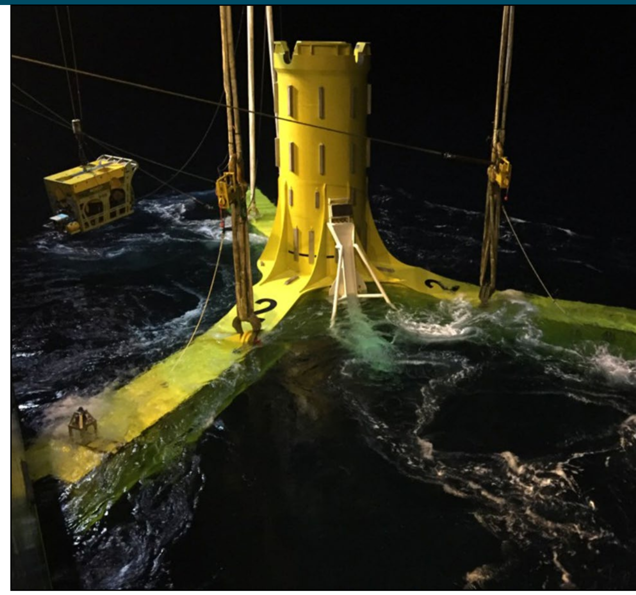

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Changes in Benthic and Pelagic Habitats Caused by Marine Renewable Energy Devices

Most marine renewable energy (MRE) devices must be attached to the seafloor in some way, either through gravity foundations, pilings, or anchors, and with mooring lines, transmission cables, and devices themselves in the water column. Physical changes in benthic and pelagic habitats have the potential to alter species occurrence or abundance at a localized scale, lead to some level of habitat loss, provide opportunities for colonization by non-native species, alter patterns of ecological succession, modify ecosystem functioning, and affect behavioral responses of marine organisms. The transformation of the seafloor and/or water column habitat to new hard substratum because of the presence of the MRE devices may also lead to artificial reef effects or changes in animal behavior.

While there is no indication that MRE devices affect marine habitats differently than other structures currently and historically placed in the ocean, regulators and stakeholders may continue to have concerns.

6.1.
IMPORTANCE OF THE ISSUE

The potential changes in marine habitats induced by MRE may be similar to those of other industries that interact with the seabed and/or have water column or surface expression, like offshore wind farms (OWFs), oil and gas platforms (OGPs), navigation buoys, or communication cables. Regulators and stakeholders have raised concerns about several effects on marine habitats caused by these other industries (e.g., modification of benthic and pelagic habitats, artificial reef effect, biofouling by non-native species). As Want and Porter (2018) wrote, "with a general trend towards stricter statutory environmental controls, the onus will be on the MRE industry to demonstrate minimal disturbance." Deploying single MRE devices and/or arrays of devices in a sustainable way means assuring that environmental risks related to a change in habitat (especially habitats for threatened or endangered species) are identified at each site, avoided, managed, and/or mitigated. Experience at OWFs provides evidence that local biodiversity may drastically change in the vicinity of an MRE device over time, thereby modifying the resilience of the ecosystem (Causon and Gill 2018). However, because marine ecosystems are exposed to natural environmental fluctuations at various temporal and spatial scales, the ability to detect changes due to anthropogenic pressures will depend on the robustness of the survey design (Bicknell et al. 2019; Sheehan et al. 2018). In addition, the cumulative effects of activities across diverse sectors may be substantial at the scale of an MRE deployment site and will need to be taken into account to understand and manage changes in the marine environment (Causon and Gill 2018; Wilding et al. 2017).

The distribution of benthic communities is strongly influenced by the depth and characteristics of the seafloor as well as the current speed, and few studies have described the natural variability of assemblages in high-energy-flow environments (Kregting et al. 2016). The exploitation of tidal energy requires high tidal velocities that are usually associated with a seafloor dominated by coarse sediments, boulders, or rocky outcrops. Benthic communities associated with these habitats are typically stress-tolerant, opportunistic organisms that are highly influenced by physical processes and natural variability, such as current velocity and sediment dynamics (Kregting et al. 2016; O'Carroll et al. 2017a). These environments are often rich in biodiversity and there are concerns that the turbulent wake of a tidal turbine might alter the local benthic communities (Kregting et al. 2016; O'Carroll et al. 2017a). The wake may also alter the phytoplankton and primary production in the water column, especially near large-scale arrays that may have the potential to change the hydrodynamics of the ambient flow (Schuchert et al. 2018). Laying cable may prove challenging in such environments, compared to those that feature a soft-sediment seafloor, and pose risks of damaging benthic habitats (Taormina et al. 2018).

Any structure left long enough in the marine environment has the potential to be colonized by fouling organisms and then act as an artificial reef by attracting fish and other mobile animals; MRE devices are no different, especially because of their seabed moorings and associated infrastructures (Alexander et al. 2016). While a single tidal turbine or wave energy converter (WEC) has a relatively limited ecological footprint, an array of devices may act as a network of interconnected artificial reef, in a way similar to that of OWFs (Causon and Gill 2018). This reef effect may spread at the ecosystem scale, with yet-to-be-identified effects on the structure and functioning of local and regional food webs (Raoux et al. 2017).

As the worldwide economy keeps growing and maritime shipping lanes expand, dispersion and propagation of non-native species is becoming a more prominent issue for the marine environment, especially in nearshore habitats. MRE devices may act as "stepping stones" for many of these non-native species to colonize new places and cross biogeographical barriers (Adams et al. 2014; Wilding et al. 2017). The connectedness of deployment sites with harbors and marinas, more particularly those where non-native species have been documented to occur, is an important consideration to keep in mind during the initial planning of a project (Bray et al. 2017).

6.2.
SUMMARY OF KNOWLEDGE THROUGH 2016

Before 2016, there were only a few deployed wave and tidal devices, notably the SeaGen tidal turbine in Northern Ireland, the OpenHydro tidal generator in the Orkney Islands of Scotland, European Marine Energy Centre tidal devices in the Orkney Islands, and the Lysekil WEC in Sweden. OWFs have been found to be reasonably comparable to MRE devices in terms of their effects on artificial reef and benthic habitats (Kramer et al. 2015), and they were used as a surrogate for many of the analyzed effects of wave and tidal devices in 2016. Additional structures in the ocean, such as fish aggregating devices, offshore oil platforms, sunken vessels, artificial reefs, and navigation buoys, were also used as surrogate devices for predicting the effects of MRE devices on benthic habitats (Arena et al. 2007; Clynick et al. 2008; Kramer et al. 2015; Page et al. 1999; Vaselli et al. 2008; Wehkamp and Fischer 2013).

By 2016, several studies showed no impacts of MRE devices or OWF locations on benthic communities or species abundance (De Backer et al. 2014; Lindeboom et al. 2011, 2015; Wilhelmsson et al. 2006). Other studies examining benthic communities at the deployed OpenHydro tidal device in the Orkney Islands, Scotland, found increased abundance and diversity of fish and predators over time compared to a control site (Broadhurst et al. 2014; Broadhurst and Orme 2014). Benthic organisms and fish at the Lysekil WEC project site in Sweden were found to have higher biomass, density, species richness, and species diversity than the reference location because of the increased structural complexity of the seabed at the foundations, although the results were not statistically significant (Langhamer 2010; Langhamer and Wilhelmsson 2009).

At the SeaGen tidal turbine, organisms including mussels, barnacles, brittle stars, crabs, and more, have been found to colonize structures on the seafloor and in the water column (Keenan et al. 2011). Colonization of the vertical structure of offshore wind pilings by species such as blue mussels (*Mytilus edulis*) led to the creation of new habitats and thus colonization by other benthic organisms and reef fish (Krone et al. 2013; Maar et al. 2009). Keenan et al. (2011) also reported that benthic communities were different during each subsequent survey at the SeaGen tidal turbine. Changes detected in benthic communities over time were attributed to temporal variability and natural processes including species competition and succession. Overall, changes in community composition were similar across all sampling stations and the reference station. Under natural ocean conditions, benthic communities undergo succession with changes in the dominant species as the communities reach a dynamic mature state. This pattern of succession and the time needed to reach the mature state must be considered when monitoring benthic communities around MRE devices to determine whether changes are natural or caused by the presence of an MRE device or array.

Concerns have been expressed about MRE devices potentially providing opportunities for non-native species to colonize new areas and spread across habitats, especially with the additional connectivity provided by MRE arrays (Adams et al. 2014; Mineur et al. 2012). Although there have been reports of non-native species colonizing underwater structures associated with offshore wind devices (Langhamer 2012), few studies have examined the mechanisms for dissemination of non-native species or suggested that MRE devices pose a higher risk for invasions than other existing marine installations (Mineur et al. 2012).

The *2016 State of the Science* report (Copping et al. 2016) identified the following data gaps and priorities for future research regarding changes in habitats:

- Determine the effects of MRE devices (wave and tidal) in the field on benthic habitats, as opposed to relying on surrogate structures.
- Address the potential benthic and artificial reef effects from arrays or co-located wave and tidal sites to determine their cumulative impacts.
- Develop a framework of ecosystem changes that incorporates the potential for cascading effects as well as natural patterns of succession.
- Validate models of community change and artificial reef effects with field data.
- Determine whether MRE devices create novel stepping stones for non-native species.
- Monitor impacts on benthic communities at existing wave and tidal locations to evaluate and determine the extent of the response to installation and operation of MRE devices.

6.3.
KNOWLEDGE GENERATED SINCE 2016

Several different types of WECs and tidal turbines have been tested in real conditions over the last decade at various locations. However, few have stayed in the water long enough (i.e., several years) to monitor and observe persistent or long-term environmental changes caused by the presence and functioning of the device. Most of the knowledge related to changes in habitats caused by MRE devices still comes from surrogate industries like OWFs, OGPs, or power and communication cables (Dannheim et al. 2019), as well as from a few modeling studies. However, the hard and sturdy structures of most OWFs and OGPs span the entire water column from the seafloor to the surface, while most MRE devices are either bottom-mounted without surface expression or floating and attached to the seafloor by mooring structures (e.g., Figure 6.1). Knowledge transfer from surrogate industries thus depends on the context.

Two main types of changes for the benthic and pelagic habitats are generated by MRE devices (Figure 6.1): damaging effects (e.g., trenching, footprint effect) and creation of habitats (e.g., biofouling, artificial reef, reserve effect). These habitat changes may also lead to indirect effects, for example facilitating the propagation of non-native invasive species.

6.3.1.
ALTERATION OF EXISTING HABITATS AND RECOVERY TIMEFRAMES

The installation and operation of MRE devices may lead to alteration and/or loss of existing benthic habitats, for example during cable installation or due to turbulence and scouring around device and mooring foundations.

Trenching and Digging for Installation of Devices and Cables

There is currently a great diversity of tidal turbine and WEC technology designs, most of them floating or bottom-mounted. The loss of benthic habitat due to the footprint of anchors and foundations is widely acknowledged by decision-makers, particularly when vulnerable marine ecosystems or other fragile habitats have been identified during the siting process and avoidance and mitigation measures are taken (Greaves and Iglesias 2018). Cable laying to link MRE devices to an offshore substation and/or the onshore grid may lead

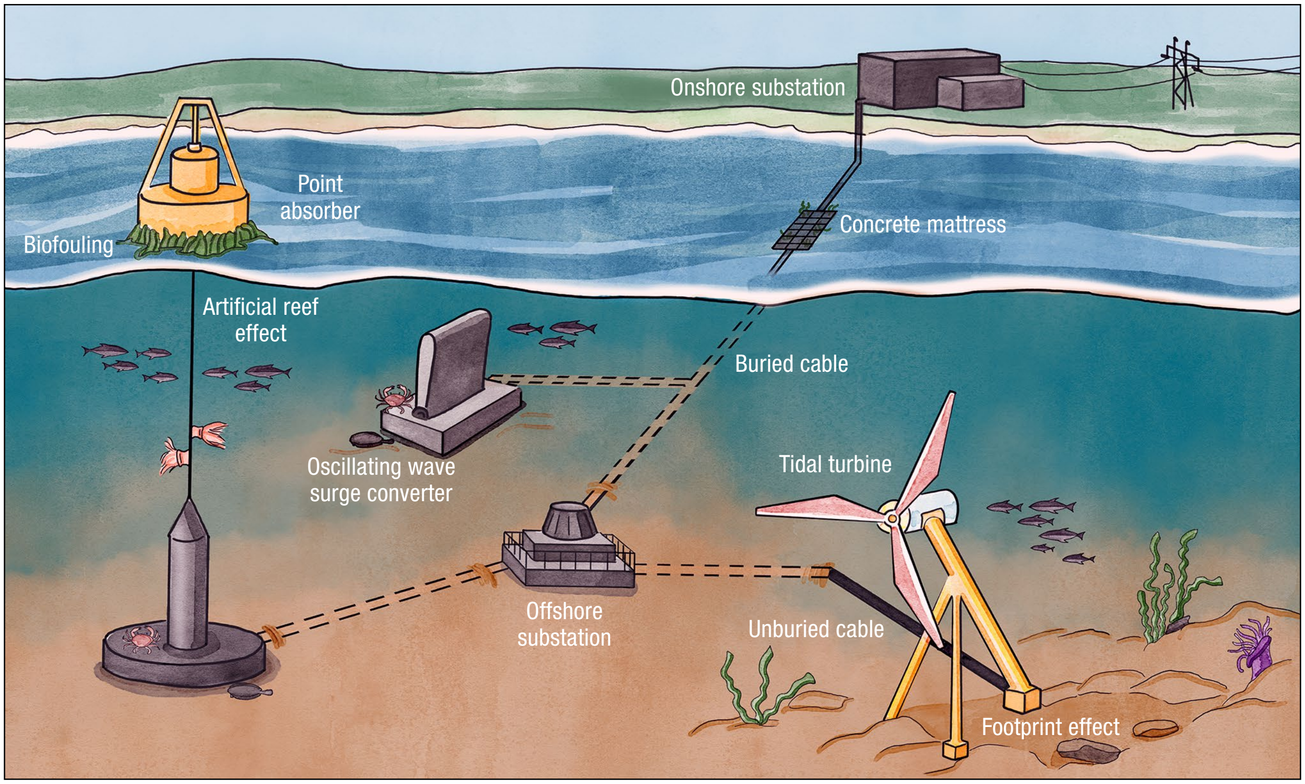

Figure 6.1. Schematic of various wave and tidal energy devices, and associated equipment, and their potential effects on the benthic and pelagic habitats. (Illustration by Rose Perry)

to direct disturbance or alteration of a much larger area of benthic habitats (i.e., following a path on the seafloor hundreds to thousands meters long), even though the physical disturbance of the seabed is very limited compared to other human activities, such as bottom fishing or deep-sea mining (Taormina et al. 2018). The cable-laying method used depends on the nature of the seafloor, and each method may result in different spatial and temporal scales of damage (Kraus and Carter 2018; Taormina et al. 2018).

Jetting and ploughing are among the favored methods for burying cables in soft sediments, the former resulting in a much wider disturbance strip than the latter (100 to 2000 m and 2 to 8 m respectively; Kraus and Carter 2018; Taormina et al. 2018). Depending on the wave and current dynamics, turbidity resulting from cable laying can persist for several days, thereby limiting the available light for primary producers, reducing prey detectability for fish and filtration efficiency for suspension-feeders. However, these effects are short-term, and resuspended sediment tends to settle in a matter of days (Taormina et al. 2018). Habitat recovery is site-specific, but seafloors where jetting or ploughing have been used to lay cables have shown rates of full recovery to pre-trenching benthic communities from two weeks to six years, similar to recovery rates for the sediment itself (Kraus and Carter 2018 and examples therein; Sheehan et al. 2018; Taormina et al. 2018). A subsequent effect of cables buried in the sediment is the localized increase in temperature at the cable-sediment interface, which has unknown consequences for benthic organisms (Taormina et al. 2018 and references therein).

Where the seafloor is dominated by unconsolidated or consolidated hard substrate, cables are usually laid on top of the sediment, sometimes encased in protective iron pipes or covered with concrete mattresses (Figure 6.2) or natural rocks (Kraus and Carter 2018; Sheehan et al. 2018; Taormina et al. 2018). In this case, disturbance is limited to the footprint of the cable itself and its protection material, unless unstabilized portions of the cable drag the surrounding seafloor if caught up in local hydrodynamic disturbances (Dunham et al. 2015; Taormina et al. 2018). Direct impacts of such methods of cable laying are the crushing, damaging, or displacement of organisms (Dunham et al. 2015; Taormina et al. 2018). However, unless cables are laid on slow-growing taxa like glass sponge reefs (Dunham et al. 2015), colonization of the iron, concrete, or rocky cable protections by encrusting organisms may lead to full recovery of the disturbed seafloor to the pre-cable state. Recovery has happened within one to eight years (Kraus and Carter 2018 and examples therein; Sheehan et al. 2018; Taormina et al. 2018), in some cases showing evidence of successful ecological successions (Sheehan et al. 2018).

The recovery timeframe for benthic communities after buried or unburied cable laying may be difficult to distinguish from natural variability (Dunham et al. 2015; Kraus and Carter 2018; Sheehan et al. 2018), and post-installation monitoring might be needed over the span of a few years to assess whether mitigation measures are necessary along the cable route. Monitoring may be required over longer periods of time in areas where fragile and/or slow-growing engineer species (e.g., seagrass meadows) cannot technically be avoided by a cable route.

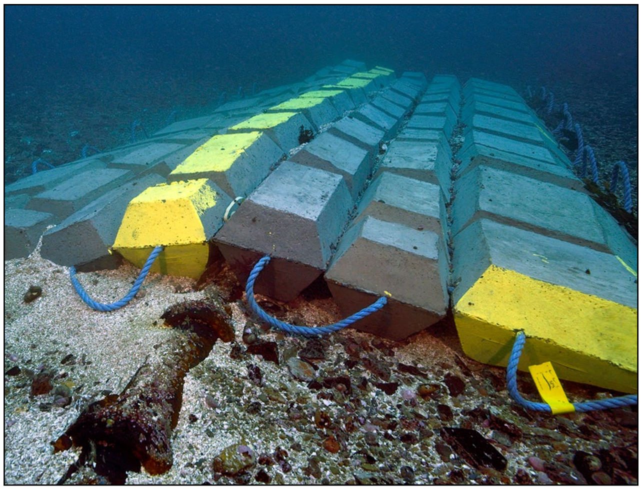

Figure 6.2. Pictures of iron shells and concrete mattresses used to protect an unburied cable at the Paimpol-Bréhat tidal turbine test site in France. The picture on the left was taken one month after the installation of the concrete mattress in 2013 (photo courtesy of Olivier Dugornay, Ifremer), and the picture on the right was taken six years later during a video survey (photo courtesy of Ifremer).

Scouring by Local Turbulences during Operation: The Footprint Effect

While the loss of seafloor habitat directly under the anchors or foundations of MRE devices is inevitable and should be mitigated during the siting process, further loss of benthic habitats during operation due to scouring by local turbulence in the immediate vicinity of the anchors and/or foundations (i.e., the footprint effect) is also a concern. This concern has been assessed and measured in real conditions involving tidal turbines (Kregting et al. 2016; O'Carroll et al. 2017a; O'Carroll et al. 2017b), concrete anchors on soft sediments (Henkel 2016), and artificial structures in an estuary (Mendoza and Henkel 2017). The last two studies particularly looked at infauna and the authors did not find any statistically significant differences in species richness, diversity, or assemblage composition compared to reference sites (Henkel 2016; Mendoza and Henkel 2017). However, the sediment mean grain size significantly varied and the abundance of organisms was slightly higher in sediments closer to the structures in the estuary setting (Mendoza and Henkel 2017).

The three former studies focused on epifaunal communities on rocky habitats around the SeaGen tidal turbine in Strangford Lough, Northern Ireland (Kregting et al. 2016; O'Carroll et al. 2017a; O'Carroll et al. 2017b). Benthic communities were highly variable within the study area, and covered a large spectrum of successional stages (Kregting et al. 2016; O'Carroll et al. 2017a). Although the epifauna in the area directly under the blades and legs of the turbine was significantly more variable than farther away from the turbine (O'Carroll et al. 2017a), seasonal variability significantly affected epifaunal communities regardless of the station (O'Carroll et al. 2017b). It is thought that at this particular site, as well as in other high-velocity-flow environments favorable to tidal energy developments, epifaunal communities are highly resilient and mainly composed of mosaics of opportunistic species adapted to great physical disturbance (Kregting et al. 2016; O'Carroll et al. 2017a, 2017b). While the authors noticed a negative effect of SeaGen on epifaunal organisms in the immediate vicinity of the turbine, probably due to the increased local turbulences that kept benthic communities at an early successional stage, the effect quickly dissipated with distance from the turbine (i.e., one rotor diameter away; O'Carroll et al. 2017a). The footprint effect of a tidal turbine on benthic communities is thus likely to be limited to the seafloor area directly adjacent to the device (Kregting et al. 2016; O'Carroll et al. 2017a).

6.3.2. CREATION OF NEW HABITATS

MRE devices can also provide new habitats to biofouling species, have effects similar to artificial reefs and fish aggregating devices, and even act as marine reserves.

Biofouling

Biofouling is a design and engineering concern for devices because it might affect performance and maintenance schedules. No antifouling paint or coating has proven fully efficient in preventing biofouling in the long run, and placing MRE devices, foundations, and cables in the water may create new hard-bottom habitats in areas where none previously existed (Figure 6.3). Few MRE devices have been in the water long enough (i.e., several years) to characterize biofouling communities and successional rates (Want and Porter 2018), but experience at OWFs and OGPs can provide some related insight. However, the structures used by the wind energy and oil and gas industries usually provide habitats for fouling organisms from the seafloor to the surface, whereas MRE devices typically do not span the whole water column (except for their mooring structures and dynamic cables). Fouling assemblages will inevitably vary between deployment sites (geography, habitats), devices, and components (Macleod et al. 2016; Want et al. 2017), but all start with a biofilm of marine bacteria and fungi followed over time by successions of initial (e.g., barnacles, hydroids and tubeworms) then secondary (e.g., anemones, ascidians and mussels) colonizers (Causon and Gill 2018; Dannheim et al. 2019). These communities are specific to hard substrates and often follow a vertical zonation (Dannheim et al. 2019). Various successional stages may be observed within an array of MRE devices in the same way different stages of development are observed in OWFs (Causon and Gill 2018).

Some of the most common biofoulers on OWFs are mussels; they compose 90 percent of epistructural biomass in the upper zone of wind turbine foundations in some locations (Slavik et al. 2018). Prolific biofouling organisms (e.g., barnacles, serpulid worms, ascidians) often have short pelagic larval durations and may be transported to artificial structures by construction

Figure 6.3. Heavily colonized tripod of a decommissioned tidal turbine in the Orkney Islands, Scotland (left), and 25 x 25 cm quadrat showing a close-up of the biofouling organisms, mainly barnacles, sponges, and brittle stars (right). (Photos courtesy of Andrew Want, Heriot-Watt University)

and maintenance vessels (Bray et al. 2017; Wilding et al. 2017). Successful colonization by biofoulers will be influenced by natural ocean variability, the seasonal availability of larvae, and the survival rates of recruits (Langhamer 2016). Biofouling can occur relatively rapidly; bare space can be colonized to almost 90 percent within two months in some cases (Viola et al. 2018). Relatively high densities of opportunistic species were found on some WECs at the Lysekil test site in Sweden (Langhamer 2016). The overall species compositions found in the intertidal habitats provided by wind turbine foundations and oil platform pilings often resemble those of nearby natural intertidal habitats and/or local harbors (Coolen et al. 2018; Viola et al. 2018). Similarly, species composition on the deeper sections of such structures as well as on the concrete foundations of MRE devices more resemble those of local subtidal natural reefs (Coolen et al. 2018; Langhamer 2016). Maximum biodiversity has been found at intermediate depths (i.e., halfway up the water column) on the foundations of wind turbines, where disturbance is also intermediate (Coolen et al. 2018). In high-energy environments, the floating parts of WECs may not provide much of a suitable intertidal habitat for biofoulers because of the constant motion and wave impacts (Causon and Gill 2018). Ultimately, biofouling is a natural process that is nearly impossible to avoid on artificial structures deployed in marine environments.

Artificial Reef Effect

In addition to providing artificial substrate for sessile (fouling) species, MRE devices may potentially attract mobile organisms like decapods, demersal and pelagic fish, and apex predators, and in that sense have effects similar to artificial reefs or fish aggregating devices (Dannheim et al. 2019; Langhamer 2016). This effect has been measured and described within several OWFs in European waters (Methratta and Dardick 2019). Several fish species have been shown to aggregate around offshore wind turbine foundations and other artificial hard structures, benefiting from foraging on the benthic communities on the foundations and adjacent habitats (Causon and Gill 2018; Dannheim et al. 2019). By increasing the complexity of the seafloor and surrounding water, OWFs and MRE devices also provide shelter and food (e.g., fouling organisms) for aggregating species, thereby potentially leading to changes in the diversity, abundance, and size of taxa making up the local communities (Causon and Gill 2018; Dannheim et al. 2019; Langhamer et al. 2018). However, the type of device and foundation, their spacing (in the case of an array), local arrangement, and portion of water occupied are important factors controlling the impact of the artificial reef effect (Adams et al. 2014; Causon and Gill 2018; Krone et al. 2017; Langhamer 2016). At the scale of an array of MRE devices, the artificial reef effect could lead to regional changes, including a shift from soft-sediment to hard-substrate communities and, potentially, intertidal communities (Causon and Gill 2018).

The artificial reef effect may not apply to every species, as demonstrated by the case of viviparous eelpouts (*Zoarces viviparous*) at the foundations and scour protection of an OWF in Sweden, where no clear attraction or avoidance was observed or could be distinguished from natural variability (Langhamer et al. 2018). However, scour protection structures on the seabed at OWFs in the southern North Sea, as well as foundations without scour protection, have been shown to attract high numbers of benthic and demersal mobile taxa such as cod (*Gadus morhua*), wrasse (*Ctenolabrus rupestris*), and edible crab (*Cancer pagurus*), and even serve as nursery grounds for some of these species (e.g., Krone et al. 2017; van Hal et al. 2017). Tidal turbines and the foundations of wind turbines also tend to attract pelagic fish; significantly increased observations and sizes of fish schools in the wake flow and changes in the vertical distribution of fish schools in the vicinity of a turbine have been noted, although there was some variability in the depths, days, and tidal cycles (Fraser et al. 2018; van Hal et al. 2017; Williamson et al. 2019). In addition to providing food, artificial structures may also provide flow refuges for pelagic fish (Fraser et al. 2018).

Recent studies have also demonstrated that power cables and associated armoring structures between MRE devices and substations may act as smaller artificial reefs as they are colonized and create new habitats (Bicknell et al. 2019; Taormina 2019; Taormina et al. 2018). Once past the first stages of biofouling, cable structures and their new epifaunal communities attract mobile macro- and megafauna (Taormina et al. 2018). This effect was observed on cables laid at a wave test site in Cornwall, England, where the abundance of pollack and saithe (*Pollachius* spp.) was higher around the cables than in the surrounding natural habitats (Bicknell et al. 2019). The reef effect is expected to be stronger on soft sediments (if cables are not buried) than where cables are laid on top of or among natural rocky reefs (Taormina et al. 2018), thereby creating small local reefs and hubs of biodiversity. However, if the cable protections are of a different structure than the surrounding natural reef (e.g., concrete mattresses vs. boulders), different species assemblages and reef effects may result (Sheehan et al. 2018).

The reef effect of artificial structures can be considered to be ecologically positive because the artificial reef increases habitat complexity and functions as an additional food source, refuge for endangered species, and nursery ground (Krone et al. 2017; Langhamer et al. 2018; Loxton et al. 2017; Raoux et al. 2017; Taormina et al. 2018). Conversely, these structures can also lead to negative effects by facilitating the introduction of non-native species or causing important shifts in local communities (Dannheim et al. 2019; Loxton et al. 2017). The nature and importance of the effects may vary according to the location of the deployment, the existing ecosystem, and natural habitats (Loxton et al. 2017).

Reserve Effect

The reserve effect is defined as the condition in which habitats and marine communities in the vicinity of a device or array of devices are *de facto* protected from fishing when exclusion zones are in place (Alexander et al. 2016). This effect can be beneficial; it promotes the potential recovery of local populations of some vulnerable species and benefits local fisheries if spillover is observed in the wider surrounding (non-protected) area around the devices (Coates et al. 2016). This reserve effect has already been confirmed, with various degrees of success, around some OWFs such as those in the North Sea (Coates et al. 2016; Krone et al. 2017; van Hal et al. 2017). For example, three years after the exclusion of bottom fisheries, fragile benthic communities within an OWF showed subtle changes toward recovery, and the authors suspected illegal trawling in the no-fishery area prevented far more significant changes from being observed (Coates et al. 2016). Nonetheless, significant increases in edible crab, wrasse, and cod populations were observed within the exclusion zone of other OWFs compared to open areas nearby (Krone et al. 2017; van Hal et al. 2017), suggesting that exclusion zones around MRE devices may act as large-scale refugia for vulnerable organisms, potentially those that are of commercial value.

While it might take several years to observe a significant reserve effect during recovery within an exclusion zone around MRE devices (Causon and Gill 2018; Coates et al. 2016), models can help understand the extent of this effect. Alexander et al. (2016) used an Ecopath with Ecosim (EwE) and Ecospace modeling approach to investigate the implications of artificial reef and exclusion zone effects in relation to MRE devices. The model showed a substantial increase in the biomass of several taxa within the exclusion zone, but not much over-spilling outside of the MRE area (Alexander et al. 2016). However, the authors highlighted some noticeable caveats of their study (e.g., fixed rectangular spatial map, coarse spatial scale, binary habitat type assignment to species) that would need to be addressed before generalizing similar approaches (Alexander et al. 2016). Similarly, Raoux et al. (2019) used an EwE model to simulate the potential reef effect by an OWF, its reserve effect, and the combined reef and reserve effect. The results showed an overall limited reserve effect at the ecosystem level, because of the relatively small size of the fishery closure area.

6.3.3. ADDITIONAL INDIRECT EFFECTS

The environmental effects discussed above are direct changes to marine habitats associated with MRE devices. These changes can become ecologically significant beyond the physical boundaries of the area of deployment (Krone et al. 2017; Slavik et al. 2018) or trigger a diversity of indirect effects and cascading processes locally, such as increases in biomass or recruitment of non-native invasive species (Causon and Gill 2018; Dannheim et al. 2019). However, these indirect effects have not been documented for MRE developments at this time and are presented here as a summary of discussions within the MRE and OWF communities.

Facilitation of Non-Native Species Dispersion

While biofouling of an exposed surface in the water is a natural process, it can also facilitate the installation of non-native species. Most non-native invasive species are organisms that have been moved around maritime traffic lines by ballast water and have established themselves on harbor structures (piers, pilings, docks) and nearby shallow-water reefs. This phenomenon has already been described for OWFs in Europe and OGPs in California (e.g., Coolen et al. 2016, 2018; van Hal et al. 2017; Viola et al. 2018) and is a potential concern regarding MRE devices (Dannheim et al. 2019; Loxton et al. 2017; Want et al. 2017), even if non-native species have yet to be reported to occur on MRE devices already deployed offshore (Want et al. 2017; Want and Porter 2018).

Studies of OWFs and OGPs have shown that non-native species are mainly found on structures occupying the upper water column, similar to intertidal habitats (Coolen et al. 2018; Viola et al. 2018), and that some of these organisms exhibit habitat preferences different from related native species, which allows them to occupy different ecological niches and avoid direct competition (Coolen et al. 2016). However, the development of native communities seemed to inhibit the recruitment of non-native species on OGP pilings in southern California (Viola et al. 2018) and marine cables in the English Channel (Taormina 2019). In the OGP piling case, the authors also demonstrated that anthropogenic disturbance (e.g., maintenance by scraping) enhanced the colonization by non-native species for at least 15 months, unless maintenance was timed to occur after the peak of the reproductive season (Viola et al. 2018). Non-native invasive species will be more likely to colonize parts of MRE devices that stay on the surface (e.g., surface attenuators) or occupy the top section of the water column (e.g., point-absorber buoys, oscillating water columns, overtopping devices, tidal lagoons), thereby providing environmental conditions similar to intertidal habitats (Causon and Gill 2018). For example, while underwater cables and their armoring structures on the seafloor can act as artificial reefs, there is very little evidence of colonization by non-native species (Taormina et al. 2018). In fact, only three occurrences of non-indigenous sea squirts were recorded during five years of monitoring along the cable route at Wave Hub, Cornwall, in the United Kingdom (UK) (Sheehan et al. 2018), and the densities of two non-native species along the cable at Paimpol-Bréhat in Brittany, France, became similar to those measured on the natural surrounding seafloor six years after the installation of the cable (Taormina 2019).

New MRE sites, especially large arrays of devices, are believed to provide new habitats for biofouling and artificial reef non-native species and could potentially act as stepping stones between already colonized areas and new natural habitats (Adams et al. 2014; Bray et al. 2017; Loxton et al. 2017). Like other biofouling organisms, non-native species might be transported to the energy extraction sites via construction and maintenance vessels (Bray et al. 2017; Wilding et al. 2017); however, a more likely means of introduction may be the towing of MRE devices to local harbors for maintenance, where non-native species are present and are likely to colonize (Loxton et al. 2017; Want et al. 2017). The use of biophysical models along with pelagic larval durations of known non-native species may help predict the connectedness of sites with local habitats and harbors (Adams et al. 2014; Bray et al. 2017; Vodopivec et al. 2017). Such models have shown that potential MRE and OWF sites in Scotland and the Adriatic Sea could provide suitable habitats for pelagic larvae produced in local harbors or nearshore habitats that would otherwise have perished offshore, *de facto* improving their survival rate (Adams et al. 2014; Bray et al. 2017; Vodopivec et al. 2017). These sites could, in turn, act as source populations and allow species to disperse further, potentially across natural biogeographical barriers (Adams et al. 2014). Siting and device maintenance need to be thought through carefully to prevent such connectedness between harbors and MRE sites for non-native species.

Local and Regional Increase in Biomass and Organic Matter

So far, the increases in local and regional biomass and changes in food webs due to the biofouling and artificial reef effects of MRE devices are mostly hypotheses and a matter of modeling approaches, because such effects may take years, if not decades, to be observed through environmental monitoring. Benthic food webs are predicted to benefit from MRE devices and OWFs through litter falls, i.e., the deposition of feces and dead organisms from fouling and aggregating organisms that enrich sediments (Causon and Gill 2018; Langhamer 2016; Slavik et al. 2018). Local enrichment of organic matter is more likely to occur near WECs and wind turbines, especially because of associated mussel growth (Langhamer 2016), rather than near tidal turbines where hydrodynamic forces may be too strong to favor local accumulations of organic matter. An increase in benthic biomass would in turn benefit higher trophic levels, up to apex predators, thereby potentially intensifying the reef effect (Raoux et al. 2017).

Two recent studies have used an EwE modeling approach (Alexander et al. 2016; Raoux et al. 2017), respectively conducted for periods of 25 years at an MRE site and 30 years at an OWF while increasing the biomass of targeted benthic and fish compartments (Figure 6.4). Both studies showed that the biomass and local food webs changed significantly within the model areas, especially with an increase in mussel biomass leading to a rise in detritivory in the food web (Raoux et al. 2017). In the case of the OWF, the total system biomass increased by 40 percent after 30 years (Raoux et al. 2017). In addition, the approach by Alexander et al. (2016) added an Ecospace component to predict changes beyond the MRE area, showing that the biomass changes were mainly occurring inside the area, rather than outside of it.

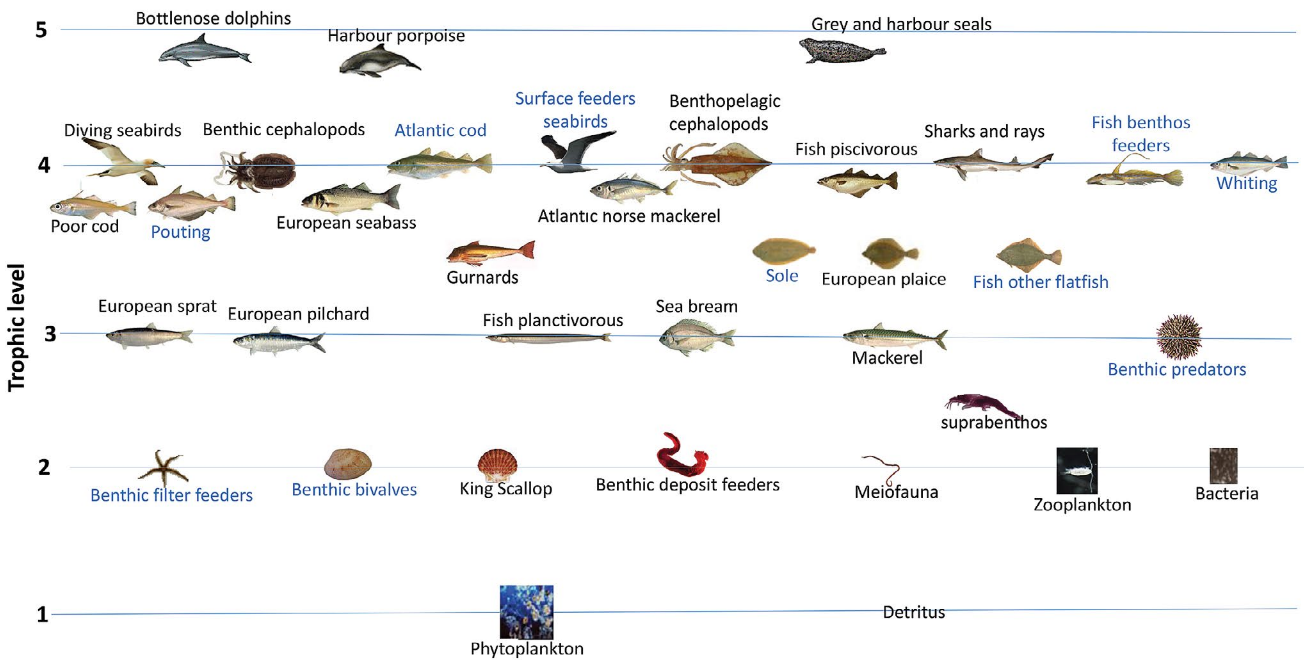

Figure 6.4. Functional groups used in an Ecopath with Ecosim model, arranged by trophic levels on the y-axis and benthic/pelagic coupling across all trophic levels on the x-axis. Functional groups in blue had their biomasses set to their accumulated maximum during the modeling approach. (From Raoux et al. 2017)

Effects of Oceanographic Changes

Other indirect effects of WECs and tidal turbines on marine habitats are the local and regional effects that changes in flow created by MRE devices (see Chapter 7, Changes in Oceanographic Systems Associated with Marine Renewable Energy Devices), especially arrays, could have on benthic and pelagic organisms. A habitat suitability modeling approach demonstrated that barnacles would largely respond negatively to the reduction in bed-shear stress generated by tidal turbine farms, whereas edible crabs would respond positively (du Feu et al. 2019). However, these effects are thought to be mainly restricted to the direct vicinity of tidal arrays, similar to the footprint effect, and farfield effects on benthic communities are unlikely (du Feu et al. 2019; Kregting et al. 2016).

Changes in flow and hydrographic conditions due to MRE devices (see Chapter 7, Changes in Oceanographic Systems Associated with Marine Renewable Energy Devices) may add a level of variability in local and farfield phytoplankton dynamics and processes (Dannheim et al. 2019). The idea is that local disturbances in the wake of devices would modify the stratification, thereby increasing vertical mixing and turbidity, which in turn would either increase the phytoplankton primary production because of higher nutrient availability, or lower it because of lack of light (Dannheim et al. 2019; Floeter et al. 2017). The question was recently addressed using biogeochemical models in the context of large-scale tidal turbine arrays: 66 MW, 800 MW, and 8 GW (Schuchert et al. 2018; van der Molen et al. 2016). Model results suggested the loss of up to 25 percent of local phytoplankton concentrations, although well below the natural seasonal variations (Schuchert et al. 2018), as well as negligible farfield effects in the case of an 800 MW tidal array, or increase in farfield phytoplankton primary production with a less-realistic 8 GW tidal array (van der Molen et al. 2016).

Extreme biofouling by filter-feeding organisms on device components is also thought to modify local hydrodynamics and phytoplankton processes. Slavik et al. (2018) used a biogeochemical model to investigate the question in relation to OWFs. Model results suggested losses of up to 8 percent of regional annual primary productivity due to increased filtration by epifauna, with the maximum loss occurring within the OWFs (Slavik et al. 2018). However, biofouling on MRE devices is not expected to reach levels observed on wind turbine foundations, because they do not provide as much habitat throughout the water column as their wind counterparts (Causon and Gill 2018).

6.4. RESEARCH AND MONITORING NEEDS TO RESOLVE THE ISSUE

This literature review has highlighted several gaps in our knowledge that need to be addressed to advance our understanding of the risks associated with changes in benthic and pelagic habitats. Often, monitoring and research programs are disconnected from one another, so the results from one program do not necessarily contribute to answering questions asked by another (Dannheim et al. 2019; Loxton et al. 2017). Benthic and pelagic communities change over time (e.g., seasonal variability, succession stages, post-disturbance resilience), and long-term studies are required to understand their ecological processes (Langhamer 2016; Taormina et al. 2018; Wilding et al. 2017). However, there is little understanding of appropriate spatial and temporal scales for environmental impact assessment (EIA) and monitoring in relation to MRE, or of the suitable thresholds of undesirable consequences (Wilding et al. 2017).

Stakeholders need justified guidelines for the levels of biodiversity, as well as the assemblages and scales to be considered (Wilding et al. 2017). This holds true for native communities as well as for potentially invasive organisms that may constitute part of the biofouling and artificial reef taxa (Loxton et al. 2017). There are gaps to fill concerning the composition of biofouling assemblages on MRE devices and aggregating species found around devices, their geographic distribution, connectivity, and dispersion abilities (Adams et al. 2014; Bray et al. 2017; Want and Porter 2018), so that regulators can knowingly assess risk and develop biosecurity measures to prevent the spread of non-native invasive species (Loxton et al. 2017).

Underwater visual surveys are very useful approaches for observing changes in species and habitat composition and distribution on and around MRE devices, either through scuba diver surveys, unmanned video transects, or cameras mounted on static structures (Bender et al. 2017). However, the high-energy environments and presence of structures and cables in the water often make for challenging conditions, and methods may need to be refined (e.g., Sheehan et al. 2020). Even greater challenges associated with image-based surveys are the amount of footage that needs to be processed to extract ecologically relevant information and the need for optimized protocols (e.g., Taormina et al. 2020).

The potential impact of localized temperature increase caused by electric cables on infauna communities is an aspect of environmental effects on benthic organisms that has not been addressed much yet (Taormina et al. 2018). Infauna communities constitute important food sources for benthic and demersal organisms like flatfish. However, considering the narrow footprint of the cables and the expected low levels of thermal radiation, this impact may turn out to be insignificant. Nonetheless, it needs to be tested, at least through modeling studies, especially in the case of larger arrays of devices.

Different types of modeling approaches (e.g., biogeochemical, food web, habitat suitability) were recently used to address several questions related to changes in benthic and/or pelagic habitats due to MRE devices and/or OWFs (Adams et al. 2014; Alexander et al. 2016; Bray et al. 2017; du Feu et al. 2019; Raoux et al. 2017; Schuchert et al. 2018; Slavik et al. 2018; van der Molen et al. 2016). Such modeling efforts need to be pursued, because models help answer questions that are difficult to address with monitoring and field observations and on a reasonable time scale. Multispecies and trophic interaction models are particularly valuable, but trickier to implement, because they may require physiological and ecological data that are not yet available (Schuchert et al. 2018).

The effects of partial and complete decommissioning of MRE devices are still unclear. As highlighted earlier, devices left long enough in the water will create habitat colonized by biofoulers and act as artificial reefs, thereby enhancing local biodiversity, so partial decommissioning could be favored. However, devices may also facilitate the establishment of invasive species and total decommissioning may be recommended (Coolen et al. 2018; Sheehan et al. 2018). Both options have benefits and drawbacks that will most likely be weighed on a case-by-case basis, but regulators will need guidelines for preferable options given certain circumstances (Fowler et al. 2018; Sheehan et al. 2018).

6.5. GUIDANCE ON MEASURING CHANGES IN BENTHIC AND PELAGIC HABITATS CAUSED BY MRE

Before-after-control-impact (BACI) analyses are among the best-suited survey designs for measuring changes over spatial and temporal anthropogenic impacts like the deployment of MRE devices (Smokorowski and Randall 2017; Wilding et al. 2017). Such analyses are particularly effective when impacts are important and/or long-lasting, and less effective when changes are variable or gradual (Wilding et al. 2017). Some authors, especially in the case of tidal turbine arrays, recommend an asymmetrical BACI survey design, in which there are more control stations than impact stations (O'Carroll et al. 2017a). Other survey designs, like a before-after-gradient design, are equally suitable for MRE development sites (Bailey et al. 2014; Ellis and Schneider 1997). In any case, it is important that good quality baseline data be collected to provide information about the natural variability within the survey area (Bicknell et al. 2019).

Some authors have highlighted the difficulty involved in characterizing the temporal natural variability of benthic and pelagic ecosystems and differentiating such variability from impacts induced by MRE devices when impact assessment and monitoring surveys only span a couple of years (Wilding et al. 2017). Extreme changes (either natural or anthropogenically induced) are more likely to be detected over a short survey timeframe, while subtle changes are more likely to take longer to observe. Some authors recommend that monitoring studies last more than three years to enable accurate measurement of extreme and subtle changes (Wilding et al. 2017), if not six to eight years to cover the recovery timeframe of some cable sites (Kraus and Carter 2018; Sheehan et al. 2018; Taormina et al. 2018).

In addition, attention needs to be given to the extent of the spatial scale to provide enough strength in detecting potential impacts (Bicknell et al. 2019). The diversity and spatial variability of benthic habitats are more likely to be characterized if the baseline sampling design during the EIA process involves a large-scale regular-spaced grid supplemented with randomly selected additional stations, in order to identify local patches and gradients in habitats and communities

(Kregting et al. 2016; O'Carroll et al. 2017b; Wilding et al. 2017). Follow-up monitoring surveys may sample a subset of the baseline survey as long as they cover the diversity of habitats and communities initially identified (O'Carroll et al. 2017b; Wilding et al. 2017).

Using a modeling approach may be helpful in highlighting some potential changes in benthic and/or pelagic habitats and species that can then be specifically looked for. Habitat suitability models (e.g., MaxEnt) are particularly valuable when it comes to identifying areas that feature the appropriate ecological requirements for a species to establish itself, and these models may help track the settlement of non-native species (Adams et al. 2014; du Feu et al. 2019). Regarding pelagic communities such as nekton organisms, parametric models (e.g., state-space model) work best for detecting changes, time-series models and semi-parametric models are better fitted for quantifying such changes, and nonparametric models are preferred for forecasting changes (Linder and Horne 2018; Linder et al. 2017). Among food web models, the EwE modeling approach is one of the most easily accessed and commonly used approaches for modeling human-induced ecosystem-wide changes over long periods of time, particularly in data-poor systems like MRE sites (Alexander et al. 2016; Raoux et al. 2017). However, many other model types also exist, such as size-based models (Rogers et al. 2014) or agent-based models (Fulton et al. 2015). Modelers interested in MRE would benefit from consulting with experienced ecological and fisheries modelers to determine what approach would be better suited given their specific questions and the available data. Experience drawn from modeling associated with an ecosystem approach to fisheries or coastal management would also suggest that an ensemble modeling approach is likely an effective option to pursue given the current levels of uncertainty (Cheung et al. 2016; Fulton et al. 2019).

6.6. RECOMMENDATIONS

While several questions have been addressed over the four years since publication of the previous State of the Science report, numerous authors have highlighted recommendations for conducting research and monitoring to reduce the uncertainty around some of the changes in benthic and pelagic habitats and to move the industry forward (Bray et al. 2017; Dannheim et al. 2019; Linder and Horne 2018; Loxton et al. 2017; Macleod et al. 2016; O'Carroll et al. 2017b; Wilding et al. 2017). Suggestions for the path forward include the following:

- Define relevant spatial and temporal scales for EIAs and monitoring surveys.
- Identify justified and acceptable thresholds for changes in benthic and pelagic environments, including the extent of loss or the level of colonization by biofouling and artificial reef organisms.
- Use modeling approaches to define habitat suitability and connectedness during the siting process.
- Characterize the diversity and ecological characteristics of biofouling communities and common non-native biofouling and artificial reef species.
- Use (transfer) as much as possible knowledge and lessons learned from other offshore industries such as offshore wind, oil and gas extraction, and fisheries.
- Identify the cumulative effects of MRE devices and other activities occurring in the same area, especially relative to the artificial reef, reserve, and stepping stone effects.

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NOTES

Changes in Benthic and Pelagic Habitats Caused by Marine Renewable Energy Devices

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REPORT AND MORE INFORMATION

OES-Environmental 2020 State of the Science full report and executive summary available at: *https://tethys.pnnl.gov/publications/state-of-the-science-2020*

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