



Inventory of the biofouling community on the first offshore solar energy farm in the North Sea

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ABSTRACT

Offshore renewable energy is rapidly expanding in the North Sea. Offshore solar is a new renewable energy technology currently developing in the Netherlands, with the first offshore tests of four years at rough conditions being successfully completed. Submerged parts of offshore renewable energy devices get fully colonised by fouling organisms, which take advantage of the new artificial habitats. In this study, we conducted an exploration of the fouling fauna that colonised the floaters of a solar farm installed for the first time in offshore waters in the Netherlands. The biofouling attached to the underwater parts of 18 floaters forming 3 clusters were sampled in a quantitative way by scuba divers. Macrofauna species and biomass were quantified in all samples. In total, 47 different taxa, including 12 non-indigenous species, were identified to occur on the floaters. Arthropoda (mainly individuals of the genus *Jassa*) was the most abundant phylum, while Mollusca (mainly blue mussel *Mytilus edulis*) showed the largest biomass. No significant differences in abundance nor biomass were observed between the two months of installation. Non-indigenous species were more abundant in number but contributed less to the total biomass compared to native species on the examined floaters, regardless of the month they were installed. The findings of this study suggest that, like any offshore artificial structure, offshore solar farms could act as stepping-stones for the spread of species. However, long-term monitoring is needed to confirm these results and gain a more comprehensive understanding of the development of fouling fauna on offshore solar farms.

1. Introduction

Offshore renewable energy (ORE) devices are rapidly proliferating in the North Sea. Offshore wind farms are currently the most successful ORE devices, with thousands of turbines installed in different countries across the North Sea. While the offshore wind industry continues thriving, other offshore renewable technologies are also developing. Floating solar energy has made significant strides in the last years and is emerging as a promising ORE (Vlaswinkel et al., 2023), while governments are beginning to develop policies to enable the expansion of offshore solar. In the Netherlands, 70 GW of offshore wind and 45 GW of offshore solar energy are expected by 2050 (“TNO Innovation for life: Floating solar panels,” 2024). Offshore solar offers potential for co-location with offshore wind farms, allowing more efficient use of marine space and infrastructure (Ali et al., 2024), thereby supporting a more sustainable and integrated offshore energy landscape. As marine spatial planning evolves and governments begin to support innovative

blue energy technologies, offshore solar is poised to become a key player in meeting renewable energy targets with minimal spatial conflicts.

The increased number of offshore artificial structures affects the marine environment in various ways, including the introduction of electromagnetic fields, noise, and changes in hydrodynamics (Risch et al., 2024; Gill et al., 2020; Hutchison et al., 2021; Hutchison et al., 2020; Scott et al., 2018; Wilhelmsson et al., 2006). One of the most apparent effects of ORE devices is habitat provisioning through the addition of hard substrates. These new habitats are rapidly colonised by multiple fouling organisms (De Mesel et al., 2015; Degraer et al., 2020; Mavraki et al., 2023), a phenomenon known as the artificial reef effect (Langhamer, 2012). Fouling organisms increase local biodiversity (Inger et al., 2009; Knorn et al., 2024; Mavraki et al., 2020a), and attract mobile species that feed on them (Mavraki et al., 2021; Reubens et al., 2014; Reubens et al., 2011). They can further alter the local food webs (Mavraki et al., 2020a; De Borger et al., 2025) by reducing primary producers (Mavraki et al., 2020b; Slavik et al., 2019) and releasing

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organic materials, such as faeces and pseudofaeces, which increase the organic content on the seabed even kilometres away of the structures (De Berger et al., 2021; Ivanov et al., 2021). This organic material contributes to benthic food webs, providing energy to benthic organisms (Coates et al., 2014; Gergs et al., 2011), with some communities relying heavily on this resource (Norling and Kautsky, 2007). Overall, fouling organisms can significantly impact the marine ecosystem.

While the fouling fauna composition in the southern North Sea is well-studied on bottom fixed offshore wind turbines and oil and gas platforms (Coolen et al., 2022; Dannheim et al., 2025), less is known about communities on floating offshore structures. In the northern North Sea, studies exist on wave and tidal energy converting devices (Nall et al., 2017; Want et al., 2023, Want et al., 2017), buoys (MacLeod et al., 2016) and floating offshore wind turbines (Karlsson et al., 2021). Regarding floating solar in the North Sea, an earlier study of the developing fouling fauna diversity was carried out (Mavraki et al., 2023); however, this solar farm was deployed in shallow, nearshore waters and the community composition differs between coastal and offshore locations, since the sea dynamics are much rougher offshore (Kröncke et al., 2018; MacLeod et al., 2016; Reiss et al., 2006).

ORE structures may also act as stepping-stones, connecting previously isolated natural habitats via species migration across generations (Adams et al., 2014; Dauvin, 2024; Fowler et al., 2020). These structures can facilitate the spread of both native (Coolen et al., 2015; Henry et al., 2018; van der Molen et al., 2018) and non-indigenous species (Adams et al., 2014; Dauvin, 2024; De Mesel et al., 2015), by providing habitat in areas lacking natural hard substrates and accelerating species dispersal (Coolen et al., 2016; McLean et al., 2022).

In this study, we collected fouling fauna from the first offshore solar farm in the Dutch North Sea. Our aim was to investigate the fouling community composition underneath the floaters that were placed in two different seasons and identify differences in community composition with time. Furthermore, we assessed the presence of non-indigenous species and quantified their relative abundance and biomass in comparison to native species.

2. Methodology

2.1. Sampling area

The solar farm sampled in this study consisted of a set of 28 individual rigid, plastic (polymer) floaters that carry the solar panels, deployed at the Offshore Test Site in The Netherlands located 12 km off the coast of The Hague (52.17335° N - 4.119883° E, WGS84; Fig. 1). This location is dedicated to testing new innovations offshore, where they can undergo lengthy tests in rough, offshore conditions. The depth in the area ranges between 12 and 20 m, with the sea-surface temperature typically varying between 6 and 22 °C (winter and summer temperatures) and salinity values usually ~32–35 PSU. The floaters are flexibly interconnected with each other, which makes the entire structure move like a blanket on the sea surface.

In between the individual floaters are gaps that let light pass through and allow for water to splash between them, naturally cleaning the solar panels. The solar farm is anchored to the seabed at 22 m depth. Fouling fauna can colonise the entire submerged surface area of the individual floaters, which together covered a total area of 35 × 17 m. No anti-fouling measures or fouling removal activities were applied to the floaters.

Sampling was conducted on the 4th of September 2023 simultaneously at three interconnected floater clusters (B, C and D), each consisting of 7 individual floaters (Fig. 2). All floaters were sampled at the same date and time. Floater cluster A was not sampled as it had been placed in the water only 18 days before sampling took place, which was considered too short for the purpose of this study. The sampled floaters had been placed in 2 steps, first placing 9 and then 12 floaters. Floaters B5 to B7, C5 to C7 and D5 to D7 were placed in the water on the 20th



Fig. 1. Location of the offshore solar farm at the Offshore Test Site 12 km offshore The Hague, the Netherlands.



Fig. 2. Schematic overview of the solar farm layout during the time of sampling (September 2023). The numbers in the centre indicate the number of days that the floaters were in the water on the day of the sampling, and if a floater has an asterisk (*) then it was sampled. Floaters of column A were excluded from the sampling. The arrow indicates the current direction in relevance to the floater deployment.

January 2023, while floaters B1 to B4, C1 to C4 and D1 to D4 were installed on the 23rd of June 2023 (Fig. 2), where floater numbers refer to their location in the farm design and not the timing during which they were installed. The floaters were designed, constructed, installed and operated by Oceans of Energy (<https://oceansofenergy.blue>, Sassenheim, The Netherlands). In December 2023, the floaters were decommissioned and towed back to shore.

2.2. Sampling process and sample analysis

A total of 18 scrape samples were collected from three floater clusters (B, C and D), each comprising floaters installed in January and June 2023 (9 samples from each deployment). Specifically, 5 samples were taken from cluster B (3 January, 2 June 2023), 7 from C (3 January, 4 June) and 6 from D (3 from each deployment) (Fig. 2). These floater clusters were selected due to their prolonged submersion in the water, promoting fouling fauna growth.

Scrape samples were collected by two scientific-commercial scuba divers using a frame (0.05 m²) and a putty knife to scrape fouling organisms from the underside of each floater, following methods similar to Coolen et al. (2020a). One diver held the frame, while the other photographed it, then scraped the content into a pre-numbered 0.5 mm mesh net. Each net held up to maximum three separate samples, isolated from each other by cable ties. Once full with samples, the nets were passed to surface support and replaced with new nets for the collection of the following samples. On board, the nets were stored in buckets filled with sea water, which was refreshed regularly to prevent overheating.

All samples were collected during a single dive window and transported to port, where all collected fouling organisms were carefully removed from the nets and transferred into plastic containers (2.5–5 L) filled with >2 x sample volume of 99 % ethanol. Ethanol was renewed after 24 h to assure the preservation of the samples until laboratory analysis.

In the lab, samples were sieved through a 212 µm sieve and sorted into taxonomic groups. If a species exceeded 200 individuals in a sample, all individuals were retained in a sample residue. The remaining residue was sub-sampled using a Motoda-box sample splitter (Motoda, 1959) until each species group had 100–200 individuals. Sub-samples were processed separately; remaining residues were stored without further processing. Sub-sample levels (e.g., 1/2, 1/4, etc.) were recorded for identification.

Organisms were identified to the lowest possible taxonomic level under a stereomicroscope, using the World Register of Marine Species (Worms, 2016) as a reference. Juveniles or individuals with incomplete body parts were identified to higher taxonomic levels. For each taxon, abundance (number of individuals or colony size in cm²) and biomass (wet weight (WW), dry weight (DW) and ash-free dry weight (AFDW)) were recorded. DW and AFDW were measured only when WW exceeded 0.01 g. DW was obtained by drying specimens at 60 °C for 72 h and subsequently equilibrating them to room temperature in a desiccator before weighing. AFDW was measured after incineration at 560 °C for a minimum of 4 h, followed by equilibration to room temperature in a desiccator prior to final weighing.

For colonial species, the abundance was assessed as surface area (cm² of the attached part of the colony per sample), as individual counts could not be accurately estimated. These species were excluded from biomass analysis but included in the taxonomic inventory for descriptive purposes.

2.3. Data preparation and analysis

If a taxon was identified above the species level and only one species from that group was present in the sample, it was grouped with that species. If multiple species from the same taxon were present in the sample, the abundance of the higher-level taxon was proportionally distributed among those species, assuming no additional species were present. Taxa identified at higher levels were retained at that level only when no lower level taxa from the same taxon were found in the sample. In this way, the number of taxa that were identified on a high taxonomic level was reduced.

To identify differences in abundance and biomass (AFDW) of fouling fauna on the floaters between the different months of installation, two negative binomial generalised linear models (GLM) were conducted. Negative binomial models were selected instead of a standard Poisson GLM due to evidence of overdispersion in the data. The models included the month of installation as a fixed effect and were fitted using the `glm.nb()` function from the MASS package (Ripley et al., 2013) in R (version 4.4.2).

3. Results

In total, 47 unique taxa (from which 28 were identified to the species level) were detected on the sampled floaters, with 35 taxa identified on

the floaters that were installed in January and 38 taxa on floaters deployed in June 2023 (Table 1). There was a small difference between the numbers of unique taxa that colonised the floaters that were introduced in the water in different times (9 taxa on the floaters of January and 12 taxa on the floaters of June). Unique taxa identified on the floaters installed in January included the arthropod *Achelia echinata*, the tubeworms *Ficopomatus enigmaticus*, and *Hydroides ezoensis*, the brittle star class Ophiuroidea, the sea urchin *Psammechinus miliaris*, the polychaete family Serpulidae, the insect *Telmatogeton japonicus* and the bryozoan *Tricellaria inopinata*. Conversely, unique data observed on the floaters installed in June included the bryozoan.

Alcyonidium, the ascidian class Ascidiacea and the genus *Molgula*, the sea star *Asterias rubens*, the Japanese skeleton shrimp *Caprella mutica* and the family of this shrimp Caprellidae, the hydrozoans *Calycella syringa* and *Clytia hemisphaerica*, the amphipod *Stenothoe valida*, the polychaete *Syllidia armata*, the nudibranch genus *Corambe* and the polychaete family Terebellidae. From the taxa identified on species level, 12 (≈ 43 %) are known non-indigenous species in the Netherlands (Bos et al., 2016), while 19 taxa belong to a taxonomic level (above species), where it is not possible to determine whether they are native or not (and thus were characterised as NA). The other 16 species (≈ 57 %) are native for the Dutch North Sea.

The rest of the analysis was focused on the non-colonial taxa. The mean abundance of non-colonial individuals per m² occurring on the floaters deployed in January (14,902 ± 6560 standard error) was larger than of the floaters mounted in June (11,561 ± 5766). However, this difference was not considered significant based on the GLM. Arthropoda (mainly represented by amphipods belonging to the genus *Jassa*) was the most abundant phylum on all the floaters introduced to the water during both installation times (Fig. 3 – mean and standard error of abundance per m² in January: 95,306 ± 37,040; in June: 74,564 ± 34,815). The second most abundant phylum in both of the months of installation was Mollusca (floaters introduced in January: 5978 ± 3837; in June: 3758 ± 2994).

The overall biomass of the fouling fauna from the floaters installed in January (mean AFDW and standard error: 29.94 ± 14.8 g m²) was higher than that from those deployed in June (7.89 ± 4.91 g m²). This difference was again not statistically significant based on the GLM. The phylum Mollusca (mainly represented by blue mussel *Mytilus edulis*) was the most dominant on the floaters irrespective of the month of deployment (Fig. 4), with a mean AFDW of 206.27 ± 85.68 g m² (January 2023) and 53.38 ± 31.75 g m² (June 2023). The phyla with the second highest biomass were Platyhelminthes (1.671 ± 0.81 g m²) from the January deployed floaters and Arthropoda (0.656 ± 0.0977 g m²) for the ones installed in June accounted for it.

Finally, data analysis revealed that non-indigenous species were more abundant than native species on floaters deployed both in January and June (Fig. 5). However, when exploring the same results but with the relative biomass as factor, the percentage of native species surpassed that of non-indigenous species.

4. Discussion

This study explored the fouling fauna composition occurring on the first demonstrator offshore solar farm in the Dutch North Sea. Arthropoda was consistently the most abundant phylum, irrespective of the duration for which the floaters remained in the water or the season in which they were deployed. Mollusca, and specifically blue mussels *Mytilus edulis*, contained the highest biomass on the floaters introduced in both months (Table 1). Both abundance and biomass were greater on floaters deployed in January than on those installed in June, indicating that the longer time since deployment had led to slightly more advanced communities with higher individual growth, whereas the June-associated communities were likely still in the early stages of development. No statistically significant differences in abundance and biomass were detected between the floaters installed in the two different months.

Table 1

List of taxa collected on the floaters that were installed in January and in June. The total number of individuals per m² collected per month (or total surface area covered by the colonial taxa per m²), and total ash-free dry weight (AFDW in g m⁻²) are presented. The status indicates whether they are native species for the Dutch North Sea or non-indigenous (NIS), while the NAs indicate that the taxonomic level did not allow for the status identification. The phyla with an asterisk (*) include the colonial taxa, which were not included in further analysis.

Taxon	Mean abundance of individuals (n m ⁻² or cm ² m ⁻²) ± SE		Mean AFDW (g m ⁻²) ± SE		Status
	January	June	January	June	
Annelida					
<i>Eunereis longissima</i>	2.22 ± 0.80	4.44 ± 1.60	0.0004 ± 0.0001	0.02 ± 0.008	Native
<i>Ficopomatus enigmaticus</i>	2.22 ± 0.80	–	0	–	NIS
<i>Hydroides ezoensis</i>	2.22 ± 0.80	–	0	–	NIS
Myrianida	6.67 ± 2.39	68.89 ± 32.39	0	0	NA
Nereididae	8.89 ± 3.19	8.89 ± 3.19	0	0	NA
Serpulidae	2.22 ± 0.80	–	0	–	NA
<i>Syllidia armata</i>	–	2.22 ± 0.80	–	0	NIS
Terebellidae	–	2.22 ± 0.80	–	0	NA
Arthropoda					
<i>Achelia echinata</i>	4.44 ± 1.60	–	0	–	Native
<i>Austrominius modestus</i>	161 ± 96.4	298.6 ± 108.53	0	0	NIS
Balanomorpha	2.22 ± 0.80	–	0	–	NA
<i>Balanus crenatus</i>	1215.9 ± 946.95	1725.76 ± 598.27	0	0	Native
<i>Bodotria scorpioides</i>	22.22 ± 6.27	106.67 ± 38.32	0	0	Native
<i>Caprella equilibra</i>	2160 ± 1183.97	3043.55 ± 1890.89	0.06 ± 0.03	0.044 ± 0.03	NIS
<i>Caprella mutica</i>	–	1000.89 ± 317.07	–	0.01 ± 0.003	NIS
Caprellidae	–	88.89 ± 31.93	–	0	NA
<i>Jassa herdmani</i>	2626.59 ± 2164.3	254.67 ± 178.37	0.02 ± 0.01	0.0002 ± 0.0002	Native
<i>Jassa marmorata</i>	88,682.5 ± 37,282.6	67,623.1 ± 33,857.94	0.47 ± 0.11	0.60 ± 0.113	NIS
<i>Monocorophium acherusicum</i>	301.78 ± 101.42	124.44 ± 26.71	0	0	NIS
<i>Perforatus perforatus</i>	113.87 ± 68.44	144.48 ± 41.53	0	0	Native
<i>Pilumnus hirtellus</i>	4.44 ± 1.06	80 ± 25.34	0	0.0004 ± 0.0001	Native
<i>Pseudocuma (Pseudocuma) simile</i>	8.89 ± 3.19	37.78 ± 9	0	0	Native
<i>Stenothoe valida</i>	–	35.56 ± 12.77	–	0	Native
<i>Telmatogeton japonicus</i>	2.22 ± 0.80	–	0	–	NIS
Chordata					
Ascidiacea	–	15.56 ± 5.59	–	0.12 ± 0.04	NA
<i>Botryllus schlosseri</i>	2.22 ± 0.80	2.22 ± 0.80	0	0	NIS
<i>Molgula</i>	–	4.44 ± 1.06	–	0	NA
<i>Molgula manhattensis</i>	8.889 ± 3.19	2.22 ± 0.80	0	0.03 ± 0.01	NIS

Table 1 (continued)

Taxon	Mean abundance of individuals (n m ⁻² or cm ² m ⁻²) ± SE		Mean AFDW (g m ⁻²) ± SE		Status
	January	June	January	June	
Molgulidae	31.11 ± 11.18	20 ± 7.18	0.34 ± 0.12	0.30 ± 0.11	NA
Echinodermata					
<i>Asterias rubens</i>	–	44.44 ± 15.96	–	0.049 ± 0.02	Native
Ophiuroidea	2.22 ± 0.80	–	0	–	NA
<i>Psammechinus miliaris</i>	11.11 ± 3.19	–	0.19 ± 0.07	–	Native
Mollusca					
Corambe	–	42.22	–	0	NA
Mytilidae	2.22 ± 0.80	–	0	–	NA
<i>Mytilus edulis</i>	5846.67 ± 1382.98	3657.78 ± 1079.97	206.27 ± 30.78	53.38 ± 11.41	Native
Nudibranchia	129.56 ± 20.41	57.78 ± 14.54	0	0.002 ± 0.0008	NA
Nemertea					
Nemertea	64.44 ± 12.60	15.56 ± 3.12	0.58 ± 0.16	0.29 ± 0.08	NA
Platyhelmyntes					
Platyhelmyntes	2887.78 ± 415.43	2415.56 ± 535.78	1.67 ± 0.29	0.39 ± 0.09	NA
Bryozoa*					
<i>Amathia</i>	300	1100	0	0	NA
<i>Bugulina stolonifera</i>	20	100	0	0	NIS
<i>Conopeum reticulum</i>	20	40	0	0	Native
<i>Electra pilosa</i>	3920	4660	0	0	Native
Cnidaria*					
<i>Alcyonium</i>	–	400	–	0	NA
Campanulariidae	340	3600	0	0	NA
<i>Calycella syringa</i>	–	2800	–	0	Native
<i>Clytia hemisphaerica</i>	–	1200	–	0	Native
Tubulariidae	960	23,600	0	0	NA

However, notable variations in biomass, as measured by ash-free dry weight (AFDW), were observed. Although the statistical model did not identify these differences as significant, the observed trend may reflect the limited sample size, which may have reduced the power to detect true differences. Finally, when abundance was used as the metric, non-indigenous species represented the highest proportion relative to native species on the floaters installed in both of the months; however, this pattern was reversed when biomass was considered as the basis for the comparison.

The blue mussel *Mytilus edulis*, showed the largest biomass on the floaters installed in both seasons (mean AFDW of *M. edulis* in January 206.27 g m⁻² and in June 53.38 g m⁻² – Table 1). When a small-sized offshore solar farm of 1 MW (ca. 6400 m²) is deployed, then mussel biomass on the floaters of the entire farm could reach a total of approximately 1,320,128 g AFDW in less than a year after installation. This conclusion is drawn from the AFDW results for mussel biomass on the floaters installed in January. The substantial mussel colonisation may indirectly influence the surrounding marine environment. Mussels can become dislodged and fall to the seafloor beneath the floaters, a phenomenon already observed under mussel farms (Mascorda-Cabre et al., 2024). These accumulated mussels and mussel shells could create reef-like habitats on the seafloor, supporting local ecosystem services (Coral et al., 2025) and facilitating the settlement and growth of larvae (Buschbaum, 2001; Commito et al., 2014). Over time, this process may lead to the formation of mussel beds underneath the solar farms, further enhancing habitat complexity, providing shelter for diverse species, and promoting local biodiversity (Bridger et al., 2022; Degraer et al., 2020;

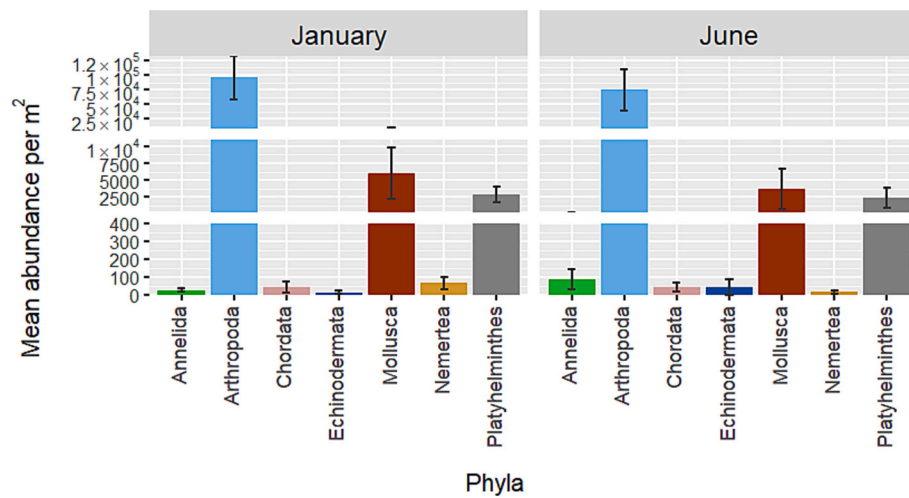


Fig. 3. Mean abundance with standard errors of the different non-colonial phyla per m² colonising the floaters in the two different months of installation (January and June 2023).

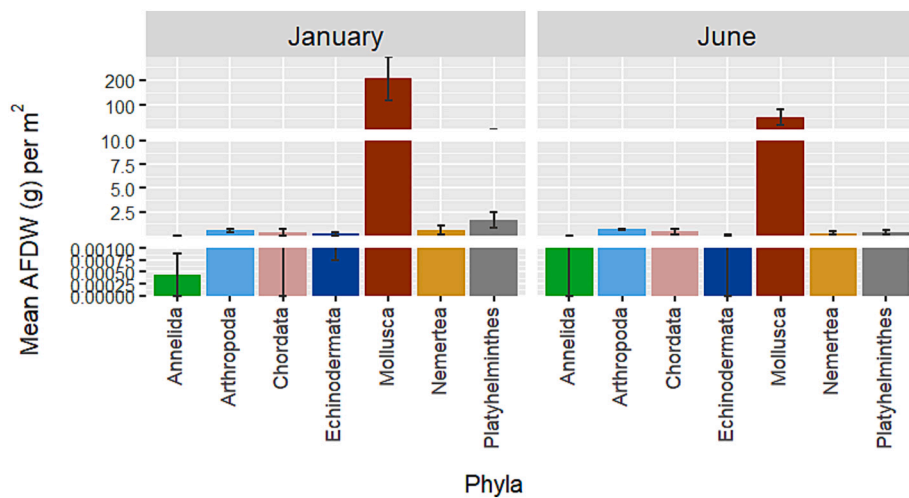


Fig. 4. Mean ash-free dry weight (AFDW in grams) per m² with standard errors of the different non-colonial phyla occurring on the floaters installed in the two different months (January and June 2023).

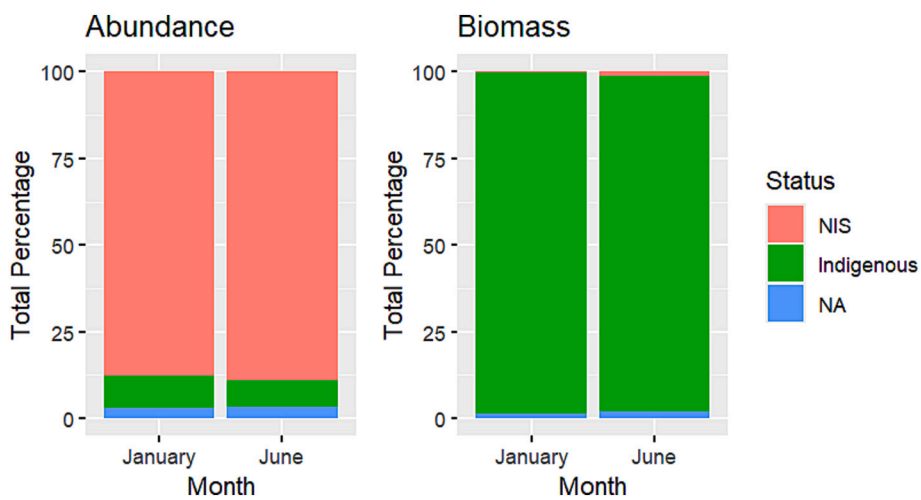


Fig. 5. Relative percentages of the abundance (left) and biomass (right) of non-indigenous (red) and native species (green) on the floaters installed in January and June 2023. The NAs (blue) represent the taxa that could not be identified to the species level and, thus, cannot be determined as native or non-indigenous. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Krone et al., 2013). Furthermore, this reef formation could enhance the presence of commercially important species, while it could increase food web complexity and connectivity (Coral et al., 2025). Mussel bed formation will be highly dependent on local hydrodynamic conditions and sediment transport processes, as strong currents can displace mussel shells over considerable distances, while active sand waves may result in burial of mussels beneath sediment layers (Donker, 2015; Widdows et al., 1998).

However, it is important to note that mussel biomass is unlikely to be evenly distributed across the entire solar farm. A recent preliminary model studying the interactions between offshore solar farms and mussel growth focused on farms of up to 1 km² in the Dutch North Sea, suggested that mussel growth varies across the farm area (Nalmpanti et al., 2025). Higher biomass tends to occur at the edges of the farm, particularly in areas aligned with dominant current directions, while lower growth is observed toward the centre of larger farms (Nalmpanti et al., 2025). Based on this study, the observed pattern is driven by differences in food availability, as mussels at the edges benefit from a continuous supply of chlorophyll-a-rich water, while mussels in the centre experience more depleted water due to upstream filtration. This spatial heterogeneity should be considered when estimating overall biomass and evaluating potential ecological impacts. Whether this pattern actually happens at large-scale solar farms still needs to be investigated.

The floaters in this study were placed in the water at different seasons, winter (January) and early summer (June) 2023, while sampling took place in September 2023. As the floaters remained in the water for less than a year before sampling, the fouling communities developed on them can be considered to be at an early succession stage. The short-term community succession on artificial structures is strongly influenced by the timing of installation, given that larval supply in temperate regions is inherently season-dependent (Schröder et al., 2006). This seasonality can in turn influence the species composition colonising the available substrata (Kerckhof et al., 2010). For the dominant phyla colonising the floaters (Arthropoda in terms of abundance and Mollusca in terms of biomass), seasonal variation in floater deployment did not seem to affect their presence or distribution. Mussels in the southern North Sea undergo spawning between late April and late June (Sprung, 1983). This spawning period has allowed the colonisation of mussels on all the floaters under study, even the ones deployed in June. Conversely, the most frequently observed amphipod species on the floaters, *Jassa marmorata* and *Jassa herdmani*, exhibit continuous year-round reproductive activities (Beermann, 2013; Beermann and Purz, 2013). Therefore, the season/timing of floater deployment did not have a significant effect on the occurrence of the most prevalent taxa colonising the floaters.

Community seasonality patterns have been observed multiple times on anthropogenic structures, such as offshore wind turbine foundations (De Mesel et al., 2015; Kerckhof et al., 2010) and also on a coastally located pilot solar farm (Mavraki et al., 2023). The floaters of this study were deployed at different times and contained almost the same number of unique species. The presence of some species only on the floaters installed in one of the two seasons could be explained by the seasonal patterns that these species exhibit, or by inter- and intra-specific competition for space. The Japanese skeleton shrimp *Caprella mutica* has established populations in temperate areas, while its population peaks between June and October (Ashton et al., 2010). This species was found only on the floaters installed in June 2023. *Caprella mutica* does not have free-swimming larvae and expands its distribution by attaching on floating structures (Minchin and Holmes, 2007). Thus, it is possible that it occupied the available habitat of the floaters installed in June and expanded its population on those floaters, where competition of the available habitat was smaller. On the other hand, *Psammechinus miliaris*, a sea urchin that was only observed on the floaters deployed in January 2023, produces young individuals in December–January (Jensen, 1969), which then colonised the only available habitat at the time.

Multiple studies have suggested that offshore anthropogenic structures, including ORE devices, facilitate the settlement and potential reproduction of indigenous and non-indigenous species, functioning as stepping stones that enhance their distribution (Adams et al., 2014; Boehlert and Gill, 2010; Coolen et al., 2020b; De Mesel et al., 2015; Leclerc et al., 2020; Taormina et al., 2020; Vaselli et al., 2008). In the current study, the relative contribution of non-indigenous species to the overall fouling community differed depending on the factor under which it was studied. When looking at the relative abundance of non-indigenous species, it was clear that they were exceeding that of native species. This result was caused by the abundant presence of mainly one non-indigenous species, the amphipod *Jassa marmorata* (defined as NIS in Bos et al., 2016), which reached abundances higher than 80,000 individuals m⁻² on the examined floaters. The other non-indigenous species found in this study, were similar to the ones observed at the intertidal zones of wind turbine foundations in the southern North Sea (Dauvin, 2024; De Mesel et al., 2015). When, however, biomass was the main factor under which the percentage of non-indigenous species was examined, the native species exceeded the non-indigenous ones. This was mainly due to the presence of the blue mussel *Mytilus edulis*, which might colonise the floaters with half the abundance of *Jassa marmorata*, but its individuals are much heavier than that of the amphipods.

Non-indigenous species can be opportunistic and act as early colonisers of new artificial substrata, while they can affect the composition of fouling communities by outcompeting other fouling species (Sheehy and Vik, 2010). In areas where hard substratum is limited, such as the offshore southern North Sea, the introduction of artificial hard substrates may favour the recruitment of non-indigenous species over native ones, promoting the stepping-stone effect (Airoldi et al., 2015). With the continuous installation of ORE devices, the spread of non-indigenous species could rapidly expand, especially at depths that have been observed to accommodate non-indigenous species, such as the intertidal zone (Kerckhof et al., 2011). This higher occurrence of non-indigenous species at intertidal parts of ORE could be explained by the fact that most of these species are typically found in coastal habitats and they seem to be capable of surviving in offshore environments as long as suitable habitat is present (Kerckhof et al., 2011).

During decommissioning, the floaters were towed to Scheveningen and removed from the water. With future floating solar initiatives, the effect that decommissioning might have on the spread of non-indigenous species should be considered. Recent advice has highlighted the potential risk that the relocation of floating wind turbines could have on the spread of non-indigenous species between different eco-regions (ICES, 2025). Although the floaters studied here were only transported across short distances, future installations might be towed further offshore, and the risks of transporting native and non-indigenous species might increase.

When considering the importance of floating offshore solar for the spread of non-indigenous and indigenous species alike, it is important to take into account the high numbers of already present artificial structures in the North Sea. For example, tens of thousands of shipwrecks, offshore wind turbines, oil and gas platforms, navigational buoys and other hard substrates are present (Coolen et al., 2020b). Many of these substrates can be found on the water surface, creating conditions comparable to those provided by floating solar. In light of this broader network of potential stepping-stones, the specific contribution of floating solar to this phenomenon remains unclear.

5. Conclusions

This study analysed the composition of fouling fauna on the first offshore solar farm in the Dutch North Sea, highlighting the significant abundance of Arthropoda and biomass of Mollusca on all the floaters, independently of the season of installation. However, the floaters installed in January had generally higher abundance and biomass

compared to the ones deployed in June, as more time was available for the biofouling communities to develop and for the individuals to grow. In total, 47 unique taxa were detected on the sampled floaters, with 35 taxa identified on the floaters that were installed in January and 38 taxa on floaters deployed in June 2023.

The findings provide a first indication that offshore solar farms may serve as stepping stones for the distribution of non-indigenous species, although the significance of this effect should be tested based on abundance and biomass together with the opportunistic behaviour of the non-indigenous species present. Further research is required to strengthen the evidence of these results and to gain a deeper understanding of the (long-term) development of fouling fauna on floaters of offshore solar farms. Finally, attention should be given to the cumulative effects of offshore renewable energy infrastructure. When considered together, wind farms, solar farms and other artificial structures may not only influence species dispersal patterns but also offer opportunities to enhance ecosystem services, such as food provisioning as offshore wind farms can provide an attractive habitat to multiple commercial species due to the increased food and shelter availability (Hooper et al., 2017). Adopting a holistic, ecosystem-based perspective will be essential to maximize the environmental benefits while mitigating potential risks associated with large-scale offshore developments.

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CRedit authorship contribution statement

Ninon Mavraki: Writing – original draft, Project administration, Formal analysis, Conceptualization. **Oscar G. Bos:** Writing – review & editing, Validation, Data curation. **Babeth van der Weide:** Data curation. **Oliver Bittner:** Data curation. **Brigitte M. Vlaswinkel:** Writing – review & editing, Project administration, Funding acquisition. **Melina Nalmpanti:** Writing – review & editing. **Joop W.P. Coolen:** Writing – review & editing, Methodology, Funding acquisition, Data curation.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

All data used for this research are presented in the manuscript.

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References

Adams, T.P., Miller, R.G., Aleynik, D., Burrows, M.T., 2014. Offshore marine renewable energy devices as stepping stones across biogeographical boundaries. *J. Appl. Ecol.* 51, 330–338. <https://doi.org/10.1111/1365-2664.12207>.
 Airoidi, L., Turon, X., Perkol-Finkel, S., Rius, M., 2015. Corridors for aliens but not for natives: effects of marine urban sprawl at a regional scale. *Divers. Distrib.* 21, 755–768. <https://doi.org/10.1111/ddi.12301>.

Ali, F., Etemad-Shahidi, A., Stewart, R.A., Sanjari, M.J., Hayward, J.A., Nicholson, R.C., 2024. Co-located offshore wind and floating solar farms: a systematic quantitative literature review of site selection criteria. *Renewable Energy Focus* 50, 100611. <https://doi.org/10.1016/j.ref.2024.100611>.
 Ashton, G.V., Burrows, M.T., Willis, K.J., Cook, E.J., 2010. Seasonal population dynamics of the non-native *Caprella mutica* (Crustacea, Amphipoda) on the west coast of Scotland. *Mar. Freshw. Res.* 61, 549–559.
 Beermann, J., 2013. Ecological Differentiation among Amphipod Species in Marine Fouling Communities: Studies on Sympatric Species of the Genus *Jassa* Leach, 1814 (Crustacea, Amphipoda).
 Beermann, J., Purz, A.K., 2013. Comparison of life history parameters in coexisting species of the genus *Jassa* (Amphipoda, Ischyroceridae). *J. Crustac. Biol.* 33, 784–792. <https://doi.org/10.1163/1937240X-00002190>.
 Boehlert, G.W., Gill, A.B., 2010. Environmental and ecological effects of ocean renewable energy development. *Oceanography* 23, 68–81. <https://doi.org/10.2307/24860713>.
 Bos, O.G., Gittenberger, A., de Boois, L.J., van Asch, M., van der Wal, J.T., Cremer, J., van der Hoorn, B., Pieterse, S., Bakker, P.A.J., 2016. Soortenlijst Nederlandse Noordzee. <https://doi.org/10.18174/401117>.
 Bridger, D., Attrill, M.J., Davies, B.F.R., Holmes, L.A., Cartwright, A., Rees, S.E., Cabre, L.M., Sheehan, E.V., 2022. The restoration potential of offshore mussel farming on degraded seabed habitat. *Aquaculture, Fish and Fisheries* 2, 437–449. <https://doi.org/10.1002/aff2.77>.
 Buschbaum, C., 2001. Selective settlement of the barnacle *Semibalanus balanoides* (L.) facilitates its growth and reproduction on mussel beds in the Wadden Sea. *Helgol. Mar. Res.* 55, 128–134. <https://doi.org/10.1007/s101520100070>.
 Coates, D.A., Deschutter, Y., Vincx, M., Vanaverbeke, J., 2014. Enrichment and shifts in macrobenthic assemblages in an offshore wind farm area in the Belgian part of the North Sea. *Mar. Environ. Res.* 95, 1–12. <https://doi.org/10.1016/j.marenvres.2013.12.008>.
 Commito, J.A., Commito, A.E., Platt, R.V., Grupe, B.M., Piniak, W.E.D., Gownaris, N.J., Reeves, K.A., Vissicelli, A.M., 2014. Recruitment facilitation and spatial pattern formation in soft-bottom mussel beds. *Ecosphere* 5. <https://doi.org/10.1890/ES14-00200.1>.
 Coolen, J.W.P., Bos, O.G., Glorius, S., Lengkeek, W., Cuperus, J., van der Weide, B., Agüera, A., 2015. Reefs, sand and reef-like sand: a comparison of the benthic biodiversity of habitats in the Dutch Borkum reef grounds. *J. Sea Res.* 103, 84–92. <https://doi.org/10.1016/j.seares.2015.06.010>.
 Coolen, J.W.P., Lengkeek, W., Degraer, S., Kerckhof, F., Kirkwood, R.J., Lindeboom, H.J., 2016. Distribution of the invasive *Caprella mutica* Schurin, 1935 and native *Caprella linearis* (Linnaeus, 1767) on artificial hard substrates in the North Sea: separation by habitat. *Aquat. Invasions* 11, 437–449. <https://doi.org/10.3391/ai.2016.11.4.08>.
 Coolen, J.W.P., Bittner, O., Driessen, F.M.F., van Dongen, U., Siahaya, M.S., de Groot, W., Mavraki, N., Bolam, S.G., van der Weide, B., 2020a. Ecological implications of removing a concrete gas platform in the North Sea. *J. Sea Res.* 166, 101968. <https://doi.org/10.1016/j.seares.2020.101968>.
 Coolen, J.W.P., Boon, A.R., Crooijmans, R., van Pelt, H., Kleissen, F., Gerla, D., Beermann, J., Birchenough, S.N.R., Becking, L.E., Luttkhuizen, P.C., 2020b. Marine stepping-stones: connectivity of *Mytilus edulis* populations between offshore energy installations. *Mol. Ecol.* 29, 686–703. <https://doi.org/10.1111/mec.15364>.
 Coolen, J.W.P., Vanaverbeke, J., Dannheim, J., Garcia, C., Birchenough, S., Krone, R., Beermann, J., 2022. Generalized changes of benthic communities after construction of wind farms in the southern North Sea. *J. Environ. Manag.* 315.
 Coral, C., Kornau, L.M., van der Heide, T., Coolen, J.W.P., Witbaard, R., Bouma, T.J., Christianen, M.J.A., 2025. Subtidal temperate reefs in marginal seas enhance biodiversity, food web complexity, and ecosystem stability. *Mar. Ecol. Prog. Ser.* 764, 15–32. <https://doi.org/10.3354/meps14883>.
 Dannheim, J., Kloss, P., Vanaverbeke, J., Mavraki, N., Zupan, M., Spielmann, V., Degraer, S., Birchenough, S.N.R., Janas, U., Sheehan, E., Teschke, K., Gill, A.B., Hutchison, Z., Carey, D.A., Rasser, M., Buyse, J., van der Weide, B., Bittner, O., Causon, P., Krone, R., Faasse, M., Wrede, A., Coolen, J.W.P., 2025. Biodiversity information of benthic species at Artificial structures – BISAR. *Sci Data* 12, 604. <https://doi.org/10.1038/s41597-025-04920-1>.
 Dauvin, J.C., 2024. Do offshore wind farms promote the expansion and proliferation of non-indigenous invertebrate species? *Mar. Pollut. Bull.* 206. <https://doi.org/10.1016/j.marpolbul.2024.116802>.
 De Borger, E., Ivanov, E., Capet, A., Braeckman, U., Vanaverbeke, J., Soetaert, K., 2021. Offshore windfarm footprint of sediment organic matter mineralization processes. *Front. Mar. Sci.* 8, 632243. <https://doi.org/10.3389/fmars.2021.632243>.
 De Borger, E., van Oevelen, D., Mavraki, N., De Backer, A., Braeckman, U., Soetaert, K., Vanaverbeke, J., 2025. Offshore wind farms modify coastal food web dynamics by enhancing suspension feeder pathways. *Commun Earth Environ* 6, 330. <https://doi.org/10.1038/s43247-025-02253-w>.
 De Mesel, I., Kerckhof, F., Norro, A., Rumes, B., Degraer, S., 2015. Succession and seasonal dynamics of the epifauna community on offshore wind farm foundations and their role as stepping stones for non-indigenous species. *Hydrobiologia* 756, 37–50. <https://doi.org/10.1007/s10750-014-2157-1>.
 Degraer, S., Carey, D.A., Coolen, J.W.P., Hutchison, Z.L., Kerckhof, F., Rumes, B., Vanaverbeke, J., 2020. Offshore wind farm artificial reefs affect ecosystem structure and functioning: a synthesis. *Oceanography* 33, 48–57.
 Donker, J., 2015. Hydrodynamic Processes and the Stability of Intertidal Mussel Beds in the Dutch Wadden Sea. University of Utrecht, Utrecht.
 Fowler, A.M., Jørgensen, A., Coolen, J.W.P., Brabant, R., Rumes, B., Degraer, S., 2020. The ecology of infrastructure decommissioning in the North Sea: what we need to know and how to achieve it. *ICES J. Mar. Sci.* 77, 1109–1126. <https://doi.org/10.1093/icesjms/fsz143>.

- Gergs, R., Grey, J., Rothhaupt, K.-O., 2011. Temporal variation in zebra mussel (*Dreissena polymorpha*) density structure the benthic food web and community composition on hard substrates in Lake Constance, Germany. *Biol. Invasions* 13, 2727–2738. <https://doi.org/10.1007/s10530-011-9943-8>.
- Gill, A.B., Degraer, S., Lipsky, A., Mavraki, N., Methratta, E., Brabant, R., 2020. Setting the context for offshore wind development effects on fish and fisheries. *Oceanography* 33, 118–127.
- Henry, L.A., Mayorga-Adame, C.G., Fox, A.D., Polton, J.A., Ferris, J.S., McLellan, F., McCabe, C., Kutti, T., Roberts, J.M., 2018. Ocean sprawl facilitates dispersal and connectivity of protected species. *Sci. Rep.* 8. <https://doi.org/10.1038/s41598-018-29575-4>.
- Hooper, T., Beaumont, N., Hattam, C., 2017. The implications of energy systems for ecosystem services: a detailed case study of offshore wind. *Renew. Sust. Energy Rev.* 70, 230–241. <https://doi.org/10.1016/j.rser.2016.11.248>.
- Hutchison, Z.L., Gill, A.B., Sigra, P., He, H., King, J.W., 2020. Anthropogenic electromagnetic fields (EMF) influence the behaviour of bottom-dwelling marine species. *Sci. Rep.* 10. <https://doi.org/10.1038/s41598-020-60793-x>.
- Hutchison, Z.L., Gill, A.B., Sigra, P., He, H., King, J.W., 2021. A modelling evaluation of electromagnetic fields emitted by buried subsea power cables and encountered by marine animals: considerations for marine renewable energy development. *Renew. Energy* 177, 72–81. <https://doi.org/10.1016/j.renene.2021.05.041>.
- Inger, R., Attrill, M.J., Bearhop, S., Broderick, A.C., Grecian, W.J., Hodgson, D.J., Mills, C., Sheehan, E., Votier, S.C., Witt, M.J., Godley, B.J., 2009. Marine renewable energy: potential benefits to biodiversity? An urgent call for research. *J. Appl. Ecol.* 46, 1145–1153. <https://doi.org/10.1111/j.1365-2664.2009.01697.x>.
- Ivanov, E., Capet, A., De Borger, E., Degraer, S., Delhez, E.J.M., Soetaert, K., Vanaverbeke, J., Grégoire, M., Grégoire, G., 2021. Offshore wind farm footprint on organic and mineral particle flux to the bottom. *Front. Mar. Sci.* 8.
- Jensen, M., 1969. Breeding and growth of *Psammechinus miliaris* (Gmelin). *Ophelia* 7, 65–78. <https://doi.org/10.1080/00785326.1969.10419289>.
- Karlsson, R., Tivefålh, M., Duranovi, I., Kjøllhamar, A., Murvold, K.M., 2021. Artificial hard substrate colonisation in the offshore Hywind Scotland pilot park. *Wind Energy Science* 1–18.
- Kerckhof, F., Degraer, S., Norro, A., Rumes, B., 2011. Offshore intertidal hard substrata: a new habitat promoting non-indigenous species in the Southern North Sea: an exploratory study. In: *Offshore wind farms in the Belgian Part of the North Sea: Selected findings from the baseline and targeted monitoring*. In: Degraer, S., Rumes, B., Brabant, R., Vigin, L. (Eds.), Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit. Royal Belgian Institute of Brussels, pp. 27–37.
- Kerckhof, F., Rumes, B., Jacques, T., Degraer, S., Norro, A., 2010. Early development of the subtidal marine biofouling on a concrete offshore windmill foundation on the Thornton Bank (southern North Sea): first monitoring results. *Underw. Technol.* 29, 137–149. <https://doi.org/10.3723/ut.29.137>.
- Knorn, A.H., Teder, T., Kaasik, A., Kreitsberg, R., 2024. Beneath the blades: marine wind farms support parts of local biodiversity – a systematic review. *Sci. Total Environ.* 935. <https://doi.org/10.1016/j.scitotenv.2024.173241>.
- Kröncke, I., Becker, L.R., Badewien, T.H., Bartholomä, A., Schulz, A.C., Zielinski, O., 2018. Near- and offshore macrofauna communities and their physical environment in a South-Eastern North Sea sandy beach system. *Front. Mar. Sci.* 5. <https://doi.org/10.3389/fmars.2018.00497>.
- Krone, R., Gutow, L., Joschko, T.J., Schröder, A., 2013. Epifauna dynamics at an offshore foundation - implications of future wind power farming in the North Sea. *Mar. Environ. Res.* 85, 1–12. <https://doi.org/10.1016/j.marenvres.2012.12.004>.
- Langhamer, O., 2012. Artificial reef effect in relation to offshore renewable energy conversion: state of the art. *Sci. World J.* 2012, 1–8. <https://doi.org/10.1100/2012/386713>.
- Leclerc, J.C., Viard, F., González Sepúlveda, E., Díaz, C., Neira Hinojosa, J., Pérez Aranedá, K., Silva, F., Brante, A., 2020. Habitat type drives the distribution of non-indigenous species in fouling communities regardless of associated maritime traffic. *Divers. Distrib.* 26, 62–75. <https://doi.org/10.1111/ddi.12997>.
- MacLeod, A.K., Stanley, M.S., Day, J.G., Cook, E.J., 2016. Biofouling community composition across a range of environmental conditions and geographical locations suitable for floating marine renewable energy generation. *Biofouling* 32, 261–276. <https://doi.org/10.1080/08927014.2015.1136822>.
- Mascorda-Cabre, L., Hosegood, P., Attrill, M.J., Sheehan, E.V., 2024. Biogenic reef creation and biodiversity enhancement by an offshore longline mussel farm. *Ecol. Indic.* 167. <https://doi.org/10.1016/j.ecolind.2024.112708>.
- Mavraki, N., Degraer, S., Moens, T., Vanaverbeke, J., 2020a. Functional differences in trophic structure of offshore wind farm communities: a stable isotope study. *Mar. Environ. Res.* 157, 104868. <https://doi.org/10.1016/j.marenvres.2019.104868>.
- Mavraki, N., Degraer, S., Vanaverbeke, J., Braeckman, U., 2020b. Organic matter assimilation by hard substrate fauna in an offshore wind farm area: a pulse-chase study. *ICES J. Mar. Sci.* 77, 2681–2693.
- Mavraki, N., Degraer, S., Vanaverbeke, J., 2021. Offshore wind farms and the attraction–production hypothesis: insights from a combination of stomach content and stable isotope analyses. *Hydrobiologia* 848, 1639–1657. <https://doi.org/10.1007/s10750-021-04553-6>.
- Mavraki, N., Bos, O.G., Vlaswinkel, B.M., Roos, P., de Groot, W., van der Weide, B., Bittner, O., Coolen, J.W.P., 2023. Fouling community composition on a pilot floating solar-energy installation in the coastal Dutch North Sea. *Front. Mar. Sci.* 10. <https://doi.org/10.3389/fmars.2023.1223766>.
- McLean, D.L., Ferreira, L.C., Benthuisen, J.A., Miller, K.J., Schläppy, M.L., Ajemian, M. J., Berry, O., Birchenough, S.N.R., Bond, T., Boschetti, F., Bull, A.S., Claisse, J.T., Condie, S.A., Consoli, P., Coolen, J.W.P., Elliott, M., Fortune, I.S., Fowler, A.M., Gillanders, B.M., Harrison, H.B., Hart, K.M., Henry, L.A., Hewitt, C.L., Hicks, N., Hock, K., Hyder, K., Love, M., Macreadie, P.I., Miller, R.J., Montevecchi, W.A., Nishimoto, M.M., Page, H.M., Paterson, D.M., Pattiaratchi, C.B., Pecl, G.T., Porter, J. S., Reeves, D.B., Riginos, C., Rouse, S., Russell, D.J.F., Sherman, C.D.H., Teilmann, J., Todd, V.L.G., Trembl, E.A., Williamson, D.H., Thums, M., 2022. Influence of offshore oil and gas structures on seascape ecological connectivity. *Glob. Chang. Biol.* 28, 3515–3536. <https://doi.org/10.1111/gcb.16134>.
- Minchin, D., Holmes, J.M.C., 2007. The first record of *Caprella mutica* Schurin, 1935 (Crustacea: Amphipoda) from the east coast of Ireland. *Ir Nat J* 28, 321–323.
- Motoda, S., 1959. Devices of simple plankton apparatus. *Memoirs of the faculty of fisheries Hokkaido University* 1–2, 73–94.
- Nall, C.R., Schläppy, M.L., Guerin, A.J., 2017. Characterisation of the biofouling community on a floating wave energy device. *Biofouling* 33, 379–396.
- Nalmpanti, M., Dinandra, T., Maar, M., Larsen, J., Vlaswinkel, B., 2025. Offshore solar farms as habitats for *Mytilus edulis*: a preliminary modelling study on mussel growth, distribution, chlorophyll-a uptake and bio-deposition in the North Sea. *Mar. Environ. Res.* 107372. <https://doi.org/10.1016/j.marenvres.2025.107372>.
- Norling, P., Kautsky, N., 2007. Structural and functional effects of *Mytilus edulis* on diversity of associated species and ecosystem functioning. *Mar. Ecol. Prog. Ser.* 351, 163–175. <https://doi.org/10.3354/meps07033>.
- Reiss, H., Meybohm, K., Kröncke, I., 2006. Cold winter effects on benthic macrofauna communities in near- and offshore regions of the North Sea. *Helgol. Mar. Res.* 60, 224–238. <https://doi.org/10.1007/s10152-006-0038-3>.
- Reubens, J.T., Degraer, S., Vincx, M., 2011. Aggregation and feeding behaviour of pouting (*Trisopterus luscus*) at wind turbines in the Belgian part of the North Sea. *Fish. Res.* 108, 223–227. <https://doi.org/10.1016/j.fishres.2010.11.025>.
- Reubens, J.T., De Rijcke, M., Degraer, S., Vincx, M., 2014. Diel variation in feeding and movement patterns of juvenile Atlantic cod at offshore wind farms. *J. Sea Res.* 85, 214–221. <https://doi.org/10.1016/j.seares.2013.05.005>.
- Ripley, B., Venables, B., Bates, D.M., Hornik, K., Gebhardt, A., Firth, D., 2013. *Package MASS*. Springer, New York. <https://doi.org/10.32614/CRAN.package.MASS>.
- Risch, D., Marmo, B., van Geel, N.C.F., Gillespie, D., Hastie, G., Sparling, C.E., Onoufriou, J., Wilson, B., 2024. Underwater noise of two operational tidal stream turbines: A comparison. In: Popper, A.N., Sisneros, J.A., Hawkins, A.D., Thomsen, F. (Eds.), *The Effects of Noise on Aquatic Life*. Springer International Publishing AG, pp. 157–178.
- Schröder, A., Orejas, C., Joschko, T., 2006. Benthos in the Vicinity of Piles: FINO 1 (North Sea).
- Scott, K., Harsanyi, P., Lyndon, A.R., 2018. Understanding the effects of electromagnetic field emissions from marine renewable energy devices (MREDS) on the commercially important edible crab, *Cancer pagurus* (L.). *Mar. Pollut. Bull.* 131, 580–588. <https://doi.org/10.1016/j.marpolbul.2018.04.062>.
- Sheehy, D.J., Vik, S.F., 2010. The role of constructed reefs in non-indigenous species introductions and range expansions. *Ecol. Eng.* 36, 1–11. <https://doi.org/10.1016/j.ecoleng.2009.09.012>.
- Slavik, K., Lemmen, C., Zhang, W., Kerimoglu, O., Klingbeil, K., Wirtz, K.W., 2019. The large-scale impact of offshore wind farm structures on pelagic primary productivity in the southern North Sea. *Hydrobiologia* 845, 35–53. <https://doi.org/10.1007/s10750-018-3653-5>.
- Sprung, M., 1983. *Reproduction and fecundity of the mussel Mytilus edulis at Helgoland (North Sea)*. *Helgoländer Meeresun.* 36, 243–255.
- Taormina, B., Percheron, A., Marzloff, M.P., Caisey, X., Quillien, N., Lejart, M., Desroy, N., Dugornay, O., Carlier, A., 2020. Succession in epibenthic communities on artificial reefs associated with marine renewable energy facilities within a tide-swept environment. *ICES J. Mar. Sci.* 77, 2656–2668. <https://doi.org/10.1093/icesjms/faa129>.
- TNO Innovation for life: Floating solar panels, 2024.
- van der Molen, J., García-García, L.M., Whomersley, P., Callaway, A., Posen, P.E., Hyder, K., 2018. Connectivity of larval stages of sedentary marine communities between hard substrates and offshore structures in the North Sea. *Sci. Rep.* 8. <https://doi.org/10.1038/s41598-018-32912-2>.
- Vaselli, S., Bulleri, F., Benedetti-Cecchi, L., 2008. Hard coastal-defence structures as habitats for native and exotic rocky-bottom species. *Mar. Environ. Res.* 66, 395–403. <https://doi.org/10.1016/j.marenvres.2008.06.002>.
- Vlaswinkel, B., Roos, P., Nelissen, M., 2023. Environmental observations at the first offshore solar farm in the North Sea. *Sustainability* 15, 6533. <https://doi.org/10.3390/su15086533>.
- Want, A., Crawford, R., Kakkonen, J., Kiddie, G., Miller, S., Harris, R.E., Porter, J.S., 2017. Biodiversity characterisation and hydrodynamic consequences of marine fouling communities on marine renewable energy infrastructure in the Orkney Islands archipelago, Scotland, UK. *Biofouling* 33, 567–579. <https://doi.org/10.1080/08927014.2017.1336229>.
- Want, A., Goubard, A., Jonveaux, S., Leaver, D., Bell, M.C., 2023. Key biofouling organisms in tidal habitats targeted by the offshore renewable energy sector in the North Atlantic include the massive barnacle *Chirona hameri*. *J. Mar. Sci. Eng.* 11, 2168. <https://doi.org/10.3390/jmse11112168>.
- Widdows, J., Brinsley, M.D., Salkeld, P.N., Elliott, M., 1998. Use of annular flumes to determine the influence of current velocity and bivalves on material flux at the sediment-water interface. *Estuaries* 21, 552–559. <https://doi.org/10.2307/1353294>.
- Wilhelmsson, D., Yahya, S.A.S., Öhman, M.C., 2006. Effects of high-relief structures on cold temperate fish assemblages: a field experiment. *Mar. Biol. Res.* 2, 136–147. <https://doi.org/10.1080/17451000600684359>.