimal strategy across all habitats, in terms of minimum number of points and projection method.

2.5 Sampling effort at the site level

Once the optimal point count strategy is adopted to efficiently capture benthic community composition within an image (which could be considered as a replicate), the second step was to determine the most relevant sampling area, *i.e.* the total area observed at the site level for a given habitat (defined as number of images * quadrat size).

To assess this optimum sampling area, we first applied the optimal point count method (defined in the previous part) to all the analysable images of one site collected during one survey (*i.e.* site B sampled in September 2015, 110 images, which corresponds to the largest dataset). These image analyses were performed using the free software PhotoQuad (Trygonis and Sini, 2012). A benthic category was assigned to each projected point, and the percentage cover was estimated for each encountered category. The biological categories were determined at the lowest possible taxonomic level (*i.e.* species when possible). For natural bottom and concrete mattresses, 55 and 21 photos of 625 cm^2 were analysed respectively, and 34 photos of 400 cm^2 were analysed for iron half-shells. For the rest of the procedure, only the biological categories were considered in order to focus on the composition of the benthic communities.

Then, we used Monte-Carlo simulations to construct curves of taxonomic similarity-area for each type of habitats, a straightforward approach to determine adequate sampling size (Weinberg, 1978; Kronberg, 1987; Schmera and Eros, 2006). For a given sampling area (*n* images), two independent sets of *n* images were randomly chosen from the total data set. Bray-Curtis similarity indices were calculated to compare the diversity sampled in each of these 2 sets. This process was repeated 1,000 times for each level of sampling area. We then produced habitat-specific (*i.e.* natural bottom, mattress, iron half-shell) similarity-sampling area curves using the package CommEcol (Schneck and Melo, 2010) in RStudio (v 1.0.0143) by plotting mean estimates of Bray-Curtis similarity for each level of sampling effort. The nonlinear relationship between similarity and the sampling area was modelled using the function *nls* of the R package *stats.* We defined the optimum sampling area as the number of survey images associated with the asymptotic point of the similarity-sampling area curve, *i.e.* when increasing sample number only marginally increases between-sample similarity (by less than 0.1%).

2.6 Taxonomic resolution

The CATAMI classification developed for underwater image analysis, combines a coarse-level taxonomy and the integration of organism morphology for the identification of benthic taxa (Althaus *et al.*, 2015).We tested this classification frame by examining how it affects diversity patterns obtained with the finest taxonomic frame that we could provide.

We used the same data set (110 images site B, September 2015) that served to determine the optimum sampling area at the site level. All the taxa identified at the lowest taxonomic level are labelled to corresponding CATAMI groups. Thus, two different community matrices were created, corresponding to two different taxonomic resolutions: the lowest taxonomic level, hereafter called LTL and the CATAMI resolution. Resemblance matrices were computed for both resolutions by calculating Bray-Curtis similarities between samples. The two similarity matrices were visually compared by computing two nMDS (non-metric Multi-Dimensional Scaling) ordinations with Rstudio (v 1.0.0143). Potential correlation between the LTL and the CATAMI matrices were examined using Spearman's rank correlation coefficient and the significance of the relationship was determined with the Monte-Carlo permutation routine RELATE of the PRIMER program (Clarke & Warwick 1994).

2.7 Bibliographic review

In order to compare our results (regarding point density, projection method and sampling area) to published protocols, a targeted bibliographic review was performed. We searched for peer-reviewed scientific publications and technical reports that used the point count method to characterise benthic communities, using online databases and internet search tools (*i.e.* Science Direct, Google Scholar, ResearchGate). In particular, we looked for studies that cited the original papers describing the CPCe (Kohler and Gill, 2006) and the PhotoQuad image-annotation software (Trygonis and Sini, 2012). We systematically classified the published methods in terms of point density (number of point cm^{-2}), projection method (random, stratified-random, regular), quadrat size (in m^2), sampling area per site (quadrat size in m^2 * number of replicates), nature of the studied community and the estimated mean size of the targeted taxa. Also, by considering an average analysis time of 0.18 minutes for each point projected (based on timed image scoring), we estimated the time needed to analyse a single picture (0.18 min pt^{-1} * density of point * quadrat size) and a full set of pictures of a site (0.18 min pt^{-1} * density of point * sampling area per site) in each study.

3. Results

3.1 Point count optimisation at the image level

Figure 4 presents the aggregated results across all the point densities simulated (from 5 to 500 points per image) to determine the scoring effort required per image to reach a satisfactory accuracy for each habitat type (*i.e.* natural bottom, mattress, half-shell) and each type of point projection (random or stratified-random). Across all simulations, the CV of the estimated percentage cover of taxa decreases rapidly as the number of points and/or the occurrence of the benthic categories increase. This reflects that percentage cover estimates are more accurate for a high density of point and/or for more abundant benthic categories (common taxon). For instance, across all investigated habitat and projection methods, ~50 point scores are sufficient to achieve a CV ≤ 0.25 for abundant taxa (percentage cover > 20%). For a given point score strategy (point density and projection method), the accuracy of percentage cover estimate varies according to the habitat considered, in particular for rare taxa (percentage cover < 10%). To reach a CV value of 0.25 for ‘5% percentage cover’ categories, 322, 345 and 342 randomly projected points are needed, for half-shell, mattress and natural bottom, respectively (Table 1). When using stratified-random projection, the number of points needed dropped to

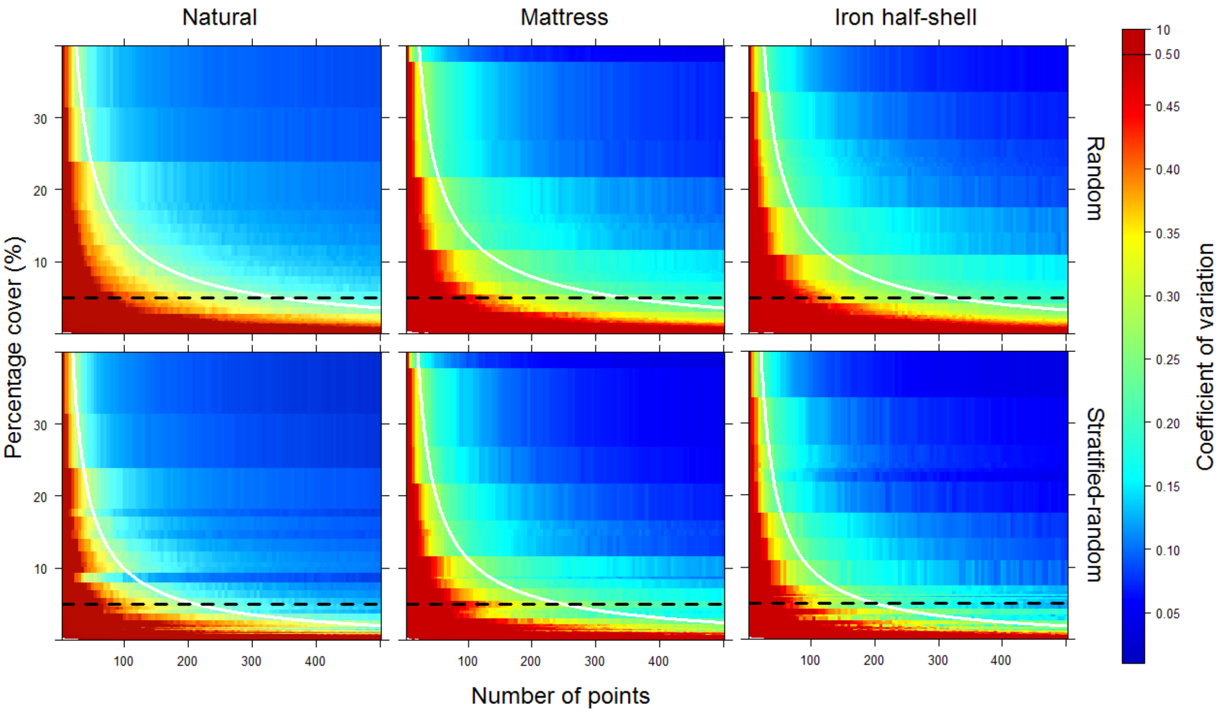

Figure 4: Change in Coefficient of variation (CV) of percentage cover estimates as a function of number of points scored per image (x-axis) and actual percentage cover of benthic categories (y-axis). The 6 panels correspond to the two different projection methods (*i.e.* random and stratified-random) and the three different habitats (*i.e.* natural, mattress, half-shell). CV, represented by the colour scale, indicates the proportion of variation around mean cover estimates (the smaller the CV, the more accurate the estimate). The white thick line delineates CV values of 0.25. The black horizontal dotted line represents benthic categories with a percentage cover of 5%. We defined the optimal number of points in each scenario as the intersect between these two lines.

199, 248 and 211, respectively. Beyond that, improving the accuracy of percentage cover estimates of rare categories is costly in terms of scoring effort since approximately 50% and 300% extra points are required to reduce CV to 0.2 and 0.1, respectively (Table 1). Consequently, the optimal method that fulfils our criteria (*i.e.* CV of 0.25 for rare taxa) requires 248 points per picture of 625 cm² (rounded to 250 points *i.e.* 0.4 pt cm^{-2}) using a stratified-random projection.

Table 1: Number of points required to reach a CV of 0.1, 0.2 and 0.25 for 5% cover benthic categories, the two different projection methods and the three different habitats.

		Natural		**Mattres**		**Iron half-shell**	
Percentage cover	Coefficient of variation	Stratified-random	Random	Stratified-random	Random	Stratified-random	Random
5%	**0.1**	727	1733	873	1526	783	1502
	0.2	290	529	351	517	288	490
	0.25	**211**	**342**	**248**	**345**	**199**	**322**

3.2 Sampling area at the site level

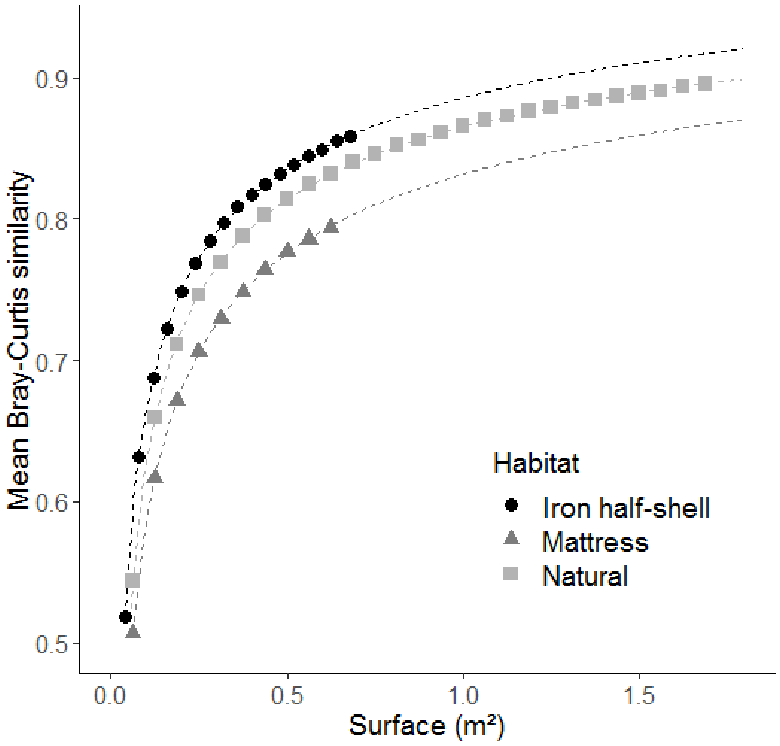

Figure 5: Evolution of the mean Bray-Curtis similarity between two equal subsamples (see Methods) in function of the sampling area (m²) for the three different habitats.

For the three investigated habitats, relationships between the taxonomic similarity between samples and the sampling effort (number of image scored) result in similar typical accumulation curves (Figure 5). The asymptote was reached slightly faster for half-shell than for mattress and natural bottom. According to our criteria (scoring an additional image represents a benefit as long as the similarity index is improved by more than 1%), the required sampling areas are 0.36 m² (corresponding to 9.05 pictures) for the half-shell, 0.55 m² (8.85 pictures) for the mattress and 0.52 m² (8.35 pictures) for the natural bottom (Table 2).

Table 2: Number of pictures and corresponding sampling area required to reach the asymptotic point of the similarity-area curve for each habitat.

	Number of pictures	**Area (m²)**
Natural	9.35	0.52
Mattress	8.85	0.55
Iron half-shell	9.05	0.36

3.3 Fitting taxonomic resolution

The analysis of pictures taken at site B in September 2015 using the lowest possible taxonomic level (LTL) underlines 44 distinct biological categories across communities of natural bottom, mattress and iron half-shell, mainly dominated by red algae (encrusting and foliose) and ascidians (solitary and colonial). nMDS analysis shows a clear taxonomic difference between the community settled on natural bottom and those developing on artificial (mattress and half-shell) habitats (Figure 6A). When using the CATAMI classification, the number of biological categories drops from 44 to 27 (a decrease of 39%). Despite this coarser taxonomic resolution, the corresponding nMDS (Figure 6B) shows a very similar pattern to the one obtained with the LTL classification. However, the visual comparison needs to be treated carefully considering the moderate stress values of the different nMDS representations. Spearman's correlation coefficient between the two patterns of taxonomic similarity is high ($\rho=0.986$) and the permutation routine confirms this correlation as significant ($p<0.05$).

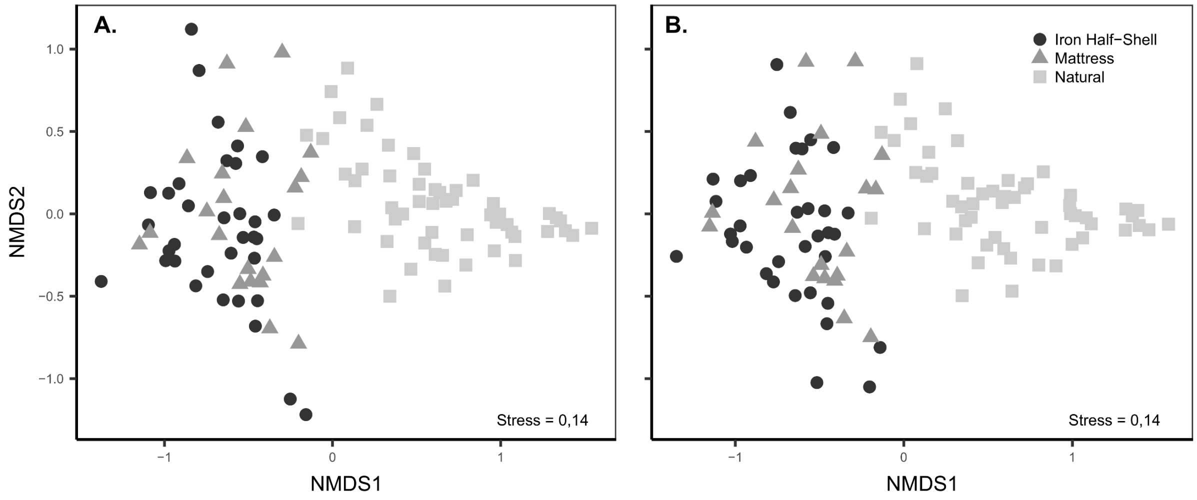

Figure 6: nMDS (non-metric MultiDimensional Scaling) of Bray-Curtis similarities of benthic community composition from underwater images of site B in September 2015. Benthic organisms were described (A) at the lowest possible taxonomic level or, (B) using the coarser CATAMI classification. Each point represents a single picture.

3.4 Comparison of image-processing protocols

We examined the methodologies of a total of 44 papers (published from 2004 to 2018) using point scores on seafloor imagery to characterise benthic communities (SI 1). The protocols are heterogeneous both in terms of points density (from 0.001 to 1 pt cm^{-2}) and sampling area (from 0.05 to 90m²). The random projection is used more frequently (57%) than the regular (23%) or stratified-random (19%) projections. Overall, the density of points decreases as quadrat size and/or sampling area increased (Figure 7A and 7B). A trend emerging from our review suggested that the smaller the mean size of the targeted taxa, the higher the

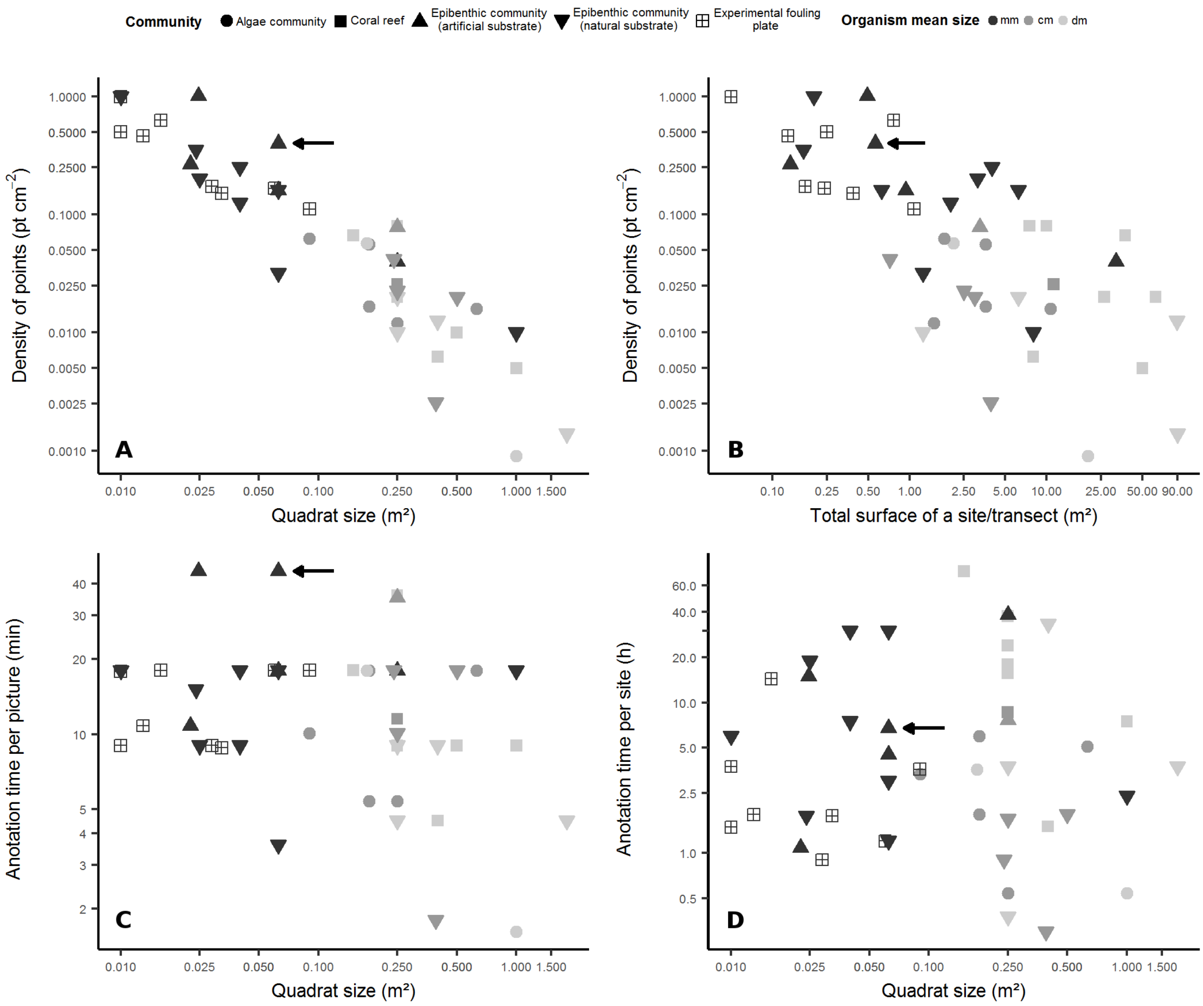

Figure 7: Synthesis of image-processing protocols from our review of published studies: Density of points scored as a function of (A) quadrat size and (B) of site/transect total surface area (*i.e.* number of image * quadrat size); and estimated time allocated to analyse (C) a single image or (D) a full set of images (*i.e.* all the picture of a site/transect) as a function of quadrat size. Each point represents a study, where shapes symbolise different community types and colours reflect the mean estimated size of target organisms. The black arrow highlights the position of our study.

density of points and the smaller the sampling area per site. The method we have adopted in the present study (*i.e.* 0.4 points cm^{-2} and 0.36 to 0.52m^2 sampled per site) is consistent with litterature (Figure 7B). Nevertheless, when considering the quadrat size of 0.0625m^2, our point density lies at the upper range of reported point densities (Figure 7A). The estimated time needed to analyse one picture and a full set of pictures of a site/transect does not show any clear trends depending on the quadrat size (Figure 7C and 7D). The method we adopted shows the highest time to analyse one picture (45 minutes) regardless of the quadrat size, but the time needed to analyse a full set of pictures of a site/transect is more consistent with those of other studies (Figure 7C and 7D).

4. Discussion

Studies of benthic communities based on underwater imagery are faced with a similar challenge: the need to strike a compromise between time-efficient imagery processing and extraction of accurate estimates of benthic community composition so as to robustly detect ecological changes (Van Rein *et al.*, 2009; Molloy *et al.*, 2013). Our stepwise optimisation protocol provides an effective means to rationalise image processing trade-offs in terms of i) time allocated to images annotation, ii) accuracy reached in percentage cover estimates and iii) taxonomic resolution. This method can easily be transposed to survey other reefs or man-made structures (wind turbine foundations, pipelines *etc.*) in coastal areas by accounting for study specifics. Indeed, our study offers guidelines to account for study-specific features related to targeted communities (displaying different properties such as mean individual size or spatial distribution pattern), spatiotemporal scales (from local-scale to broad-scale) and objectives (repeated monitoring surveys, biodiversity or anthropogenic pressure assessment, non-indigenous species survey *etc.*) when designing and implementing protocols for underwater image sampling and analysis.

4.1 Accounting for study-specific benthic community properties

Although our bibliographic review reveals that the subsampling strategies chosen to process images are heterogeneous and rarely justified, it appears that all these protocols follow some tacit rules that rationalise quadrat/image size as well as the density of points used to annotate images. Interestingly, the review reveals a negative relationship between quadrat size and point density such that the larger the surface of the quadrat, the lower the density of points. This consistent trend across published studies suggests that, both quadrat size and point density are empirically adjusted so as to match the morphological properties of the targeted communities, especially mean organisms' size. Accordingly, when large taxa (*i.e.* > 10 cm) are targeted, large quadrats are used and high densities of points are not needed to effectively assess their relative abundances. For example, Dumas *et al.* (2009) used quadrat of 1 m² and a density of points of 9.10^{-4} pt cm^{-2} to describe megafauna associated with coral reef habitat. Conversely, in order to study macrofauna community of intertidal shore (dominated by barnacles), Dias *et al.* (2018) used quadrat size of 100 cm² and a density of points of 1 pt cm^{-2}.

Besides organism size, another important ecological feature to account for, is the spatial distribution of taxa. Indeed, taxa are rarely uniformly distributed in nature (*i.e.* homogeneous distribution) and rather exhibit different degrees of clustering (*i.e.* heterogeneous distribution of the individuals). This can impact the effectiveness of spatially-structured sampling methods (Cochran, 1946; Dutilleul, 1993; Legendre *et al.*, 2002; McGarvey *et al.*, 2016) such as the mode of point projection. The literature shows that stratified-random sampling design performs better than random sampling design to estimate relative abundance of taxa (*i.e.* higher accuracy in cover estimates in our case) for communities exhibiting clustered taxa (Cochran, 1946; McGarvey *et al.*, 2016). When a community tends towards a homogeneous spatial distribution pattern (*i.e.* no clustering of biological organisms), the different methods tend to perform equally. Consequently, whatever the spatial pattern of the community, stratified-random designs are always at least as accurate as the random sampling designs (Cochran, 1946), which explains why the sampling effort required with random projection is always higher than with the stratified-random projection to reach a similar precision. Nevertheless, although random designs gives wider confidence interval of the percentage cover, these are unbiased, in the sense that they will not be impacted by spatial pattern of the taxa (McGarvey *et al.*, 2016). Thus, the absence of regularity in spatial distribution patterns of organisms has incited some authors to generalize the use of the random design at the expense of stratified-random (Dethier *et al.*, 1993; McGarvey *et al.*, 2016), which may explain why random projection remains the most widely used projection method in the literature.

In our study, we identified that spatial clustering of the surveyed taxa influenced the accuracy of our estimates at two different spatial scales, namely within each image and across images at the site level. At the image scale, the stratified-random projection significantly reduces image processing time as the number of points required to accurately estimate percentage cover is up to 38% smaller than with the random projection. Nevertheless, the optimal point density showed between-habitat differences that are more pronounced with the stratified-random projection than with the random projection. Since we determined the optimal number of points in a consistent way across habitats (to reach a satisfactory accuracy for rare categories associated with a 5% cover), the fact that a given accuracy is reached with a higher point density on mattresses with respect to natural or half-shell habitats can only be explained by a difference in spatial patterns of the rare categories. Indeed, our exhaustive picture analyses (dedicated to image sampling strategy) showed that rare benthic categories (with a cover inferior than 10%) occurred in more numerous and smaller patches on mattress habitat (17.9 ± 7 patches of 0.58 ± 0.3 cm^2, results not showed) than on the two other habitats (respectively 9 ± 2 patches of 1.6 ± 1 cm^2 for natural habitat and 9.4 ± 7 patches of 1.4 ± 0.7 cm^2 for the iron half-shell; results not showed). This suggests a more homogeneous spatial repartition of categories (*i.e.* a lower level of clustering) on the mattress habitat. Consistently with the statements exposed above (Cochran, 1946; McGarvey *et al.*, 2016), accurate estimating of percentage cover of rare taxa on mattress habitat requires the highest number of points with stratified-random projection.

At the site scale, we found that the sampling areas required to accurately describe benthic communities are habitat-specific, which reflects the influence of heterogeneity of the community's structure across images. Specifically, a larger sampling area is required to reach accurate estimating of community composition on mattress and natural habitats relative to half-shell habitat. Since our optimisation approach is based on taxonomic similarity between images within a site, a larger optimum sampling area means that the photographs are more different from each other, or in other words, that the spatial distribution of communities is more heterogeneous (*i.e.* more clustered repartition at the scale of sites). Such observations are in agreement with recent simulations that showed that a larger sampling area was required to achieve an equivalent level of precision for clustered distributions relative to homogeneously-distributed communities (Perkins *et al.*, 2016). To summarise, accurate estimating of community composition and percentage cover of rare taxa requires a higher point density and a larger sampling area on mattresses relative to natural and half-shelf habitats. These are the consequences of a more homogenous spatial distribution of rare taxon within each image (which is consistent with the homogeneous nature and flatness of each single concrete unit), while the community appears more variable across images at the site scale (which is consistent with the fact the exposition of concrete units to the current is variable).

4.2 Distribution of sampling efforts across nested spatial scales

Image-based monitoring of seafloor communities often follows a hierarchical survey design that encompasses nested spatial scales, for instance, from regional survey zones, to local sites within region, transects within site (across heterogeneous conditions) and all the way down to individual images (Perkins *et al.*, 2018). According to the scope and the scale of the study, different strategies can be adopted to prioritise scoring effort across theses nested scales.

For instance, broad-scale studies tend to favour number of images scored per site rather than point density per image (Brown *et al.*, 2004; Molloy *et al.*, 2013; Perkins *et al.*, 2016). This intuitive choice is usually driven by the need to detect significant changes across large spatial scales (*i.e.* across multiples sites within large geographical domains, Brown et al., 2011; Perkins et al., 2018), for instance across a large gradient of stressors, either local human impacts such as the effects of long-term fishing activities or climate change. In the same way, successful detection of site-specific changes through time requires high accuracy at the site level (increased number of pictures) rather than at the image level (Larsen *et al.*, 2001; Elston *et al.*, 2011). For example, studies by James *et al.* (2017) and Marzloff *et al.* (2018) focused on a broad spatial scale spanning several bioregions in eastern Australia (> 2,000 m of coastline) in order to study taxonomic changes on a benthic reef across a subtropical to cold temperate environmental gradient. Consequently, they made a significant effort at the site level (minimum of 30 photos of 2 m^2 analysed per site) to characterise broad community changes across sites, whereas point score information recorded at the image scale was finally downgraded to presence or absence of targeted taxa in their analyses.

On the contrary, local-scale studies often require detection of quantitative differences in taxonomic composition across fine-scale heterogeneous conditions rather than across remote sites. For example, studies on artificial habitats (such as ours) generally examine differences in community composition between natural and artificial habitats in a given area, or between different artificial habitats. To detect such subtle changes in the occurrence of particular species, scoring effort is usually put at the image level by increasing point density. For example, Gestoso *et al.* (2018), who focused on non-indigenous species within fouling communities (macrofauna) worked with experimental plates submitted to different treatments. They compared a few number of pictures, only 5 quadrats of 0.01 m^2 for each conditions, but placed the emphasis on the image analysis, with a point density of 1 point cm^{-2}.

In the present study, both the scope (comparing communities of different habitats) and the spatial scale (within an area < 10 km) of our survey led us to first design the description of images to accurately estimate the cover of taxa at the smallest scale , and secondly to assess the sampling effort at site scale to encompass local benthic diversity.

4.3 Relevant taxonomic sufficiency

Identification of benthic taxa from underwater imagery is difficult and often cannot be performed at a high level of taxonomic resolution, even by specialists. Consequently, using a suitable taxonomic classification is critical to annotate benthic taxa from underwater imagery. It is necessary to adapt the classification scheme in agreement with the objective of the study. In our case, we showed consistent differences in community composition between the artificial (half-shell and mattress) and natural habitats regardless of the taxonomic resolution used. Thus, while the CATAMI classification is coarser than the LTL, with 39% less taxa (27 and 44 taxa, respectively), it provides sufficient taxonomic resolution to detect community-level changes. For instance, a clear difference in taxonomic composition was highlighted between artificial and natural habitats epibenthic communities (with both classifications), and a decrease of taxonomic resolution does not significantly impact the output of our community analysis. Similarly, James et *al*. (2017) showed that CATAMI performed as well as well-resolved classifications when assessing large-scale changes in benthic community structure. Nevertheless, these authors only demonstrated the robustness of CATAMI to characterise broad-scale changes in community structure. Thus, in the present study, we comfort these conclusions by showing that the CATAMI image annotation scheme is also effective in characterising local-scale changes in community composition across different hard habitats.

Our results corroborate with several studies on taxonomic sufficiency that showed that identification at high taxonomic level (*i.e.* family level) allows reliable spatiotemporal analysis of benthic communities (Warwick, 1988; Urkiaga-Alberdi *et al.*, 1999; De Biasi *et al.*, 2003; Doerries and Van Dover, 2003). Warwick (1993) explains these results by the fact that the family level often brings together organisms showing similar major functional traits, which are supposed to react similarly to environmental fluctuations. Here, we are working with a resolution even coarser than family taxonomic rank, but a similar hypothesis can be applied to the different morphotype groups we used in the CATAMI classification. In our case, it should be noted that the differences in taxonomic resolution between the two classifications are sometimes marginal. Indeed, for 45% of the taxa, the lowest possible taxonomic level identified from imagery corresponds actually to the morphotype level used with the CATAMI typology. In this sense, CATAMI classification is well adapted for image-based descriptions of benthic communities.

In addition to providing consistent results relative to a study-specific taxonomic classification, the standardised classification CATAMI can make image analysis not only faster, but also more reliable. Indeed, identification at a lower taxonomic resolution decreases misidentification risks and allows non-specialists to analyse images. These advantages make CATAMI a well-suited classification scheme in our case, and we recommend its broader application for underwater imagery annotation in order to facilitate comparisons of ecological patterns across studies.

5. Conclusions

While our optimal image-processing protocol remains specific to our case study, we believe that our stepwise strategy provides guide lines to rationally tackle the challenges inherent to image annotation. Both our specific application and the literature review of point-score analyses of underwater imagery provide general keys to consider in future studies so as to define an effective image analysis protocol. Specifically, we described the different levels (*i.e.* point score density, surveyed area at different spatial scales, taxonomic resolution) at which a study can balance out results accuracy versus time of analysis. Hereafter, we summarise the major study-specific characteristics and constraints to account for when optimising an underwater imagery processing method (Figure 8).

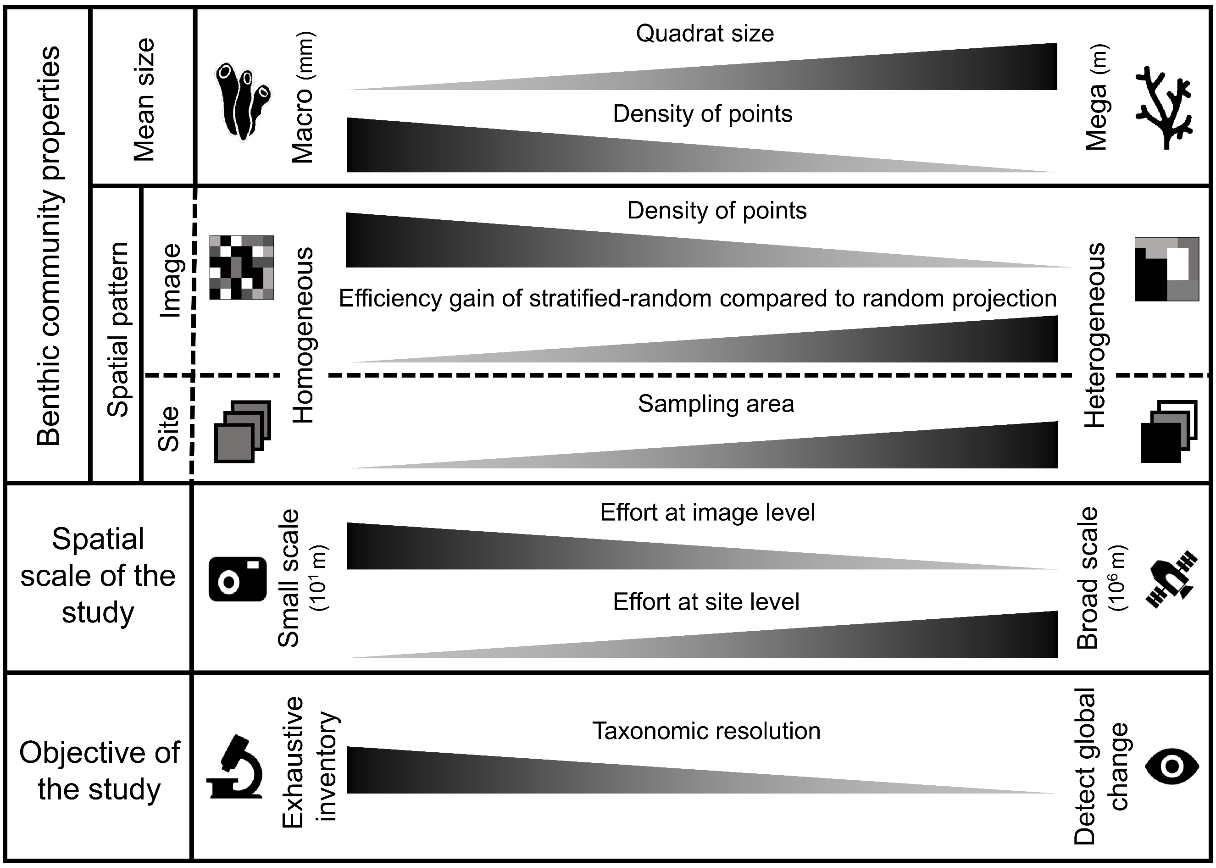

Figure 8: Summary of the different parameters to be taken into account when designing an underwater image-based sampling and analysis strategy for benthic monitoring. The figure represents how underwater surveys and image processing strategies need to address study-specific features related to targeted communities (displaying different properties such as mean individual size or spatial distribution pattern), spatiotemporal scales (from local-scale to broad-scale) and objectives (repeated monitoring surveys, biodiversity or anthropogenic pressure assessment, non-indigenous species survey *etc*.).

A first consideration in designing the image scoring protocol depends on the ecological attributes of the targeted community, in particular the mean size of targeted taxa. For instance, communities dominated by megafauna/flora taxa (*e.g.* coral reefs, kelp forests *etc*) can be studied using larger quadrats and a reduced density of points per picture, while communities with smaller taxa are most effectively sampled using a protocol similar to ours. While the mean size of targeted organisms is usually well apprehended when designing image sampling and analysis strategies, less attention is paid to their spatial distribution patterns. Spatial distribution patterns are rarely quantified prior to sampling and are difficult to apprehend as they depend on a wide range of biotic and abiotic factors, which explains why it is rarely take into account. Our results characterise how distribution patterns can affect the effectiveness of the sampling strategy. We demonstrate that sampling designs can benefit from accounting for any prior knowledge available from previous surveys or ecological knowledge about spatial patterns in the targeted community. When no information is available about spatial patterns, we overall recommend the use of a stratified-random projection as a more time-efficient and reliable method than random-projection. An effective description of image content requires to test and validate the density of point with respect to the desired accuracy of occurrence estimation. In our case, the high point density is explained by the fact that the investigated macroepibenthic sessile communities are probably one of the most difficult models for image-based study due to a high number of rare taxa with a low patchiness (*i.e.* homogeneous repartition) and the dominance of small and encrusting individuals.

The aim of our study, which focused on fine-scale changes in epibenthic communities where mean organism size is small (~10mm), led us to define our optimum method following a stepwise approach. This method can serve as a general guideline for other image-based benthic studies even though other approaches can be considered for broader-scale studies. For example, Perkins *et al.* (2016) simultaneously optimised the number of pictures per site and the density of points along transects, albeit *in silico* using an artificial data set. Because their approach requires a comprehensive knowledge of the study ecosystem across large spatial scales, such optimisation procedure cannot easily be applied in real-world case studies. Thus, we suggest that prioritising between a higher density of points and a larger sampling area depends on the spatial scope of the ecological study. For local-scale studies, efforts should first be put at the smallest scale of observation by increasing the effort on image analysis, while for broad scale-studies the largest scale of observation should be optimised by increasing the total area.

Besides all these case-specific considerations for instance concerning the spatial scale of the study, the level of accuracy required to tackle an ecological question will impact the design of the imagery protocol. For instance, in our study we arbitrarily optimised image scoring so as to reach high accuracy for benthic categories with a percentage cover ≥5%, which may not be relevant to all studies. When the study objective is only to detect substantial variations in benthic community composition, this accuracy criterion can for instance be set for benthic categories with higher percentage covers (*e.g.* 10%, 20%), which would significantly decrease the required point density. Thus, it is essential to explicitly define *a priori (i.e. before designing and implementing the image scoring protocol)* the degree of accuracy required to tackle the ecological question(s) at stake... When such an explicit accuracy target has not been set beforehand, it is critical to assess the quality and robustness of the biological information extracted from underwater imagery to avoid any false ecological interpretations and/or flawed analyses.

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Supplementary information 1: List of the 44 papers (published from 2004 to 2018) using point scores on seafloor imagery to characterise benthic communities from the literature review.

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Chapter 3

Succession of epibenthic communities on artificial reefs associated with marine renewable energy facilities within a tide-swept environment

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Abstract

Although colonisation of artificial structures by epibenthic communities has been documented in numerous case studies, our understanding of those deployed in high energy hydrodynamic environments is limited. In this context, this study aims to characterise the epibenthic colonisation of different structures associated with a tidal test site located in a high-energy hydrodynamic environment. Using 4 years of underwater image-based surveys, we characterised changes though space and time in the taxonomic composition of epibenthic assemblages colonising natural habitat and two kinds of artificial structures. Our results highlighted that the two artificial habitats presented an overall similar trend in the ecological successions but their communities tended to differ at the late stages of the succession. The deployment of these artificial structures resulted in the addition of stable substrata in an environment where natural hard substrates are highly mobile and strongly exposed to sediment abrasion. Although, epibenthic communities colonizing artificial habitats are unlikely to have reached their climax at the end of our survey, these supported structurally-complex taxa facilitating an overall increase in local diversity. We were able to quantify how epibenthic communities can significantly vary in high-energy coastal environment, and our final survey suggests that the ecological succession was still in progress 5 years after deployment of artificial reefs. This therefore highlights the need to maintain a long-term continuous survey of coastal artificial reef habitats to understand in greater detail ecological successions and temporal variability.

Keywords

Benthic community ; marine renewable energy; artificial reef; succession; non-indigenous species; underwater imagery

1. Introduction

When submerged in seawater, hard substrates are colonised by epibenthic organisms which form the so-called “biofouling”. These organisms are diverse and often dominated by marine invertebrates (*e.g.* Arthropoda, Tunicata, Bryozoa, Annelida, Porifera *etc.*), and macroalgae (*e.g.* Chlorophyta, Rhodophyta and Ochrophyta). Colonisation of bare substrates can be described in time as an ecological succession, *i.e.* a sequence of stepwise changes in assemblage composition until community composition eventually reaches a persistent state, called climax (Clements, 1916; Connell and Slatyer, 1977). Early pluricellular eukaryote colonisers are usually pioneering species, which are gradually replaced by longer-lived morphologically-complex species as the colonising community transitions towards its climax (Clements, 1916). Numerous biotic and abiotic factors condition the composition, the succession of epibenthic assemblages and the needed time to reach climax (Falace and Bressan, 2000). Biotic factors include competition and trophic interactions between organisms (Connell and Slatyer, 1977) while non-biotic factors such as substratum properties (*e.g.* surface type, material, texture, slope *etc.* ; Falace and Bressan, 2000) and environmental conditions (*e.g.* temperature, light, pH, salinity, currents *etc.* ; Bowden et al., 2006; Falace and Bressan, 2000; Pérès and Picard, 1964) depend on implantation site characteristics.

For several centuries, humans have deployed artificial structures on the bottom of aquatic ecosystems for different applications (Lima *et al.*, 2019). These structures constitute artificial reefs as they mimic certain characteristics of natural reefs via provision of hard-substrate and shelters to living organisms (Thierry, 1988; Bohnsack *et al.*, 1991; Jensen *et al.*, 2000a). Among these artificial reefs, two main types can be differentiated: *i*) those intentionally designed and installed for their ecological effects (*e.g.* ecosystems conservation/restoration, fish stocks enhancement/management *etc.*; Jensen et al., 2000) and *ii*) those deployed for another primary purpose, such as oil rigs, breakwaters, or Marine Renewable Energy (MRE) facilities (*e.g.*

windfarms, tidal turbines and wave energy converters) (Wilson and Elliott, 2009; Langhamer, 2012; Lima *et al.*, 2019). Worldwide, the number of MRE structures rapidly increases to meet the increasing demand for renewable energy to mitigate global anthropogenic climate change (Copping *et al.*, 2014; Lindeboom *et al.*, 2015; Coolen *et al.*, 2018). In addition to providing carbon neutral energy, MRE structure colonisation by benthic organisms is considered as an extra positive environmental benefit (Langhamer, 2012; Copping *et al.*, 2016). For instance, when installed on soft bottoms MRE facilities directly increase substrate structural complexity and hence facilitate colonisation by organisms previously absent, which leads to an overall increase in local diversity (De Mesel *et al.*, 2015). Moreover, epibenthic communities developing on MRE facilities can provide important food sources for commercial fish and crustacean species that also colonise artificial reefs (Pickering and Whitmarsh, 1997; Reubens *et al.*, 2011; Krone *et al.*, 2013a). Furthermore, certain epibenthic organisms can create complex tri-dimensional biogenic structures (*e.g.* kelps, gorgonians *etc.*) and further increase habitat heterogeneity (Pickering and Whitmarsh, 1997). Conversely, artificial structures may also facilitate the introduction/expansion of non-indigenous sessile species by creating new connectivity routes via a stepping-stone process (Mineur *et al.*, 2012; Adams *et al.*, 2014). For example, in the Adriatic sea, artificial structures along sedimentary coastlines were shown to harbour 3 times more non-indigenous ascidian than natural rocky reefs or artificial structures built close to rocky coastlines affecting their spread at regional scales (Airoldi *et al.*, 2015). Also, the massive spread of the non-indigenous green algal species *Codium fragile ssp. tomentosoides* in the Mediterranean sea is suspected to have been favoured by the high number of coastal breakwaters (Bulleri and Airoldi, 2005).

Although colonisation of MRE structures by epibenthic communities has been documented in several case studies, our knowledge is limited concerning artificial structures deployed in high energy hydrodynamic environments (*i.e.* with current velocities > 1.5 m s^{-1})

that are specifically targeted for tidal energy extraction (Copping *et al.*, 2016; Quillien *et al.*, 2018). In these tide-swept environments, physical processes can considerably influence epibenthic community structure, as organisms are exposed to high physical stress, which can dominate over interspecific biological interactions (Sousa, 1979a; Dean and Connell, 1987). Although assemblages associated with these tide-swept environments have been described as highly diverse (Connor *et al.*, 2004; Kregting *et al.*, 2016), few studies have monitored their long term composition to characterise their variability. This is mainly due to logistical constraints associated with field sampling in these environments: sampling essentially relies on scuba diving as remote grabs are ineffective on hard substrates (Sheehan *et al.*, 2010), and diving operations are restricted to narrow time windows as they are only achievable during slack tides.

In this context, this study aims to characterise more fully epibenthic colonisation of different MRE structures located in a high-energy hydrodynamic environment. We specifically focussed on ecological succession of these epibenthic communities using several years of underwater image-based surveys undertaken by divers. We also carefully scrutinised the temporal dynamics of two different non-indigenous species and one regionally-emblematic foundation species.

2. Methods

2.1 Study area

The study area encompasses a 15 km-long submarine power cable (8 MVA - 10 kVDC) set up in 2012 to connect the tidal test site of Paimpol-Bréhat (Brittany, France; Figure 1) managed by Electricité De France (EDF). The implantation site is characterised by major tidal currents (up to 3 $m.s^{-1}$ during Spring tides) and the bottom is dominated by hard substratum (pebbles and rocks) and highly mobile shell debris and coarse sand. Because of these characteristics, 11 km of cable are unburied and fully protected with nested cast iron half-shells

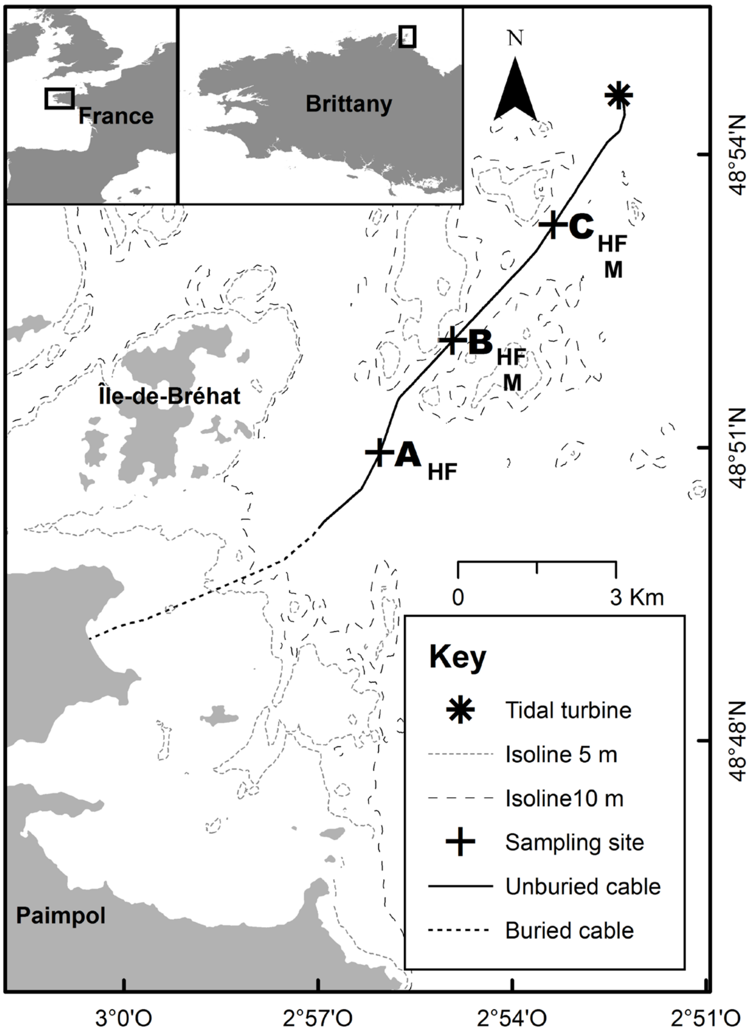

Figure 3: Map of the study area off the northern coast of Brittany in western France (top-left and top-centre panels), which shows the location of the three stations (A, B and C) on the unburied section of the subsea power cable. At each site, acronyms specify the types of studied artificial habitats (*i.e.* HF for Half-Shell and M for mattress).

(50 cm long, 15 cm diameter). The cable is also stabilised by 120 concrete mattresses (6 m long, 3 m wide, Figure 2A) installed in 2013, preventing any displacement caused by high hydrodynamic conditions.

Three sites (A, B and C) located along the cable route (Figure 1) were annually surveyed. Sites B and C included both concrete mattresses and half-shells, while site A only had half-shells. Due to several setbacks in the commissioning progress of the project, no electric current has transited through the cable so far and associated protection structures have actually acted as simple artificial reefs.

2.2 Sites characterisation

The three sites span similar depth ranges (between 18 and 20 m). To characterise the hydrodynamical characteristics of each sites more thoroughly, three environmental variables were extracted based on GPS positions: mean and maximum residual current velocity above the seafloor (in $m.s^{-1}$) were derived from a 2010-2015 climatology from simulations using the MARS3D hydrodynamic model (Lazure and Dumas, 2008) and seafloor topography was used to characterise exposure to residual current (ranging from 0°, when the seafloor is sheltered, to 180°, when it is fully exposed to dominant current).

2.3 Image acquisitions

Using underwater imagery performed by scuba divers, benthic communities were monitored at each site over six campaigns carried out: in September 2014, March and September 2015, September 2016, September 2017 and March 2018. All sites were surveyed during each campaign, except for site A in September 2017 due to bad weather conditions. Hereafter, campaigns occurring in September and March are referred to as “summer” and “winter”, respectively. At each site and at each date, high-definition photographs of benthic communities were systematically taken on the two artificial habitats that protect the cable and on the neighbouring natural bottom, as follows:

iv) each side of each 50 cm long iron half-shell on a marked 10 m transect;

v) same 16 regularly spaced concrete blocks (8 of 47x38 cm and 8 of 47x20 cm) of the mattress;

vi) a minimum of 20 25x25 cm quadrats randomly placed on the neighbouring natural habitat 10 m apart from the cable route in order to avoid any potential influence of artificial structures.

The 4 year survey produced a total of 1,482 images (Table 1). Photographs were taken at a resolution of 37 million pixels per image with a Nikon D810 inside a Ikelite underwater housing, with a 20 mm lens and 2 Keldan LED lights (105W, 9000 lumens). All images of half shells were calibrated with a scale bar.

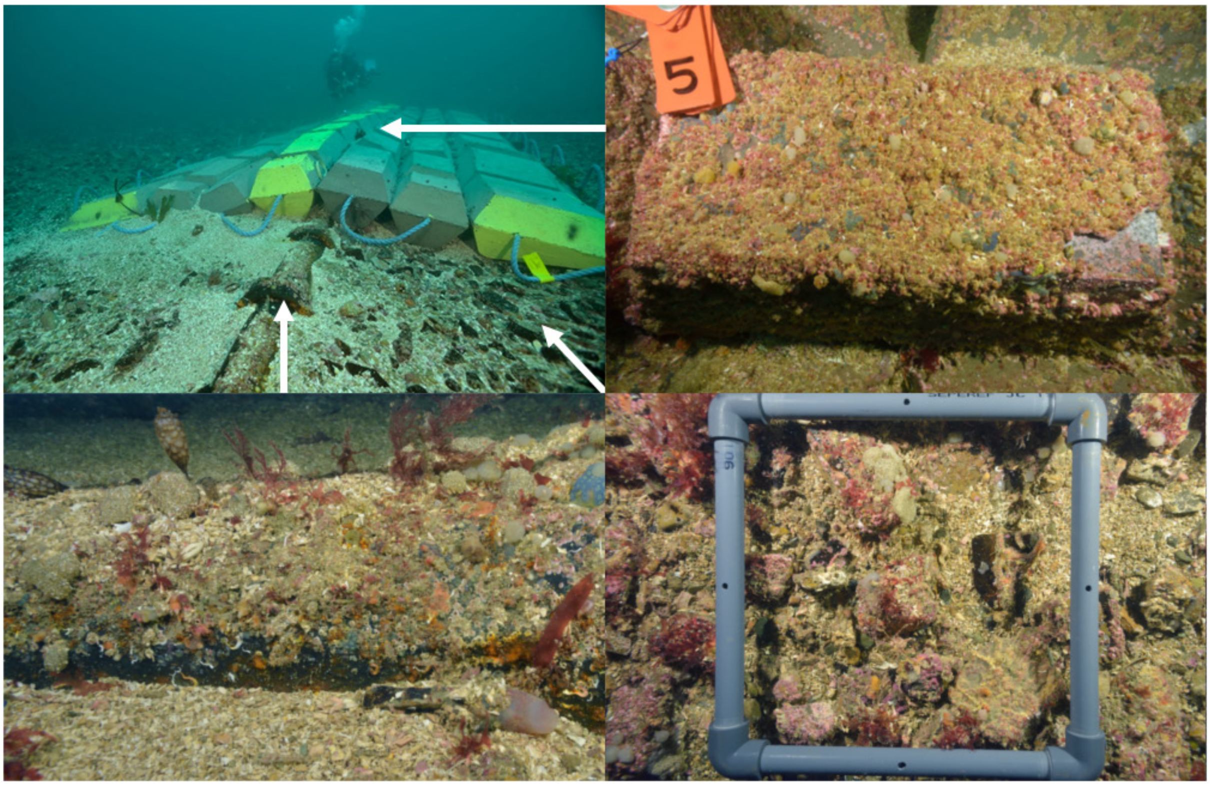

Figure 2: Overall view of one of the survey sites including cast-iron half-shells, a concrete mattress (freshly installed) and natural habitat (top-left); Close-up views of one of the mattresses concrete units (top-right), one cast-iron half- shell (bottom-left), and one of the quadrats placed on the natural habitat (bottom-right) (courtesy: Olivier Dugornay).

2.4 Image analyses

***Benthic community*:** Images were described following a scoring methodology adapted from the point count method (Pielou, 1974) and tested with a subset of the whole available data base (see Chapter 2). Briefly, for each combination of habitat, site and campaign, 10 images were chosen randomly among the available set of images. To score images, an area of 625 cm² was cropped on ‘natural’ and ‘mattress’ habitat images. For half-shell habitat, an area of 500 to 625 cm² was cropped within each picture as a 625 cm² surface was not always reachable. Within these cropped areas, 0.4 points.cm^{-2} (*i.e.* 250 points for an area of 625 cm²) were projected with the random-stratified projection method. Then, each projected point was manually assigned to a benthic category (biological or substratum type). Biological categories were labelled using the highest resolution of the CATAMI (Collaborative and Automated Tools for Analysis of Marine Imagery) classification (Althaus *et al.*, 2015). This classification combines coarse taxonomy levels and organism morphology to identify benthic taxa from underwater imagery (Althaus *et al.*, 2015). The percentage cover of each category was then calculated as the ratio between the number of points attributed to this category and the total number of points. These image analyses were performed using the free software PhotoQuad (Trygonis and Sini, 2012).

Table 1: Summary of the total number of pictures sampled at the different sites and campaigns and details concerning the number of pictures analysed to study community-level changes and dynamics of the three target species. Note that Site A was not surveyed during the 2017 summer campaign due to adverse weather conditions, and that pictures from the 2016 summer campaign could not be analysed for the community-level study due to poor image quality.

Campaign	**Number of pictures sampled**			**Number of pictures analysed**	
	Site			**Community study**	**Species of particular interest**
	A	**B**	**C**		
Summer 2014	58	98	85	80	241
Winter 2015	71	114	110	80	295
Summer 2015	68	109	96	70	273
Summer 2016	61	97	97	-	255
Summer 2017	-	89	94	50	183
Winter 2018	58	90	87	70	235
Total	316	597	569	350	1482

Some sets of images were excluded from this analysis for two reasons. The poor image quality of the September 2016 campaign could have biased the analysis and were thus excluded. Also, on half-shells of site C, images taken from September 2015 onwards showed very small analysable surfaces due to a smothering of coarse sand, which prevented any proper image analysis for this habitat. Overall, a total number of 350 images were analysed (Table 1).

Target species: Image scoring was adjusted for three target species so as to specifically study their temporal dynamics. These three species were chosen both for ecological and practical reasons: (*i*) the slipper limpet *Crepidula fornicata* and the stalked sea squirt *Styela clava* are two non-indigenous species, and the kelps *Laminaria sp*. are an important keystone species in the region; and (*ii*) these organisms are easily recognisable from imagery because of their large size and conspicuous aspect. Contrary to the image scoring strategy described above to analyse the whole community, all available images across all campaigns were exhaustively analysed by counting all visible individuals belonging to these three species (Table 1). In order to analyse changes in density estimates over time for each target species (ind.m^{-2}), estimated abundances were standardised by each image surface area. In total, 1,482 images were analysed for this task.

2.5 Data analyses

A permutational multivariate analysis of variance (PERMANOVA, Anderson, 2001) was used to determine if epibenthic community composition significantly differed across 3 factors: *i*) sites (fixed, 3 modalities), *ii*) campaigns (fixed, 5 modalities) and *iii*) habitats (fixed, 3 modalities). Pairwise tests were used when relevant, to further explore significant community changes in space and time. Prior to the PERMANOVA, homoscedasticity was tested across all combinations of factors using PERMDISP (Anderson, 2006; SI 1). As a complementary exploratory approach, changes in epibenthic communities were also visualized using non-metric multidimensional scaling (nMDS; Clarke and Ainsworth, 1993). All multivariate analyses were based on Bray-Curtis similarity matrices, computed without any prior transformation because no dominant taxa were present. Differences in target species densities were characterised across habitats and campaigns using non-parametric Kruskal-Wallis tests. When significant, pairwise comparison tests using Bonferoni correction were applied. Data analyses were performed using the *vegan* package (Oksanen *et al.*, 2018) within the R environment using Rstudio interface (RStudio Team, 2015). Graphics were produced using the *ggplot2* package (Wickham, 2016).

3. Results

3.1 Site characterisation

Sites A, B and C present similar mean current velocity (0.53, 0.57 and 0.48 $m.s^{-1}$ respectively) but more contrasted maximum current velocity (3.13, 3.53 and 2.83 $m.s^{-1}$ respectively). Exposure to residual current increases with distance from the coast, from an exposure of 87° at site A to 97° and 113° at sites B and C, respectively.

3.2 Epibenthic community dynamics

Across all pictures analysed, a total of 36 taxa from 10 phyla were identified (SI 2). Benthic assemblage composition was significantly different across all factors "Habitat", "Campaign", "Site" as well as across all levels of interactions between these factors, which suggests that benthic communities did vary spatially throughout the survey following habitat-specific dynamics (Table 2).

Table 2: Results of PERMANOVA based on Bray Curtis similarities in epibenthic community composition, which we characterised using taxa relative cover percentage. The PERMANOVA tested for the effects of habitat (Half-Shell, Mattress and Natural), site (A, B and C), campaign (summer 2014, winter 2015, summer 2015, summer 2017 and winter 2018) and all levels of interactions. Significant values at P (perm) ≤ 0.05 are shown in bold.

Factor	df	SS	MS	Pseudo-F	P (perm)
Habitat	2	29.1	14.5	162.3	**0.001***
Site	2	9.2	4.6	51.1	**0.001***
Campaign	4	10.1	2.5	28.2	**0.001***
Habitat:Site	3	4.0	1.3	15.0	**0.001***
Habitat:Campaign	8	7.3	0.9	10.2	**0.001***
Site:Campaign	7	2.2	0.3	3.6	**0.001***
Habitat:Campaign:Site	8	2.0	0.2	2.8	**0.001***
Residuals	301	27.0	0.1		
Total	335	90.9			

Pairwise comparisons within Habitat x Campaign x Site interactions highlighted significant differences across a major part of possible combinations (Table 3). The communities of natural habitat were always significantly different between the 3 sites. Also, regardless of the site considered, communities associated with natural habitats were always significantly different from those associated with the two artificial habitats. Within artificial habitats, epibenthic communities were significantly different between mattress and half-shell habitats, except during the 2014 and 2015 summer campaigns. Importantly, across all sites and habitat types, community composition changed significantly over time. Only natural habitat communities at site B did not significantly change between campaigns.

Table 3: Summary of pairwise PERMANOVA test conducted based on Bray Curtis similarities of epibenthic taxa relative cover percentage (N= 595 combinations). Significant values at $P \leq 0.05$ are shown in bold. S = Summer; W = Winter.

Pairwise Comparisons

Within natural habitat		**Within artificial habitat**		**Artificial vs natural**	
Site A	*P*	**2014S**	*P*		*P*
All combinations	***<0.05***	Half-Shell site B vs Mattress site C	*>0.05*	**All combinations**	***<0.05***
Site B		All other combinations	***<0.05***		
All combinations	*>0.05*	**2015S**			
Site C		Half-Shell site B vs Mattress site C	*>0.05*		
All combinations	***<0.05***	Half-Shell site B vs Mattress site B	*>0.05*		
		Half-Shell site B vs Half-Shell site A	*>0.05*		
Between sites		All other combinations	***<0.05***		
All combinations	***<0.05***	**2015W-2017S-2018W**			
		All combinations	***<0.05***		
		Between campaign			
		All combinations	***<0.05***		

Overall, these results highlight *i*) clear differences in community structure between artificial and natural habitats; *ii*) differences in community structure between the three sites; and *iii*) larger temporal changes in communities colonising artificial habitats relative to those found on natural hard substrates.

The MDS clearly discriminates between natural communities at sites A and C (along the second axis) while communities at site B appear more transitional (Figure 3). Natural habitats were dominated by sheet like red macroalgae at site A as opposed to encrusting algae (mainly brown) and encrusting bryozoans at site C (Figure 4). At site B natural substrate hosted both sheet like red algae and encrusting brown algae at similar coverage (Figure 4). Temporal variations in natural habitat communities, even if statistically significant at sites C and A, were less pronounced than those associated with communities on artificial habitats (Figure 3 and 4).

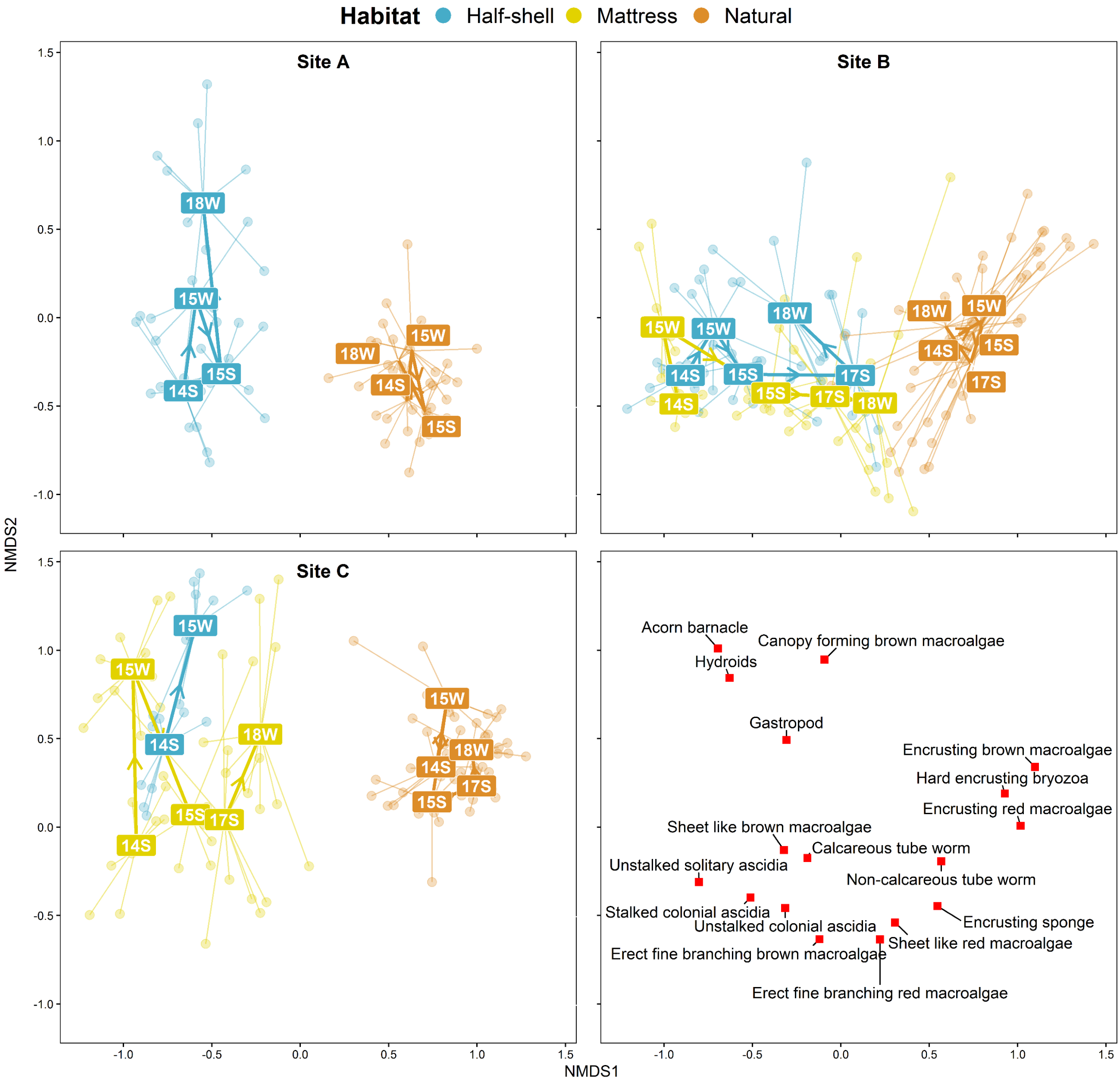

Figure 3: Non-metric Multidimensional Scaling (nMDS) based on Bray-Curtis similarities in community composition between samples. Each point represents an image, and thin lines connect all images scored from the same 'Habitat, Site and Campaign' combination to their centroid; bold lines and arrows show the mean temporal trajectories for each site and habitat. For clarity purposes, the nMDS is shown independently for each survey site (A, B and C). The 50% most frequent epibenthic taxa and the 70% best fitting with the axes were displayed in the bottom right panel. 2D Stress = 0.18; S = Summer; W = Winter.

All sites considered, the first axis of the MDS clearly distinguishes between artificial and natural habitats (Figure 3). Taxonomic similarity between artificial and natural habitats increased with time at site B but remained low at sites A and C (Figure 3, Figure 5).

Half-shells and mattresses hosted fairly similar communities at the beginning of the survey, which are characterised by high proportions of unstalked solitary ascidians and other types of ascidians (Table 3, Figures 3 and 4). Note that if the point count method does not quantify understorey organisms, we observed from the imagery that this matrix of ascidians mainly developed on top of barnacle mats during the summer of 2014. From the summer 2015 campaign onwards, community composition started to diverge between the two artificial habitats (Figure 5). Indeed, the cover of the different ascidians declined in both habitats but were essentially substituted by sheet-like red macroalgae and hydroids on half-shells, whereas mattresses were mostly overgrown by diverse types of macroalgae. Canopy forming algae (*i.e.*

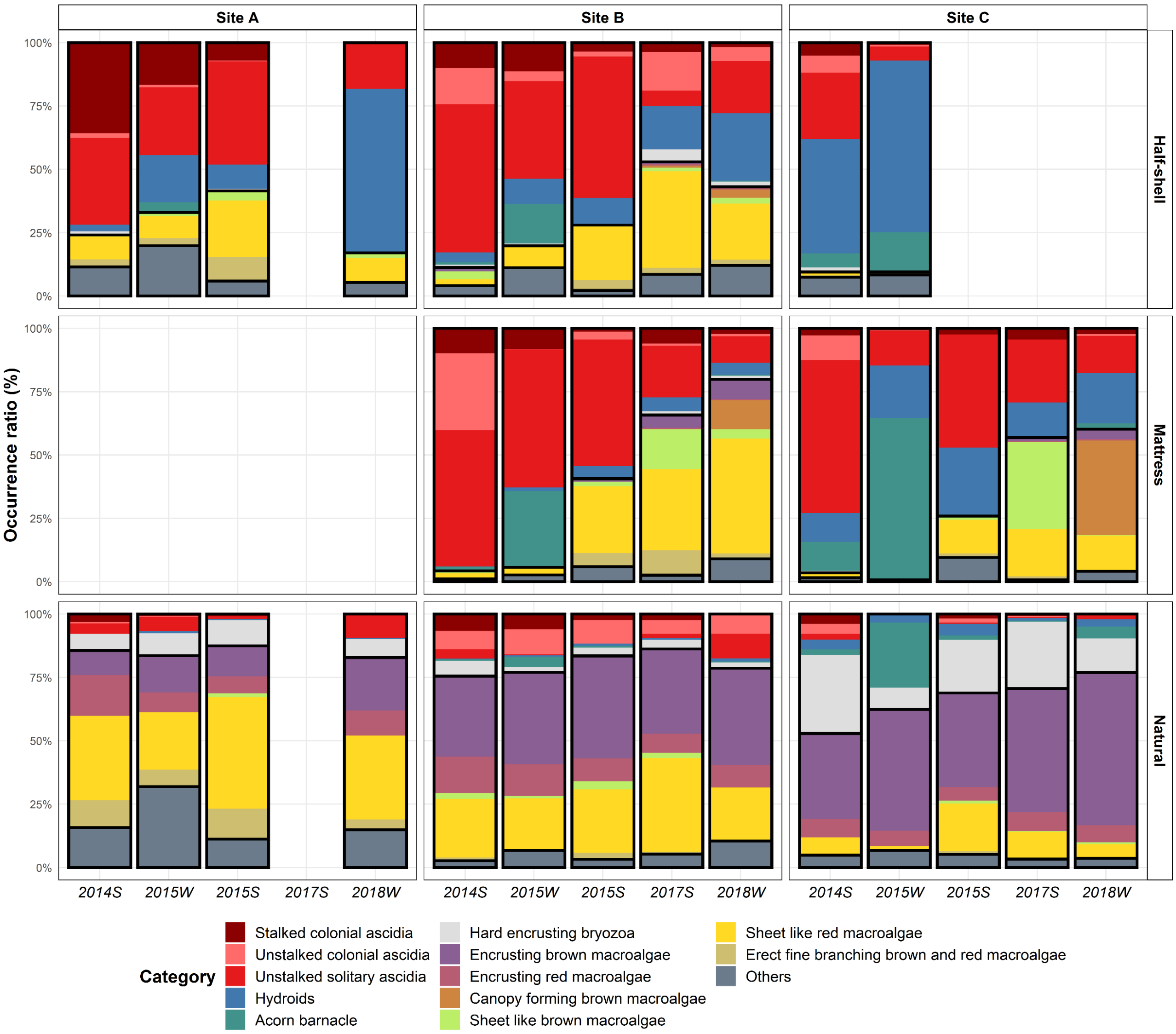

Figure 4: Temporal changes in mean relative percentage cover estimates for epibenthic taxa on each of the studied Habitats (Half-Shell, Mattress and Natural) and at each sites (A, B and C). Only taxa with mean relative cover percentage > 5% were displayed, all others were agglomerated as “Others”. Black lines separate algal and animal taxa as well as “Others”. S = Summer; W = Winter.

kelps) only started to appear on mattresses during the last survey (Figure 4). It should be noted that during the winter 2015 campaign, a massive settlement of barnacles was recorded on both artificial habitats at sites B and C and natural habitat at site C, but the population was not observed during the following campaigns (Figure 3).

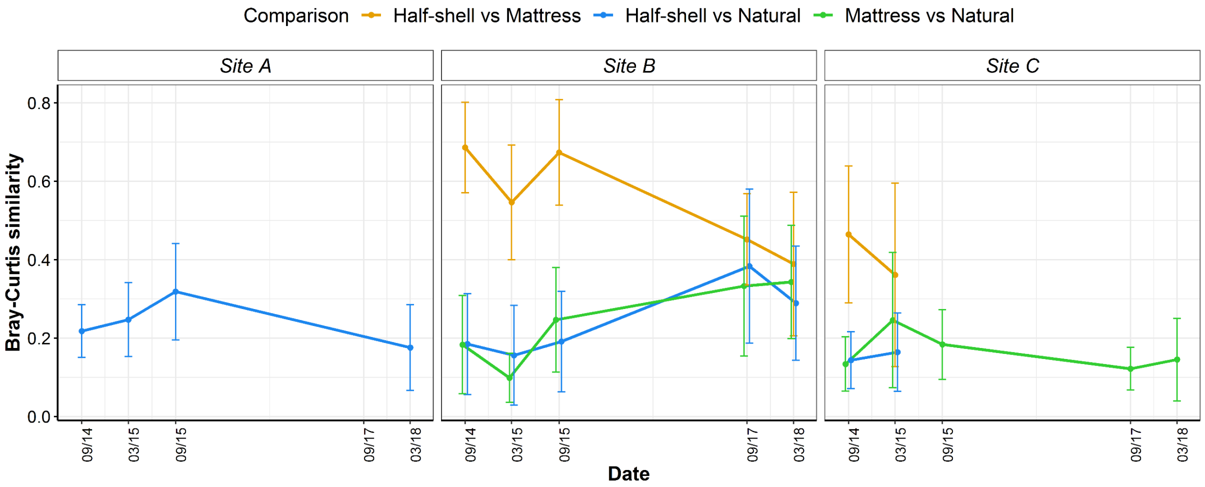

Figure 5: Temporal evolution of mean Bray-Curtis similarities of epibenthic assemblage composition between the different habitats. The mean similarity was computed from all possible combinations of images between the different habitats at each campaign and at the three survey sites (A, B and C). Error bars represent standard deviation.

3.3 Target species dynamics

On natural habitat, densities of the three target species were constant throughout the period except for *C. fornicata*, where the density decreased over time at site C (Figure 6).

The two non- indigenous species *C. fornicata* and *S. clava* exhibited quite similar temporal dynamics on artificial habitats. Their densities were higher on artificial habitats relative to natural habitats during the first years of the survey before declining to levels similar to those on natural habitats (Figure 6). Concerning *C. fornicata*, densities on half-shell habitats at site A and B were higher than those on natural habitats until winter 2015. Afterwards, these decreased below 1 ind m^{-2}, which corresponds to the population density estimates on natural habitats. For *S. clava*, densities measured at all sites were globally higher on half-shell and mattress habitats than on natural habitats and converged with time towards similar densities, starting from summer 2016 for the mattress habitat and from winter 2018 for the half-shell habitat.

The foundation species *Laminaria sp.* presented a completely different temporal dynamic (Figure 6). Kelp densities measured on natural and half-shell habitats remained consistently low (<1 ind.m^{-2}) throughout the surveys and at all three sites. However, densities observed on mattress habitats steadily increased over time. At the beginning of the survey (*i.e.* summer 2014 and winter 2015), *Laminaria sp.* was scarce on mattresses (< 1 ind.m^{-2}) but starting from summer 2015 density was always higher on mattresses than on natural and half-shell habitats, with maximum values reaching 185.2 ± 43.3 ind.m^{-2} at site C during the summer of 2017.

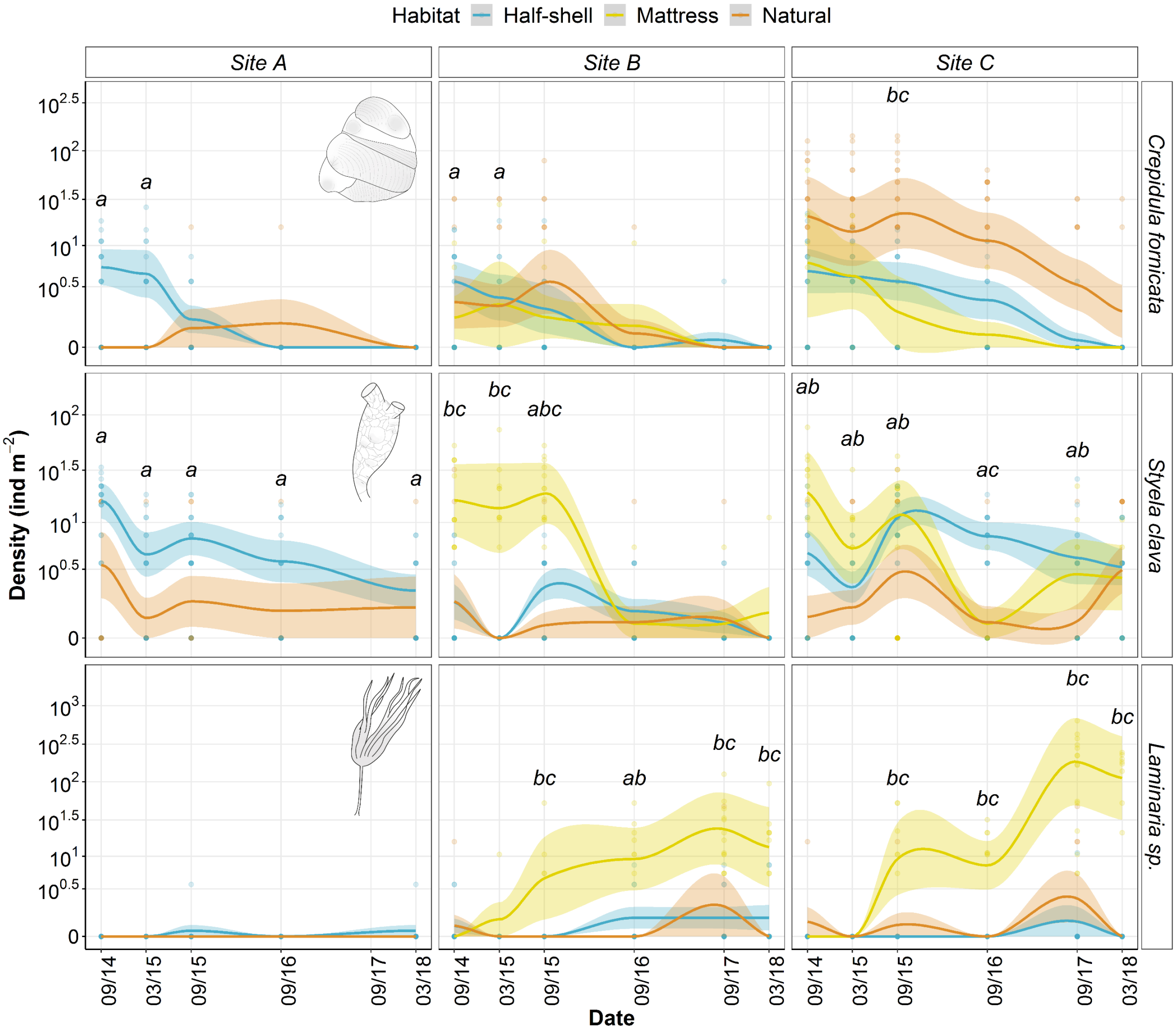

Figure 6: 2014-2018 temporal changes of the densities of the three target species (*C. fornicata, S. clava* and *Laminaria sp.*) at each site (A, B and C) and on each habitat (Half-Shell, Mattress and Natural). Points represent single image density estimates and curves represent mean habitat-specific trends smoothed out using a loess (local polynomial regression fitting). The envelopes surrounding these average trends represent 95% confidence intervals. Note that we used a logarithmic scale for clarity. Letters indicate significant differences in target species density between habitats during a given campaign based on pairwise comparison tests: a = significant difference in density between Half-Shell and Natural habitats, b= between Mattress and Natural habitats and c = between Half-Shell and Mattress habitats.

4. Discussion

Based on a four-year survey using high-resolution underwater imagery, we characterised habitat-specific changes though space and time in the taxonomic composition of epibenthic assemblages in a tide-swept environment. In particular, we described the ecological succession on two kinds of artificial habitats. Our results specifically highlighted an overall similar 4-year trend in the ecological successions across both artificial habitats. However, differences in community composition became significant between these two habitats at the latter stages of ecological succession.

4.1 Spatial heterogeneity

While our three sites are only 2.5 km apart and located at similar depths, they host significantly different benthic communities on natural habitat. Encrusting organisms dominated at site C, as opposed to erect taxa at site A while community composition at site B seemed median relative to the other two more contrasted sites This pattern may be explained by the relative exposure to residual currents that increase from site A to site C. This suggests that a physical stress gradient influences epibenthic communities along the cable route, from the coast to open sea. Our observations are in line with those made concerning the SeaGen tidal test site in the Strangford Narrows, where encrusting communities were found to be associated with high-energy hydrodynamic locations, while erect communities dominated in more sheltered areas (O’Carroll *et al.*, 2017b). Indeed, because of their morphology, encrusting taxa are less exposed to shear stress and abrasion relative to erect taxa that protrude into the water column and have a reduced point of attachment to the substratum (Vogel, 1994). They are consequently more easily swept-away by high frictional flow or abrasion *via* mobile sediments (Daly and Mathieson, 1977; Palmer and Palmer, 1977; Vogel, 1994). Hydrodynamic conditions, in addition to cause abrasion and direct removal of epibenthic taxa, alter the stability of the pebbles which serve as fixation points for these taxa, thus increasing the overall stress. Indeed, when the overturn frequency of the substratum (*i.e.* pebbles and boulders) is high, development of erected taxa it prevented (Osman, 1977; Sousa, 1979a).

To summarise, sites A, B and C presented increasing hydrodynamic conditions from coast to the open sea, leading to an increase of stress on epibenthic communities by *i*) direct removal, *ii*) abrasion by sediment and *iii*) substratum instability.

4.2 Patterns of ecological succession

Our survey started 2 years after the installation of half-shells and 1 year after the installation of concrete mattresses. Thus, the first months of bare substrate colonisation, which are often associated with rapid changes in benthic macrofauna assemblages (Wahl, 1989), were not monitored.

While final community composition differed across the two different artificial habitats (*i.e.* half-shell and mattress habitats), the overall temporal trend in terms of ecological succession patterns are similar over the 4-year survey. Although mattresses and half-shells were not immersed at the same time, they hosted similar epibenthic communities at the beginning of the survey. In the summer of 2014, barnacle mats covered by an ascidian matrix dominated in both artificial habitats. Six months later, in winter 2015, ascidian communities had almost entirely disappeared from both artificial and natural habitats and were replaced by newly-settled mats of barnacles. This “community reset” was probably due to a detachment of previous barnacle mats (and associated attached ascidians) after the end of their natural life cycle and/or because of intense abrasion due to harsh winter conditions (*i.e.* storms). Ascidian overgrowth on barnacles may have contributed to their death, prior to their detachment. Indeed, ascidians often grow inside barnacle orifices (Russ, 1980; Yakovis *et al.*, 2008). From summer 2015 onwards, barnacles did not dominate the artificial habitats. Ascidians (especially unstalked and solitary ones) persisted but were gradually replaced or overgrown by different erect macroalgae (especially on mattresses) and hydroids (especially on half-shells) towards the end of the 4-year survey.

Both barnacles and ascidians are well-known to be early colonisers on a wide variety of artificial hard substrates (Brault and Bourget, 1985; Henschel *et al.*, 1990; Hatcher, 1998; Andersson *et al.*, 2009; De Mesel *et al.*, 2015). On the other hand, kelps are commonly described as a characteristic species of late stages of ecological succession (Carter *et al.*, 1985; Hirata, 1986) while hydroids can both occur as a transient or a permanent species in epibenthic communities (Forteath *et al.*, 1982; Boero and Fresi, 1986; De Mesel *et al.*, 2015). Ecological successions described on both mattresses and half-shells in our study are classic in the sense that epibenthic communities are initially dominated by pioneering taxa (*i.e.* barnacles and ascidians), which are then gradually outcompeted by long-lived and morphologically-complex taxa (*i.e.* macroalgae and hydroids). This succession likely results from a facilitation cascade (Figure 7 ; Altieri et al., 2007). Facilitation cascades can be summarised as a chain of foundation species involved in a hierarchy of positive interactions (*e.g.* mutualism or facilitation). In our study, the “primary” substratum, *i.e.* new artificial substrate, gets colonised by barnacles which are the first foundation species as they form a “secondary” substratum allowing the settlement of a wide diversity of taxa (Brault and Bourget, 1985; Henschel *et al.*, 1990). Consequently, a variety of ascidians can colonise the “secondary” substratum engineered by barnacles to create a “tertiary” substratum. Ascidians, together with remaining barnacles, then allow for the settlement of more complex and long-lived taxa, such as kelp or hydroids. A facilitation cascade dynamic conducted by the same taxa (*i.e.* barnacles, ascidians, and macroalgae) was shown on cockle shells of the White sea (Yakovis *et al.*, 2008; Yakovis and Artemieva, 2017). While our time series stopped when the first signs of colonisation by kelp and hydroids were observed, it is expected that these complex habitat-formers will also facilitate the settlement of a new range of taxa (Norderhaug *et al.*, 2002; Christie *et al.*, 2003; Di Camillo *et al.*, 2017).

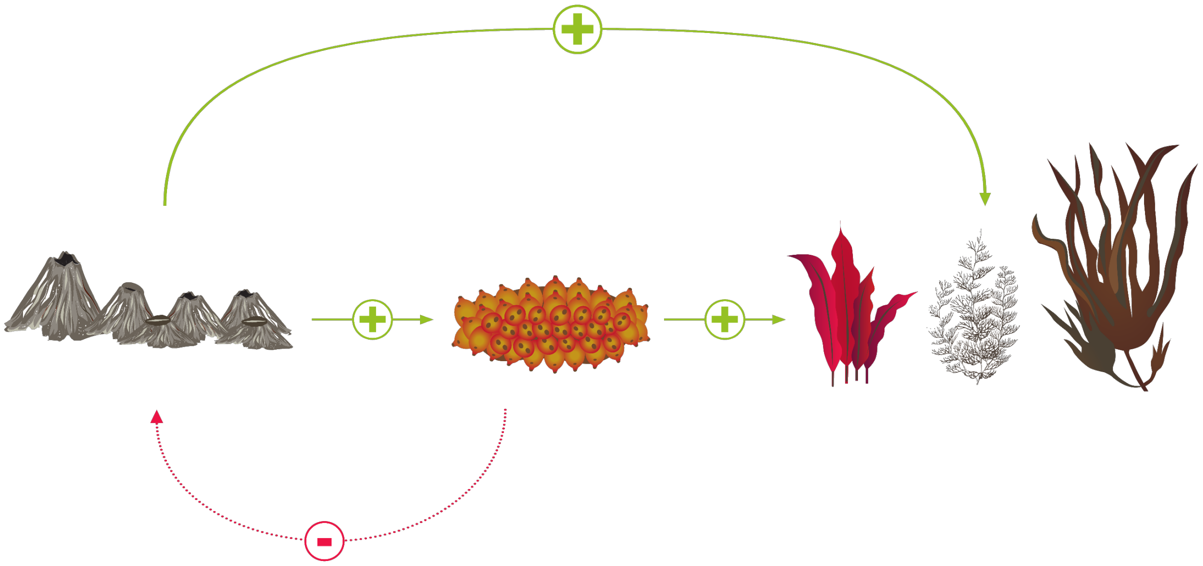

Figure 7: Conceptual diagram of the facilitation cascade that occurred on artificial habitats of the Paimpol-Bréhat tidal test site. Barnacles were the first colonisers of the barren artificial habitats and form a secondary substratum that facilitates the settlement of a variety of ascidians. Ascidians, together with the remaining barnacles, then allow for the settlement of more complex and long-lived taxa, such as various macroalgae, kelp, hydroids etc. Although barnacles had facilitated ascidian settlement, their overgrowth on barnacles may have contributed to the disappearance of barnacles.

The community composition observed on mattresses and half-shells during winter 2018 indicated that their ecological successions had still not reached an equilibrium, although these two artificial substrates were installed 5 and 6 years before, respectively. Reaching the climax state can take up to 11 years for epibenthic communities (Whomersley and Picken, 2003). Nevertheless, taxa considered as indicators of late stages of ecological successions dominated the community of our artificial habitats at the end of our survey, giving an idea of the characteristics of their future climax stages.

4.3 Implications for non-indigenous species

Numerous examples showed that introduction of new artificial habitats on marine environments can contribute to introduction or propagation of non-indigenous species (Bulleri and Airoldi, 2005; Vaselli *et al.*, 2008; Mineur *et al.*, 2012; Airoldi *et al.*, 2015; De Mesel *et al.*, 2015). Non-indigenous species are often opportunistic and act as early colonisers of new artificial habitats, which directly contribute to their invasive success (Mineur *et al.*, 2012; De Mesel *et al.*, 2015). In this study, the densities of the two non-indigenous species *C. fornicata* and *S. clava* were higher on both artificial habitats than on natural habitats during the first stages of the ecological succession. Nevertheless, their densities rapidly decreased with time to reach similar levels to those of natural habitats, after one year of survey for *C. fornicata* and after two years for *S. clava*. These temporal dynamics shown by the two species are typical of pioneer species on artificial habitats. *C. fornicata* settled rapidly on the freshly installed mattresses, on half-shells and on conspecific individuals but was never observed to settle onto other epibenthic species. Conversely, *S. clava* was able to settle on the "secondary substratum" formed by other epibenthic taxa already settled, as reported in other locations (Lützen, 1999). The quicker disappearance of *C. fornicata* can thus be explained by a stronger competition for space (*e.g.* with barnacles and ascidia), as the proportion of available artificial substratum drastically declined with time. On the contrary, *S. clava*, which can settle on other species, can persist in time as availability of primary substratum is not a limiting factor. Nevertheless, its density eventually decreased due to spatial competition processes with other species of the community.

Although the densities of these two species on artificial habitats decreased within a few years, they possibly persisted long enough for individuals to become sexually mature and reproduce. Indeed, *S. clava* can reach sexual maturity within 10 months (Lützen, 1999) while *C. fornicata* female and male individuals can reach maturity within 2 and 1 year post-settlement, respectively (Richard, 2005). Consequently, we showed that such artificial habitats can act as a stepping stone by allowing non-indigenous species to settle and successfully reproduce even if they do not persist locally in the long term. In our case however, there is no risk of a stepping stone effect since *C. fornicata* and *S. clava* have been present on surrounding biogeographic areas for decades (Mineur *et al.*, 2012). Furthermore, the presence of hard substratum is not limiting around the Paimpol-Bréhat's submarine power cable (which was installed between several rocky shelves; Figure 1). Nevertheless, our understanding of the mechanisms and implications of artificial structures acting as stepping stones remains to be elucidated (Mineur *et al.*, 2012; Copping *et al.*, 2016; Dannheim *et al.*, 2019) This needs particular attention from the scientific community, especially *i*) for artificial structures installed in soft sediment area where hard substratum is limiting, and *ii*) at the geographical distribution limits of invasive species.

While *C. fornicata* and *S. clava* rapidly colonised new substrate and declined, their dynamics could have been different if these substrates had been in a different environment. Indeed, artificial substrates installed within soft-sediment environments disproportionally favour non-indigenous over native species (Airoldi *et al.*, 2015). Also, note that we only targeted these two species because they were easily recognizable on underwater images. Other non-indigenous species could exhibit different colonisation dynamics and for instance outcompete indigenous species in the long term. For example, another study comparing trends of epibenthic communities colonisation on natural and artificial (aluminium sheet metal, polystyrene, PVC and rubber), showed a declining proportional abundance of native compared to non-indigenous species over time (Tyrrell and Byers, 2007).

4.4 Towards different climaxes

Despite a common trend in terms of community succession on both artificial substrates during the first 2 years of the survey, epibenthic communities exhibited contrasted habitat-specific trajectories at the end of the four-year survey. During the two last campaigns, high proportions of hydroids taxa dominated on half-shells while erect macroalgae, especially *Laminaria sp.* were dominant, alongside with hydroids, on mattresses. These observations suggest that each artificial habitat moves towards its own specific climax state, both of them being different from the climax of the surrounding natural habitat. These different trajectories may be facilitated by a combination of drivers:

1. First, substrate type can largely influence epibenthic community composition. Numerous studies highlight that seafloor characteristics (*e.g.* texture, complexity, composition

and colour) impact epibenthic communities colonising artificial habitats (Hixon and Brostoff, 1985; Glasby, 2000). Concrete and steel, which constitute mattresses and half-shell, respectively, were reported to be colonised by different epibenthic communities (Andersson *et al.*, 2009). Concrete, which has a rougher surface than steel (Andersson *et al.*, 2009), can display similar patterns of colonisation than natural rocky surfaces (Foster, 1975; Sousa, 1979b). On the contrary, steel supports different epibenthic communities to concrete and natural reef habitats (Andersson *et al.*, 2009; Ushiama *et al.*, 2016), as its smoother surface makes epibenthic species settlement more challenging. (Andersson *et al.*, 2009).

2. Furthermore, in such a high-energy environment, substratum stability can also play an important role in shaping the trajectory of colonising epibenthic communities. At the three sites, the natural habitat is essentially composed of a majority of pebbles in different proportions with only a few boulders. The strong tidal currents of the area can regularly destabilise these pebbles making them a highly unstable habitat for benthic macrofauna and preventing the development of erect and complex taxa (Osman, 1977; Sousa, 1979a). On the other hand, the artificial habitats considered in this study are massive and cannot be displaced by currents, offering more stability and allowing the development of more complex communities.

3. In addition to stability, habitat elevation above neighbouring sediments directly determines local exposure to abrasion. We can indeed hypothesise that the higher the habitat, the more sheltered the epibenthic communities from sediment abrasion. Natural habitats are by definition at the level of the sea bottom, whereas half-shells and mattresses sit respectively at ~15 cm and ~40 cm above. As explained previously, erect taxa are less adapted than encrusting organisms to this frictional stress caused by abrasion (Vogel, 1994). Natural habitats, which are more exposed to abrasion, showed higher proportions of encrusting organisms than the two artificial habitats. Conversely, erect taxa are more abundant on more sheltered (*i.e.* elevated) habitats such as moderately-elevated half-shells, and even more abundant on concrete

mattresses. The high colonisation of hydroids, mainly *Sertularia sp*., at the apex of the half-shells is linked to the tolerance of this species to periodic submergence and scouring by sand (Connor *et al.*, 2004). Mattress habitats are the least exposed to abrasion and consequently housed the most complex communities including large canopy-forming algae.

To summarise, *i*) natural habitats dominated by unstable pebbles are well-exposed to sediment scouring, *ii*) half-shells constitute a stable cast iron habitat moderately exposed to sediment scouring due to moderate elevation and *iii*) mattresses constitute a stable concrete habitat marginally exposed to sediment abrasion as their anchor point is high above adjacent sediments (Figure 8).

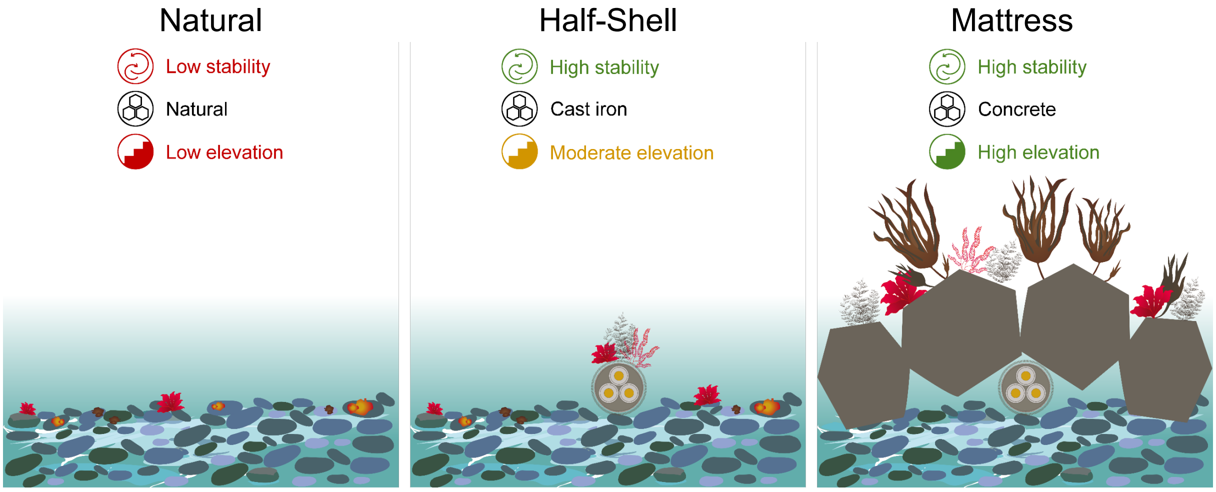

Figure 8: Conceptual diagram of the epibenthic colonisation of the three different habitats of the Paimpol-Bréhat tidal test site at the end of our survey. Natural habitats (left) are dominated by unstable pebbles and are highly exposed to sediment scouring; the epibenthic community is thus characterised by encrusting taxa. Half-shells (middle) constitute a stable cast iron habitat moderately exposed to sediment scouring due to moderate elevation; the epibenthic community is thus characterised by erect taxa with moderate structural complexity (*i.e.* hydroids). Mattresses (right) constitute a stable concrete habitat marginally exposed to sediment abrasion as their anchor point is high above adjacent sediments; the epibenthic community is characterised by various erect taxa with complex morphology (*i.e.* kelps).

4.5 An environment under high pressure

Tide-swept benthic environments, such as Paimpol-Bréhat tidal test site, are specifically exposed to a strong physical stress caused by pebbles/boulders overturning and abrasion by moving sediment (Sousa, 1979a, 1979b; Dean and Connell, 1987). In these environments, epibenthic community structures are more likely influenced by physical processes than by biological interactions (Dean and Connell, 1987; Sousa, 1979a). Physical disturbances can regularly free space for recolonization, as the case of the "community reset" observed during our winter 2015 campaign. These disturbances can interrupt successional sequences so that epibenthic communities form an heterogeneous mosaic of species assemblages at different ecological successional states (Osman, 1977; Palmer and Palmer, 1977; Sousa, 1979a, 1979b). Deployment of stable artificial habitats in such an environment submitted to high physical pressure somehow lead to the occurrence of habitat patches, whereas epibenthic succession can reach more complex ecological states than the community on natural habitats.

5. Conclusions

Deployment of artificial structures in the Paimpol-Bréhat tidal test site resulted in the addition of stable substrata in an environment where natural hard substrates are highly mobile and strongly exposed to sediment abrasion due to strong hydrodynamic conditions. These safe houses of stability allow for structurally-complex epibenthic communities to flourish, which facilitates an overall increase in local diversity as lack of stable natural hard substrates limits the development of mature epibenthic communities. Nevertheless, epibenthic communities colonizing artificial habitats are unlikely to have reached their climax at the end of our four-year survey. Because we quantified how epibenthic communities can significantly vary in high-energy coastal environment, and because our final surveys suggested that the ecological succession was still in progress 5 years after deployment of artificial reefs, our study highlights the need to maintain survey in the long-term of coastal artificial reef habitats to more fully understand the ecological successions and temporal variability.

Acknowledgements

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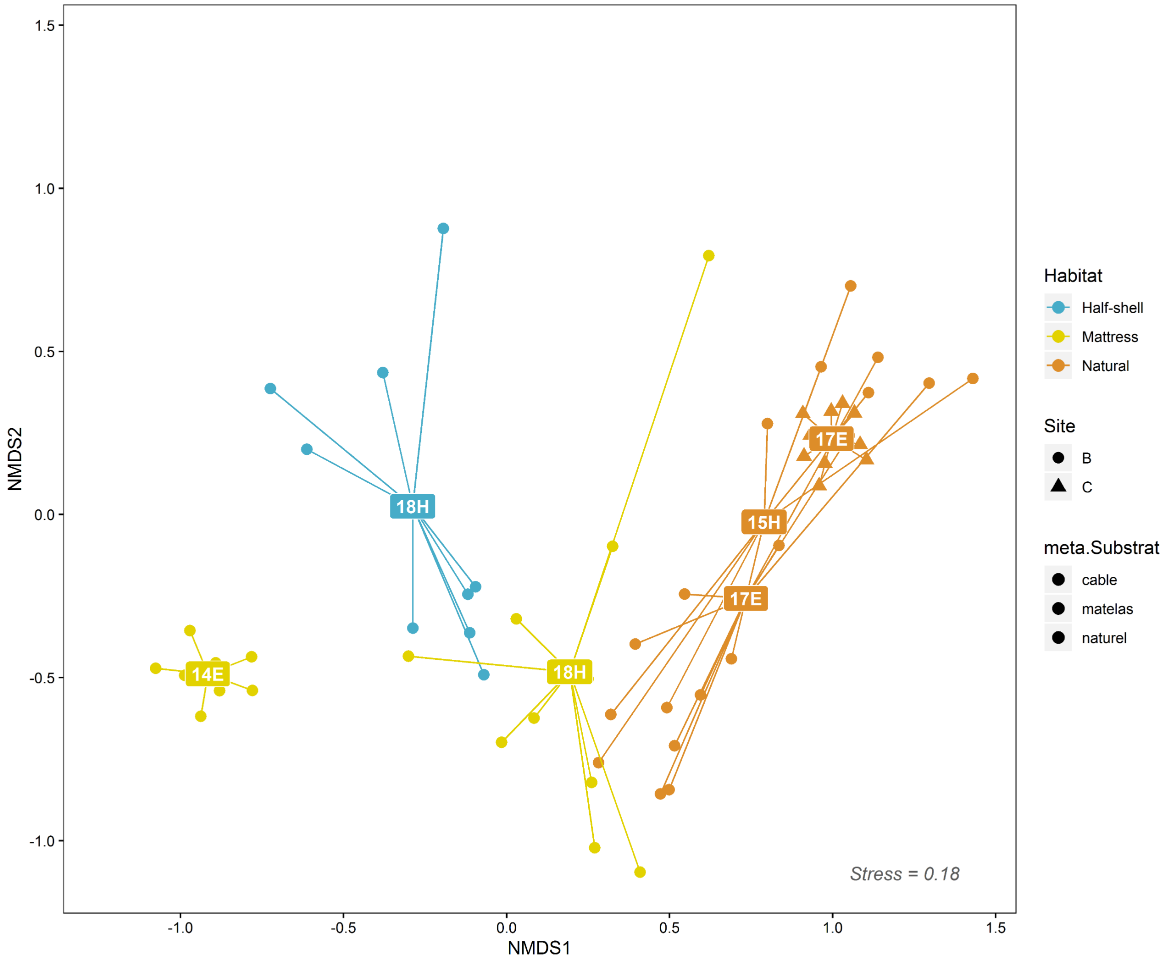
Supplementary information 1: Results of the pairwise homoscedasticity test (PERMDISP) between the different groups of images (c.f. Table; only significantly different combinations are shown) and non-metric Multidimensional Scaling (nMDS) based on Bray-Curtis similarities in community composition between samples with only the groups of images showing significant differences of variance being displayed (c.f. Figure). The spatial segregations of these different groups allow us to perform a PERMANOVA, although homoscedasticity conditions are not fully validated.

Pairwise Comparison										
Site	**Habitat**	**Campaign**		**Site**	**Habitat**	**Campaign**	**diff**	**lwr**	**upr**	**P (adj)**
C	Naturel	2017E	/	B	Half-shell	2018H	*-0.22*	*-0.44*	*-0.01*	***0.03***
B	Naturel	2017E	/	B	Mattress	2014E	*0.21*	*0.00*	*0.42*	***0.04***
C	Naturel	2017E	/	B	Mattress	2018H	*-0.23*	*-0.44*	*-0.02*	***0.01***
C	Naturel	2017E	/	B	Naturel	2015H	*-0.23*	*-0.44*	*-0.02*	***0.01***
C	Naturel	2017E	/	B	Naturel	2017E	*-0.24*	*-0.45*	*-0.03*	***0.00***
All other combinations										*>0.05*



Supplementary information 2: List of the different taxa described during image annotation of the different habitats of the Paimpol-Bréhat tidal test site. The CATAMI classification was used to describe the different taxa (Althaus et al., 2015), only the category "Calcareous tube worms" was added. The number of pictures on which each taxa was observed is indicated (total number of pictures analysed: 350). Cloe-up photos were taken by Xavier Caisey.

CATAMI classification					**Number of pictures**
Macroalgae					
	Encrusting				
		Brown			177
		Red			159
	Erect fine-branching				
		Brown			42
		Red			114
	Filamentous				
		Red			3
	Large Canopy Forming				
		Brown			18
	Sheet like				
		Brown			95
		Red			291
Sponge					
	Encrusting				109
	Erect form				14
	Massive form				
		Balls			5
		Cryptic			2
Cnidaria					
	Colonial anemones				
		Corallimorphs			34
		Zoanthids			14
	Corals				
		Octocorals			
			Fleshy		
				Arborescent	1
		Stony corals			
			Solitary		
				Attached	1
	Hydroids				216
	True anemones				
		Other anemones			87

Bryozoa			
	Hard		
		Branching	1
		Encrusting	193
Worms			
	Polychaetes		
		Calcareous tube worms	122
		Non-calcareous tube worms	39
Echinoderms			
	Sea cucumbers		
		Benthic	1
Mollusc			
	Bivalves		7
	Chitons		2
	Gastropods		130
Crustacea			
	Barnacles		
		Acorn	115
	Crab		
		True crab	8
	Hermit crabs		5
	Prawns/Shrimps/Mysids		7
Ascidians			
	Stalked		
		Colonial	189
		Solitary	45
	Unstalked		
		Colonial	162
		Solitary	258
Fish			
	Bony fishes		7

Chapter 4

Renewable energy homes for marine life: habitat potential of a tidal energy project for benthic megafauna

Under review in Marine Environmental Research.

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Abstract

An increasing number of offshore structures are being deployed worldwide to meet the growing demand for renewable energy. Apart from energy production, these structures can also provide new artificial habitats to a diversity of fish and crustacean species. This study characterises how concrete mattresses, which stabilise the power cable of a tidal energy test site, can increase habitat capacity for benthic megafauna. Our three-year monitoring, which relied on both dive-based and video-based surveys, revealed that these mattresses provide a eligible habitat for 6 species of large crustaceans and fish. In particular, we identified that two commercially valuable species, *i.e.* the edible crab *Cancer pagurus* and the European lobster *Homarus gammarus,* progressively occupied these new artificial habitats throughout the course of the project. The shape and the number of shelters available below individual mattresses largely determine the nature and the extent of colonisation by mobile megafauna. Local physical characteristics of the implantation site (*e.g.* substratum type, topography, exposition to current *etc.*) significantly impact the amount and the type of shelters provided by the concrete mattresses. Thus, to characterise the habitat potential of artificial structures precisely , it is crucial to take into account (*i*) the design of the structures themselves, and (*ii*) consider how they will interact with the local environmental conditions when deployed on the seafloor.

Keywords

Artificial reef ; marine renewable energy ; crustacean ; ichthyofauna ; habitat complexity

1. Introduction

Artificial reefs are man-made structures placed on the sea bed in aquatic habitats for different purposes, for instance to mimic characteristics of natural reefs such as substrate and/or shelter provision to associated organisms (Thierry, 1988; Bohnsack *et al.*, 1991; Jensen *et al.*, 2000a). Development of artificial reefs locally increases both hard substratum availability and habitat heterogeneity (especially when deployed on soft-sediment bottoms), which can consequently lead to higher densities and biomass of fish and decapods (Bohnsack *et al.*, 1994; Bombace *et al.*, 1994; Langhamer and Wilhelmsson, 2009).

The magnitude of the enhancement of associated benthic diversity by artificial reefs depends both on reef properties and on local environmental characteristics. Colonisation success depends on artificial reef shape and size, constitutive material, orientation and degree of complexity (that directly determine habitat) and refuge availability (Ferreira and Coutinho, 2001; Charbonnel *et al.*, 2002; Sherman *et al.*, 2002; Hackradt *et al.*, 2011). A range of local environmental factors (*e.g.* neighbouring habitat type, hydrological features, amplitude of seasonal variation) can significantly influence the amount and the diversity of colonising organisms (Bohnsack *et al.*, 1991; Bombace *et al.*, 1994; Godoy *et al.*, 2002; Noh *et al.*, 2017). A long-standing scientific debate persists between two dominant theories regarding the role of artificial reefs for mobile fauna: (i) the “attraction hypothesis” and (ii) the “production hypothesis” (Lima *et al.*, 2019). The first assumes that artificial reefs only attract specimens from nearby ecological communities, without increasing overall biomass production (Bohnsack, 1989) while the latter advocates that artificial reefs increase abundance and biomass of associated species by enhancing habitat and food availability (Pickering and Whitmarsh, 1997; Polovina and Sakai, 1989). Lima et al. (2019) highlight that, despite several decades of scientific observations and experiments on the subject, separating the reef effect and the effects of changing environmental and socioeconomic conditions remains complex, impacting the assessment of artificial reefs performance.

Artificial reefs can be divided into two types: *i*) structures designed and installed specifically for their reef properties (for a variety of reasons *e.g.* ecosystems conservation/restoration, fish stocks enhancement, fisheries management *etc.*; Jensen, 2002) and *ii*) structures deployed for other purposes, such as oil rigs, breakwaters, or Marine Renewable Energy (MRE) facilities (Wilson and Elliott, 2009; Langhamer, 2012; Lima *et al.*, 2019). MRE facilities and associated structures (*e.g.* protection structures, submarine power cables, foundations, turbines *etc.*) are not only colonised by a variety of benthic organisms including algae, sessile epifauna and mobile macrofauna but also mobile megafauna (*i.e.* fish and decapods). A diversity of fish and crustacean species can settle on artificial reefs deployed as part of MRE facilities (see Wilhelmsson and Langhamer, 2014 for a review). For example, commercially valuable crustacean species such as the European lobster (*Homarus gammarus*) or the edible crab (*Cancer pagurus*) can shelter around the foundations of offshore wind (Hooper and Austen, 2014; Krone *et al.*, 2017) or wave farms (Langhamer and Wilhelmsson, 2009). Thus, such reef effects can represent an ecological benefit of MRE, since artificial structures generally host higher diversity, densities and biomass of benthic organisms than the surrounding soft bottoms (Langhamer and Wilhelmsson, 2009; Broadhurst and Orme, 2014). Wilson and Elliott (2009) estimated that in the long term, a wind-turbine facility provides 2.5 times the amount of habitat relative to the initial loss during the installation process, even though this new habitat may be of a different character to the initial one. When their deployment requires the implementation of new exclusion areas for fishing, MRE may thus act as a refuge for commercially-exploited populations, with potential spill-over benefits for adjacent stocks and fisheries (Lindeboom *et al.*, 2011, 2015). However, the long-term reef effect associated with MRE facilities remains poorly characterised (Langhamer and Wilhelmsson, 2009;

Lindeboom *et al.*, 2015; Copping *et al.*, 2016), especially within high hydrodynamic energy areas (as tidal energy sites; Copping et al., 2016).

The purpose of this study is to enhance our current understanding of the reef effects associated with MRE facilities using a French-based tidal energy test site as a case study. We specifically examined the habitat capacity of concrete mattresses that stabilise an unburied submarine power cable that connects the test site to the mainland. Scuba divers monitored the abundance of several species of fish and crustaceans over a 2 years period to (1) characterise the reef effect associated to the MRE facilities, and more specifically (2) to understand how the physical characteristics of the artificial mattresses in interaction with the adjacent natural seafloor can determine the diversity and the abundance of the associated megafauna.

2. Methods

2.1 Study site

The study site consists of a 15 km-long submarine power cable (8 MVA - 10 kVDC) laid in 2012 by Electricité de France (EDF). to connect the tidal test site of Paimpol-Bréhat to the mainland (Brittany, France; Figure 1). Due to several setbacks in the project development, no electric current transited through the cable during the course of this study. In 2013 EDF deployed 15 km of cable, 11 of which were unburied and stabilised by 120 concrete mattresses due to local seafloor characteristics (dominance of pebbles and presence of boulders ; Figure 2.A) to prevent any cable displacement due to high hydrodynamic conditions. These mattresses were installed at depths ranging from 15 to 33 m and for the majority approximately 50 m apart (with some 200 m apart). The 6 m-long and 3 m-wide mattresses were made up of 73 concrete blocks linked together by an array of polypropylene rope with a maximum thickness of 0.3 m, and a weight of ~10 t.

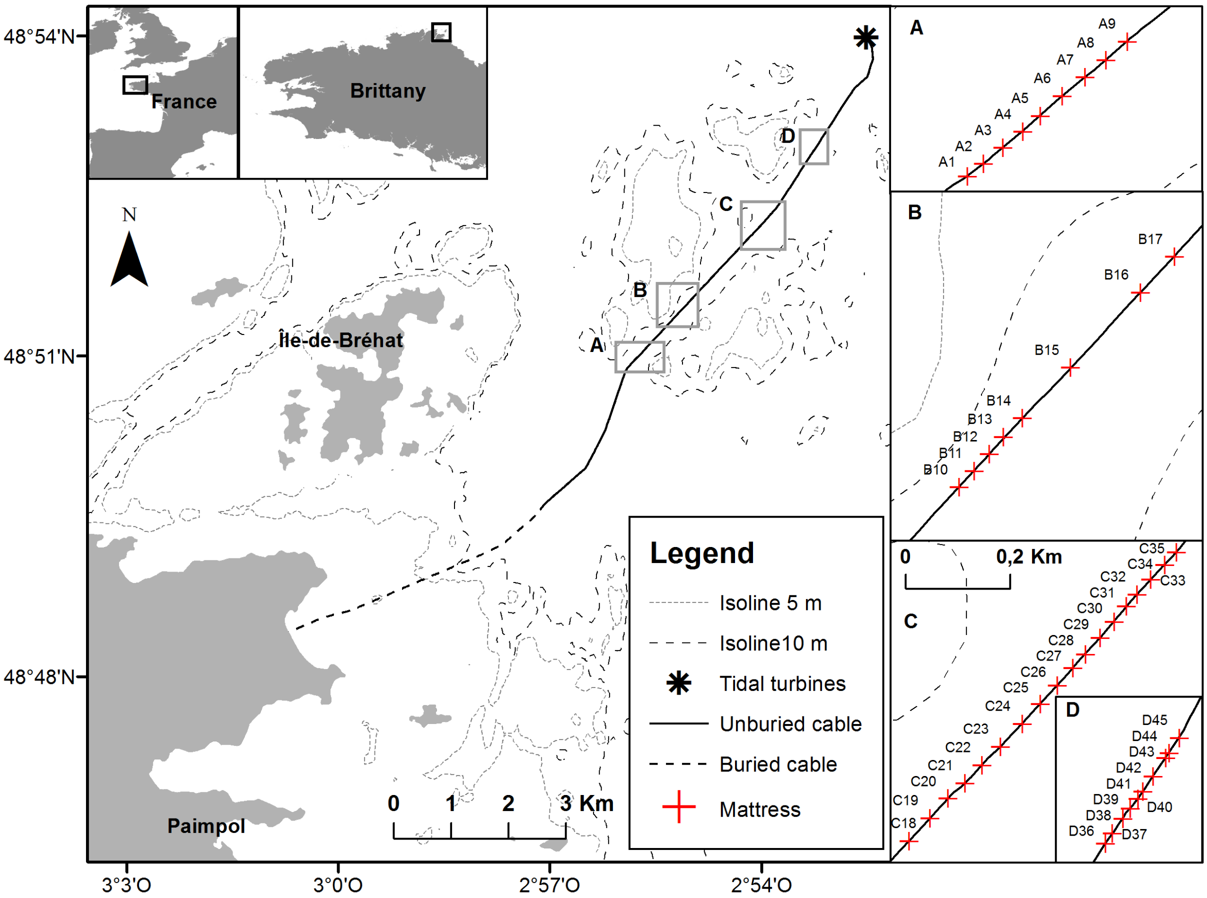

Figure 1: Map of the study area off the north coast of Brittany in Western France (top-left and top-centre panels). The thicker black line indicates the location of the power cable that connects the Paimpol-Bréhat tidal turbine test site to the mainland. Zones A, B, C and D (right) correspond to the four areas where concrete mattresses were surveyed by scuba divers.

2.2 Target species

A set of 6 benthic megafauna species frequently observed around concrete mattresses during preliminary surveys, were surveyed: the crustaceans *Homarus gammarus* (European lobster) and *Cancer pagurus* (edible crab) and the benthic or demersal fish, *Conger conger* (European conger), *Labrus bergylta* (Ballan wrasse), *Trisopterus luscus* (whiting pout) and *Trisopterus minutus* (poor cod).

2.3 Sampling strategy

Between June 2015 and June 2017, 45 different concrete mattresses in the 15-20 m depth range were surveyed by divers within four different areas along the cable (Zones A, B, C and D; Figure 1). These four areas are located within a channel surrounded by several rocky shelves.

A total of 45 mattresses were surveyed in June 2015, 30 in September 2015 (zone A, D and part of the zone C) and 37 in June 2016 and 2017 (zone A, C and D; Table 1). All surveys were performed at slack tides and during daytime hours, *i.e.* between 8 am and 8 pm. During each survey, two divers inspected each mattress: the first diver moved slowly along the entire perimeter of the mattress while examining all the cavities and counting the abundance of the 6 target species. Simultaneously, the second diver followed the first one around the mattress and recorded a video using a GoPro Hero 4® camera to provide some additional observations of the mobile fauna and the environment (substratum bottom type, frequency and forms of cavities *etc.*). Hereafter, a "sample" refers to all these pieces of information recorded for a given mattress, during a given campaign.

Table 1: Summary of the concrete mattresses and zone surveyed during each campaign.

Campaign	Number of Mattresses	Zone
June 2015	45	A-B-C-D
September 2015	30	A-C(only C25 to C35)-D
June 2016	37	A-C-D
June 2017	37	A-C-D

2.4 Environmental variables

The substratum surrounding each mattress and the types and number of cavities were estimated from each video. Based on video records, each concrete block located around the mattress edges was assigned to one of three substratum categories: sand (Snd), pebbles (Pbl) or boulders (Bld). For each mattress, substratum properties were estimated as a ratio of each of these three categories across all concrete blocks (Table 2). Two different types of cavity that can provide habitats to mobile fauna underneath the mattresses were identified: the "holes", which correspond to small triangular cavities between two concrete blocks along the mattress width (Figure 2.B & E; Table 2); and the "caves", which correspond to cavities formed below the mattress when it does not touch the seafloor (Figure 2.C, D & F; Table 2). Any free space of at least 10 cm height below a concrete block is considered as a cave.

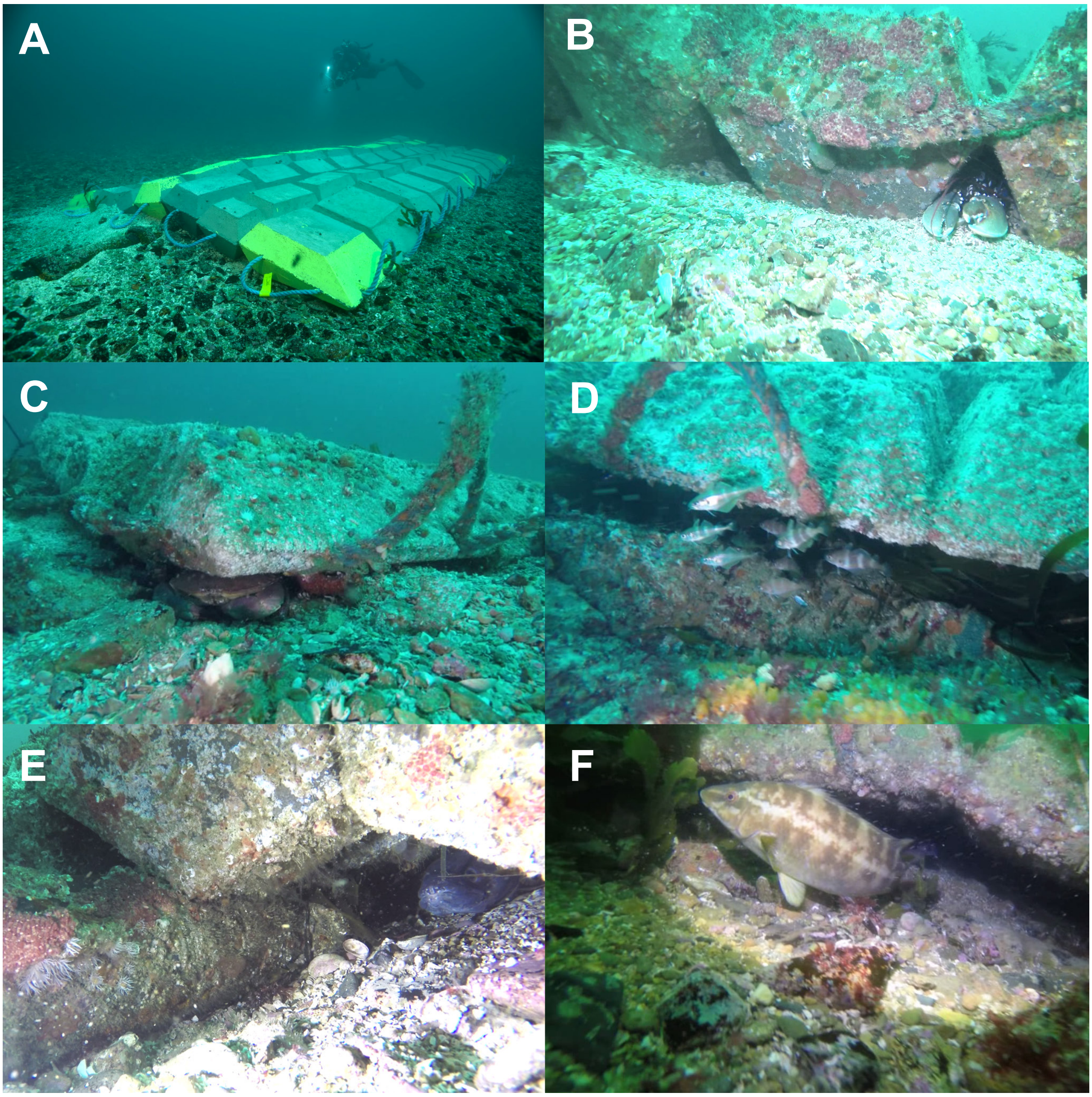

Figure 2: (A) Overall view of a concrete mattress a few weeks after its installation on the Paimpol-Bréhat tidal test site power cable; (B) *Homarus gammarus* within a “hole”, *i.e.* a small triangular cavity between two concrete blocks; (C) *Cancer pagurus* within a “cave”, *i.e.* a >10 cm high cavity formed below the mattress; (D) school of *Trisopterus luscus* close to a mattress “cave” ; (E) *Conger conger* within a “hole” ; (F) *Labrus bergylta* close to a mattress “cave’.

In addition, the following environmental variables were extracted from a Geographic Information System (GIS) database using the position of each mattress (Table 2): (1) bottom residual current velocity (computed from a 2010-2015 climatology extracted from the MANGA500-MARS3D hydrodynamic model); (2) seafloor facet exposure to residual current (computed using seafloor aspect and residual current direction and ranging from 0°, when the seafloor is sheltered, to 180°, when it is fully exposed to dominant current) and (3) linear distances from each mattress to the closest 5 m and 10 m depth isobaths. These variables were selected out of a wider panel to avoid redundant variables or variables with insufficient resolution or biological relevance.

Table 2: Summary of available biological and environmental variables, either measured in situ or from video footage, or derived from bathymetric map or from the MARS3D hydrodynamic model.

	Variable	**Unit**	**Origin**	**Mean**	**Min**	**Max**
Biological data	*H. gammarus*	count	*in situ*	1.1	0	5
	C. pagurus	count	*in situ*	1.53	0	5
	C. conger	count	*in situ*	1.34	0	4
	T. luscus	count	video	2.81	0	67
	T. minutus	count	video	0.28	0	5
	L. bergylta	count	video	1.11	0	6
Environmental data	Bathymetry	m	*in situ*	18,1	15	20
	Sand proportion	%	video	16%	0%	68%
	Pebble proportion	%	video	25%	0%	36%
	Boulder proportion	%	video	3%	0%	89%
	Number of holes	count	video	10.39	4	12
	Number of caves	count	video	9.73	0	28
	Bottom-current velocity	$m\ s^{-1}$	GIS	0.71	0.65	0.99
	Exposure	°	GIS	89.12	12.7	163.85
	Distance to 5 m isobath	m	GIS	498.3	149.7	791.2
	Distance to 10 m isobath	m	GIS	243.8	62	403.5

2.5 Biological data

In addition to *in situ* abundance counts performed by divers, video counts were performed for the 6 target species (Table 2). To avoid multiple counts of single individuals of *T. luscus* and *T. minutus* given their high mobility, the maximum number of individuals occurring in a single snapshot of the full video was recorded. In order to investigate species-specific sheltering preferences, the positions of each counted individual with respect to the mattress was noted as either *i*) inside a “hole”, *ii*) inside a “cave”, or *iii*) free-moving outside any cavities.

For *H. gammarus*, *C. pagurus* and *C. conger*, we used *in situ* counts by divers for multivariate analyses (Table 2), video counts underestimating both of these species abundance (SI 1) due to their cryptic behaviours. For the 3 other target species *L. bergylta*, *T. luscus* and

T. minutus, multivariate analyses were based on video counts (Table 2) since *in situ* counts were not so accurate due to their high mobility and their tendency to form dense schools (SI 1). All video analyses were performed using the Ifremer ADELIE© Software V2.0 according to the methodology developed and tested by Dufournaud (2018).

Due to poor footage quality, only 109 out of the 149 videos could be fully analysed and were used to perform multivariate analyses. To add some additional data about the cavity preferences of the target species, locations (either inside a "hole", inside a "cave", or free-moving out of any cavities) in which each specimen were detected were studied for each species using the full set of 149 videos.

2.6 Data analysis

Three main types of statistical analyses were performed, namely: *i*) non-parametric Kruskal-Wallis tests to explore the temporal variations of megafauna abundance and diversity, *ii*) principal component analysis (PCA) to study patterns in megafauna assemblage compositions, and *iii*) redundancy analysis (RDA) to examine how variability in megafauna community relates to environmental conditions.

From results collected on the 30 mattresses surveyed during the four campaigns, a non-parametric Kruskal-Wallis test was applied to study temporal variations of (1) the abundance of each target species, (2) the total abundance of all species (with and without *T. luscus*) and (3) the species richness per mattress. When appropriate, a *post-hoc* test using Bonferroni correction was performed to study pairwise differences between campaigns. We then characterised variability in megafauna composition between samples using a PCA. Finally, to related community variability to changes in environmental variables (Table 2), we performed a Redundancy Analysis (RDA; (Legendre and Legendre, 1998)). We used Draftsman's plots to detect significant correlations (>0.7) between covariates and keep only a subset of environmental variables prior to analysis. Using a Monte-Carlo permutation test (999 permutations), a forward selection process was performed to identify environmental variables that best correlate to observed variability in community composition. In order to reduce the weight of abundant school-forming species such as *Trisopterus sp.*, a logarithmic transformation was applied to the abundance data set before all multivariate analyses. Environmental variables were normalised prior to RDA analysis. Data analysis was performed with Rstudio (RStudio Team, 2015) using the *vegan* (Oksanen *et al.*, 2018) and *ggplot2* (Wickham, 2016) packages.

3. Results

3.1 Temporal variation

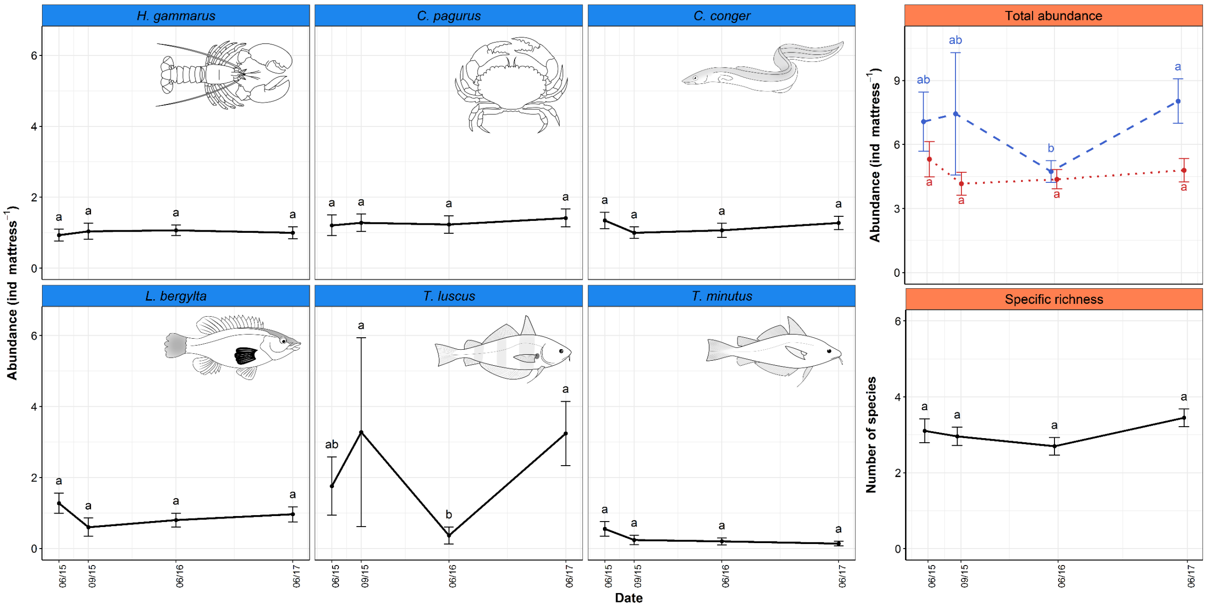

Figure 3: Changes between June 2015 and June 2017 in the mean abundance of the 6 different target species (average number of individuals per mattress ± standard errors), in the total surveyed abundance per mattress (blue dashed line: all species taken together, red dotted line: all species except the highly abundant *Trisopterus luscus*) and in the specific richness per mattress. Points labelled with different letters mean that the values are significantly different. Only the 30 mattresses surveyed at each campaign were considered.

Although substrate colonisation by megafauna vary slightly over time for individual mattresses (SI 2), mean abundances across the 30 mattresses surveyed during all the campaigns did not significantly change over time for *H. gammarus* (Kruskall-Wallis test, P = 0.92), *C. pagurus* (Kruskall-Wallis test, P = 0.79), *C. conger* (Kruskall-Wallis test, P = 0.71), *L. bergylta*

(Kruskall-Wallis test, P = 0.18), and *T. minutus* (Kruskall-Wallis test, P = 0.22; Figure 3). Only *T. luscus* showed significant abundance changes between campaigns (Kruskall-Wallis test, P = 2 .10^{-5}; Figure 3) with a lower abundance in June 2016 compared to September 2015 and June 2017. Overall individual abundance (across all six taxa) per mattress significantly changed between campaigns (Kruskall-Wallis test, P=0.05; figure 3) with a lower total abundance in June 2016 relative to June 2017, which can be attributed to changes in *T. luscus*. Indeed, when excluding *T. luscus*, there was no significant temporal change in total megafauna abundance (Kruskall-Wallis test, P=0.85; figure 3). Specific richness per mattress did not significantly change overtime either (Kruskall-Wallis test, P = 0.28; Figure 3).

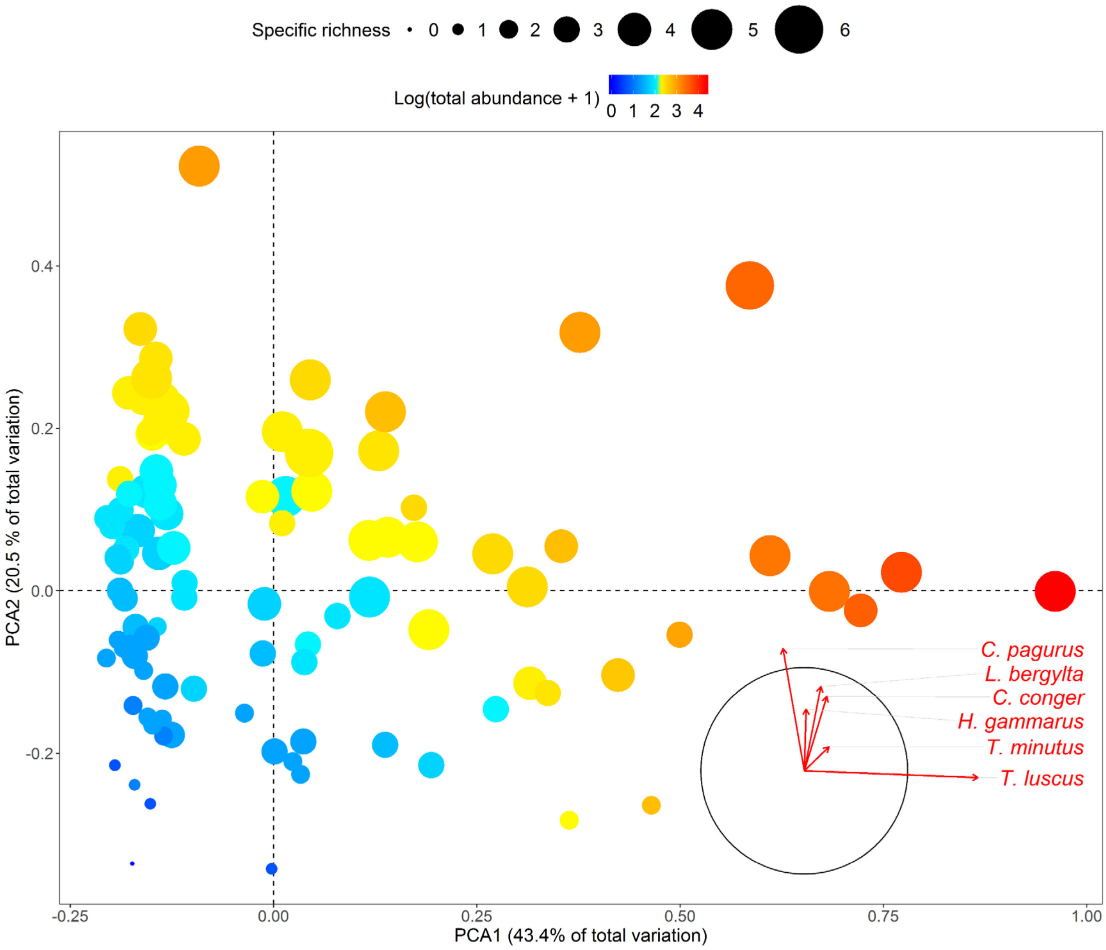

Figure 4: Principal Component Analysis (PCA) of log transformed abundance data for the 6 target species. Each point represents a sample (*i.e.* a concrete mattress during a given campaign). Point size is proportional to species richness and colour indicates total megafauna abundance. Vector overlays show how species abundance correlate to the two first principal components.

3.2 Patterns in community composition / assemblage composition

Out of the 109 fully analysed samples, only 3 did not shelter any individuals of the target species. *T. luscus* was the most abundant species (306 individuals counted in total), followed by *C. pagurus* (167 individuals), *C. conger* (146 individuals), *H. gammarus* and *L. bergylta* (each 121 individuals) and finally *T. minutus* (30 individuals).

The first two PCA axes capture 63.9% of the total variation (Figure 4). Axis PCA1 (43.4% of total variation) is positively correlated to *T. luscus* abundance while the abundance of 4 of the 6 species, mainly *C. pagurus* and then *L. bergylta* and *C. conger* positively correlates with axis PCA 2 (20.5% of total variation; Figure 4). The colour and size codes used to visualise samples on the PCA also illustrates that both specific richness and total abundance per mattress are positively correlated with the two first axes (Figure 4). PCA ordination highlights a large gradient of colonisation among samples, from low-abundance and low-richness samples (in the bottom left) to samples characterised by a high level of colonisation (in the top right of the plot).

Table 3: Environmental variables selected in the RDA as well correlated to the variability in the abundance of the 6 target species colonising concrete mattresses at the Paimpol-Bréhat tidal test site cable (Monte Carlo permutation test in RDA with 999 permutations; $p < 0.05$). High correlation ($r > 0.5$) between environmental variables and the first two RDA axes are highlighted in bold.

			Explained		Correlation	
Environmental variable	F-value	p-value	λ	%	RDA1	RDA2
Cave	13.555	**0.001**	0.21	21%	**0.77**	0.11
Depth	5.837	**0.002**	0.09	9%	-0.47	0.12
% Pebble	3.759	**0.006**	0.06	6%	0.06	**0.83**
Exposure	3.445	**0.008**	0.05	5%	0.34	-0.2
% Boulder	2.857	**0.027**	0.04	4%	**0.64**	-0.2
Holes	2.667	**0.037**	0.04	4%	0.17	**0.61**
Total			0.49	49%		

In the RDA (Figure 5), the environmental variables that best correlate to the variability in the megafauna composition are, in order of importance, number of caves, depth, percentage of pebbles, exposure to current, percentage of boulders, and finally number of holes (Table 3). These 6 variables count for 49% of the explained variability in megafauna composition (Table 3; axis 1 and axis 2 explains 22.15% and 3.7% of the total variation, respectively, Figure 5).

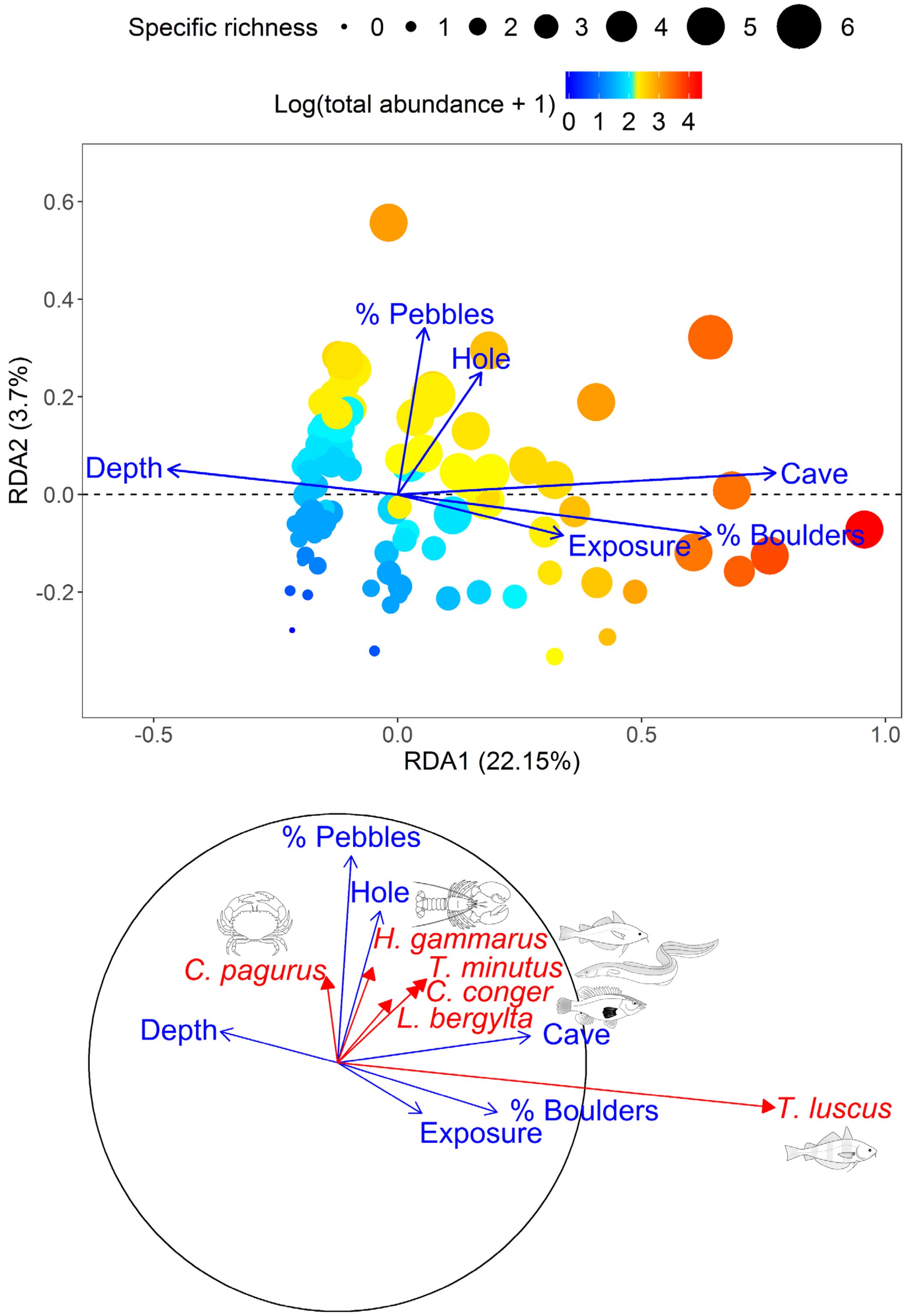

Figure 5: Redundancy analysis (RDA) ordination plots of axes 1 and 2 showing (A) samples (*i.e.* a concrete mattress during a given campaign, points) in relation to environmental variables (blue arrows); and (B) target megafauna species (red arrow) in relation to environmental variables (blue arrows). A different scaling was used for each panel, so environmental variables projection on the RDA should be used to reconcile both parts of the Figure. Axes 1 and 2 together explain 25.85% of the total taxonomic variation. Point size corresponds to associated specific richness and point colour to associated total megafauna abundance.

Results of the RDA (Figure 5) are consistent with those of the PCA (Figure 4). Number of caves present below the mattresses and percentage of boulders correlate positively to RDA axis 1, while number of holes and percentage of pebbles mainly correlate to RDA axis 2 (Figure 5, Table 3).

Note that samples with high abundance of *T. luscus* (to the right of the RDA) are associated with relatively high numbers of caves and percentages of boulders (Figure 5). On the other hand, high abundances of *C. pagurus* and *H. gammarus* occur on mattresses with high numbers of holes and percentage of pebbles. *L. bergylta*, *C. conger* and *T. minutus* are correlated with high number of caves and holes (Figure 5). Finally, samples with low diversity and low total abundance exhibited relatively small numbers of both types of cavity (holes or caves) and low percentages of pebbles and boulders (Figure 5).

3.3 Habitat preference

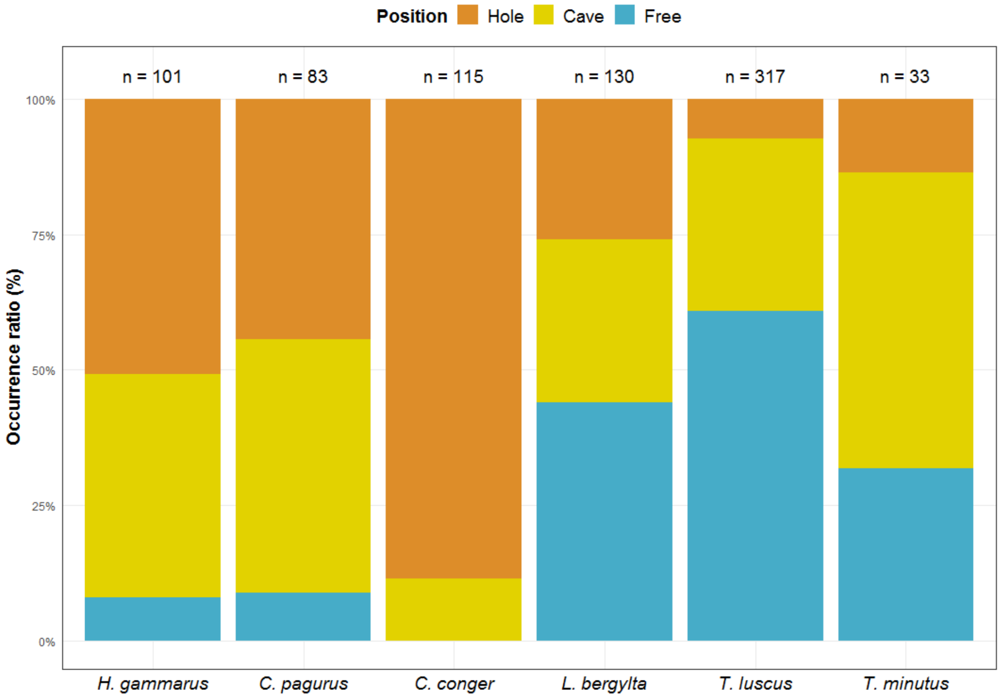

Figure 6: Relative frequency of locations (either inside a “hole”, inside a “cave”, or free-moving out of any cavities) in which the 6 target species were detected, based on video analyses of 149 samples.

According to video data, conger shows a clear preference to hole cavity for sheltering (88,5% of sheltered individuals observed on videos are in holes),whereas the two species of

Trisopterus sheltered more in the cave cavities (for sheltered individuals, respectively 80% for *T. luscus* and 81.4% for *T. minutus* found in caves; Figure 6). Conversely, the edible crab, the European lobster and the Ballan wrasse do not show any clear preference towards any of the two cavity types and were found to shelteri in both (for sheltered individuals, respectively 51.2%, 44.8% and 53.9% are in cave; Figure 6).

4. Discussion

By combining *in situ* visual census by divers and video analysis, our results help to characterise how MRE facilities can enhance benthic megafauna diversity by providing artificial reefs. Specifically, we *i*) characterised the habitat potential of concrete mattresses deployed to anchor an unburied power cable, *ii*) discussed how interactions between the artificial reef and the natural substrate conditions the effectiveness of the 'reef effect' and *iii*) to a lesser extent highlighted some ecological preferences of 6 target species.

4.1 Habitat potential of cable stabilizing structures

ORE structures create additional potential habitat for benthic megafauna, as shown by several studies on colonisation (Langhamer and Wilhelmsson, 2009; Reubens *et al.*, 2011; Wilhelmsson and Langhamer, 2014; Krone *et al.*, 2017). On the subsea power cable of Paimpol-Bréhat, concrete mattresses offer a suitable habitat for large crustaceans and fish, at least for the 6 species targeted during our three-year monitoring. This result corroborates with previous studies that showed that these 6 targeted species are known to be attracted by a number of artificial hard substrates, either associated with MRE facilities (Langhamer and Wilhelmsson, 2009; Reubens *et al.*, 2011; Krone *et al.*, 2017) or with other types of man-made structures (Jensen *et al.*, 1994, 2000b; Charbonnel *et al.*, 2000; Fabi *et al.*, 2004; Santos *et al.*, 2005; Castège *et al.*, 2016). Although we did not quantify their abundance, other species of benthic megafauna (including fish from the Blenniidae or Gobiidae, families and other wrasses species such as *Labrus mixtus* and *Ctenolabrus rupestris*, and crustaceans like *Galathea sp*. and *Necora puber)* were also regularly observed and seem to find a suitable habitat within the mattresses.

In our study, one mattress was on average inhabited by 1 to 2 individuals of *C. conger*, *H. gammarus* and *C. pagurus*, corresponding to species-specific density around 0.1 ind m^{-2}. By extrapolating our results in terms of density, the 120 mattresses stabilising the power cable, contain the following of around 128 *H. gammarus*, 155 *C. conger*, 163 *C. pagurus*, 117 *L. bergylta* and 318 *Trisopterus sp*. inhabiting these structures. These density values were smaller than others reported in the literature. Krone et al. (2017) show that scour protections of a wind turbine foundation (1 m high and around 30 m diameters) in the German Bight (North Sea) were inhabited by several thousand of *C. pagurus* individuals (being ~7 ind m^{-2}). Similar kinds of scour protections were found to host dense schools of *T. luscus* on a Belgian wind farm (Reubens et al., 2011). With an average density estimated at 14 ind m^{-2}, total *T. luscus* population within the windfarm was estimated at about 22,000 individuals. Langhamer and Wilhelmsson (2009) highlighted colonisation of fish and crustacean on wave energy foundations (1 m high and 3 m diameters) in the North Sea, with a mean density of around 5 edible crabs per foundation, being ~0.7 ind m^{-2}. Finally, Jensen et al. (1994) estimated that in the Poole Bay artificial reef, each reef unit (1 m high and 4 m diameters) made up of several blocks (40 x 20 x 20 cm) sheltered between 2 and 3 *H. gammarus* individuals (*i.e.* density up to 0.25 ind m^{-2}). The fact that density values found in our study were smaller than those reported in the literature may be mainly explained by the difference of surrounding natural habitats. In areas with a high dominance of soft sediment (*e.g.* the North sea), the number of shelters provided by natural habitat surrounding artificial reef is very low. Thus, a lot of individuals are constraint to shelter in these artificial reefs, creating a a stronger attraction effect. In our case, natural hard substrate providing shelters are present in the wider area giving other options for this species. Another point can come from the basic shape of concrete mattresses, which are less complex than scour protections or wave-energy foundations.

However, the abundance of the target species were constant during our two-year monitoring. This absence of temporal variation suggests that *i*) colonisation of mattresses by mobile megafauna reached a plateau in less than 2 years after their deployment (first campaign was in June 2015 *i.e.* 2 years after the deployment of the mattresses) and *ii*) target species are permanent, rather than temporary residents of the mattresses. Colonisation of artificial reefs by megafauna has been reported to occur rapidly. Jensen et al. (1994) show that *H. gammarus*, *C. pagurus, T. luscus* and different species of wrasses can colonise artificial reefs within 3 weeks of their deployment. Concerning, the steady occurrence of the target species around the mattresses, these may be explained by their mobility biological traits. Concerning *H. gammarus*, two modes of behaviour may exist: a mobile phase, with migration between different reefs, and a territorial phase where lobster individuals stay in close proximity to a chosen site/shelter (Jensen *et al.*, 1994). *H. gammarus* can be highly loyal to its refuge, as showed by Jensen et al. (1994): 21% of lobsters caught on a reef unit stayed on it for more than 100 days. *Labrus bergylta* and other *Labridae* are also territorial species dwelling in the vicinity of an identified reef unit (Jensen *et al.*, 1994; Villegas-Ríos *et al.*, 2013). Results from a mark-recapture programme suggest that a wide proportion of *Trisopterus luscus* individuals are bound to the same artificial reef units, which serve as a “home reef” (Fowler *et al.*, 1999).

The degree of colonisation of individual mattresses appears highly dependent on the number and type of available shelters. Both these features condition how an artificial reef can artificially enhance the carrying capacity of the local environment (Bohnsack, 1989; Eggleston *et al.*, 1992; Pickering and Whitmarsh, 1997). As each species exhibit specific habitat preferences, the variety of shelters also largely explains the species composition of artificial reefs (Smith *et al.*, 1979; Chandler *et al.*, 1985; Anderson *et al.*, 1989; Beets and Hixon, 1994;

Pickering and Whitmarsh, 1997). Optimisation of MRE facilities through basic designs (*e.g.* with creation of manufactured holes of different sizes) has been shown to enhance their attractivity for benthic species (Langhamer and Wilhelmsson, 2009). The concrete mattresses of Paimpol-Bréhat were not designed to effectively provide additional habitat for marine fauna but to stabilise the submarine power cable and prevent fishing gear hooking. The two types of shelters identified, holes and caves host different groups of species. While *L. bergylta* shelters in both type of cavities, *Trisopterus sp.* show a clear habitat preference for caves. *Trisopterus sp*. are known to colonise rocky habitats with numerous and wide cavities such as caves, crevasses or wrecks for shelters against tidal current (Jensen *et al.*, 1994; Krone *et al.*, 2013b). Consequently, they shelter to a limited extent in holes and favour wide caves that can host a whole school. This schooling behaviour conditions the species preference for larger caves, as highlighted by the high correlation between *T. luscus* abundance and the number of caves available below concrete mattresses. Our results also highlight that holes constitute the preferred habitat for *C. conger*. This solitary species is known to shelter in narrow cavities, the holes of the mattress constitute narrow and linear shelters which fit perfectly the shape of the adults, compared to the caves which are too wide. Adult European lobster individuals use physical shelters to avoid predators and being swept by strong tidal currents (Addison and Lovewell, 1991). Given the high tidal currents that can occur at the Paimpol-Bréhat tidal site (up to 2.5 m s^{-1}), mattresses thus provide an adequate shelter to lobsters. Lobsters typically select dark shelters that fit their body size closely (sometimes with physical contact; Wahle et al., 2013). Although *H. gammarus* shows a subtle preference for holes, this species is found in both cavities in equal proportions, suggesting that narrow caves can also be appealing to lobsters. Less information is available concerning sheltering behaviour and preferences of *C. pagurus* but the species has been reported to compete with lobsters for shelter so it is likely to display similar habitat preferences (Richards and Cobb, 1986). Contrastingly to the European lobster, *C. pagurus* individuals are known to escape predators by rapidly burrowing themselves in sandy habitats (Hudon and Lamarche, 1989). This burrowing behaviour may allow *C. pagurus* to colonise a wider variety of mattresses relative to lobster, including those exhibiting high proportions of soft sediments.

Among our target species, three groups can be discerned based on their use of the habitats created by the artificial reef: (*i*) solitary and nocturnal species found in shelters during the day, such as *C. conger*, *H. gammarus* and *C. pagurus*; (*ii*) solitary and diurnal species found in shelters during the night, such as *L. bergylta*; and (*iii*) gregarious species that display a nocturnal activity, such as the two species of *Trisopterus*. These different diel behaviours suggest a possible shift in the occupation of mattresses, as observed for other artificial (Santos *et al.*, 2002) and natural reefs (Mallet *et al.*, 2016; Myers *et al.*, 2016). This day/night shift may introduce a bias in our counting procedure. Considering that all diving surveys occurred during the daytime, counts of diurnal species may have been underestimated as these might had temporarily left the mattresses and their close proximity during their period of activity. On the contrary, the counts of nocturnal species were more accurate because individuals are motionless within the different cavities of the mattresses during the day. Furthermore, *Trisopterus sp.* proceeds to tidal migration in addition to day/night cycle, which may be another source of bias when evaluating its occupancy rate. Schools of *T. luscus* are found to be more congregated and closer to artificial reef units during high current speeds (>0.3 m s^{-1}), and to be more dispersed and further from the reef during low current speeds (<0.1 m s^{-1}; Fowler et al., 1999). Considering that all diving surveys occurred during slack tides (*i.e.* low speed currents), the abundance of *Trisopterus sp.* were possibly underestimated. In order to study in greater detail these community changes at the mattress scale resulting from the activity rhythms of the different megafauna species, the use of continuous video recording over several tide and day/night cycles could be useful (Weiss *et al.*, 2009; Aguzzi *et al.*, 2013; Mallet *et al.*, 2016).

4.2 Interaction of artificial reef with local environment

Differences in megafauna colonisation originate from the variability in local environmental conditions around each mattress: interaction between mattress and heterogeneous natural bottom directly influences the number and types of cavities available. Previous studies showed that environmental variables, such as bottom types, depth, hydrodynamic conditions, sediment dynamics or distance to natural reef, significantly impact the colonisation of artificial reefs (Ambrose and Swarbrick, 1989; Bohnsack *et al.*, 1991; Bombace *et al.*, 1994; Foster *et al.*, 1994; Godoy *et al.*, 2002; Noh *et al.*, 2017).

Our results reveal that the higher the proportion of boulders, which is positively correlated with high hydrodynamic conditions, the higher the number of caves. Indeed, the presence of boulders creates an irregular seafloor topography and prevent the edges of mattress from fitting flush with it, thus creating overhanging space under the mattresses *i.e.* caves. From this point of view, Alexander (2013) showed that the shape of the seafloor underneath a flat concrete block plays an important role as it directly impacts the volume that could be colonised. On the other hand, the number of holes available increases with the percentage of pebbles. However, in the presence of pebbles, caves do not form under the mattresses, but holes remain as open cavities between the adjacent concrete blocks. Although less colonised than mattress with caves, these holes provide shelters to *C. pagurus*, *H. Gammarus*, *C. conger* and *L. bergylta*. Finally, when the seafloor is dominated by sand and shell debris, the number of cavities is very low because, *i*) the flatness of the seafloor prevents the creation of caves, and *ii*) accretion of sand and shell debris caused by the presence of the mattress often lead to smothering the holes reducing their availability. To summarise, holes constitute a narrow and deep cavity inherent to the way mattress is manufactured, but its availability can be impacted by the degree that it is filled by the sediment, which depends on local sediment dynamics. Caves constitute more or less narrow cavities with large openings, which only originate from the interaction between mattress and the local topography of the site. Here, we only differentiated these two types of cavities, but a more precise description of their physical features (*e.g.* through the use of quantitative criteria such as depth, size and shape of the entrance) could allow a better comprehension of mattress reef properties (Alexander, 2011).

Colonising an artificial reef also depends on distance to the close presence of natural reefs (Jessee *et al.*, 1985; Ambrose and Swarbrick, 1989). The closer to existing natural hard habitat, the higher the probability for artificial reefs to attract transient species (Campos and Gamboa, 1989; Potts and Hulbert, 1994). When deployed closely to existing natural reefs, artificial reefs essentially extend the amount of hard habitat with direct benefits for hard-substrate species recruitment (Danner *et al.*, 1994). In our case, it is unlikely that distance to natural reefs influences mobile megafauna composition on mattresses. Since mattresses are installed between large shallow rocky shelves, their distance to natural rocky habitat are considered as low (less than 1km) and therefore relatively homogeneous. Computing precise distances between each mattress and the nearest natural reef would be difficult due to the very complex shape of the 10m and 5m isobaths in this area.

Finally, these structures associated with submarine power cables are not a classic artificial reef in the sense that the current transiting through cables generates electromagnetic fields. A lack of knowledge still exists concerning the impact of these anthropogenic electromagnetic fields on marine fauna (Taormina *et al.*, 2018 ; see Chapter 1). This can potentially impact species capable of electroreception and/or magnetoreception through effects on predator/prey interactions, avoidance/attraction behavior, navigation/orientation capabilities or induce physiological and developmental effects (Copping *et al.*, 2016). Over the course of this study, no electric current transited through the cable and the mattresses thus acted as a classic artificial reef, but further investigations should be conducted once electrical current passes through.

5. Conclusions

Although the concrete mattresses deployed to anchor the submarine power cable were not specifically designed to act as a refuge for marine fauna, a three-year monitoring study (*both in situ* and using videos) shows that they offer a suitable and stable habitat for at least 6 species . Interactions between local seafloor and hydrodynamic characteristics (substratum type, topography, exposition to current *etc.*) and artificial reef units directly condition the variety and the availability of shelters. In our study, these two factors, *i.e.* shelters shape and availability, largely determine the degree of colonisation by mobile megafauna. Consequently, in order to characterise in detail the habitat potential for megafauna of MRE structures, it is critical to both, optimise the design of the artificial structures, and anticipate how they will interact with local environmental site characteristics .

Acknowledgements

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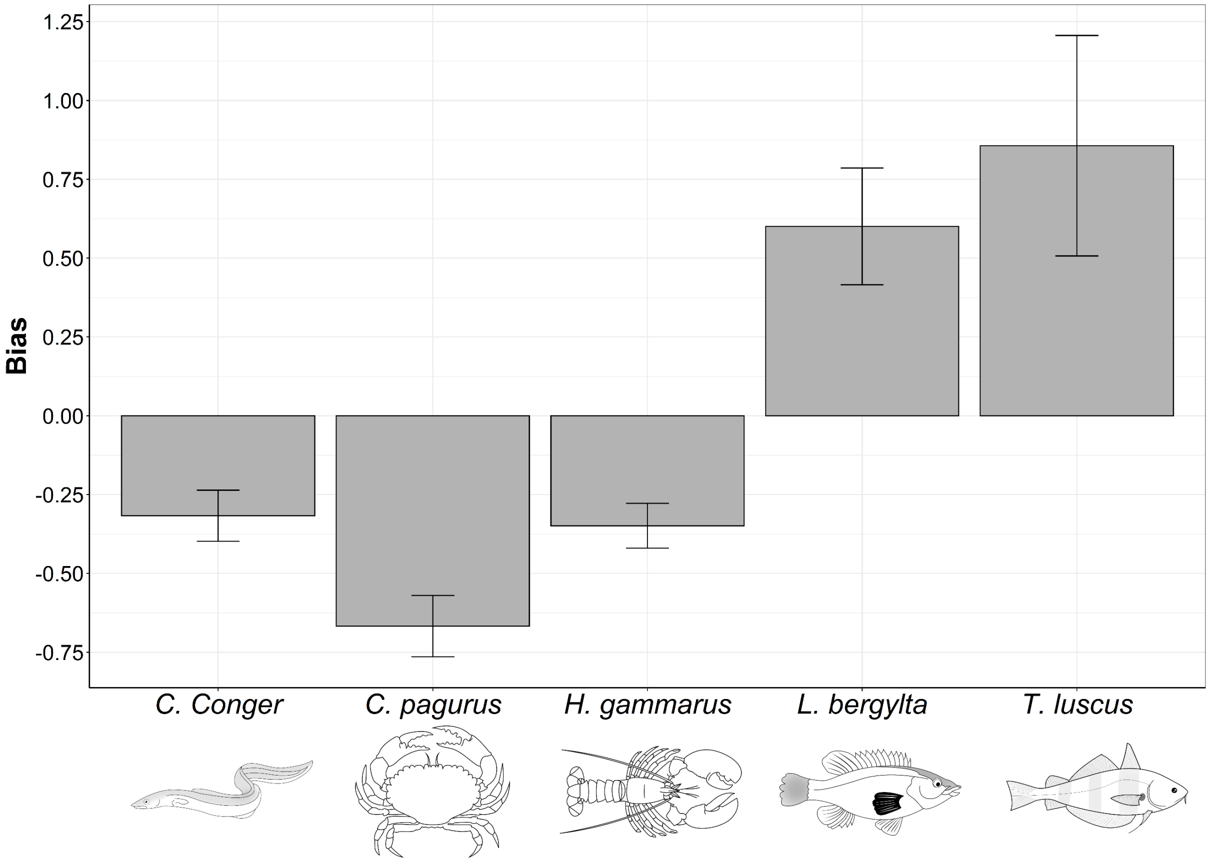
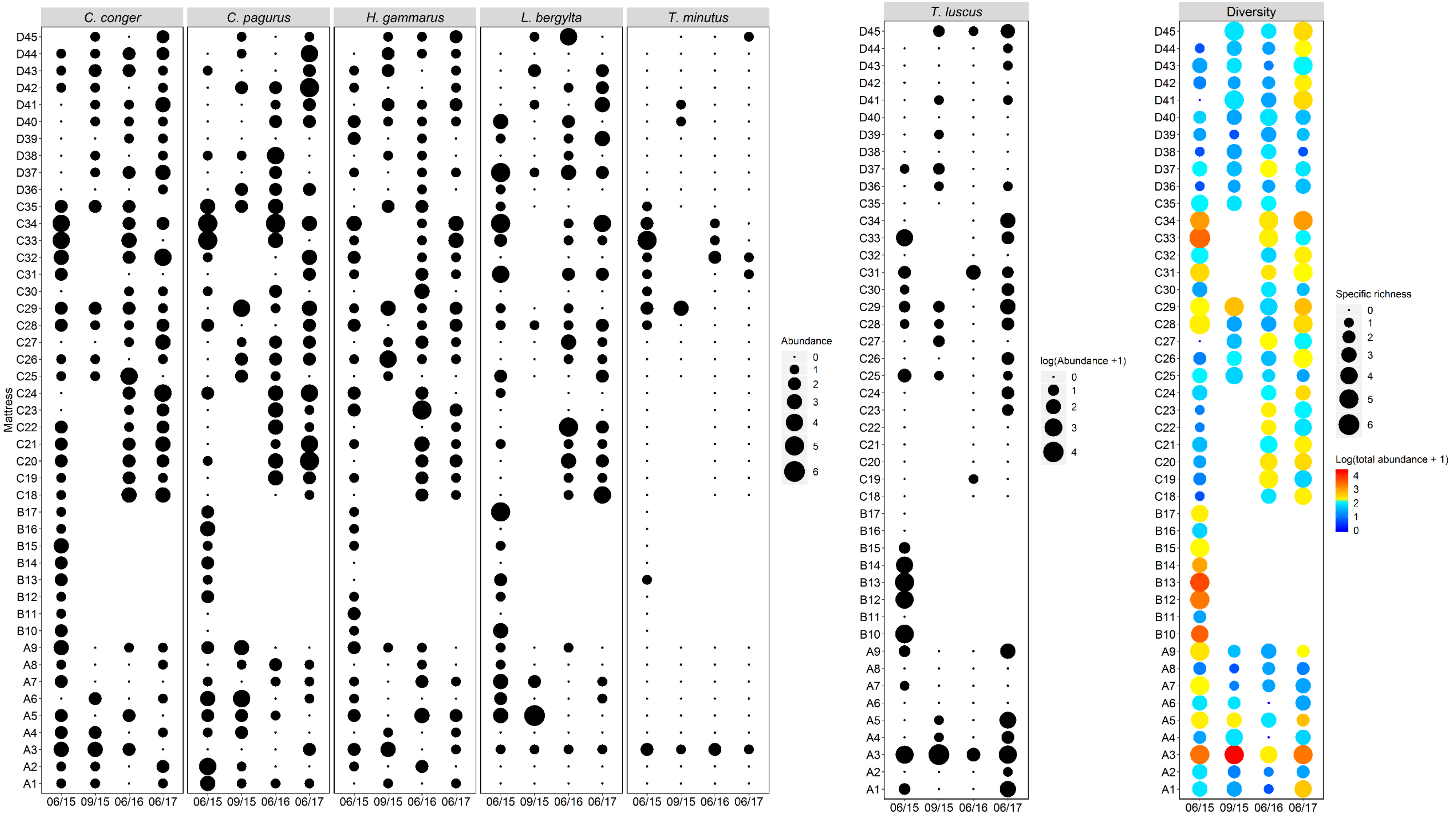
Supplementary information 1: Megafauna species count bias between visual census by divers and video analysis. The bias was calculated for each sample (*i.e.* a concrete mattress during a given campaign) as a ratio with the following formula: Bias $_{i,j}$ = (count of the species *j* on the sample *i* using video analysis - count of the species *j* on the sample *i* using visual sensus) / mean count per sample of the species *i* (*i.e.* mean presented in Table 2) . A bias inferior to 0 indicates that video analysis underestimates the count of the species, while a bias superior to 0 indicates that visual census underestimates the count of the species. Results for counts of *T. minutus* were not represented here. These results are based on the work of Dufournaud (2018).



Supplementary information 2: Variation of the colonisation of each concrete mattress during the four campaigns: for each sample (*i.e.* a concrete mattress during a given campaign) the abundance of each of the 6 target species (left) as well as the specific richness and the total abundance (right). Abundance of *T. luscus* and total abundance were expressed using a logarithmic transformation to create a useable graphic representation.



Chapter 5

Impact of magnetic fields generated by AC/DC submarine power cables on the behaviour of juvenile European lobster (*Homarus gammarus*)

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Abstract

The number of submarine power cables using either direct or alternating current is expected to increase drastically in coming decades. Data concerning the impact of magnetic fields generated by these cables on marine invertebrates are scarce. In this context, the aim of this study was to explore the potential impact of anthropogenic static and time-varying magnetic fields on the behaviour of recently settled juvenile European lobsters (*Homarus gammarus*) using two different behavioural assays. Here we showed that juvenile lobsters did not exhibit any change of behaviour when submitted to an artificial magnetic field gradient (maximum intensity of 200 µT) compared to non-exposed lobsters in the ambient magnetic field. Additionally, no influence was noted on either the lobsters' ability to find shelter or modified their exploratory behaviour after one week of exposure to anthropogenic magnetic fields (225 ± 5 µT) which remained similar to those observed in control individuals. It appears that static and time-varying anthropogenic magnetic fields, at these intensities, do not significantly impact the behaviour of juvenile European lobsters. Nevertheless, to form a complete picture for this biological model, further studies are needed on the other life stages as they may respond differently.

Keywords

Anthropogenic impact ; behaviour ; *Homarus gammarus* ; magnetic field ; submarine power cable

1. Introduction

Submarine power cables are used worldwide for numerous applications: to connect autonomous grids, to supply power to islands, marine platforms or subsea observatories, and to carry power generated by marine renewable energy installations (offshore wind farms, tidal and wave turbines). In 2015, almost 8,000 km of HVDC (High Voltage Direct-Current) cables were present on the seabed worldwide, 70% of which were in European waters (Ardelean and Minnebo, 2015). The number of submarine power cables, using either direct (DC) or alternating current (AC), is expected to increase dramatically in the coming decades. This rise is in part due to an increase in grids connecting islands and archipelagos, and also to the development of marine renewable energy projects. Indeed, marine renewable energy development is a possible solution to the global increasing demand for renewable energy in order to combat climate change (Copping *et al.*, 2014).

Submarine power cables, like any other man-made installation or human activity at sea may temporarily or permanently impact the marine life and habitats through habitat damage or loss, noise, chemical pollution, heat emission, risk of entanglement, introduction of artificial substrates and the creation of reserve effects (Taormina *et al.*, 2018 ; see Chapter 1). Among all these potential environmental incidences, one of the main concerns is related to the emission of electromagnetic fields (EMF), which are generated by the electric current flowing through power cables. EMF can be divided into electric fields (measured in volts per meter, $V.m^{-1}$) and magnetic fields (MF, measured in µT). EMF characteristics vary greatly as a function of the cable type (distance between conductors, load balance between the three phases in the cable, *etc.*) just as much as the power and type of current, *i.e.* DC *vs.* AC (DC producing a static MF and AC a time-varying MF; Copping et al., 2016; Ohman et al., 2007). Electric fields are generally confined inside cables because of the armouring whereas MF are not. The MF strength increases with current flow and rapidly declines with distance from the cable

(Normandeau Associates Inc. *et al.*, 2011). The MF produced at the surface of the cable by either DC or AC cables can be highly heterogeneous, with intensity ranging from 1 to 3,200 µT (Bochert and Zettler, 2006; Normandeau Associates Inc. *et al.*, 2011).

Numerous marine species harness the Earth's geomagnetic field for orientation and migration, including elasmobranchs (rays and sharks), teleosts, mammals, turtles, mollusks and crustaceans (Kirschvink, 1997; Willows, 1999; Walker *et al.*, 2002; Lohmann *et al.*, 2008; Durif *et al.*, 2013; Lohmann and Ernst, 2014; Cresci *et al.*, 2017). Consequently, anthropogenic MF can potentially impact species capable of magnetoreception through effects on predator/prey interactions, avoidance/attraction behaviours, navigation/orientation capabilities or induced physiological and developmental effects (Copping *et al.*, 2016). Data concerning anthropogenic MF impacts on invertebrates are scarce, and existing studies have reported minor or non-significant impact of anthropogenic EMF (Bochert and Zettler, 2004; Woodruff *et al.*, 2012, 2013, Love *et al.*, 2015, 2017b; Hutchison *et al.*, 2018).

The European lobster (*Homarus gammarus*) is widely distributed along the continental shelf in the North-East Atlantic from Morocco to near the Arctic Circle. This species is heavily exploited in some areas and represents great economic value. In 2016, the global catch was estimated at 4,713 t (Source = FAO FishStat). European lobsters show a preference for rocky habitats which provide shelters (Childress and Jury, 2007). Consequently they are frequently observed within artificial reefs, including those related to marine renewable energy installations and their submarine power cables (Krone *et al.*, 2013b; Hooper and Austen, 2014). This behavioural trait can lead to extended MF exposures which may induce stress for the lobster. Although two experimental studies showed low impact of EMF exposure on the behavioural activity of a similar species, the American lobster (*Homarus americanus*; Hutchison et al., 2018; Woodruff et al., 2013), no study has focused on the European lobster so far. Furthermore, no attention has been paid to early developmental stages of either of these species, which can be assumed to be more vulnerable to disturbances than adult specimens.

In this context, the aim of this study was to explore the potential impact of anthropogenic MF produced by either AC or DC submarine power cables on the behaviour of recently settled European lobster juveniles. To address this question, we studied using two different behavioural assays (*i*) the avoidance/attraction effect of anthropogenic MF and (*ii*) the effect of an extended MF exposure on their exploratory behaviour and ability to find a shelter.

2. Methods

2.1 Specimens' origin and maintenance

European lobster juveniles (N=203) at development stages VI-VIII were used in this study. The offspring came from six berried females purchased from a local lobster dealer, close to Bergen and transferred May 2018 to the Institute of Marine Research Austevoll station (N60°05′15.36″, E5°15′54″). Hatching followed the set-up described by Agnalt et al. (2017), although the filtrated seawater was from 160 m depth (showing a constant salinity of 34.7 ppt) and heated to a temperature of 14 °C. Once reaching stage IV, the post-larvae were transferred and raised individually in single compartments. The compartments were maintained inside a tank (1.5x1.5 m with 1 m depth of water with a flow of 30 L min^{-1}) with seawater at 14 °C in continuous flow at a 16:8 h light:dark cycle. The lobsters were fed daily with dry feed OTOHIME C2 (PTC Japan) or frozen shrimp. The postlarva stage IV, which still had a swimming behaviour, continued their growth to stage V (*i.e.* juvenile), and then became fully benthic. To induce normal claw development (Govind and Pearce, 1989), grained sand was added to each individual unit at stage IV and V. Only juveniles with two intact claws were used in these experiments. Exposure treatment and testing described below took place in a separate room than the one used to rear the lobster juveniles. This experiment was carried out following The Code of Ethics of the World Medical Association for animal experiments.

2.2 Helmholtz coils

To produce artificial magnetic fields, Helmholtz coils designed by MAPPEM Geophysics© (http://www.mappem-geophysics.com/) were used. The coil (1.5x1.5x1.0 m) was designed to produce time varying (*i.e.* AC) or static (*i.e.* DC) magnetic fields with intensities reaching about 230 µT, which is comparable to those produced by high power submarine cables (based on data calculated by the French transmission system operator RTE, 200 µT corresponds to the intensity found at 1 m of a 1000 A DC power cable and at less than 50 cm for a 780 A AC power cable). The coils created (*i*) an area of homogeneous magnetic fields in the center, and (*ii*) an area of decreasing magnetic field gradient in the periphery (SI 1).

2.3 Avoidance/attraction test

In order to study the avoidance/attraction potential of anthropogenic MF on juvenile lobsters, individuals were tested under three MF gradient configurations: (*i*) with a time varying MF gradient (hereafter called AC MF, N=30), (*ii*) with a static MF gradient (hereafter called DC MF, N=31) and (*iii*) with ambient MF (*i.e.* control treatment, N=31).

Long rectangular raceways made with white opaque walls (125x14x7 cm) were placed across the MF intensity gradient area, either AC or DC (Figure 1.A). For control treatment, the coil was turned off, resulting in the absence of any MF gradient inside the raceway. Within the raceway, four different zones were defined *a posteriori* (Figure 1.A): High Magnetic Field 1 (HMF1), High Magnetic Field 2 (HMF2), Low Magnetic Field 3 (LMF3) and Low Magnetic Field 4 (LMF4). Each raceway was filled with 3 cm of seawater (at 12 ± 1 °C; the seawater was replaced between each trial). To observe shelter seeking behaviour, two grey and opaque half-cylinder shelters (2.50x7.50x1.25 cm), open on both sides, were positioned at each end of the raceway (at 2.5 cm from the wall; Figure 1.A). Thus, one shelter was positioned in the high MF end of the raceway, and the other one in the low MF end of the raceway. Although *H. gammarus* is a nocturnal animal, more active during the night, the test was performed with day-light conditions, in order to stimulate their sheltering behaviour. The luminosity intensity was measured at 5 different points along the raceway (SI 2) using a spectrophotometer (Ocean Optics FLAME-S-UV-VIS).

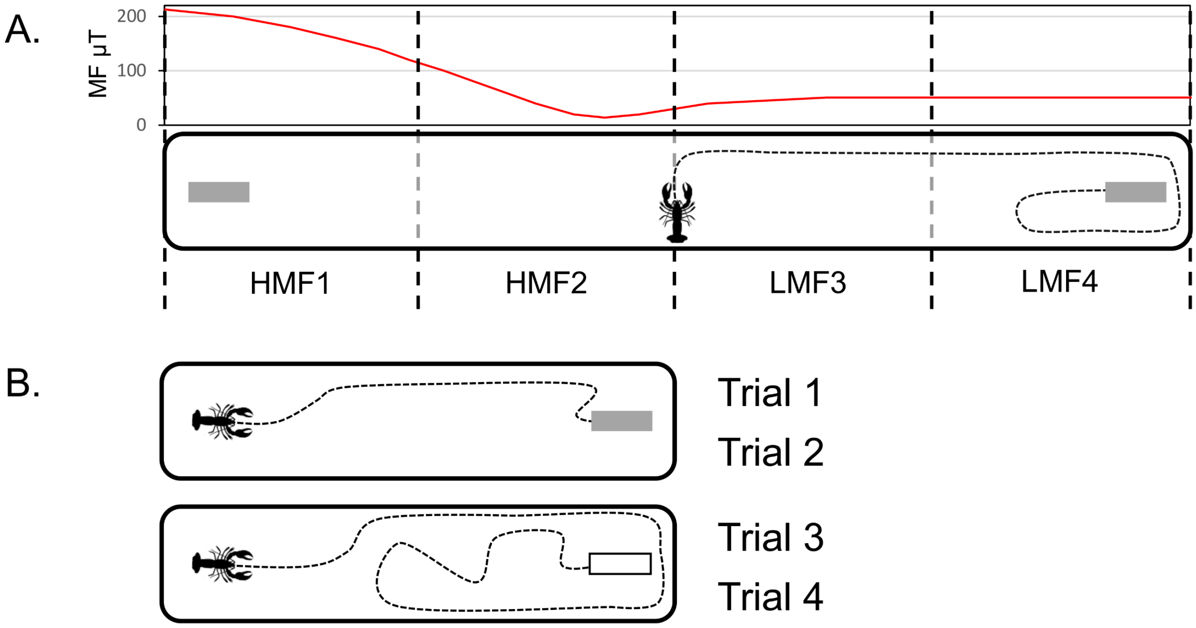

Figure 1: Experimental setup (A) Avoidance/attraction test: raceway of 125x14x7 cm with two half-cylinder shelters were used at each side. Zones were labelled into 4 different zones depending on the intensity of the magnetic field: HMF1: High Magnetic Field 1, HMF2: High Magnetic Field 2, LMF3: Low Magnetic Field 3, LMF4: Low Magnetic Field 4 ; the magnetic field gradient generated for the AC and DC treatments is shown at the top ; (B) Post-exposure test: raceway of 66x14x7 cm was used, one shelter was positioned at one end, four consecutive trials were performed, trials 1 and 2 with grey opaque shelter, trials 3 and 4 with white opaque shelter. The Dotted line represent possible paths of movement of lobsters. All figures to scale except lobster representation.

The behavioural tests were carried out by carefully placing each lobster inside a circular ring (5x5 cm) at the center of the raceway (mid distance between the two shelters). After 10 minutes of acclimation, the lobster was released by removing the ring, and the animal's behaviour was then recorded over a 45-minute period with a GoPro hero 5 Black (1080 p, 25 fps) placed above the raceway. No one was present in the experimental room during video-tracking of lobster behaviour. The lobsters used in the experiments had never been used in prior experimentation. Each individual (N=92) was tested once and treatments were randomized. Between 16 and 32 different individuals were tested per day. All experiments were undertaken between 11 am and 4 pm. The carapace length of each lobster was measured after the test.

Each video was analyzed with the video tracking software Ethovision XT (Noldus ©). From each footage, we extracted *i*) the time the lobster took to find the shelter (in minutes; when the lobster did not enter any shelter, a maximum time was assigned *i.e.* 45 mins), *ii*) the time spent inside the two different shelters and the four raceway zones when outside shelters (as a percentage), *iii*) the total distance travelled overall and per zone (distances are expressed in Carapace Length CL, in order to avoid any bias of the specimens' size on the distance travelled), *iv*) the mean velocity in overall and per zone (in CL s^{-1}) and *v*) the movement/immobility ratio (*i.e.*, when outside a shelter, the ratio between the time when the lobster moved and the total time) overall and per each zone.

2.4 Exposure treatments

To study juvenile lobster exploratory and shelter seeking behaviour after MF exposure, 111 individuals were exposed to the following treatments for one week prior to the test: (*i*) time varying MF (hereafter called AC MF, N=38, MF = 225 ± 5 µT), (*ii*) static MF (hereafter called DC MF, N=35, MF = 225 ± 5 µT) or (*iii*) ambient MF (*i.e.* control treatment, N=38). During the exposure, lobsters were maintained in separate units (7.0x3.5x7.0 cm) within a tank (40x30x10 cm) which was placed in a homogeneous MF area. The tank was filled with 8 cm of seawater at 12 ± 1°C in current flow (0.85 L min^{-1}). The room was submitted to a 9:15 h light:dark cycle, and the lobsters were fed daily with dried food or frozen shrimp alternately.

After one week of exposure, the ability to find a shelter of each lobster was assessed following the method described by Cresci et al. (2018). To do so, rectangular raceways with white opaque walls (66x14x7 cm) were used (Figure 1.B). Raceways were placed in the MF homogeneous area used to exposure, and filled with 3 cm of seawater (at 12 ± 1 °C; the water was entirely replaced between each trial). A half-cylinder shelter (2.50x7.50x1.25 cm) was positioned at one end of the raceway. As for the attraction/avoidance test, the test was performed with day-light conditions, in order to stimulate their sheltering behaviour.

For each trial, one lobster was released at the end of the raceway (opposite the shelter) and the behaviour of the animal was recorded for 30 minutes with a GoPro hero 5 Black (1080 p, 25 fps) placed above. The lobsters used in the experiments had never been tested before. To study their learning abilities, each lobster performed 4 consecutive trials of 30 minutes using two different colored opaque shelters open on both sides: grey shelters for the first two trials and white shelters for the last two trials (Figure 1.B). The lobsters used in this experiment were different from those used in the "attraction/avoidance test". Treatments were randomized for each individual. Between 6 and 8 different individuals per day were tested and all experiments were achieved between 11 am and 4 pm. The carapace length of each lobster (N=111) was measured after the trials.

Each video was analyzed *posteriori* with the video tracking software Ethovision (Noldus ©). We extracted *i*) the time the lobster used to find the shelter (in min; when the lobster did not enter the shelter, the maximum time was assigned *i.e.* 30 min), *ii*) the total distance travelled (for the same reasons than for the avoidance/attraction test, the distances are expressed in Carapace Length CL), *iii*) the mean velocity (in CL s^{-1}) and *iv*) the activity ratio (*i.e.*, when outside the shelter, the ratio between the time where the lobster moves and the total time).

2.5 Statistical analysis

Results are given as mean ± standard error. We tested the data for normality assumption using the Shapiro-Wilk test as well as variance homoscedasticity by examining graphed residuals. When possible, two-way repeated-measures ANOVA (RM-ANOVA) with the intra-subject factor "zone" (for avoidance/attraction test) or "trial" (for exposure experiment) and the inter-subject factor "treatment", were used to study the different behaviour (*i.e.* the time to find a shelter, the total distance travelled, the mean velocity and the activity ratio) of the lobsters. For each RM-ANOVA, variance-covariance matrix sphericity was verified using Mauchly test. When significant, p-values were re-calculated using the Greenhouse-Geisser correction. Non-parametric rank test of Kruskal-Wallis was used when the use of RM-ANOVA was not possible. Finally, to compare the proportions of time spent in the different shelters or in the different zones of the raceway, permutational analysis of variance (PERMANOVA) with euclidian distance was applied. The statistical analyses were performed using RStudio (V 3.4.3; RStudio Team, 2015) with the packages vegan (Oksanen et al., 2018), lme4 (Bates et al., 2015), Rmisc (Hope, 2013) and ggplot2 (Wickham, 2016).

3. Results

3.1 Avoidance/attraction test

Once released, lobsters typically headed in one direction until they made contact with the side of the raceway, then, progressed exploring the area in either direction by feeling the raceway wall using their antennae. Once lobsters perceived or made physical contact with one of the shelters, 68.5% of them entered it and remained there until the end of the test. Lobsters which never entered a shelter during the test, usually spent part of their time exploring the raceway, before staying immobile in a corner of the raceway until the end of the test.

All treatments taken together, 87% of the lobsters entered at least one of the shelters, the first entrance occurred on average 13.8 min after the beginning of the test. This time did not

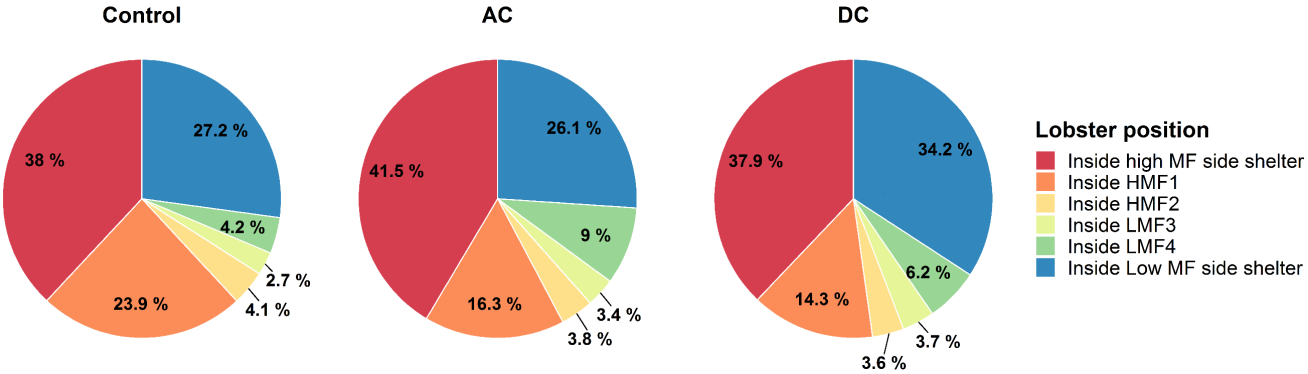

Figure 2: Effect of the magnetic field gradient on attraction/avoidance behavior of the European lobster (*Homarus gammarus*). Percentage of time spent in the two different shelters and in the different zones of the raceway. HMF1: High Magnetic Field 1, HMF2: High Magnetic Field 2, LMF3: Low Magnetic Field 3, LMF4: Low Magnetic Field 4. The three treatments were Control: coil off (n=31); AC: coil on in alternative current mode (n=30); DC: coil on in continuous current mode (n=31).

differ significantly between treatments (14.4 ± 2.7 min for Control; 14.4 ± 2.6 min for AC and 12.6 ± 2.7 min for DC; Kruskal-Wallis test P=0.96).

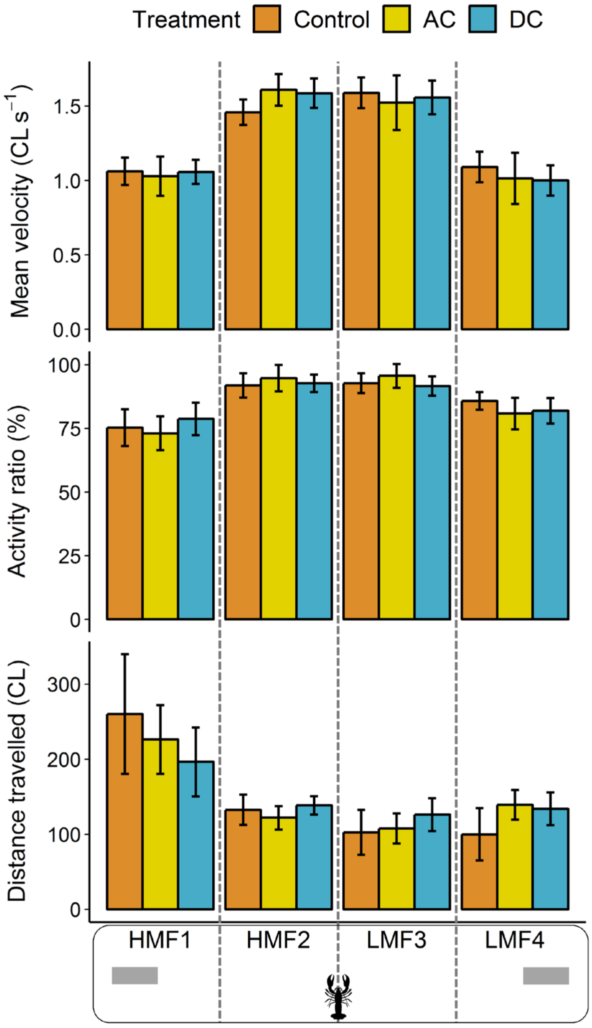

Figure 3: Effect of the magnetic field gradient on the behavior of the European lobster (*Homarus gammarus)*. CL: Carapace length of lobster, HMF1: High Magnetic Field 1, HMF2: High Magnetic Field 2, LMF3: Low Magnetic Field 3, LMF4: Low Magnetic Field 4. Error bars represent the 95% confidence-interval corrected for interindividual variability (Loftus and Masson, 1994). The three treatments were Control: coil off (n=31); AC: coil on in alternative current mode (n=30); DC: coil on in continuous current mode (n=31).

In all three treatments, lobsters spent more time inside the shelters (68 ± 3.5 % of the time) than outside. Across all treatments, lobsters spent more time in the high MF-shelter end (38.9 ± 4.5 % of its time, Figure 2) than in the low MF-shelter (29 ± 4.2 % of its time, Figure 2). When outside shelter, in all treatments, lobsters spent twice as long in the high MF end of the raceway (*i.e.* zones HMF1 and HMF2, 21.9 ± 2.9 % of time outside shelters) than in the low MF end of the raceway (*i.e.* zone LMF3 and LMF4, 9.7 ± 1.4 % of time outside shelters; Figure

2). The proportion of time spent in the low MF side shelters, high MF side shelter and in the different area did not change across treatments (PERMANOVA, df=2, pseudo-F=0.39, P=0.82, Figure 2).

Within the entire raceway and within each zone of the raceway, the total distance travelled, the mean velocity and the activity ratio of the lobsters did not differ significantly between the three treatments (RM-ANOVA P>0.05 in all cases; Table 1; Figure 3).

Table 1: Summary of the different two-way ANOVAs for repeated measures on the effects of the treatment and the interaction of treatment and zone on the different behaviour of the European lobster (*Homarus gammarus*) for the attraction/avoidance test.

Effect	df	F	p.value
Mean velocity			
Treatment	89	0.15	0.86
Treatment:Zone	267	0.99	0.43
Distance travelled			
Treatment	89	1.17	0.31
Treatment:Zone	267	1.59	0.15
Activity ratio			
Treatment	89	0.75	0.48
Treatment:Zone	267	1.01	0.42

3.2 Exposure test

During the week of exposure, no mortality occurred.

Typical behaviour of lobsters during this test was similar to that observed during the avoidance/attraction test. When released, lobsters chose a direction until they made contact with the wall of the raceway, then, explored the raceway using their antennae. Once they found the shelter, they usually entered and remained there for the rest of the test. When considering all trials and all treatments together, 71.5% of the lobsters entered the shelter at least once, and among them, 77.2% did not get out for the rest of the trial after the first entrance. When a lobster did not enter a shelter, it usually spent part of its time exploring the aquarium, and eventually, remained motionless until the end of the test.

Across all treatments, a larger number of lobsters entered the grey shelter (*i.e.* trials 1 and 2, respectively 93.6% and 95.7% of the lobsters had entered the shelter) than the white shelter (*i.e.* trials 3 and 4, respectively 46.8% and 53.2% of the lobsters had entered the shelter). They also took less time to enter the grey shelter (5.6 ± 0.8 min and 4.5 ± 0.7 min for trial 1 and 2) than the white shelter (21.5 ± 1.1 min and 19.9 ± 1.2 min for trial 3 and 4; Figure 4).

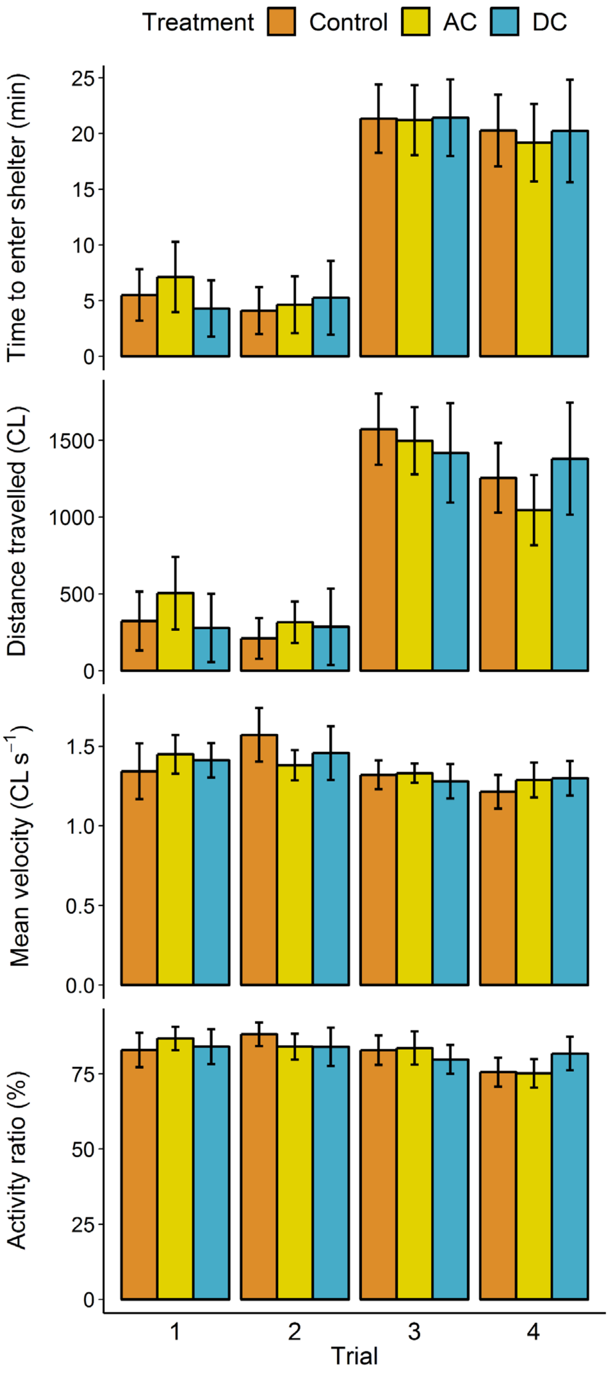

Figure 4: Effect of 1-week exposure to different magnetic fields on the behavior of the European lobster (*Homarus gammarus*) during four consecutive trials. Trials 1 and 2 were with a grey opaque shelter, trials 3 and 4 with a white opaque shelter. CL: Carapace length of lobster , Error bars represent the 95% confidence-interval corrected for interindividual variability (Loftus and Masson, 1994). The three treatments were Control: coil off (n=38); AC: coil on in alternative current mode (n=38); DC: coil on in continuous current mode (n=35).

All trials taken together and within each trial, the time to enter the shelter did not significantly change between treatments (RM-ANOVA P>0.05; Table 2; Figure 4). In the same way, the total distance travelled, the mean velocity and the movement/immobility ratio of the lobsters did not differ significantly between the three treatments (RM-ANOVA P>0.05 in all cases; Table 2; Figure 4).

Table 2: Summary of the different two-way ANOVAs for repeated measures on the effects of the treatment and the interaction of treatment and trial on the different behaviour of the European lobster (Homarus gammarus) after 1-week exposure.

Effect	df	F	p.value
Time to enter shelter			
Treatment	91	1.20	0.30
Treatment:Trial	273	0.39	0.84
Mean velocity			
Treatment	91	0.33	0.72
Treatment:Trial	273	1.42	0.22
Distance travelled			
Treatment	91	0.34	0.71
Treatment:Trial	273	1.32	0.26
Activity ratio			
Treatment	91	0.25	0.78
Treatment:Trial	273	1.43	0.20

Lobsters did not show any signs of learning in any of the treatments; *i.e.* lobsters did not take significantly less time to find the shelter in trial 2 compared to trial 1, and in trial 4 compared to trial 3 (Figure 4).

4. Discussion

H. gammarus is perceived as a vulnerable species with regards to the emission of man induced MF, since it colonizes artificial reefs created by submarine power cables. Moreover, its relatively sedentary way of life may expose them durably (Normandeau Associates Inc. *et al.*, 2011). Potential risks of artificial MF on juvenile lobsters are alteration of sheltering and exploratory behaviour or physiological and developmental effects. To date, this work constitutes the only study focusing on the impacts of MF on the early life stage of invertebrates.

4.1 Impact of magnetic fields on behaviour

We demonstrated that juvenile European lobsters do not exhibit any change of behaviour when submitted to an artificial static or time-varying magnetic field gradient (with maximum intensity of 200 µT) compared to non-exposed lobsters in the ambient magnetic field. Indeed, their exploratory behaviour (described by mean velocity, total distance travelled and activity ratio), the choice of shelter as well as the proportion of time spent in the different areas of the MF gradient were not significantly different from the ones exhibited by control lobsters.

Our results showed that lobsters were clearly attracted to one side of the raceway, whatever the treatment (*i.e.* Control, AC MF gradient or DC MF gradient). This attraction was likely due to a light gradient within the raceway and a shadow created by the Helmholtz coils. Indeed, the side that lobsters preferred was darker (illuminance: 43.1 ± 5.1 lux, SI 2) than the other side of the raceway (67.5 ± 3.1 lux, SI 2). Considering that lobsters show a strong light avoidance (Botero and Atema, 1982; Johns and Mann, 1987), this light gradient can explain this attraction. Nevertheless, we can however conclude that static and time-varying MF do not constitute a primary factor determining European lobster's exploratory and sheltering behaviour via any attraction or repulsion and is at least overridden by subtle light conditions.

Previous studies on other decapod species showed heterogeneous conclusions. Adult American lobsters (*Homarus americanus*) and Dungeness crab (*Metacarcinus magister)* did not significantly change their behaviour (*i.e.* activity and use of space) when submitted to high MF intensities in laboratory (static MF from 500 to 1,100 µT; Woodruff et al., 2013, 2012). Nevertheless, the authors highlight that results of these two studies need to be treated carefully because of a noteworthy large amount of variability between individuals, trials, and seasons. In a field study of Love et *al*. (2017), the same species (Dungeness crab) and the rock crab (*Cancer productus)* had no difficulty to cross AC power cables at intensities between 24.6 and 42.8 µT (for Dungeness crab) and between 13.8 and 116.8 µT (for Rock crab).

On the other hand, in a field study, *H. americanus* responded to MF by a subtle but significant change of its use of space during an exposure to a power cable (static MF of 65.3 µT) but which did not actually create any barrier to its displacement (Hutchison *et al*. 2018). Although the number of replicates was too small to reach robust conclusions, the edible crab *Cancer pagurus* showed an attraction to a high artificial MF (2,800 µT) in a recent laboratory experiment (Scott *et al*. 2018). Similar results were found with the freshwater crayfish *Orconectes limosus*, which was more present inside shelters submitted to a less intense artificial MF (800 µT) than in non-exposed shelters (Tański et al. 2005). Finally, the Caribbean spiny lobsters (*Panulirus argus*) showed contradictory results with a size-dependent avoidance of artificial MF (300 µT) *i.e.* only the biggest spiny lobsters avoided this artificial MF (Ernst and Lohmann 2018).

P. argus can sense the Earth's MF, probably through magnetite-based magnetoreceptors organs (Ernst and Lohmann, 2016), and use this information for navigation and homing (Lohmann, 1984, 1985; Lohmann *et al.*, 1995; Boles and Lohmann, 2003; Lohmann and Ernst, 2014). It is possible that some *Homarus sp*. populations, which migrate seasonally on shore to reproduce (Pezzack and Duggan, 1986), may possess similar sensory capacity, which could explain in part the results obtained by Hutchison *et al.* (2018). However, to date, there is no evidence proving such ability to detect MF. Ernst and Lohmann (2018) mentioned a possible ontogenic shift in the ability of the spiny lobster to respond to MF, this species may acquire or improve their magnetosense as they grow. If this ontogenic shift exists also for the European lobsters, the juveniles that did not show any significant response to artificial MF could be too young to be impacted but may respond differently once older. This point highlights the need to fully apprehend the impact of MF from power cables on *Homarus sp*. by considering its whole life cycle, and that further knowledge on their physiological ability of magneto-reception is required.

4.2 Magnetic fields exposure

In our experiments, all lobsters survived after one week of exposure to MF, whether from AC or DC (225 ± 5 µT). Also, after this exposure, the lobsters' ability to find a shelter and their exploratory behaviour (mean velocity, total distance travelled and activity ratio) remained similar to those observed in the control individuals.

Sheltering constitutes an important antipredator mechanism for juvenile lobsters in the wild. Consequently, if this behaviour is modified by any disturbance, juvenile lobster mortality may be significantly impacted. For example, Cresci et al. (2018) showed that exposure to teflubenzuron, an in-feed pharmaceutical used in salmon aquaculture, significantly impacted the sheltering behaviour of juvenile European lobsters, especially by reducing their learning abilities *i.e.* their capacity to learn the location of shelters and reach them more quickly. In the present study, lobsters did not show any signs of learning regardless of treatment. This lack of learning may be due to the young age of our lobsters (newly settled between stages VI and VIII, CL around 0.9 cm) compared to the later juveniles in the study by Cresci *et al*. (CL around 1.7 cm). Similarly, juvenile American lobsters at stage V did not show immediate learning when placed in similar conditions, *i.e.* an open area with a constant visual contact with the shelter (Bayer et al. 2017). An alternative explanation to the absence of learning in our study can be the absence of necessity to reach the shelter rapidly, *i.e.* no stress source or rewards existed in our experimental setup that could stimulate learning behaviour. A number of studies show learning ability of several species of crustacean (mainly crayfish and crabs) increase to avoid stress (*e.g.* electrical shocks) or to obtain food reward (Tomsic and Romano, 2013).

During the behavioural tests, all the lobsters had more difficulties to find the white shelter compared to the grey one whatever the treatment. Lobster vision, just as their sense of touch provided by their long antennae, are both crucial for detecting and exploring potential shelters (Bayer *et al.*, 2017; Cresci *et al.*, 2018). The high contrast of color between the grey shelter and white background of the raceway may explain why lobsters were more able to visually locate the grey shelters. On the other hand, white shelters on a white background became almost invisible to the lobsters, which had to physically touch the shelter with their antennae to detect it, in a more random process. Considering that vision and touch senses of juvenile lobsters as well as their sheltering behaviour were not impacted by a 1-week exposure to static or time-varying MF, their capacity to escape predation in the wild should remain unchanged in the presence of artificial MF of similar intensities.

In the literature, lack of significant impact of MF on survival of marine organisms was also shown by other laboratory studies using higher MF values. In a study of Bochert and Zettler (2004), the north sea prawn (*Crangon crangon*), the round crab (*Rhithropanopeus harrisii*), the glacial relict isopod (*Saduria entomon*), the blue mussel (*Mytilus edulis*) and young flounders (*Platichtys flesus*) showed no difference of survival between control animals and animals exposed to a static MF of 3,700 µT for several weeks. In the same way, early life stages of the rainbow trout (*Oncorhynchus mykiss*, 36 days with static MF of 10,000 µT or time-varying MF of 1,000 µT) and Northern pike (*Esox lucius*, around 20 days with static MF of 10,000 µT), showed no significant impact on larval and embryonic mortality despite an increase of the yolk-sac absorption rate for the exposed individuals (Fey et al. 2019a , 2019b). Nevertheless, no information about post-exposure development of this larvae was given. Despite this apparent absence of direct mortality caused by MF reported by the literature, Stankevičiūtė et al. (2019) stressed for the first time a genotoxic and cytotoxic effect of exposure to 1,000 µT AC MF on different aquatic species: the rainbow trout (larval stage, 40 days exposure), the Baltic clam (*Limecola balthica*, 12 days exposure) and the common ragworm (*Hediste diversicolor*, 12 days exposure). The degrees of genotoxicity and cytotoxicity of MF on aquatic organisms remain poorly known at present, but affected integrity of genetic information may cause a variety of diseases and disorders, including tumors (Stankevičiūtė *et al.*, 2019). In conclusion, these genetic and physiological criteria should also be considered in future studies.

4.3 Magnetic fields intensity

The MF intensities used in experimental studies previously mentioned are in most cases higher or equal to 1,000 µT, which constitute very high values of MF. The use of such intensities corresponds, in most cases, to modeled data, without any link with *in situ* measures (which is also the case for our study). The low numbers of field studies which performed MF measures *in situ*, highlighted significantly lower intensities (a maximum of 116.8 µT in study of Love et al., 2017). Although the MF intensity produced by a power cable highly depends on its characteristics, a gap seems to exist between MF intensity obtained from modelling and measured *in situ*. Hutchison et al. (2018) even showed, from *in situ* measures, that MF intensity produced by an AC power cable was significantly lower than modeled values commissioned by the grid operator. Consequently, it seems that most experimental studies dealing with MF intensities are never measured *in situ* and are probably unrealistic with respect to the majority of functioning submarine power cables. Thus, transposition of the results obtained experimentally to the field remains difficult. In a context where the number of connections, but also the individual power of submarine power cables show a quick increase, more *in situ* measurements of the MF intensity produced, which remain extremely scarce, are needed to understand more and to evaluate the impact of this perturbator on marine life.

Nevertheless, in the scope of providing accurate guidelines regarding technology used for energy transmission, threshold values of tolerance must be evaluated for the number of marine organisms by using a wide range of MF intensities, even including high intensities probably unrealistic for submarine power cables.

5. Conclusion

In our study, we showed that there was no anthropogenic MF impact on juvenile European lobsters, whether coming from DC or AC power cables with realistic intensity values. The ability to find a shelter after a 1-week exposure remained unchanged and no avoidance or attraction to anthropogenic MF can be demonstrated. However, we showed that visual cues, such as color of the shelters and a light intensity gradient affected their shelter seeking behaviour. Further knowledge on *Homarus sp.* physiological ability of magneto-reception and how this potential magneto-sense can evolve during its life is required to fully understand the impact of anthropogenic MF on this biological model.

Acknowledgements

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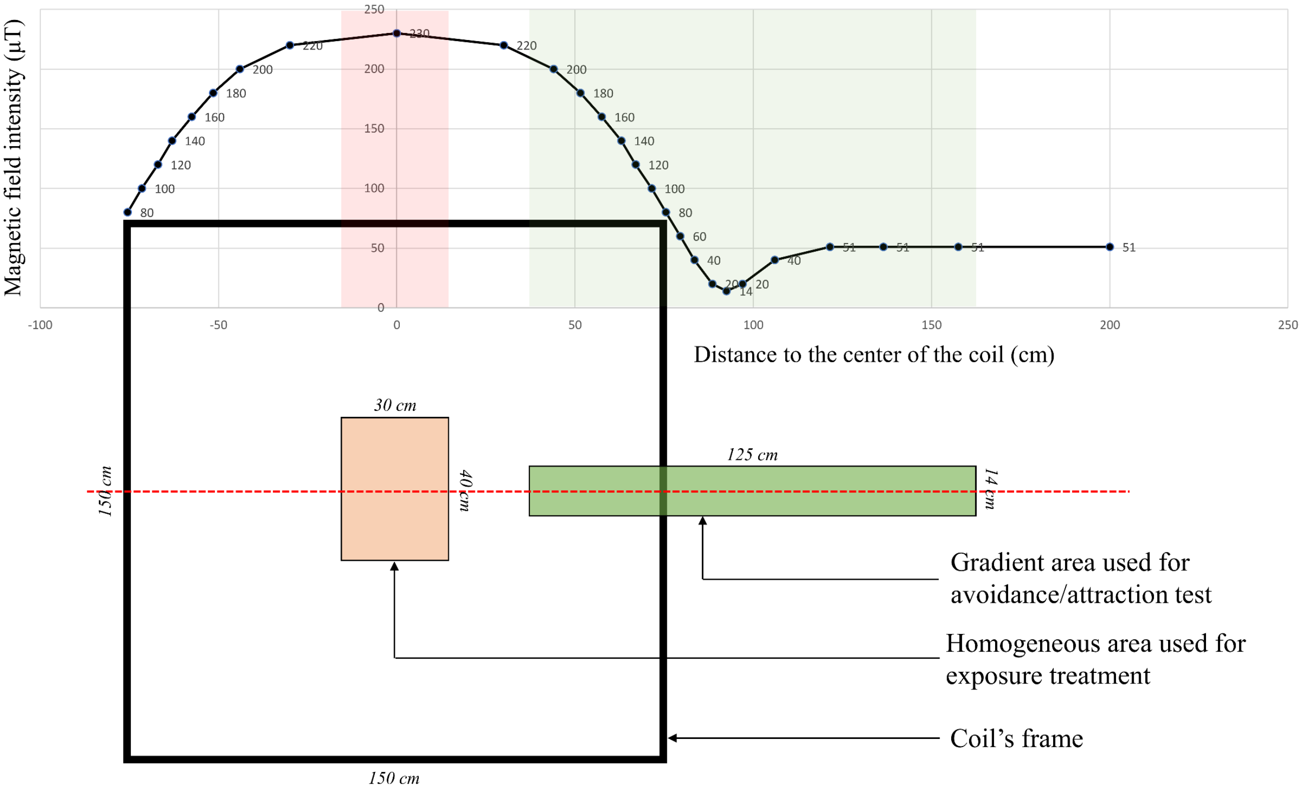
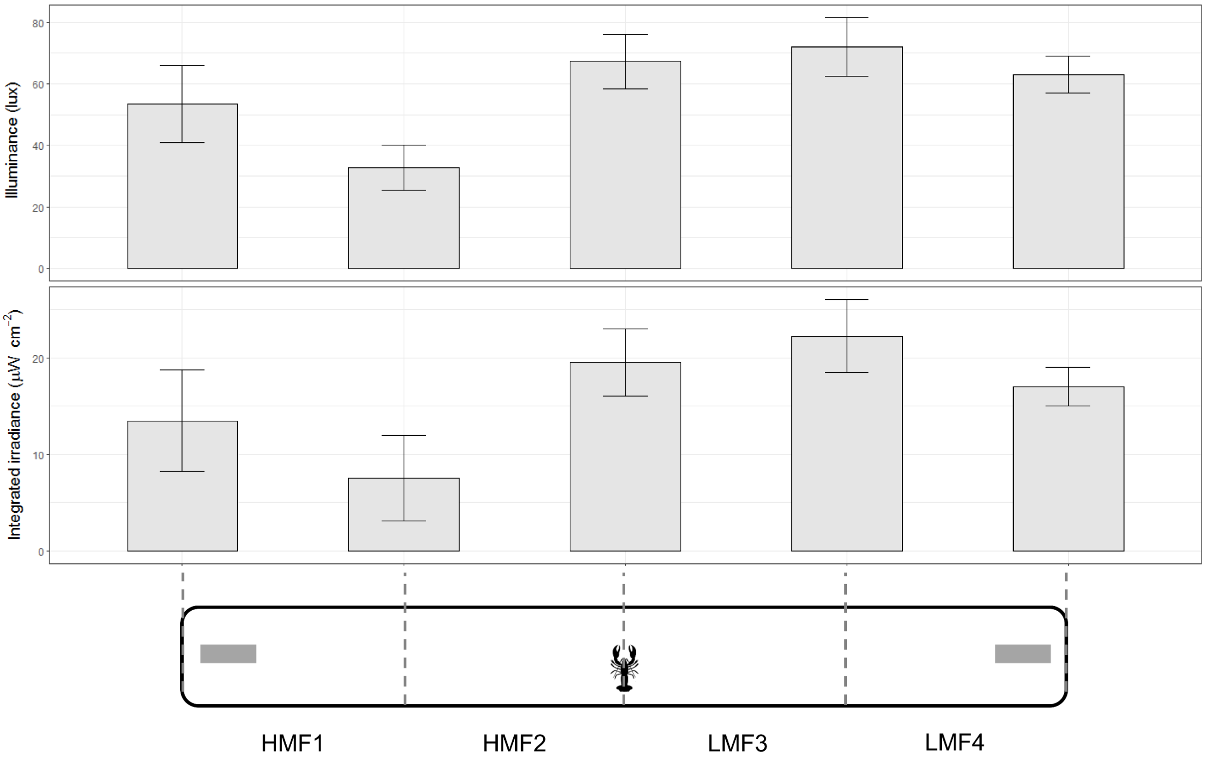
Supplementary information 1: Distribution of the magnetic field generated with the running Helmholtz coils. Measures were taken in the horizontal work area, located at equal distance between the two coils that are vertically separated by 1 m. Each coil is constituted of 600 m of wire (conductor material composed of copper with a 2.5mm² section) rolled up around a 1.5 x 1.5 m wooden frame. For DC treatment the coils were alimented with a BK Precision DC power supply (model BK-1745A). For AC treatment, coils were alimented with a single phase variable auto transformers (model RS CMV 15E-1).



Supplementary information 2: Light measurement along the raceways used for avoidance/attraction experiment. The luminosity intensity was measured at 5 different points along the raceway using a spectrophotometer (Ocean Optics FLAME-S-UV-VIS). Error bar represents the standard error. Top: illuminance, bottom: integrated irradiance.



Chapter 6

The power to protect: Do submarine cables generate unintended reserves?

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Abstract

Submarine power cables linking Jersey and France are associated with a 60 km² exclusion area where all anthropogenic activities (*i.e.* anchoring, trawling *etc.*) are forbidden in order to protect these cables. This study aims to examine the potential "reserve effect" of this exclusion area on the communities of benthic macrofauna. More precisely, we studied the potential relationships between both taxonomic and functional macrofauna diversity, and the position according to the exclusion area (*i.e.* inside *vs.* outside) as well as the fishing effort. On the English side of the exclusion area, no inside *vs.* outside differences were observed for taxonomic and functional proxies of macrobenthic assemblages. This absence of a clear pattern was linked to a very low fishing activity even for the sites located outside the exclusion area. On the other hand, for the French side of the exclusion area, study sites inside the exclusion area presented more diverse macrobenthic communities, both with taxonomic and functional indices, suggesting a reserve effect. Nevertheless, due to our unbalanced sampling design according to the spatial distribution of the fishing effort, linking this change in diversity with fishing effort data was difficult.

Keywords

Anthropogenic impact ; marine protected area ; submarine power cable ; macrobenthos ; functional diversity

1. Introduction

Submarine cables are deployed worldwide for different applications: namely for communication transfer (*e.g.* optical fibers), connecting autonomous energy grids to supply power to remote places (*e.g.* islands, marine platforms or subsea observatories), and more recently to convey power generated by marine renewable energy installations (*i.e.* offshore windfarms, tidal and wave turbines *etc.*). In order to protect these cables from damages caused by human activities such as bottom fishing or anchoring, these zones are sometimes protected by an Exclusion Area (EA) where anthropogenic activity is prohibited by the local authorities. For example, to protect 4 different cables linking the North and South Islands of New Zealand, the New Zealand authorities created an EA of approximately 236 km², where anchoring and fishing activities are prohibited (TRANSPOWER, 2011). These access restrictions can result in positive effects for ecosystems by creating a "reserve effect". Firstly, economically exploited species are protected throughout their whole lifespan which is the case for sedentary species (*e.g.* molluscs) and during the time they spend in the area for mobile species (*e.g.* fish and decapods). Secondly, the sea bottom is preserved from direct impact (scraping and ploughing of the seabed, resuspension of sediment and removal of non-targeted species) generated by bottom gears such as beam and otter trawl, dredges *etc.* (Dayton *et al.*, 1995; Thrush and Dayton, 2002). Considering that the number of submarine cables is expected to increase drastically in the coming decades due to increasing grid connections to islands, development of communications and marine renewable energy projects (Taormina *et al.*, 2018 ; see Chapter 1), a better comprehension of their potential indirect positive influence on the ecological state of marine ecosystems is essential.

Few works exist aiming to characterise the potential reserve effect induced by submarine cables. In the Hauraki Gulf (New-Zealand), a study focusing on fish communities found no significant difference in species richness inside and outside an EA associated to submarine cables (Shears and Usmar, 2006). According to the authors, this lack of response was potentially due to the late protection status (< 4 years) or to illegal fishing activities inside the protected area. Conversely, in the Gulf of Maine (United States), a study of an EA associated with a fibre-optic cable route showed a significant difference in epifaunal community structure, with engineer species being more frequent within the EA (Nenadovic, 2009).

Evaluation of reserve effect associated with EA are often based on the study of taxonomic diversity through species inventories (Villamor and Becerro, 2012). Nevertheless, as ecosystem processes depend more on functional diversity than species diversity *per se* (Nyström, 2006), these approaches are not sufficient to set conservation priorities. Thus, in order to bring further information to the efficiency of EA on marine ecosystems, approaches that shed light on functional diversity are also needed.

In this context, this study aims to examine the potential “reserve effect” of an EA associated with two submarine power cables in a coastal environment in the English Channel. We considered the communities of benthic macrofauna through a twofold approach examining both their taxonomic and functional diversity.

2. Methods

2.1 Study area

The study site is located in the Norman-Breton gulf (English Channel), between the Island of Jersey and France (Figure 1). Two unburied submarine power cables (called Normandy 1 and Normandy 2) were installed on gravelly and coarse sandy sediments in 1982 and 2000 respectively, to power Jersey with electricity (Figure 1). To prevent any damage, each of them were located within an area where all anthropogenic activities (*i.e.* anchoring, trawling etc) are forbidden within a distance of 500 m on either side of the cables. Since the two cable routes are close and run parallel, the protection measures have resulted in a global shared EA of approximately 60 km^2.

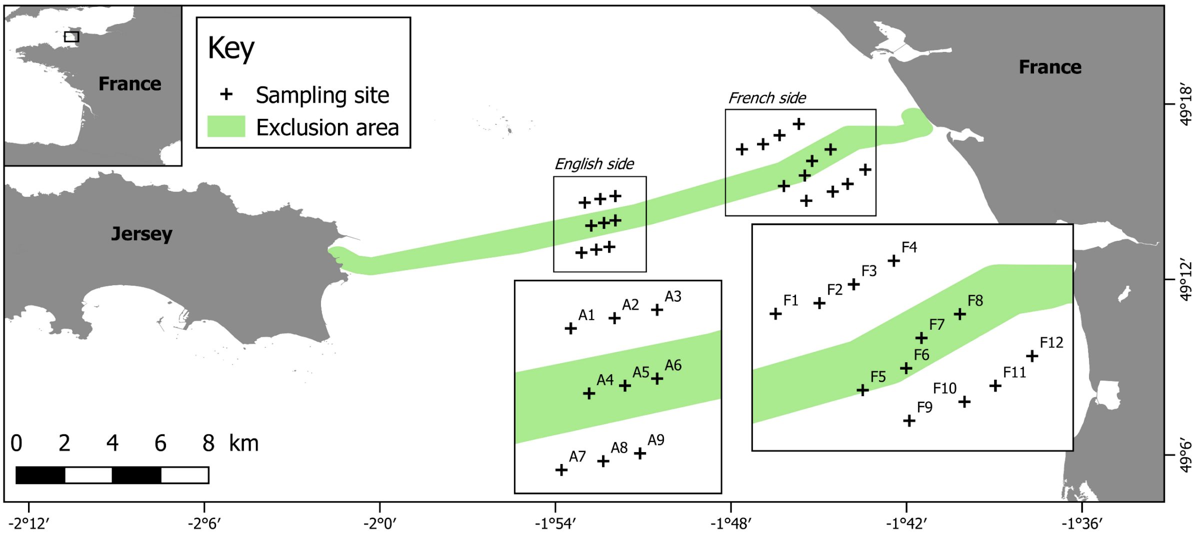


Figure 1: Map of the study area between Jersey and France, which shows the location of the 21 sample sites, distributed on the English and French side.

2.2 Sampling strategy

A total of 21 sites were studied, 12 located on the French side of the area and 9 on the English side, among them 7 are located within the EA and 14 outside (Figure 1). Due to differences of substratum properties between the French and the English side, we used a 0.1 m^2 Van Veen grab on the French side (site F1 to F12) and a 0.1 m^2 Hamon grab on the English side (Site A1 to A9). At each site, 4 samples were performed: the first was stored for granulometric analysis and the 3 others were sieved on 2 mm and preserved on a 4% formaldehyde solution for post analysis of macrofauna. Sampling occurred in October 2017and March 2018 for the French and English sides respectively.

2.3 Sample processing

Granulometric analysis: Samples kept for granulometry were washed using freshwater to remove salt before being stored in an oven for 48 h at 60°C. Afterwards, the different fractions of sediment were separated using test sieves (12 different mesh; from 2,000 to 40 µm) stacked on an automatic sieve shaker left for shaking for 20 min. Each sieve's residual was then weighed and the contribution of each size fraction (*i.e.* gravel, sand and mud) was calculated for each sample. Due to logistical problems, samples of sites F5 to F7 were not analysed.

Macrofauna analysis: Samples preserved in formaldehyde were washed in fresh water for at least 2 h prior to any manipulation. Macrofauna individuals (> 2 mm) were sorted, identified to the lowest possible taxonomic level using a binocular magnifier and microscope and counted. The density of each species (ind.m^{-2}) in each replicate was calculated by dividing its abundance by the surface of the sampling. We obtained two separate "taxa x replicate" matrices, corresponding to the French and English areas.

Table 1: Traits and corresponding modalities (and their abbreviations) used in this study.

Trait	Modalities	Abbreviations
Feeding method	Suspension feeder	F_Susp
	Surface deposit feeder	F_Dep_Surf
	Subsurface deposit feeder	F_Dep_Sub
	Predator	F_Pred
	Scavenger	F_Scav
	Grazer	F_Graz
	Parasit	F_Paras
Maximum size	Very little	S_V_Small
	Little	S_Small
	Medium	S_Med
	Large	S_Lar
Life span	Short	L_Short
	Medium	L_Med
	Long	L_Long
Development mode	Direct	Dev_Dir
	Asexual	Dev_Asex
	Indirect - planktotrophic	Dev_Plankt
	Indirect - lecithotrophic	Dev_Lecit
Substrate position	Deep burrower	Sed_Deep
	Shallow burrower	Sed_Shal
	Sediment-water interface	Sed_Inter
	Emerging	Sed_Surf
Living habit	Swimmer	Mob_Swim
	Crawler	Mob_Crawl
	Burrower	Mob_Bur
	Sedentary	Mob_Sedent
	Attached	Mob_Atta
Fragility	Fragile	Fr_Frag
	Medium	Fr_Med
	Robust	Fr_Rob

2.4 Biological trait collection

For each identified taxa, information were collected to specify 7 different biological traits, each of them being declined in 3 to 7 modalities (Table 1). We selected traits that characterise the life cycle (*i.e.* maximum size, lifespan and development mode) and the behaviour (*i.e.* feeding ecology, position relative to sediment, mobility and fragility) of each taxon and that reflect key ecological processes that can vary according to the intensity of bottom anthropogenic activities. Information was collected by gathering data from the scientific literature and "taxa x traits" matrices previously built for other studies (Androuin, 2018; Bacouillard, 2019). Taxa were scored for each trait modality based on their affinity using a fuzzy coding approach (Chevenet *et al.*, 1994). Fuzzy coding allows the transformation of qualitative trait data in quantitative data in order to conduct multivariate analysis. The final results are two matrices "taxa x traits", referring to the English and the French areas. For each area, this "taxa x traits" matrix was merged with "taxa x replicate" matrix by multiplying the modality score of each taxon by its density in each replicate, giving a "trait x replicate" matrix.

2.5 Fishing effort

Fishing effort in the study area was estimated for the 2014 to 2017 period using Vessel Monitoring System (VMS) data provided by the Système d'Informations Halieutiques (SIH). A filter was applied to raw data in order to only conserve vessels that were in action of fishing and using benthic towed gears (mostly fishing dredges and otter-trawls). The mean fishing time in hours per year is thus reported on a 1' x 1' resolution grid of the studied area. By estimating that fishing vessels worked at a speed of 3 knots , and that apertures of dredges and otter-trawls are of 5 m and 70 m, respectively (Eigaard *et al.*, 2016), the mean reworked bottom surface by fishing gear per year for each 1' x 1' cell was calculated. This surface was divided by the surface of a 1' x 1' cell in order to compute a mean percentage of reworked surface per year (Figure 2). A percentage equal to 100% means that all the surface of the 1' x 1' cell was reworked once during the year, and a percentage higher than 100% means that all the surface of a 1’ x 1’ cell was reworked more than once during the year. Finally, fishing effort was associated to each station according to their GPS position (Figure 2).

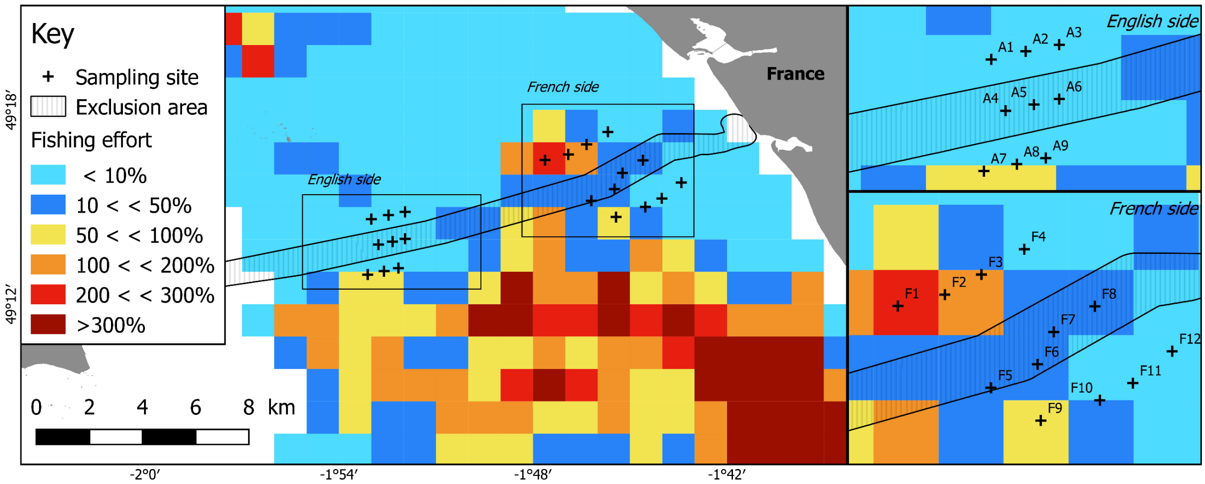


Figure 2: Map of the study area between Jersey island and France, which shows the location of the 21 study sites, distributed on the English and French side and the fishing effort on a 1’ x 1’ resolution grid computed from vessel monitoring system data.

2.6 Data analyses

Due to the fact that French and English sites were not sampled with the same grab and at the same season, results were analysed separately.

To characterise the variability in taxonomic and functional compositions of macrobenthic assemblages, Principal Component Analyses (PCA) from the different matrices were conducted using R software. Granulometric characteristics were superimposed on PCA ordinations following the "*envfit*" procedure of the vegan package. To evaluate the significance of the relations between the fishing effort or the position according to the EA (hereafter called “*Position*” in order to make the text more readable) and the functional/taxonomic assemblage composition, analyses of similarities (ANOSIM; Clarke, 1993) based on Euclidian distance matrices were computed. Matrices of species and trait modality abundances were Hellinger-transformed prior to all multivariate analyses. Hellinger transformation allows for the use of Euclidian-based methods such as PCA on frequency data (Legendre and Gallagher, 2001).

Several complementary indices describing different aspects of the taxonomic and functional diversity of the macrobenthic assemblages were computed for each replicate. Concerning taxonomic diversity, in addition to total density and specific richness, Shannon and Pielou diversity indices were calculated. For functional diversity, different indices were considered: the Functional Richness (FRic), the Functional Evenness (FEve), the Functional Originality (FOri) and the Functional Specialisation (FSpe). A brief description of these indices and their calculation method are available in supplementary material (SI 1). To study the significance of the relation between the fishing effort or the position according to the EA and these indices, two-way analyses of variance (ANOVA) were used. When appropriate, post-hoc tests using Bonferroni correction were performed to study pairwise differences between factors. Data were tested for normality assumption using the Shapiro-Wilk test as well as variance homoscedasticity by examining graphed residuals prior to any analyses.

3. Results

The sediment granulometry of the study area was quite homogenous with a dominance of sand with gravel (SI 2). Proportions of mud were very low on both the English and French sides with values consistently inferior to 0.4%, except at site A3 where it reached 3%. A total of 172 different taxa were recorded over the entire study area (SI 3), including 112 within the English side and 135 within the French side. The dominating phylum was Annelida (84 species, 48.8%) followed by Arthropoda (54 species, 31.4%) and Mollusca (27 species, 15.7 %).

From 2014 to 2017, the fishing effort was higher on the French side (*i.e.* 65 ± 5% of the surface reworked each year) than on the English side (*i.e.* 9 ± 3%; Figure 2). On both sides, the fishing effort was lower inside the EA (20 ± 2% and 4 ± 0.3% on the French and English side, respectively) than outside (88 ± 21% and 12 ± 4% on the French and English side, respectively). Considering the range of observed values, five different levels of fishing effort were generated:

i) < 10%; *ii*) between 10 and 50%; *iii*) between 50 and 100%; *iv*) between 100 and 200% and *v*) between 200 and 300%.

3.1 Taxonomic diversity

The two first axes of PCA performed on taxonomic composition of macrobenthic assemblage of the English side captured 23.7 % of the total variation. Axis PCA1 (12.6% of total variation) was positively correlated with *Crepidula fornicata* and *Nucula nucleus* abundance. Axis PCA2 (11.1% of total variation) was positively correlated with *Scoletoma*

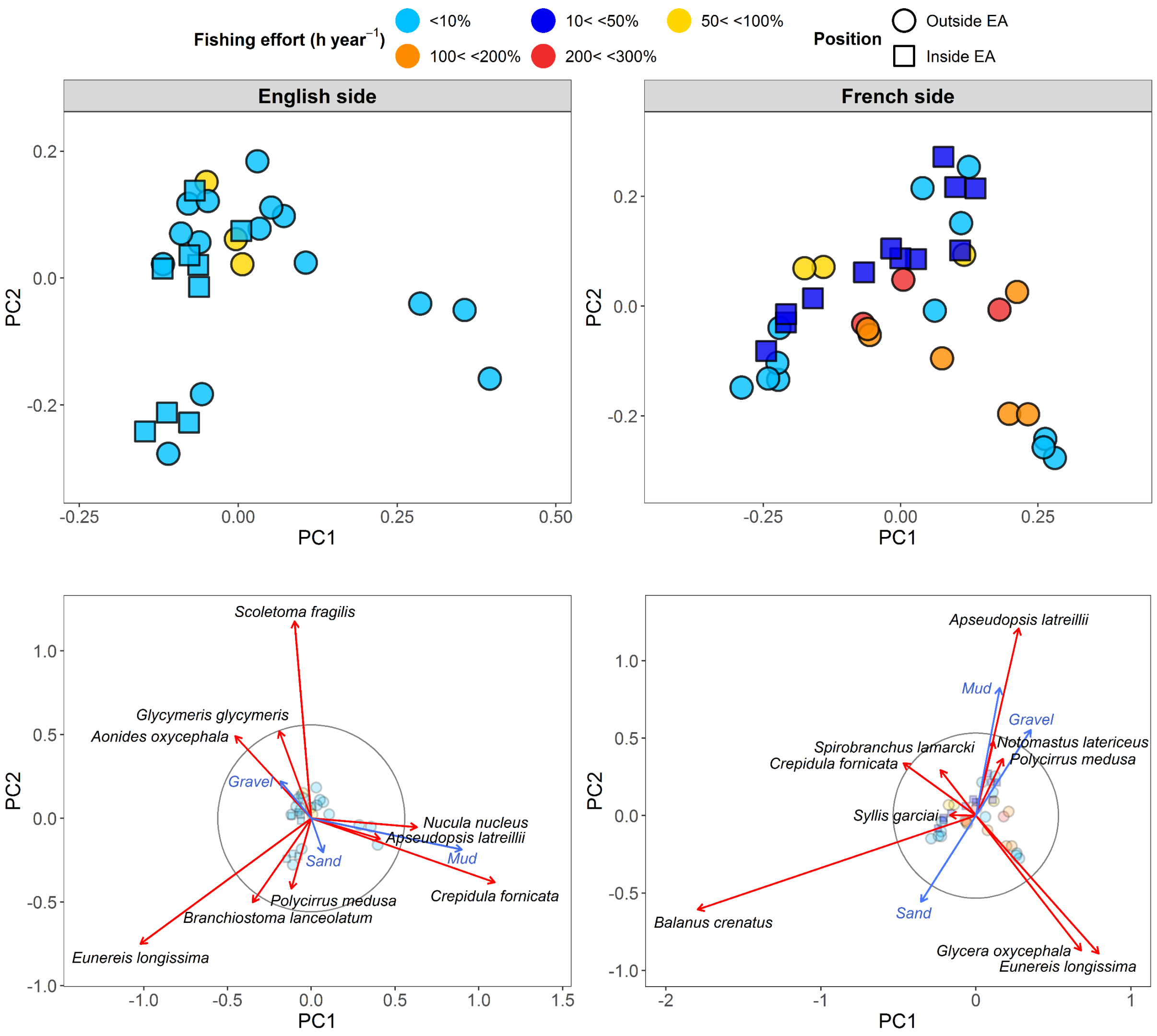


Figure 3: Principal Component Analysis (PCA) of Hellinger transformed density data of macrofauna assemblages from the English side (left) and the French side (right). Each point represents a replicate. Colour points indicate fishing effort and shape indicates the position according to the Exclusion Area (EA). Vector overlays (bottom) show how the species densities correlate with the two first principal components. Only the 5% of the species which best fit with axes 1 and 2 were displayed. Different sediment categories (sand, gravel and mud) were super-imposed onto the PCAs using the envfit procedure.

fragilis and *Glycymeris glycymeris* abundance, and negatively correlated with *Eunereis longissima* and *Branchiostoma lanceolatum* abundance (Figure 3). The proportion of mud was highly correlated with axis PCA 1. The three replicates of site A3, characterised by very high density of *C. fornicata* and higher proportions of mud, were separated from all other sites along axis PCA 1. Excluding replicates of site A3, two different assemblages can be differentiated along axis PCA 2: a sandy gravel community with *G. glycymeris* and *S. fragilis*, and a gravely sand community with *E. longissima* and *Polycirrus medusa*.

The two first axes of PCA performed on the French side assemblages captured 35.1% of total variation. Axis PCA1 (20.4% of total variation) was negatively correlated with *Balanus crenatus* abundances. Axis PCA 2 (14.7% of total variation) was positively correlated with *Apseudopsis latreillii* abundance, and negatively correlated with *Glycera oxycephala* and *Eunereis longissima* abundances (Figure 3). Mud and gravel were positively correlated with axis PCA2, while sand was negatively correlated (Figure 3).

Table 2: Results of the analyses of similarity (ANOSIM) based on Euclidian distances of Hellinger transformed macrofaunal densities of the English and French sides. The effects of position according to the exclusion area (Position) and fishing effort were tested.

Taxonomic		
English side		
Factor	*R*	*Significance*
Fishing effort	-0.25	0.9
Position	-0.04	0.64
French side		
Factor	*R*	*Significance*
Fishing effort	0.03	0.29
Position	0.01	0.4

On both sides, no site ordination patterns appear with respect to the EA (*i.e.* inside *vs.* outside) or the fishing effort. Consistently, ANOSIM tests did not show any significant taxonomic difference between the macrobenthic assemblages according to these two factors (Table 2).

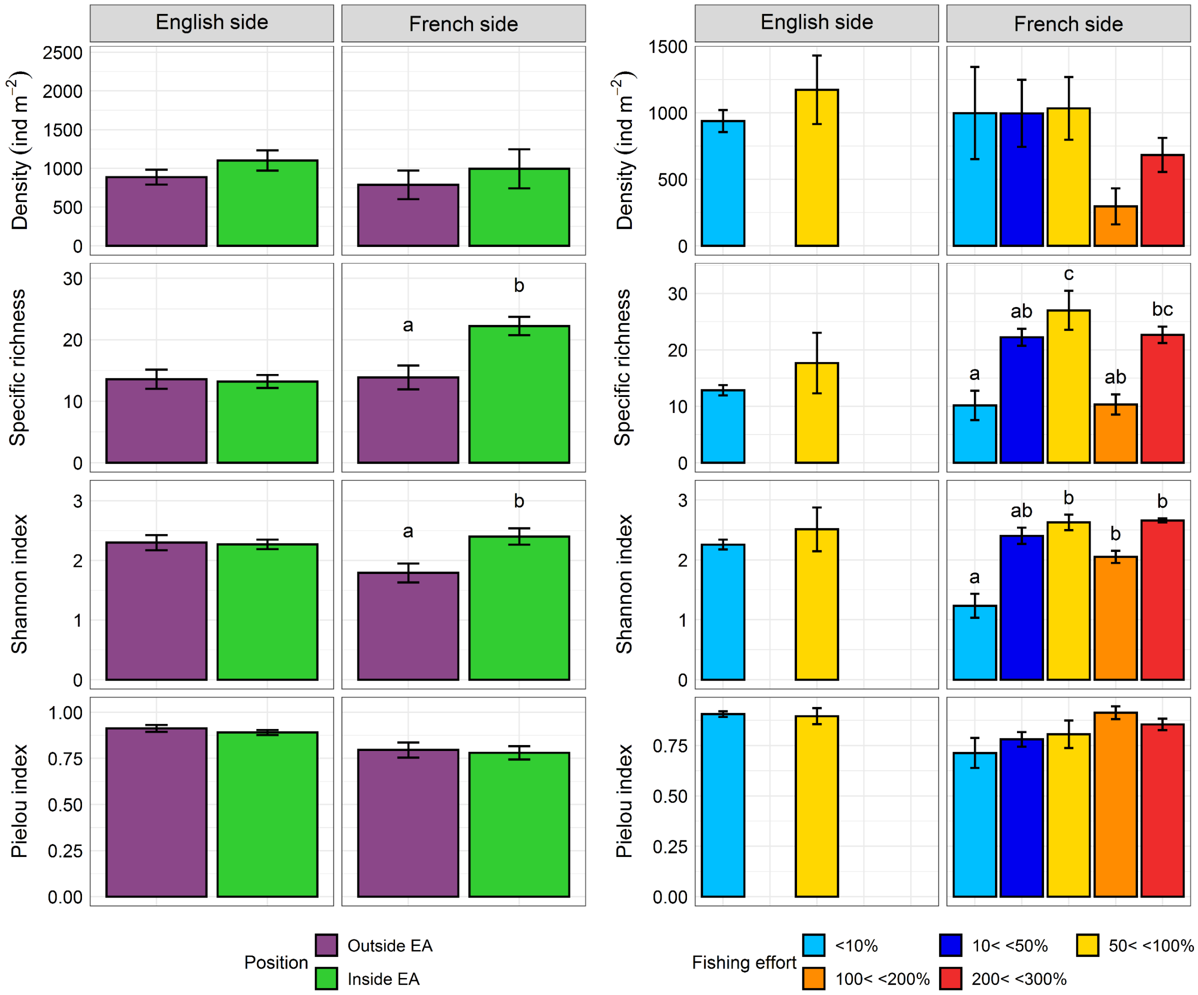


Figure 4: Effects of the position according to the Exclusion Area (EA; left) and fishing effort (right) on density, specific richness, Shannon and Pielou indices of macrofaunal communities of English and French sides. Different letters mean significant difference according to pairwise tests.

Since site A3 is located in a totally different habitat to that of the other sites due to the very high density of *C. fornicata*, it was excluded before computing the different diversity indices in order to avoid a bias caused by this structuring species. On the English side, total density, specific richness as well as Shannon and Pielou indices did not change significantly with fishing effort and position, whereas on the French side, specific richness and Shannon index showed significant differences for the two factors (Figure 4, Table 3). Mean specific richness and Shannon index were respectively 60% and 34 % higher inside the EA than outside. On the French side, the specific richness was higher when the fishing effort was between 50 and 100% and between 200 and 300% compared to effort inferior to 10% and between 100 and

Table 3: Summary of the different ANOVAs performed to detect the effects of fishing effort and the position according to the exclusion area (Position) on the density, specific richness, Shannon index and Pielou index of macrofaunal communities of the English and French side.

Taxonomic diversity

	English side				***French side***			
Index	*Factor*	*df*	*F*	*P value*	*Factor*	*df*	*F*	*P value*
Density	Fishing effort	1	2.28	0.146	Fishing effort	3	0.9	0.451
	Position	1	1.93	0.179	Position	1	0.43	0.518
Specific richness	Fishing effort	1	2.5	0.129	Fishing effort	3	7.45	**0.001**
	Position	1	0.03	0.859	Position	1	12.7	**0.001**
Shannon index	Fishing effort	1	0.94	0.342	Fishing effort	3	10.8	**0.000**
	Position	1	0.03	0.869	Position	1	11.4	**0.002**
Pielou index	Fishing effort	1	0.27	0.607	Fishing effort	3	1.99	0.137
	Position	1	0.67	0.422	Position	1	0.07	0.800

200% (Figure 4). For the Shannon index, the value was significantly higher for stations with high fishing effort (between 50 and 300%) compared to stations with fishing effort inferior to 10% (Figure 4). The distribution of the relative abundance of the different phyla did not change according to the fishing effort on the English side (Figure 5). Nevertheless, on the French side, the contribution of Annelida phylum increased with the fishing effort while the proportion of Arthropoda decreased (Figure 5).

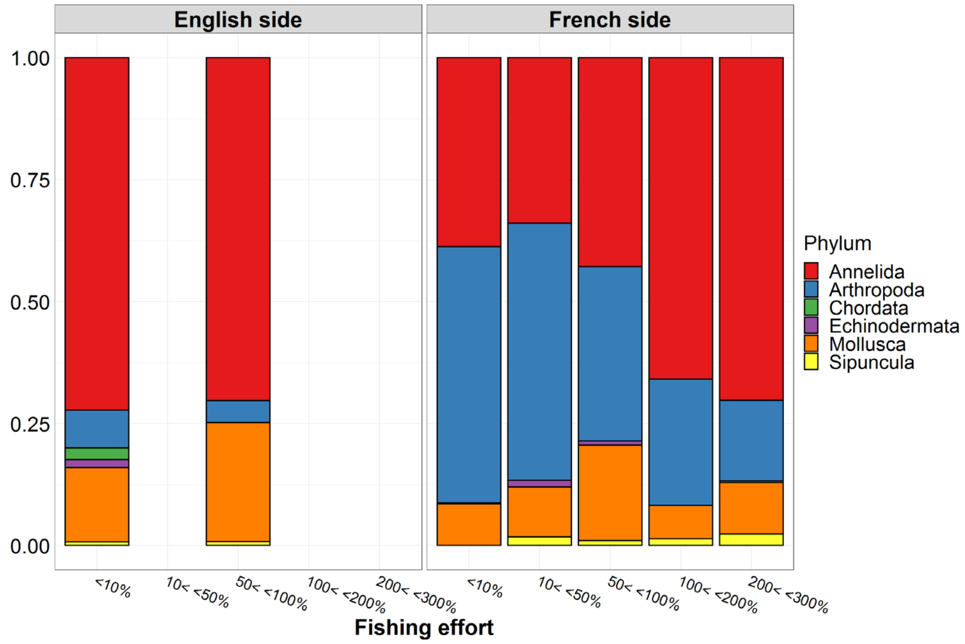


Figure 5: Mean relative abundance of the different Phyla of the macrofaunal assemblages according to the fishing effort for the English and French side.

3.2 Functional diversity

Multivariate analyses conducted on functional matrix showed similar results to those performed on taxonomic matrix. On the English side, PCA captured 52.7% of total variation. The axis PCA1 (39% of total variation) is positively correlated to large burrower organisms with a medium lifespan, and negatively correlated with robust organisms living at the sediment interface with a long lifespan (Figure 6). Axis PCA1 was highly correlated to mud

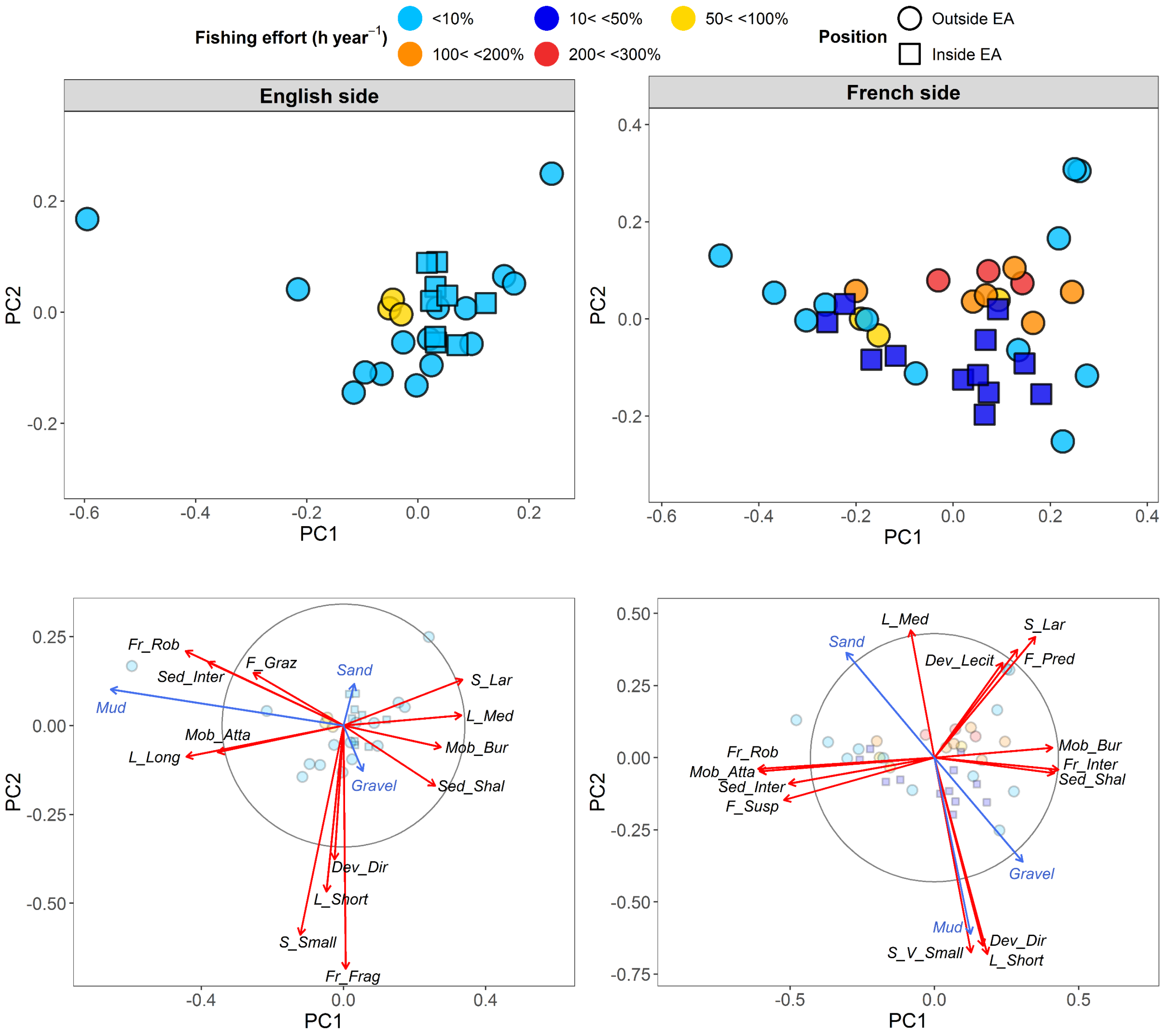


Figure 6: Principal Component Analysis (PCA) of Hellinger transformed trait-modalities densities data of macrofauna assemblages from the English side (left) and the French side (right). Each point represents a replicate. Colour points indicate fishing effort and shape points indicate the position according to the Exclusion Area (EA). Vector overlays (bottom) show how the trait-modalities density correlates with the two first principal components. Only the 5% trait-modalities which best fit with the two axes were displayed. Abbreviations used are the same as in Table 1. Different sediment categories (sand, gravel and mud) were super-imposed onto the PCAs using the envfit procedure.

proportion and separated site A3 from the other sites. Axis PCA 2 (13.7% of total variation) was negatively correlated to fragile and small organisms with short lifespan and direct development (Figure 6).

On the French side, PCA captured 75.4% of total variation of macrobenthic assemblage functional composition. Axis PCA1 (54.8% of total variation) was positively correlated to organisms characterised by an intermediate fragility and living in shallow burrows and negatively correlated with robust attached suspension-feeders. Axis PCA2 (20.6% of total variation) was positively correlated to large predators with a medium lifespan and reproducing with lecithotrophic larvae, but negatively correlated with very small organisms with direct development and short lifespan. Granulometry was correlated with axis PCA2, gravel and sand being negatively correlated while sand was negatively correlated.

For assemblages of both sides, fishing effort and position did not appear as significant explaining factors (Table 4).

Table 4: Results of the analyses of similarity (ANOSIM) based on Euclidian distances of Hellinger transformed trait-modalities' densities of the English and French sides. The effects of position according to the exclusion area (Position) and fishing effort were tested.

Functional		
English side		
Factor	*R*	*Significance*
Fishing effort	-0.19	0.8
Position	0.8	0.87
French side		
Factor	*R*	*Significance*
Fishing effort	0.05	0.21
Position	-0.06	0.78

As for taxonomic indices, site A3 was excludedbefore computing functional indices. All functional indices (*i.e.* FRic, FEve, FOri and FSpe) computed for the English side assemblages did not significantly change in function to the position and the fishing effort (Figure 7, Table 5). For the French side assemblages, the functionnal richness was significantly higher inside the EA than outside (mean increase of 38% ; Figure 7, Table 5). The functionnal richness also

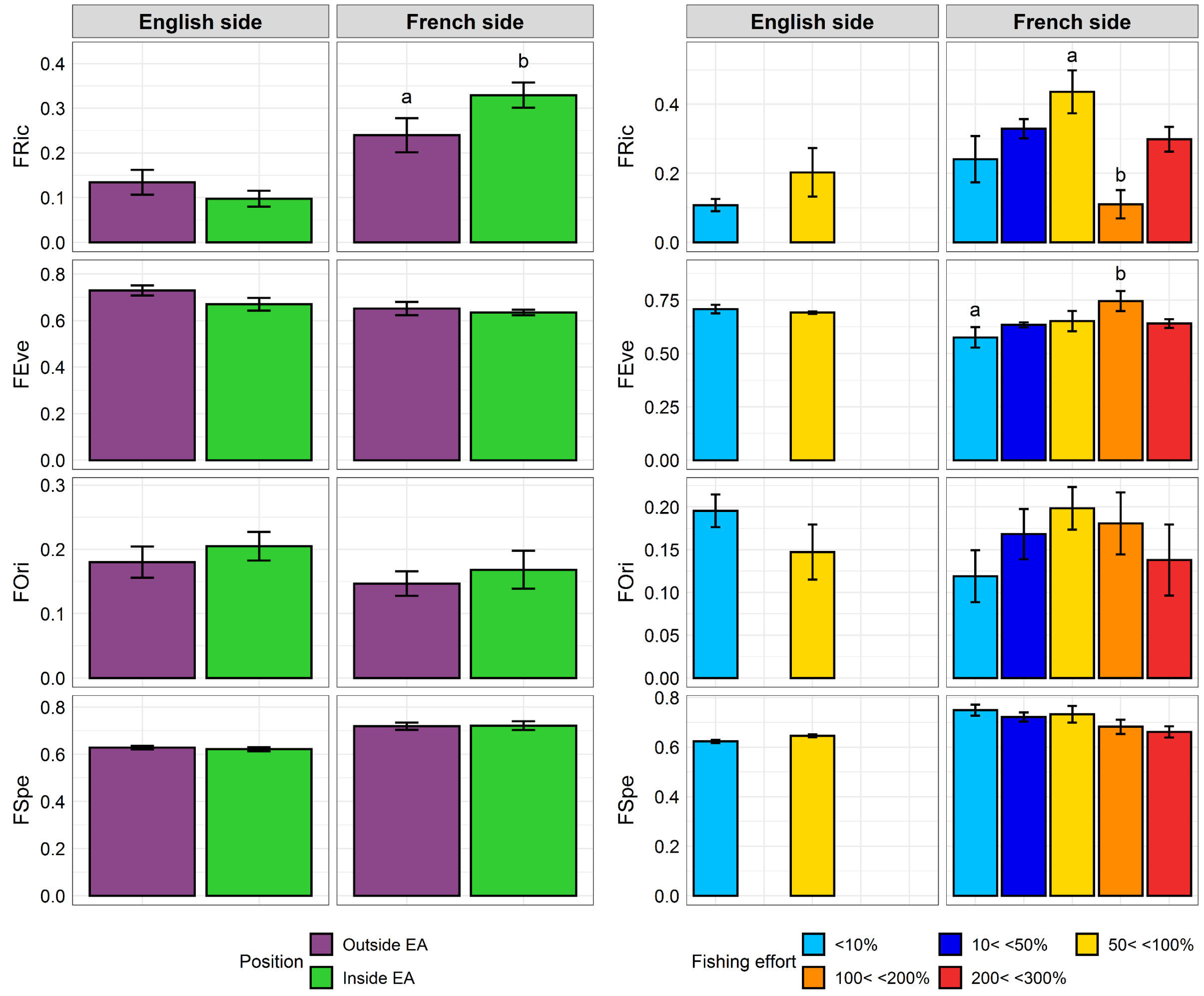


Figure 7: Effects of the position according to the Exclusion Area (EA; left) and fishing effort (right) on the functional richness (Fric), functional evenness (Feve), functional originality (Fori) and functional specificity (Fspe) of macrofaunal communities of English and French sides. Different letters mean significant differences according to pairwise tests.

varied with the fishing effort, with a significant decrease of this index between class 100-200% and class 50-100% (Figure 7, Table 5). The functional eveness was also significantly influenced by fishing effort, with significantly higher values in sites where fishing effort ranges between 100 and 200% than in sites with fishing effort inferior to 10% (Figure 7, Table 5). Other functional indices (*i.e.* FOri and FSpe) did not change independing on the position and fishing effort for assemblage of the French side.

On the French side, the relative abundance of large size species tended to rise with an increasing fishing effort (Figure 8), while the proportions of very small size species tended to

Table 5: Summary of the different ANOVAs on the effects of the position according to the exclusion area (Position) and fishing effort on the functional richness (Fric), functional evenness (Feve), functional originality (Fori) and functional specificity (Fspe) of macrofaunal communities of the English and French side. Significant P values (<0.05) are in bold.

Functional

	English side				***French side***			
Index	*Factor*	*df*	*F*	*P value*	*Factor*	*df*	*F*	*P value*
Fric	Fishing effort	1	2.42	0.136	Fishing effort	3	5.9	**0.003**
	Position	1	1.02	0.324	Position	1	4.52	**0.044**
Feve	Fishing effort	1	0.81	0.380	Fishing effort	3	4.16	**0.016**
	Position	1	2.95	0.101	Position	1	0.3	0.589
Fori	Fishing effort	1	0.54	0.469	Fishing effort	3	0.87	0.467
	Position	1	0.46	0.506	Position	1	0.4	0.534
Fspe	Fishing effort	1	1.28	0.271	Fishing effort	3	2.08	0.123
	Position	1	0.36	0.555	Position	1	0.01	0.923

decrease. Relative abundance of other traits remained unchanged with respect to the fishing effort, on both sides.

4. Discussion

4.1 Reserve effect

On the English side, no reserve effect was detectable on the macrobenthic community, neither in its structural or functional diversity. This absence of effect may be due to the low level of anthropogenic pressure in this area, even outside the EA (the mean annual fishing effort from 2014 to 2017 was inferior to 5% of the reworked bottom surface for all sites except one). Although communities highlight a relative homogeneity, some slight faunal variations can be observed. These changes likely result from fine variations of habitat characteristics (*e.g.* granulometry) and degree of natural patchiness within the "subtidal mixed sediment" habitat (according to the EUNIS classification ; Davies et al., 2004) rather than from the presence of the exclusion area. Indeed, *Nucula nitidosa*, Branchiostoma *lanceolatum*, *Glycymeris glycymeris* and *Scoletoma fragilis* characterise three different habitats, respectively: *i*) *Abra alba* and *Nucula nitidosa* in circalittoral muddy sand or slightly mixed sediment, *ii*) *Branchiostoma lanceolatum* in circalittoral coarse sand with shell gravel and *iii*) *Mediomastus*

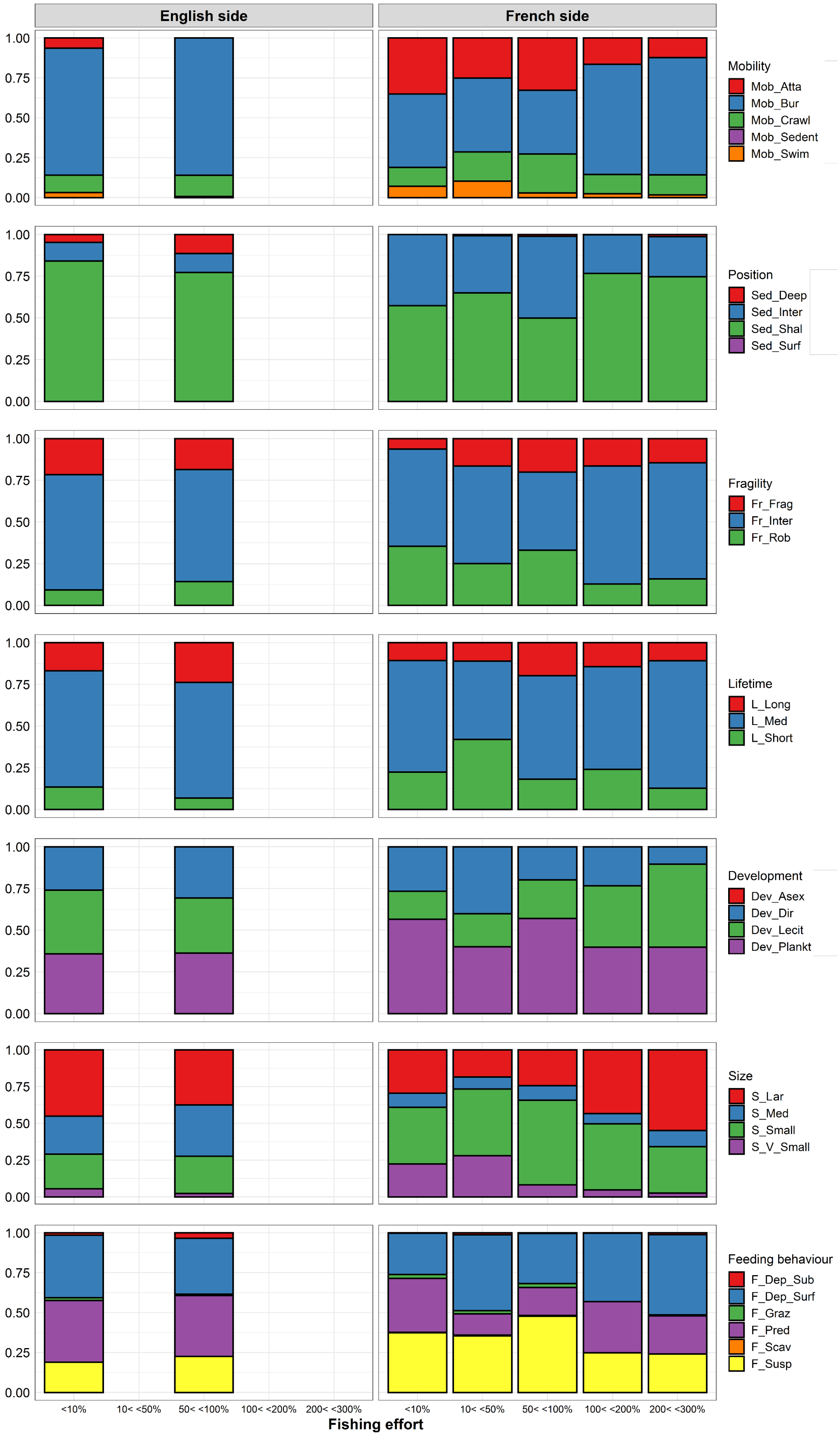


Figure 8: Mean relative proportions of trait modalities according to the fishing effort for macrofaunal assemblages of the English (left) and the French (right) sides. Abbreviations used are the same as in Table 1.

fragilis, *Lumbrineris spp*. and venerid bivalves in circalittoral coarse sand or gravel (Davies *et al.*, 2004).

Conversely, benthic communities located on the French side benefit from a reserve effect. Values of specific richness, Shannon diversity index and functional richness are all significantly higher inside the EA rather than outside. Such a result is consistent with a number of studies showing that marine reserves promote diversity of benthic macrofauna compared to areas submitted to anthropogenic activities (Thrush and Dayton, 2002; Halpern, 2003; Villamor and Becerro, 2012; van Denderen *et al.*, 2014). Recurrent perturbations by bottom fishing gears can reduce habitat complexity by both homogenisation of the substratum (Schwinghamer *et al.*, 1996) and destruction of sessile fauna (Collie *et al.*, 1997).The consequences are a decrease in species diversity (Veale *et al.*, 2000; Thrush and Dayton, 2002).

Surprisingly, relations obtained between taxonomic and functional diversity proxies and fishing effort suggest an unclear effect of the presence of the corridor of exclusion. Macrobenthic assemblages of sites preserved of or submitted to low fishing pressure (*i.e.* < 5%) showed the lowest diversity, while sites submitted to an intermediate fishing pressure (*i.e.* 50-100%) showed the highest diversity values. This trend can be partly explained by the intermediate disturbance hypothesis (Connell, 1978). According to this theory, habitat which is submitted to intermediate levels of disturbance demonstrate a coexisting species indicator of both early and late successional stages which results in an increased diversity. As empirical studies rarely demonstrate this predicted humped diversity–disturbance relationship, the intermediate perturbation hypothesis remains highly discussable (Fox, 2013), especially for marine soft-sediment communities (Thrush and Dayton, 2002). While the intermediate disturbance hypothesis described disturbance as a means for reducing resource monopolisation, Thrush and Dayton (2002) highlighted that direct competition for food or space is difficult to demonstrate as a main process in soft sediments, which make this hypothesis inappropriate.

In addition to their impact on biodiversity, marine reserves have frequently been shown to lead to an increase of megafauna (*e.g.* fish and decapod) size (Halpern, 2003; Shears *et al.*, 2006; Barrett *et al.*, 2007, 2009). In our study however, decrease of the fishing effort is associated with a decrease of the proportion of large size macrofauna species to the benefit of very small size species. This trend can be explained by the alteration of trophic interactions through a top-down control. As the density of predators (*e.g.* fish) may increase in reserves, the predation pressure on macrofauna should increase as well. For example in New-Zealand, the predation pressure on the urchin *Evechinus chloroticus* by fish and rock lobsters is approximatively 7 times higher in no-take marine reserves than outside, resulting in a lower density of urchins inside the reserves (Shears and Babcock, 2002). In our case, in areas with high fishing pressure, we can hypothesis that densities of benthic and demersal fishes are low, and that, consequently, the predation pressure on large size organisms (*i.e.* mainly Annelida of several centimetres) is reduced. Conversely, in area with less fishing effort, as the predation pressure increase, the abundance of these preys logically decreases. To confirm this hypothesis and have a better comprehension of the interactions in the ecosystem, further studies on how other biological compartments of the community (*e.g.* megafaunal community) are impacted by the fishing effort would be essential.

4.2 Different fishing gears, different impacts

In our study, the fishing effort was only quantified as the total surface reworked by fishing activities, without any distinction between the different gears used. However, depending on their technical characteristics, fishing gear (*e.g.* mass, degree of contact with the seafloor, trawling speed *etc.*) disturb the bottom in different ways (Thrush and Dayton, 2002). In the Norman-Breton gulf, fisheries mainly used dredges for bivalves and to a lesser extent, otter trawls for benthic/demersal fish. The first (aperture ~5 m) impacts a lower surface than the second (aperture ~70 m), but penetrates the seafloor more deeply than trawls (~16 cm for dredges *vs.* ~2 cm for trawl ; Hiddink et al., 2017). As the depth of penetration is highly correlated to depletion of fauna (Hiddink *et al.*, 2017), dredge fishing gear should theoretically cause more disturbance than trawls for a given reworked surface. In that case, the overall fishing effort cannot be simply considered as a sum of the surface reworked by these different gears, but each of them should be carefully weighed with their respective pressure intensity, in order to describe induced disturbances more finely.

4.3 A relative disturbance

It is important to state that, although the fishing effort was higher on the French side than the English side, pressure remained relatively low in both areas when compared to neighbouring intense fishing grounds (Figure 2; only 3 sites submitted to a fishing effort > 100%). Also in other location like in The Adriatic sea, reworked areas caused by trawl fishing can be locally superior to 1,000% (Eigaard et al., 2017). Considering the low overall fishing effort of our study area, and the low number of replicates operated within the sites presenting an major fishing effort, it is difficult to state that the between-site differences observed within the French site are clearly due to the fishing activities rather than natural spatial heterogeneity. When evaluating the effect of an exclusion area on marine communities, it is highly important to be able to differentiate 'reserve' and 'habitat' effects (Miller and Russ, 2014). Here, a better characterisation of the habitat of each site (*e.g.* site topography, more precise granulometric analysis *etc.*), as well as a more balanced sampling effort according to the different fishing effort, would have enabledmore robust conclusions to be drawn.

5. Conclusions

The aim of this study was to explore the potential reserve effect caused by an exclusion area associated to submarine power cables on benthic macrofauna communities. On the English side, no inside *vs.* outside differences were observed for taxonomic and functional proxies of macrobenthic assemblages. This absence of a clear pattern was linked to a very weak fishing activity even for the sites located outside the EA. On the French side, sites inside the EA presented more diverse macrobenthic communities, both with taxonomic and functional indices, ,suggesting a reserve effect. Nevertheless, linking this change of diversity solely with fishing effort data was difficult, especially when considering our unbalanced sampling design according to the spatial distribution of the fishing effort.

Acknowledgements

This work is sponsored by the North Sea Program (Institute of Marine Research), Région Bretagne, France Energies Marines, IFREMER, and the National Research Agency within the framework of Investments for the Future program under reference ANR-10-IED-0006-17. The authors would like to thank Aurélie Fauveau for her kind assistance.

Supplementary information 1: Brief description of the different taxonomic and functional indices used, along with details of their calculation method.

Taxonomic diversity indices:

- **Shannon-Wiener index**

Shannon-Wiener index (***H'***) accounts for both abundance and evenness of the species present.

$$H' = -\sum_{i=1}^{i=S} p_i . log_2(p_i)$$

The proportion of species *i* relative to the total number of species (*pi*) was calculated, and then multiplied by the logarithm base 2 of this proportion ($\log_2 pi$). The resulting product was summed across species (*S* being the total number of species). It varied between 0 (only one species) and 1 (all species uniformly distributed).

- **Pielou evenness index**

Pielou evenness index (***J'***) can be calculated by dividing *H'* by *Hmax* (here $Hmax = log_2 S$). Equitability assumes a value between 0 and 1 with 1 being complete evenness.

$$J' = \frac{H'}{H'_{max}} = \frac{H'}{log_2 S}$$

Functional diversity indices:

- **Functional richness**

The functional richness (***FRic***) corresponds to the volume occupied by the community in the multidimensional trait space. A high value indicates a major space occupied by the community, and thus a high functional diversity.

- **Functional evenness**

Functional evenness (***FEve***) measures the regularity of species abundances within the volume, accounting for both the evenness of abundance distribution among species and for the regularity of the functional distances among species.

- **Functional originality**

Functional originality (***FOri***) indicates the isolation of the species in the functional space occupied by a given community. *FOri* increases when the functional originality of the increase (*i.e.* when traits are not redundant).

$$FOri = \frac{\sum Abrel_i * Ori_i}{Ori_{i_{max}}}$$

Where *Abrel*$_i$ is the relative abundance of the species *i* ; *Ori*$_i$ is the originality of the species *i* and *Ori*$_{i\ max}$ is the maximal functional originality.

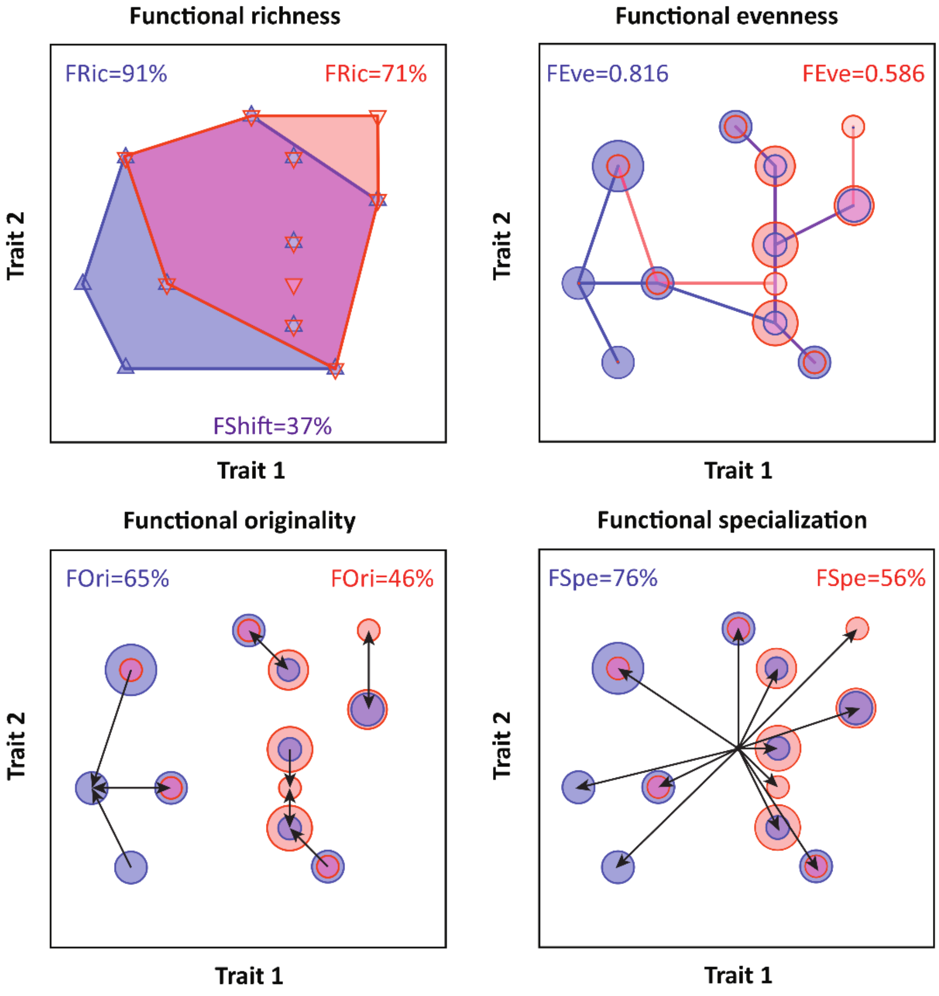
- **Functional specialisation**

Functional specialisation (***FSpe***) corresponds to the mean Euclidian distance of a species from the centre of the volume occupied by the community in the multidimensional trait space. The more *FSpe* is close to 1, the more there is a “specialist” species in the community.

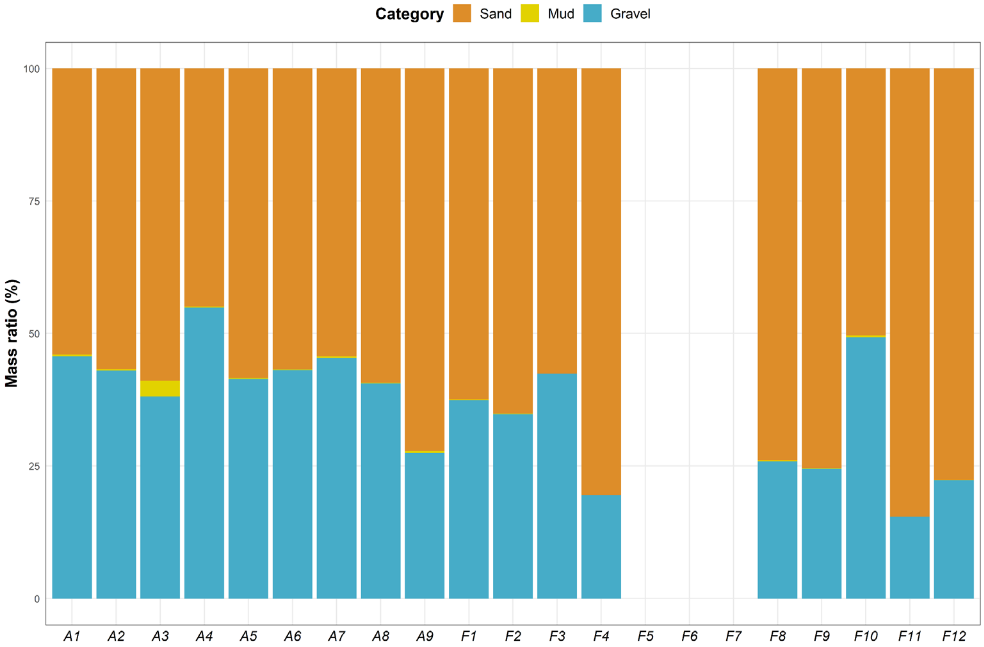
$$Fspe = \frac{\sum_{i=1} Abrel_i Spe_i}{Spe_{max}}$$

Where *Abrel*$_i$ is the relative abundance of the species *i* ; *Spe*$_i$ is the specialisation degree of the species Figure I.

Hereafter, a graphical representation of the different functional indices and potential changes after a disturbance. Species (dots) are plotted in two-dimensional functional spaces according to their respective trait values, circle sizes are proportional to species relative abundance before and after disturbance in blue and red, respectively. (From Mouillot et al., 2013)



Supplementary information 2: Granulometry characteristics of the different study sites of the English (A) and French (F) sides. Due to logistical problems, granulometry of sites F5 to F7 was not analysed.



Supplementary information 3: List of the different macrofauna taxa (respectively ordered in function of Phyllum, Class, Order and Species) encountered on French and English sides combined. The number of samples where each species was present is given in the right column (N total=63).

Annelida

Polychaeta

Eunicida

Dorvilleidae	
Protodorvillea kefersteini	11
Schistomeringos neglecta	1
Schistomeringos rudolphi	1
Eunicidae	
Eunice pennata	2
Lysidice ninetta	4
Lysidice unicornis	1
Marphysa sanguinea	2
Lumbrineridae	
Lumbrineriopsis paradoxa	3
Lumbrineris latreilli	1
Scoletoma fragilis	34
Scoletoma impatiens	3
Oenonidae	
Arabella iricolor	1

Phyllodocida

Glyceridae	
Glycera lapidum	14
Glycera oxycephala	38
Nephtyidae	
Nephtys caeca	5
Nephtys cirrosa	5
Nephtys hombergii	1
Nephtys kersivalensis	2
Nephtys longosetosa	6
Nereididae	
Eunereis longissima	31
Websterinereis glauca	8
Pholoidae	
Pholoe baltica	1

Pholoe inornata	1
Phyllodocidae	
Eteone flava	1
Eulalia aurea	1
Eulalia expusilla	3
Eulalia mustela	11
Eulalia tripunctata	5
Eulalia viridis	3
Eumida sanguinea	2
Mysta picta	1
Pseudomystides limbata	5
Polynoidae	
Gattyana cirrhosa	2
Malmgrenia arenicolae	1
Malmgrenia castanea	3
Malmgrenia ljungmani	9
Malmgrenia marphysae	4
Sigalionidae	
Pelogenia arenosa	2
Sthenelais boa	4
Syllidae	
Dioplosyllis cirrosa	1
Eurysyllis tuberculata	3
Odontosyllis fulgurans	12
Palposyllis prosostoma	4
Sphaerosyllis taylori	1
Syllis armillaris	4
Syllis garciai	29
Syllis gracilis	1
Trypanosyllis coelica	1
Sabellida	
Sabellidae	
Acromegalomma vesiculosum	1
Pseudopotamilla reniformis	1
Sabella pavonina	1
Serpulidae	
Spirobranchus lamarcki	29
Spionida	
Poecilochaetidae	
Poecilochaetus serpens	9

Spionidae	
Aonides oxycephala	34
Laonice cirrata	15
Malacoceros fuliginosus	1
Polydora ciliata	2
Pseudopolydora pulchra	1
Scolelepis (Parascolelepis) tridentata	2
Spio decorata	4
Spio filicornis	1
Spio martinensis	23
Spio symphyta	1
Terebellida	
Ampharetidae	
Ampharete baltica	2
Amphicteis midas	1
Cirratulidae	
Caulleriella A	1
Caulleriella alata	22
Caulleriella bioculata	4
Chaetozone D	2
Chaetozone gibber	2
Chaetozone zetlandica	10
Cirriformia tentaculata	7
Terebellidae	
Lanice conchilega	5
Nicolea venustula	2
Pista cristata	14
Polycirrus medusa	26
Thelepus setosus	2
Trichobranchidae	
Terebellides stroemii	3
NA	
Capitellidae	
Heteromastus filiformis	1
Notomastus latericeus	45
Maldanidae	
Leiochone leiopygos	1
Orbiniidae	
Orbinia cuvierii	1
Sabellariidae	

Sabellaria spinulosa	4
Scalibregmatidae	
Scalibregma celticum	13

Arthropoda

Hexanauplia

Sessilia

Archaeobalanidae	
Semibalanus balanoides	2
Balanidae	
Balanus balanus	1
Balanus crenatus	21
Perforatus perforatus	1

Malacostraca

Amphipoda

Ampeliscidae	
Ampelisca spinipes	3
Amphilochidae	
Apolochus neapolitanus	1
Apolochus spencebatei	1
Atylidae	
Nototropis guttatus	3
Nototropis vedlomensis	14
Bathyporeiidae	
Bathyporeia elegans	3
Bathyporeia guilliamsoniana	4
Calliopiidae	
Apherusa bispinosa	1
Caprellidae	
Phtisica marina	4
Cheirocratidae	
Cheirocratus assimilis	1
Cheirocratus intermedius	3
Cheirocratus sundevallii	4
Corophiidae	
Leptocheirus hirsutimanus	5
Leptocheirus pilosus	7
Leptocheirus tricristatus	7
Monocorophium sextonae	1
Iphimediidae	

Iphimedia obesa	1
Ischyroceridae	
Ericthonius punctatus	14
Leucothoidae	
Leucothoe lilljeborgi	4
Liljeborgiidae	
Idunella picta	2
Lysianassidae	
Lysianassa ceratina	1
Lysianassa insperata	1
Lysianassa plumosa	1
Maeridae	
Animoceradocus semiserratus	1
Maera grossimana	1
Othomaera othonis	8
Melitidae	
Abludomelita gladiosa	9
Abludomelita obtusata	3
Oedicerotidae	
Perioculodes longimanus	2
Synchelidium haplocheles	3
Synchelidium maculatum	1
Photidae	
Gammaropsis maculata	3
Unciolidae	
Unciola crenatipalma	1
Urothoidae	
Urothoe brevicornis	8
Urothoe marina	12
Decapoda	
Alpheidae	
Athanas nitescens	1
Galatheidae	
Galathea intermedia	5
Inachidae	
Macropodia rostrata	2
Paguridae	
Anapagurus hyndmanni	8
Pagurus cuanensis	5
Polybiidae	

Liocarcinus pusillus	2
Porcellanidae	
Pisidia longicornis	3
Isopoda	
Anthuridae	
Cyathura carinata	2
Arcturidae	
Astacilla longicornis	1
Cirolanidae	
Conilera cylindracea	1
Eurydice pulchra	13
Eurydice spinigera	1
Gnathiidae	
Gnathia vorax	1
Janiridae	
Janira maculosa	3
Tanaidacea	
Apseudidae	
Apseudopsis latreillii	20

Chordata

Leptocardii

NA	
Branchiostomatidae	
Branchiostoma lanceolatum	6

Echinodermata

Ophiuroidea

Amphilepidida	
Amphiuridae	
Amphipholis squamata	13
Ophiotrichidae	
Ophiotrix fragilis	1
Ophiurida	
Ophiuridae	
Ophiura ophiura	1

Mollusca

Bivalvia

Adapedonta	
Pharidae	

Ensis ensis	3
Ensis siliqua	2
Arcida	
Glycymerididae	
Glycymeris glycymeris	15
Cardiida	
Cardiidae	
Parvicardium scabrum	10
Carditida	
Astartidae	
Goodallia triangularis	4
Nuculida	
Nuculidae	
Nucula nucleus	8
Ostreida	
Ostreidae	
Ostrea edulis	1
Pectinida	
Pectinidae	
Mimachlamys varia	1
Venerida	
Mactridae	
Spisula elliptica	9
Spisula solida	7
Veneridae	
Gouldia minima	1
Polititapes rhomboides	11
Timoclea ovata	6
Venus verrucosa	1
Gastropoda	
Lepetellida	
Fissurellidae	
Diodora graeca	1
Littorinimorpha	
Calyptraeidae	
Calyptraea chinensis	21
Crepidula fornicata	16
Neogastropoda	
Buccinidae	
Buccinum undatum	3

Nassariidae	
Tritia incrassata	2
Tritia reticulata	1
Trochida	
Trochidae	
Steromphala cineraria	1
NA	
Lottiidae	
Testudinalia testudinalis	7
Polyplacophora	
Chitonida	
Callochitonidae	
Callochiton septemvalvis	1
Tonicellidae	
Tonicella marmorea	6
Lepidopleurida	
Leptochitonidae	
Leptochiton cancellatus	4
Scaphopoda	
Dentaliida	
Dentaliidae	
Antalis entalis	1
Antalis vulgaris	1
Sipuncula	
Sipunculidea	
Golfingiida	
Golfingiidae	
Golfingia elongata	2
Golfingia vulgaris vulgaris	2
Phascolionidae	
Phascolion strombus strombus	17

General discussion

Submarine power cables create various effects that can influence benthic ecosystems through direct and indirect interactions with the different associated compartments (Figure 1). In this manuscript, we particularly focused on *i*) the "reef effect", *ii*) the "reserve effect" and *iii*) the impact of electromagnetic fields associated with submarine power cables.

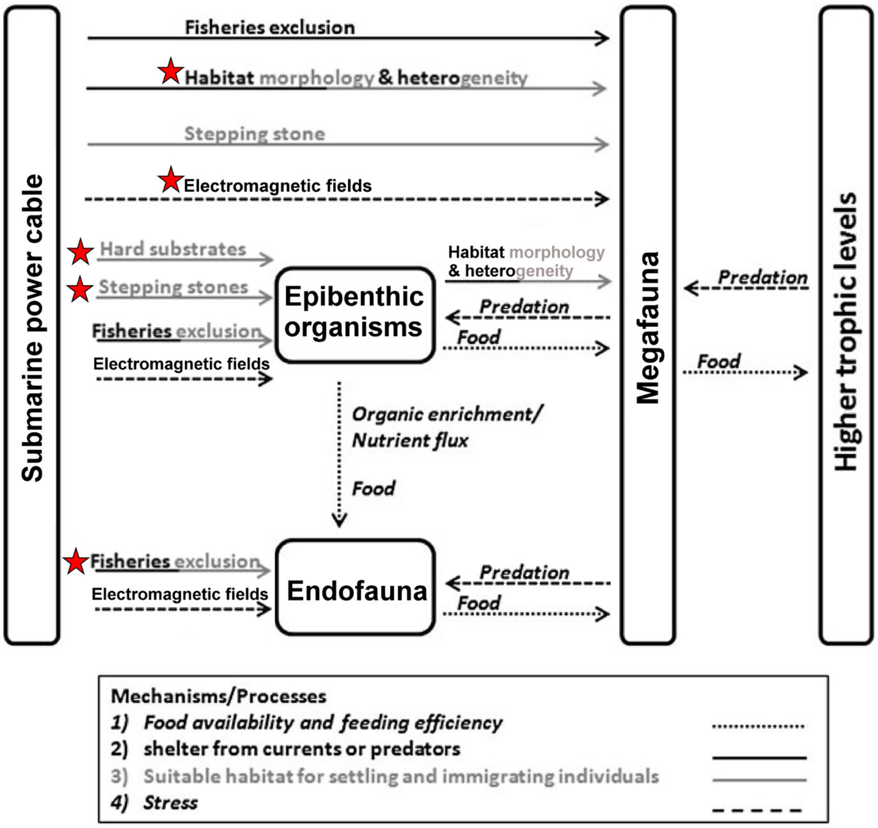


Figure 1: Schematic overview of the most important effects created by submarine power cables during the operation phase. Each mechanism/process type is indicated by a corresponding line style. Red stars indicate the particular topics covered in this manuscript (Modified from Reubens *et al.*, 2014)

Just like the strange case of Dr. Jekyll and Mr. Hyde, submarine power cables have a dual nature when considering their interactions with benthic ecosystems. Indeed, while some of their impacts are often considered as "negative" on benthic ecosystems, others can be considered as having a "positive" effect. The question that remains to be clarified is to which side the balance tilts when all these impacts are considered?

1. Dr. Jekyll

1.1 Artificial reef

1.1.1 Power cables *vs.* associated structures

Submarine power cables, when unburied, constitute a new hard substrate free for benthic colonisation creating the so-called “reef-effect”.

When talking about the reef effect associated with submarine power cables, we can discriminate *i*) **the cable itself**, naked or with its close protections (*i.e.* half-shells) and *ii*) **the associated structures**, whether for protection and stabilisation (*i.e.* mattress, rock dumping *etc.*) or for connections (*i.e.* connection hub). The cable, naked or with its close protections (hereafter simplified as “cable”), represents a long, thin cylindrical (< 20 cm diameter) artificial reef. Conversely, the associated structures can exhibit highly diverse and sometimes complex shapes (*e.g.* concrete mattress, see Figure 2 of Chapter 1 P.15 *vs.* connection hub; Figure 2.A) and are far larger (*i.e.* several cubic meters).

As both cables and associated structures constitute hard substrates, they are subject to colonisation by sessile epibenthic communities (*i.e.* fouling communities). Although both types of structures are subject to such colonisation, they can host taxonomically distinct communities, even if they are located in the same environment, due to differences in materials, elevation or shape complexity. At the Paimpol-Bréhat tidal test site, we indeed showed that concrete mattresses and half-shells associated with the cable harboured different sessile epibenthic communities although they are deployed in the same environment (see Chapter 3).

Conversely, concerning megafauna, the associated structures constitute a more efficient artificial reef than the cable itself as they are larger and more complex, offering a greater shelter potential (Figure 2.B and 2.C). In Chapter 4 for example, we showed that concrete mattresses deployed at the tidal test site of Paimpol-Bréhat were colonised by a large community of mobile

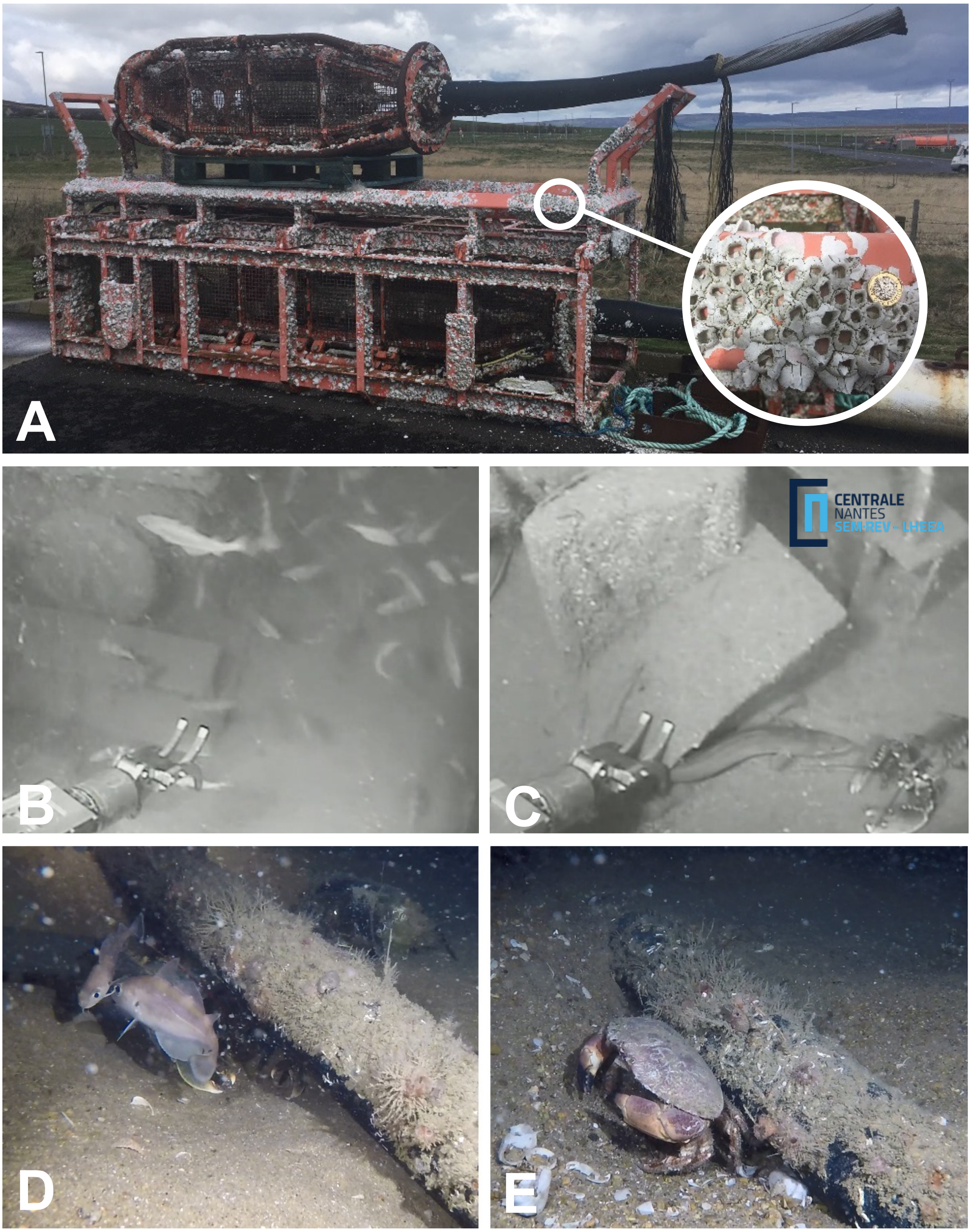


Figure 2: Different types of cables and associated structures, with benthic colonisation. (A): Two electrical connection hubs one on top of the other, used at the wave test site of EMEC (Orkney, Scotland) and close up view of barnacles colonisation after three years at sea. (B-C): Colonisation of the connection hub of the SEM-REV floating wind-turbine test-site (France) deployed on soft sediment, strong colonisation by megafauna (B: *Trisopterus sp*.; C: *Conger conger* and *Homarus gammarus*) was observed (Credits: SEM-REV test site, ECN-CNRS). (D-E):Colonisation of the laid-down cable deployed on soft sediment of the SEM-REV floating wind-turbine test-site (France), occasional colonisation by megafauna occured (D: *Trisopterus minutus* and *Homarus gammarus* ; E: *Cancer pagurus*) (Credits: © MNHN.Concarneau-René DERRIEN).

megafauna. Nevertheless, cables themselves can occasionally host megafauna individuals due to small increases in bottom complexity, for example by offering a roof under which some species (*e.g.* edible crabs) can find a shelter (Figure 2.D and 2.E).

1.1.2 Surrounding seafloor characteristics

Besides the different structures and their specificities, the potential extension of the reef effect beyond these structures greatly depends on the characteristics of the surrounding natural seafloor.

When installed in a soft sediment area, cables are usually buried in order to protect them from potential damage caused by fishing gear, anchors, strong hydrodynamic forces or storms, preventing any reef effect. Nevertheless, uncommon situations exist where the cable and/or its associated structures are not buried when deployed on a soft sediment seafloor. Cable can sometimes be simply laid down on the seafloor without burial or a protection structure, based on the view that the creation of a protected area where anthropogenic activities are prohibited will be efficient enough to prevent any kind of damage (*e.g.*, the cables Normandy 1 and 2 studied in Chapter 6). In other situations, cables can be simultaneously buried and protected by the addition of artificial structures placed on the seafloor. For example, the export cable of the SEMREV floating wind turbine test site (France) is buried along the majority of its route and, in some sections where the burial depth is considered insufficient, is protected by concrete mattresses. In such cases, unburied structures installed on soft sediments allow for the settlement of an epibenthic community otherwise absent due to the lack of natural hard substrata in the surrounding areas. For example the ATOC/Pioneer communication cable (Half Moon Bay, California), laid down in a muddy area, was colonised by a large number of *Metridium farcimen* (Kogan et al., 2006; see Figure 5.A of Chapter 1 P.25). Unburied structures also create shelters for megafauna individuals in an area with low complexity (*e.g.* Figure 2.B and 2.C), thus creating potential relay points between different natural reefs which can facilitate their movements.

On hard substrata, as burial is difficult, submarine power cables are generally laid down with their protection structures. In this case, as the presence of hard substrata is non-limiting, the importance of the reef effect can be considered as less important than on soft sediment. At its climax stage, epibenthic colonisation of artificial structures can be quite similar to that of the surrounding natural seafloor community. For example, the half-shells covering the BassLink HVDC cable between Tasmania and the Australian mainland, showed epibenthic communities similar to those of the surrounding basalt reef area 3.5 years after cable installation (See Figure 5.B Chapter 1 P.25). Nevertheless, before reaching this climax, the epibenthic community follows an ecological succession whose duration can be highly variable. In Chapter 3, we showed that the epibenthic communities associated with the artificial structures of the Paimpol-Bréhat tidal test site were not yet stabilised after 5 years of deployment. Note that the epibenthic communities present on artificial structures can be rather different to those of the surrounding hard bottom. This difference can be explained by several factors such as the

Figure 3: Colonisation of the cable deployed in a rocky environment of the SABELLA tidal test-site (France). A high density of kelps (*Laminaria hyperborea*) was observed on the naked cable (Credits: Olivier Dugornay – IFREMER).

material used or the relative elevation with respect to the seafloor For example, the unburied power cable of the SABELLA tidal test site (Brittany, France) exhibit higher densities of kelps than the surrounding rocky natural reef (Figure 3). This pattern can be mainly explained by the slightly raised position of the cable above the seafloor, sheltering young kelp recruits from sediment abrasion in comparison to the surrounding seafloor.

In the same way as for soft sediment, the shape of the associated structures can explain their habitat potential for megafauna individuals. For natural hard substrata, the interaction between artificial structures and the complex seafloor may also create additional available habitats, subsequently increasing the reef effect. In this sense, we showed in Chapter 4 that the local physical characteristics of the implantation site (*e.g.* substratum type, topography, exposure to current *etc.*) significantly impact the amount and the type of shelters provided by concrete mattresses, in turn influencing the degree of colonisation by megafauna.

1.1.3 Hybrid between soft and hard: the Paimpol-Bréhat case study

The Paimpol-Bréhat tidal test site was an interesting case study in order to provide information on the colonisation of artificial structures, as its environment presented some characteristics inherent to both soft and hard substratum areas. The local seafloor was characterised by a dominance of pebbles, the presence of coarse sand and to a lesser extent the presence of boulders. While pebbles are usually considered as a hard substratum since they allow the settlement of epibiota and prevent the presence of endofauna, their high instability as well as their low shape complexity give this habitat some properties of soft sediments. In an environmental context of strong tidal currents, the deployment of artificial structures resulted in the addition of stable substrata on bottoms where pebbles can be moved and are strongly exposed to abrasion by highly mobile particles. Consequently, both half-shells and mattresses facilitate the development of an epibenthic community with a higher structural complexity than on natural habitats in close proximity (see Chapter 3). Also, the concrete mattresses allowed the appearance of a large megafauna community in an area with low structural complexity. The extent of megafauna colonisation was due to *i*) the design of the mattresses themselves, and *ii*) their interaction with the local environmental conditions, *i.e.* irregular topography of the seafloor due to the presence of boulders (see Chapter 4).

In a nutshell, in this particular context, artificial structures allowed the installation of more complex epibenthic communities than on the natural bottom, to some extent as observed for artificial reefs installed in soft sediment areas. However, the concrete mattresses interact with the varying topographic complexity of the seafloor, creating more complex habitats for megafauna, like for artificial reefs installed on natural rocky reef.

1.1.4 Interactions between sessile epibiota and mobile megafauna

We considered sessile epibenthic and mobile megafauna communities colonising artificial structures as two distinct compartments without taking into account their potential interactions. Although our data acquisition strategies were not designed with this in mind, we discuss here how they establish relationships.

Firstly, trophic relationships probably exist as algae and animals of the epibenthic community can provide a sufficient food resource for megafauna species. A previous study demonstrated that 70% of the diet of reef fish is derived from epibiota present on the artificial structure itself *vs.* only 20-25% from the surrounding sand bottom (Johnson et al., 1994). Also, a survey of offshore windfarm populations of *Trisopterus luscus* showed that this species mainly feeds on benthic epifauna colonising the foundations (mainly small crustaceans such as *Jassa herdmani* and *Pisidia longicornis*; Reubens et al., 2011).

Secondly, some species from the epibenthic community are considered as "*autogenic engineers*" as they change their environment *via* their own physical structures (Jones et al., 1994). These species play an important role by creating new additional space and shelters for numerous macro- and megafauna species (Dayton, 1971). For example, the high degree of colonisation of the ATOC/Pioneer cable by *Metridium farcimen* (Kogan et al., 2006) increased the spatial complexity and led to a higher abundance of some fish species in close proximity to the cable. A similar pattern could also emerge for the concrete mattresses of the Paimpol-Bréhat tidal test site. Kelps of the genus *Laminaria* (see Chapter 3), a well-known engineer species which provides a large and heterogeneous habitat depending on its height (maximum of 1-2 m) and shape (Christie et al., 2003), were growing on these artificial structures. There is no doubt that, if these kelps persist over time and grow to their maximum size, they will attract a lot of additional species, including epifauna, epiphytes and fishes.

Finally, the epibenthic community can be considered as a facilitator for colonisation by megafauna individuals, mainly through *i*) trophic interaction and *ii*) habitat creation. To fully apprehend the extent of the reef effect associated with submarine power cables, and to a wider extent with all other kinds of artificial reefs, it is important to consider the whole associated ecosystem by taking into account all possible interactions between the different compartments (Figure 1).

1.1.5 An efficient artificial reef?

While submarine power cables and their associated structures constitute an artificial reef allowing the colonisation of a large range of sessile and mobile species, the true question is whether the extent of the reef effect is significant.

In comparison to other artificial reefs for which significant impacts are well established, such as shipwrecks (Krone and Schröder, 2011) wind-farm foundations (Reubens et al., 2010) or artificial reefs designed for ecosystem conservation (Jensen, 2002), cables alone can be considered as having a minor effect, if only by their spatial coverage. Indeed, even while cables can be deployed across several tens of kilometres, their limited width is expected to lead to a dilution of the associated reef effect. Consequently, cables alone will harbour an epibenthic community and will potentially play a role as a periodic relay point for several mobile megafauna species, but should not result in a drastic modification of the hosting ecosystem. Structures associated with the cable (*e.g.*, concrete mattress, rock dumping) may play a more important role. These structures, by their higher structural complexity, present similarities with the other types of artificial reefs previously mentioned, and should create significant patches allowing the sustainability of a reef community. Although a single associated structure (*e.g.* only one mattress or connection hub) may play an anecdotal role, the deployment of several units, by creating a network of artificial reefs, may have a significant influence on the surrounding area, for instance *via* unpredictable cumulative effects.

Note that, in the context of the development of marine renewable energy (MRE), the artificial structures specific to power cables will interact with diverse other artificial structures such as wind-turbine foundations or scouring protections, thus creating a wide network of diverse artificial reefs.

1.1.6 The decommissioning's issue

As submarine power cables and associated structures allow the gradual development of a new community through a reef effect, the disturbance linked to their potential decommissioning has to be considered. The complete removal of the cables and their associated structures at the end of their life expectancy, will lead to the complete removal of a specific ecosystem that took years to attain its stability (Smyth et al., 2015). This new disturbance likely acts as a 'reset' function for the community which will once again need several years to reach an ecological climax without any guarantee that the benthic community will be similar to the pre-installation community.

Commonly, legislation requires the complete removal of offshore installations at the end of their life cycle (for example for State parties to the regional OSPAR Convention). Nevertheless, some examples have demonstrated significant benefits, both environmental and commercial, of leaving the offshore structures in place, the most famous example being the "*Rigs-to-Reef*" program with oil platforms in the Gulf of Mexico (Kaiser and Pulsipher, 2005).

Concerning MRE, according to the work performed by Smyth et al. (2015), partial removal of structures associated with offshore windfarms will result in environmental and economic benefits when compared to complete removal, especially if the community created on the structures has conservation or commercial value. Considering the current increase in all different MREs, and their associated power cables, it is important to consider these different possibilities of "*Renewables-to-reefs*" (Smyth et al., 2015).

1.1.7 Underwater imagery for the study of artificial reefs

In this work, the study of the reef effect was achieved through the use of underwater images, whether photo and video, which constitute a method with several benefits. Indeed, the collection of large amounts of high-resolution information on benthic biodiversity is non-destructive, which is essential for long-term monitoring of artificial reefs. In the case of cables, it is even sometimes forbidden by industrial companies to touch them, making underwater imagery particularly well suited. Also, cameras operated by scuba divers or underwater vehicles provide access to remote sites that are difficult to sample with classic methods due to seafloor (*e.g.* hard substratum environment) or hydrodynamic (*e.g.* highly hydrodynamic environment) characteristics. Thus, it is particularly interesting in the case of MRE sites (*e.g.* tidal or wave sites; O'Carroll et al., 2017; Sheehan et al., 2018) as these sites commonly feature these harsh characteristics. Underwater imagery does not however only have advantages, indeed in terms of taxonomic precision, this method will never substitute destructive sampling, which is able to determine taxa to the specific level. Thus, if the aim of a study is to perform an exhaustive inventory of the local diversity, underwater imagery is not adequate.

Also, extracting ecologically relevant information from the large amount of raw images remains a time-consuming and somewhat laborious task. In response to the paucity of recommendations for underwater image analysis, we developed and optimised an image scoring strategy that strikes a compromise between time-efficient image annotation and accuracy to describe sessile epibenthic communities (see Chapter 2). We believe that the use of underwater imagery meets the high demand for inexpensive and time-effective tools for monitoring changes in benthic communities in the particular context of the development of marine renewable energies and should consequently be considered.

1.2 Reserve effect

In order to protect submarine power cables from damaging human activities, such cables are associated with exclusion areas where anthropogenic activities (trawl fishing, anchoring, dredging, *etc.*) are prohibited by local authorities. The size of the protected zones and the level of restriction depend on the cable installation method (buried or unburied), the number of cables present in the area, and the size of the electrical connections. These restrictions can result in positive effects for hosting ecosystems if anthropogenic activities that directly impact benthic communities previously occurred in the given area. In our study, on the protected area associated with the cables between Jersey island and France, we showed only a minor effect of the protected area on the macrobenthic communities (see Chapter 6). Such a minor effect was mainly explained by the relatively low level of anthropogenic pressure existing in the study area, even outside the protected area. Furthermore, we did not have any information about the ecological state of the area prior to the implementation of the protected area, making our conclusion on its effect on benthic community incomplete.

Protected areas associated with subsea cables present some particularities. Firstly, as such areas are designed to encompass the cable layout, they commonly have a particularly narrow and linear shape. This shape has a high perimeter-to-area ratio, which is far from being an optimal design for a protected area in conservation ecology (McLeod et al., 2009). When the perimeter-to-area ratio is high, mobile species are more likely to disperse across boundaries to

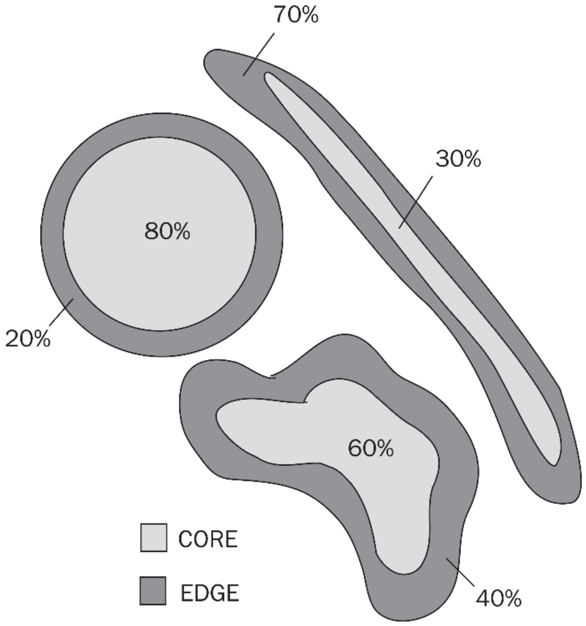


Figure 4: The edge to interior (*i.e.* core) ratio of a habitat patch is affected by patch shape. A more convoluted, irregular, or linear patch will have a higher proportion of edge. From Kennedy et al. (2003).

an unprotected area (Buechner, 1987). Congruently, a high perimeter-to-area ratio is synonymous with a high proportion of "edge habitat" (Figure 4). As edges are more likely to fall victim to noncompliant fishing activities, the ideal shape of a protected area is therefore one that minimizes edge effects while maximizing the interior protected area (McLeod et al., 2009).

While, for conservation purposes, protected areas usually focus on critical areas (*i.e.*, that are biologically or ecologically important, such as nursery grounds, spawning aggregations, and areas of high species diversity ; McLeod et al., 2009), protected areas associated with cables are not particularly designed to encompass these critical habitats. In fact, as cable layouts are commonly compelled by law to avoid these critical habitats in order not to damage them during the installation phase, they cannot benefit from the potential protection of the exclusion area. Nevertheless, such cases, in which the protected area associated with cables encompasses a critical habitat, can happen occasionally, and the protected area thus takes on a new dimension. For example, at Belle-Île-en-Mer Island (Southern Brittany, France), the presence of a submarine cable is at the origin of the single protected area which encompass part of a large Maerl bed (*i.e.* a Rhodolith accumulation), which constitutes a biogenic habitat with high ecological value (Figure 5, Dubreuil et al., 2017).

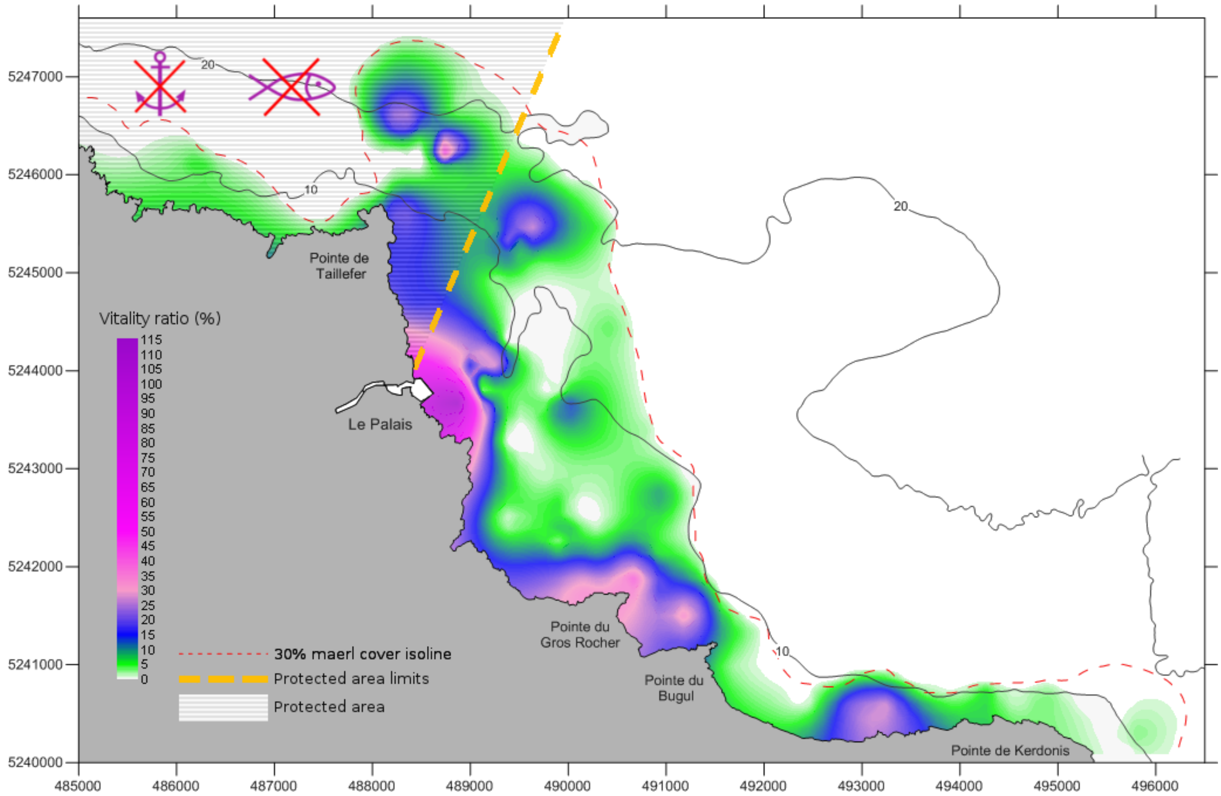


Figure 5: Vitality distribution of the Belle-Île-en-Mer maerl bed with the position of the protected area associated with the presence of several cables (Modified from Dubreuil et al., 2017).

To conclude, as the protected areas associated with submarine power cables are not designed for ecosystem conservation purpose, they are obviously not as efficient, through their design and their location, as true marine protected areas. However, even if not optimised, these protected areas can have a positive impact on marine ecosystem, through *i*) conservation, if the area was not disturbed by anthropogenic activities before the cable installation, or *ii*) restoration, if the ecosystem had poor ecological status before implementation. Finally, the benefits of these protected areas may act in synergy with the reef effect created by the artificial structures. For example, within the Dutch Egmond aan Zee offshore wind farm, the habitat heterogeneity, benthic biodiversity and the use of the area by the benthos, fishes, marine mammals and some bird species have all increased due to the protected area associated with the wind farm but also because of the reef effect of the wind turbine foundations, rockfill and cables (Lindeboom et al., 2011).

2. Mr. Hyde

2.1 Non-indigenous species: the flip-side of artificial reefs?

2.1.1 The 'stepping stone' effect

As explained in the previous section, the introduction of new artificial structures commonly results in the addition of a new epibenthic community. In some case, it can also be synonymous with the introduction of new non-indigenous species as these are often opportunistic and act as early colonisers of new artificial habitats (De Mesel et al., 2015; Frédéric Mineur et al., 2012). If these non-indigenous species persist for long enough to reproduce and emit pelagic larvae, the creation of new connectivity routes is thus possible (Adams et al., 2014; Frederic Mineur et al., 2012). The creation of these potential new connectivity routes may be created *via* a "stepping stone" process, which results from the addition of a hard substratum in an environment where it was previously absent.

Consequently, soft sediment environments are expected to be at highest risk when artificial structures are installed. But considering that within such environments, power cables are almost systematically buried, the opening of a new connectivity route *via* these structures has a low chance of occurrence. It is therefore essential to continue to bury cables as much as possible in this kind of environment. When burial is impossible, major attention must be paid to this issue.

With regard to rocky environments, when submarine power cables and associated structures are simply laid down on the seafloor, as the presence of hard substrate is not a limiting factor, the potential risk of creating a new connectivity route for non-indigenous species is very low. It remains however possible that cables and associated structures may nevertheless host non-indigenous species in higher abundance than the surrounding habitat due to characteristics specific to artificial structures (Airoldi et al., 2015), but it should not result in an expansion of their distribution range or a deep modification of the ecosystem.

"Hybrid" benthic environments, between hard bottom and soft sediment, such as the Paimpol-Bréhat tidal test site, may play a particular role in regard to this issue. As explained above for artificial reefs, these environments are commonly considered as hard habitats. But at the Paimpol-Bréhat test site, the installation of an original epibenthic community (see Chapter 3) results more from the addition of a habitat that is more stable and more sheltered from abrasion, which were the two main forcing factors, than from the addition of a hard structure *sensu stricto*. It is thus possible that this new stability/shelter offered by artificial structures benefits some non-indigenous species, offering them the possibility to open new connectivity pathways. In a nutshell, it is possible that the presence of a hard substrate does not constitute the only limiting factor to consider when looking at the stepping stone issue. According to the environmental conditions, and specifically when considering highly hydrodynamic areas, other factors such as substratum stability and elevation from the bottom may play major roles which need to be considered.

To conclude and address future prospects, it is crucial to remember that the reef effect and the potential of creating new pathways for non-indigenous species associated with submarine power cables are quite low compared to those of other kinds of artificial structures that cannot be buried. For example, in the context of the creation of MRE farms, the other artificial structures such as foundations (*e.g.* for wind turbine) or scouring protections, represent a much higher potential risk than the associated power cables due to their spatial extent and should be considered first.

2.1.2 Facilitation cascade: a secondary stepping stone?

By studying the ecological succession of the epibenthic communities colonising the artificial structures of the Paimpol-Bréhat tidal test site, we highlighted that this process was driven by a facilitation cascade (see Chapter 3; Figure 7 P.94). In other words, a succession of foundation species was involved in a hierarchy of positive facilitation interactions. The facilitation cascade may however also benefit non-indigenous species *via* the same mechanisms as for native species (Altieri et al., 2010; Gribben et al., 2019). For example, the study of facilitation cascades on cobble beaches showed that cordgrass first allows the settlement of ribbed mussels, creating in turn a nursery habitat for the invasive crab *Hemigrapsus sanguineus* (Altieri et al., 2010; Altieri and Irving, 2017; Figure 6). The presence of this nursery enhances the number of *H. sanguineus* and its coexistence with native crabs in adjacent areas outside the patches of cordgrass (Altieri and Irving, 2017).

Figure 6: Facilitation cascade of cordgrass and ribbed mussels allowing the creation of a nursery area for the invasive crab *Hemigrapsus sanguineus* (from Altieri et al., 2010).

In the case of artificial structures, this can be problematic. As we showed, through a facilitation cascade, artificial structures may host some foundation species that were absent in the surrounding habitat. These species may, in turn, allow for the settlement of non-indigenous species which can potentially lead to negative impacts on the ecosystem such as *i*) a spill-over into the area surrounding the artificial structure and *ii*) the creation of a new connectivity route through a stepping stone process. The theory underlying the opening of a new route through a stepping stone process is commonly based on the addition of a new primary substrate with new characteristics (*i.e.* hardness, stability, shelter *etc.*), enabling the settlement of non-indigenous species. This process does not consider the facilitation role of the epibenthic community that colonised the new artificial substratum, although it may play a fully-fledged role for non-indigenous species. Consequently, we can consider two different stepping stone processes for benthic species: *i*) the **primary stepping stone**: a new connectivity route is created by the addition of the artificial structure itself, and *ii*) the **secondary stepping stone**: a new connectivity route is created by the addition of facilitating species appearing because of the presence of a new artificial structure.

At the end of our survey of the epibenthic communities at the Paimpol-Bréhat site, we showed that the kelp *Laminaria sp.* started to colonise the concrete mattresses (see Chapter 4). Kelps are known to be very important structuring species; consequently if individuals persist over time on mattresses, they will allow the colonisation of a wide range of new species, and among them possibly non-indigenous species. Therefore, this again highlights the high importance of performing long-term monitoring of the epibenthic communities colonising an artificial structure, even once the climax is believed to have been reached.

2.2 Magnetic fields

2.2.1 Major concern and/or major knowledge gap?

Submarine power cables present a particularity compared to numerous other manmade structures at sea: the emission of electromagnetic fields (EMF) generated by the electric current flow passing through. The potential impact of EMF on marine life constitutes one of the major environmental concerns associated with submarine power cables. Although over the last few years, increasing attention has been paid to this question through several studies, an important knowledge gap remains (see Chapter 5). In order to help to overcome this lack of knowledge, we performed an experimental study on the impact of magnetic fields (MF) on juvenile

European lobsters (see Chapter 5). This study was one of the first to focus on the impact of MF on young life stages of invertebrates. Under our experimental design, juvenile lobsters did not exhibit any change in behaviour when submitted to an artificial MF gradient and were not influenced after one week of exposure to this MF. Our results are consistent with most of those from the literature, which have so far shown no major impacts of MF emitted by power cables on marine organisms. Concerning European lobster, uncertainties nevertheless still exist for adult individuals as no study has been conducted so far. Another interesting topic could be to investigate the potential impact of MF on the embryogenic and larval development of European lobsters, as berried females can sometimes be in direct contact with power cables when living on or within artificial reefs (*e.g.*, individuals counted under the mattress of the Paimpol-Bréhat tidal site).

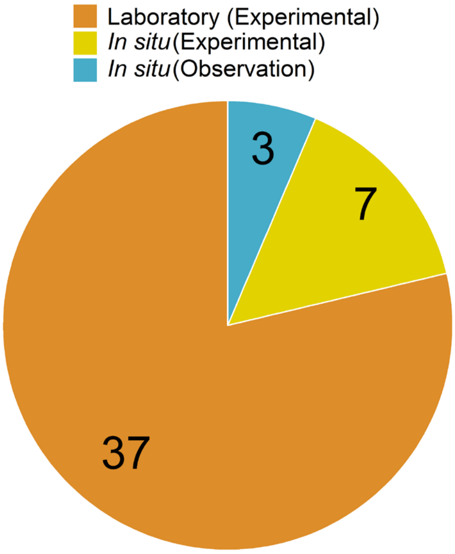


Figure 7: Approach employed by the 47 different studies (SI 1) focusing on the impacts of electromagnetic fields on aquatic life, whether in the laboratory or *in situ*.

Based on a literature review of studies on the impact of EMF on aquatic life (SI 1), it appears that the majority of studies adopted an *ex situ* experimental approach, while the number of *in situ* studies remains very low (Figure 7). The achievement of more *in situ* surveys or experiments is essential in order to fully conclude on this issue. All works we conducted on the Paimpol-Bréhat tidal test site (see Chapter 2 to 4) were performed without any electric current passing through the cable, and thus without emitted MF. Thus, it is of particular interest to continue the different megafauna and epibenthic community surveys in order to determine whether changes due to the MF (*e.g.* appearance/disappearance of particular species) are generated once the cable is connected to the tidal turbine.

2.2.2 Poorly known disturbance

It is clear that potential impacts of EMF on marine life remain a major issue but beyond that, the characterisation of the range of EMF intensities produced by existing submarine power cables is also poorly known. In experimental studies on the impacts of MF on marine life performed ex situ, the intensities used are, in most cases, higher than or equal to 1,000 µT (Figure 8), which constitute very high values. The use of such intensities is, in most cases, justified by modelled data, without any link with in situ measurements, but based on the assumption that MF can easily be simulated. Paradoxically, the low numbers of field studies performing MF measurements in situ highlighted a significantly lower range of intensities (to our knowledge, a maximum of 116.8 µT has been measured by Love et al., 2017 for a 35 kV AC cable; Figure 8). Although the MF intensity produced by a power cable highly depends on

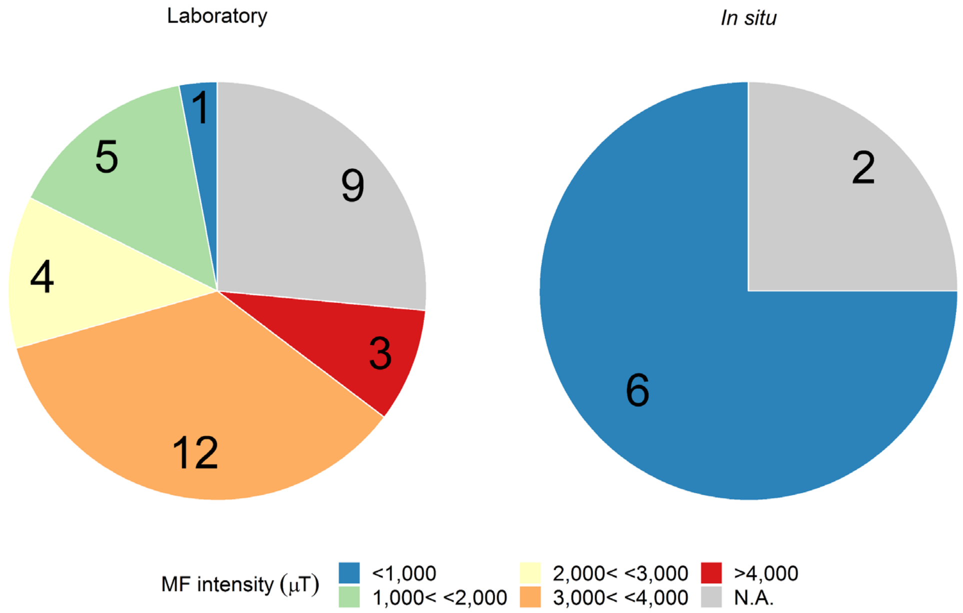


Figure 8: Magnetic field (MF) intensity used in the 42 studies (SI 1) focusing on the impact of MF on aquatic life. Studies based on laboratory experiments (left) used much higher intensities than those measured *in situ* (right).

its characteristics, a gap seems to exist between MF intensities obtained from modelling and those measured *in situ*. Hutchison et al. (2018) even showed, from *in situ* measurements, that the MF intensity produced by an alternating current power cable was significantly lower than modelled values commissioned by the grid operator. Consequently, it seems that most MF intensities used in experimental studies were never measured *in situ* and are probably unrealistic with respect to the majority of functioning submarine power cables. Even if these studies provide useful information, the transposition of results obtained experimentally to the field remains difficult. In a context where both the number of connections and the individual power of submarine power cables are quickly increasing, more *in situ* measurements of the MF intensity produced are primarily needed to better understand and evaluate the potential impact of this disturbance on marine life.

Although power cables are known to be the main man-made structure at sea that can produce EMF, other engineered devices or structures are liable to create the same kind of disturbance without any attention paid to them. For example, a rail tunnel beneath the sea (*e.g.* Channel Tunnel between France and England) produces intense EMF (and specifically electric fields) because of the high amount of electricity transiting inside through overhead lines or power cables (D'Eu, *pers. comm.*). Although such tunnels are installed deep under the seafloor, to our knowledge there is not the slightest work to quantify the EMF emitted in the hosting marine ecosystem. To be able to put into perspective the effects of EMF generated by submarine cables, it is thus very important to consider all other structures that can potentially generate electric and magnetic disturbances.

2.2.3 Electromagnetic fields: a candidate for environmental risk retirement?

Concerns about potential impacts of MRE on the marine environment contribute to slow siting and consenting of devices worldwide. These concerns are often victim to important scientific knowledge gaps which leads to heightened perceptions of risks. In order to proceed to a "retirement" of environmental risks that are likely not harmful to the marine environment, it is crucial to fill these numerous knowledge gaps.

The risks associated with EMF emission in the marine environment by power cables, which is often highlighted as an important potential impact of MRE, may constitute an interesting candidate to retirement. Indeed, as the first wave of scientific feedback tends to point to non-significant impacts on marine life, it is tempting to exclude the associated risk and to classify it as acceptable. From our point of view, feedback about the impacts of EMF is still too weak to definitively eliminate this risk. As explained previously, there is a lack of *in situ* physical characterisation of the EMF generated (which constitutes the first step to a better characterisation of the potential impacts) and an even larger lack of surveys with potentially sensitive species.

In this regard, under the Ocean Energy Systems - Environmental task, a process for retiring risks has recently been developed (Copping et al., 2019). It aims to determine which interactions of marine renewable energy devices and the marine environment constitute a low risk and may be retired, contrary to risks which need further data collection or mitigation applied to reduce the risks to an acceptable level. The first tests of this process occurred during a workshop at the EWTEC conference in Naples (2019 September) and the associated results should be published soon. The two environmental risks submitted were the effects on marine animals of *i*) underwater noise produced by marine energy devices and *ii*) of EMF from subsea power cables. We believe that such processes are essential to make progress in the scientific research on the impact of MRE and to meet the growing demand for renewable energy as quickly as possible.

3. Conclusions

Submarine power cables can thus interact in a positive and a negative way with the benthic environment, although this distinction remains subjective. Concerning the potential benefits power cables can bring to benthic ecosystems, as feedback on the effects of artificial reefs and marine protected areas is significant, associated uncertainties are quite low. On the other hand, the potential impacts of EMF and the risks of creating new connectivity routes for non-indigenous species is still poorly understood, making it difficult to draw up a complete picture. Nevertheless, in most cases, the reef effect and EMF impacts can be considered as having highly localised spatial impacts (even null for the first and reduced for the second when cables are buried) leaving only the reserve effect having a more significant spatial extent. It is important to specify that there are as many cases are there are cables, as associated impacts depend on numerous factors based on the *i*) implantation site (*e.g.* seafloor characteristics, ecosystem health before installation *etc.*) and *ii*) cable installation and characteristics (*e.g.* number and power of cables, presence of associated structures, burial of cables, size of the protected area *etc.*).

Supplementary information 1: List of the 24 publications (*i.e.* 47 study cases ; published from 1997 to 2019) on the impacts of electromagnetic fields on aquatic life used.

Ball, R. E., Oliver, M. K., and Gill, A. B. 2015. Early Life Sensory Ability — Ventilatory Responses of Thornback Ray Embryos (Raja clavata) to Predator-Type Electric Fields. Developmental neurobiology: 721–729.

Bochert, R., and Zettler, M. L. 2004. Long-term exposure of several marine benthic animals to static magnetic fields. Bioelectromagnetics, 25: 498–502.

Formicki, K., and Perkowski, T. 1998. The effect of a magnetic field on the gas exchange in rainbow trout oncorhynchus mykiss embryos (salmonidae). Italian Journal of Zoology, 65: 475–477.

Formicki, K., Sadowski, M., Tański, A., Korzelecka-Orkisz, A., and Winnicki, A. 2004. Behaviour of trout (Salmo trutta L.) larvae and fry in a constant magnetic field. Journal of Applied Ichthyology, 20: 290–294.

Greater Gabbard Offshore Winds Ltd. 2005. Greater Gabbard Offshore Wind Farm environmental statement. 672 pp.

Hays, G. C. 2013. Animal Navigation: Salmon Track Magnetic Variation. Current Biology, 23: R144–R145. Cell Press.

Hutchison, Z., Sigray, P., He, H., Gill, A. B., King, J., and Gibson, C. 2018. Electromagnetic Field (EMF) Impacts on Elasmobranch (shark, rays, and skates) and American Lobster Movement and Migration from Direct Current Cables. Sterling (VA): U.S. Department of the Interior, Bureau of Ocean Energy Management. OCS Study BOEM 2018-00. 254 pp.

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UNIVERSITE SCIENCES
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Titre : Impacts potentiels des câbles électriques sous-marins des projets d'énergies marines renouvelables sur les écosystèmes benthiques

Mots clés : Communauté benthique ; Champs électromagnétiques ; Récif artificiel ; Effet « réserve » ,Energies marines Renouvelables ; Imagerie sous-marine

Résumé :

Dans un contexte de développement rapide des projets d'énergies marines renouvelables, le but de cette thèse était d'améliorer les connaissances sur les impacts potentiels des câbles électriques sous-marins sur les écosystèmes benthiques côtiers. En se focalisant sur la phase de fonctionnement, ce travail était essentiellement dédié à la caractérisation de l'effet récif généré par ces câbles et leurs structures associées (protection, stabilisation) sur les communautés épibenthiques fixées et la mégafaune mobile. L'étude était principalement basée sur l'utilisation d'images sous-marines (photo et vidéo) prises *in situ* par des plongeurs. Ce travail a mené à des réflexions méthodologiques sur la manière la plus efficace d'analyser ce genre de données afin d'appréhender pleinement la dynamique de colonisation des structures artificielles et leur rôle d'habitat pour des espèces commerciales. Outre cet effet récif, certains organismes se retrouvent exposés à des champs magnétiques émis par les câbles électriques. Ceci m'a conduit à mesurer expérimentalement l'impact de champs magnétiques artificiels sur le comportement du homard Européen (*Homarus gammarus*) au stade juvénile. Finalement, nous avons étudié *in situ* les potentiels bénéfices pour la macrofaune benthique de l'exclusion d'activités anthropiques autour de la route de câbles électriques. Le couplage d'approches *in situ* et ex situ m'a permis de mieux appréhender les impacts environnementaux associés aux câbles électriques sous-marins. Ces résultats permettront d'améliorer l'évaluation de l'empreinte écologique des futurs raccordements électriques.

Title : Potential impacts of submarine power cables from marine renewable energy projects on benthic communities

Keywords : Benthic communities ; Electromagnetic field ; Artificial reef ; Reserve effect ; Marine renewable energy ; Underwater imagery

Abstract :

In a global context of rapid development of marine renewable energy projects, the aim of this PhD thesis was to better characterise the potential impacts of submarine power cables on coastal benthic ecosystems. The work specifically focused on the impacts associated with the operational phase. The major part of this work was dedicated to the reef effect created by these cables and their protective and stabilising structures on sessile epibenthic communities and mobile megafauna. This work was mainly based on underwater imagery, either video or photo collected *in situ* by divers. The challenge of working with underwater imagery has led me to optimise image analyses so as to effectively monitor benthic colonisation and to quantify artificial reef habitat provision to commercial species. In addition to this reef effect, colonising organisms are exposed to magnetic fields generated by the power cables. Thus, I designed an experimental study to assess the impact of realistic magnetic fields on the behaviour of juvenile European lobsters (*Homarus gammarus*). Finally, we explored the ecological impacts of excluding anthropogenic activity from the cables routes and potential benefits for benthic macrofauna. By coupling both *in situ* and *ex situ* approaches, my PhD research better characterises the environmental impacts associated with submarine power cables. These results will help to assess the ecological footprint of future power grid connections.