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PREPARING FOR OFFSHORE RENEWABLE ENERGY DEVELOPMENT IN THE MEDITERRANEAN

By

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LAURA BRAY

ABSTRACT

The development of offshore wind farms and marine renewable energy devices in the Mediterranean is central to both national, and international, energy strategies for countries bordering the Mediterranean Sea. The ecological impacts of marine renewable energy development in the Mediterranean region, although essential for policy makers, are as yet unknown. The Northern Adriatic is identified as a plausible site for offshore wind farm development. Using the wider region (Adriatic and Northern Ionian) as a case study, this thesis examines the likely impact to the marine environment if an offshore wind farm is established. Site suitability, based on wind speed, bathymetry, and larvae connectivity levels are investigated along with the plausibility of the turbines operating as artificial reefs in the area. As offshore wind farms may alter the larval connectivity and supply dynamics of benthic populations, a connectivity map was constructed to identify areas of high and low connectivity in the Adriatic Sea. The Puglia coast of Italy is a likely larval sink, and displays some of the highest connectivity within the region, suggesting potential inputs of genetic materials from surrounding populations. Considering offshore wind farms could operate as artificial reefs, an in-situ pilot project was established to simulate the presence of wind turbines. Macroinvertebrates colonized the new substrata within the first few months but were lower in abundance when compared to a natural hard substrata environment. Time, turbine location, and the material used for turbine construction all affected the macro-invertebrate communities. In addition, fish abundances, and diversity were lower around the simulated OWF foundations in comparison to a natural hard substrata environment, and no increases in fish abundance occurred around the simulated turbines when compared to reference sites of soft substrata. This observation was validated with the use of an ecosystem modelling software (Ecopath with Ecosim),

which simulated the overall ecosystem level impacts that would occur if 50 offshore monopile wind turbines were introduced to the Northern Ionian and colonized by macroinvertebrate communities. When compared to the baseline scenario (no simulated introduction of an OWF), the introduction of new habitat had no discernible impacts to the structure or functioning of the marine ecosystem. Noticeable changes to the ecosystem were only apparent if fishing restrictions were enforced in parallel with the simulated offshore wind farm; the ecosystem appears to become more structured by top down predation. In addition seabirds are also impacted by the reduction of fishing discards as a food source. These results are the first attempt to quantify the suspected benefits of offshore wind farms operating as *de-facto* marine protected areas.

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CHAPTER 1

PREDICTING THE IMPACT OF OFFSHORE WIND FARMS ON MARINE LIFE IN THE MEDITERRANEAN SEA: EVIDENCE TO DATE FROM OTHER SYSTEMS

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Contributions: I performed and wrote the literature review with feedback from SD, CA, M V-L; TS provided the Eta-Skiron model outputs. JMH-S supervised the thesis.

1.1 INTRODUCTION

The global demand for energy supply continues to increase rapidly (Kermeli et al., 2014), accelerated demographic and economic growth (Esseghir and Haouaoui Khouni, 2014), modifications in energy usage as a result of climate change (Cartalis et al., 2001), and rising demands for rural electrification in many Middle East and North Africa (MENA) countries (Tsikalakis et al., 2011) have dramatically increased the energy demands of the Mediterranean region; a trend that is set to continue (Esseghir and Haouaoui Khouni, 2014). Consequently, problems concerning the security of energy supply, and the impact of global warming and ocean acidification as a result of CO_2 emissions, have stimulated research, and development, into environmentally sustainable energy. This drive is reflected in the Horizons 2020 EU Renewables Directive (2009/28/EC), with member states being required to obtain 20% of their energy consumption from renewable energy sources by 2020 (EWEA et al., 2014). Non-EU Mediterranean countries have also recognized the need to decrease reliance on hydrocarbons and most have adopted similar policies (Tsikalakis et al., 2011).

Europe is seeing a rapid expansion of the wind energy sector on land; however, higher mean winds speeds due to a reduction in offshore surface roughness (EEA, 2009), and comparatively lower visual and noise pollution than onshore wind farms (Bilgili et al., 2011), has led to a recent expansion of marine wind farms with further planned developments particularly within the North Sea and Baltic regions (> 40GW by 2020) (Jacques et al., 2011). Currently, the Mediterranean Sea has no operational offshore wind farms (OWFs), yet this is expected imminently, as several regions have been earmarked for OWF development (De Decker et al., 2011).

The environmental effects of OWF construction in the Mediterranean are as yet unknown. The Mediterranean Sea has several distinct characteristics including minimal tidal ranges, high levels of biodiversity and endemism (Coll et al., 2010), and a high potential for range extension of alien species due to the Suez Canal acting as a corridor from the Red Sea (Rilov and Galil, 2009; Zenetos et al., 2012). The region is also exposed to a suite of coastal pressures including pollution, busy shipping lanes, eutrophication, urban development, habitat degradation, and overfishing (Micheli et al., 2013). Due to the characteristics of the region (e.g. susceptibility to alien species), impacts of existing OWFs may not be directly applicable to the Mediterranean environment. In recognizing some of this uncertainty, the aims of this chapter are threefold. Firstly, likely areas of importance for OWF construction in the Mediterranean are identified. Secondly, the biological effects of existing OWFs in Northern European Seas are systematically assessed and considered in relation to the unique conditions of the Mediterranean basin, with particular focus on areas identified as suitable wind farm locations, and finally, towards the end of the chapter, the scope of the thesis is presented based on the knowledge gaps identified by the literature review.

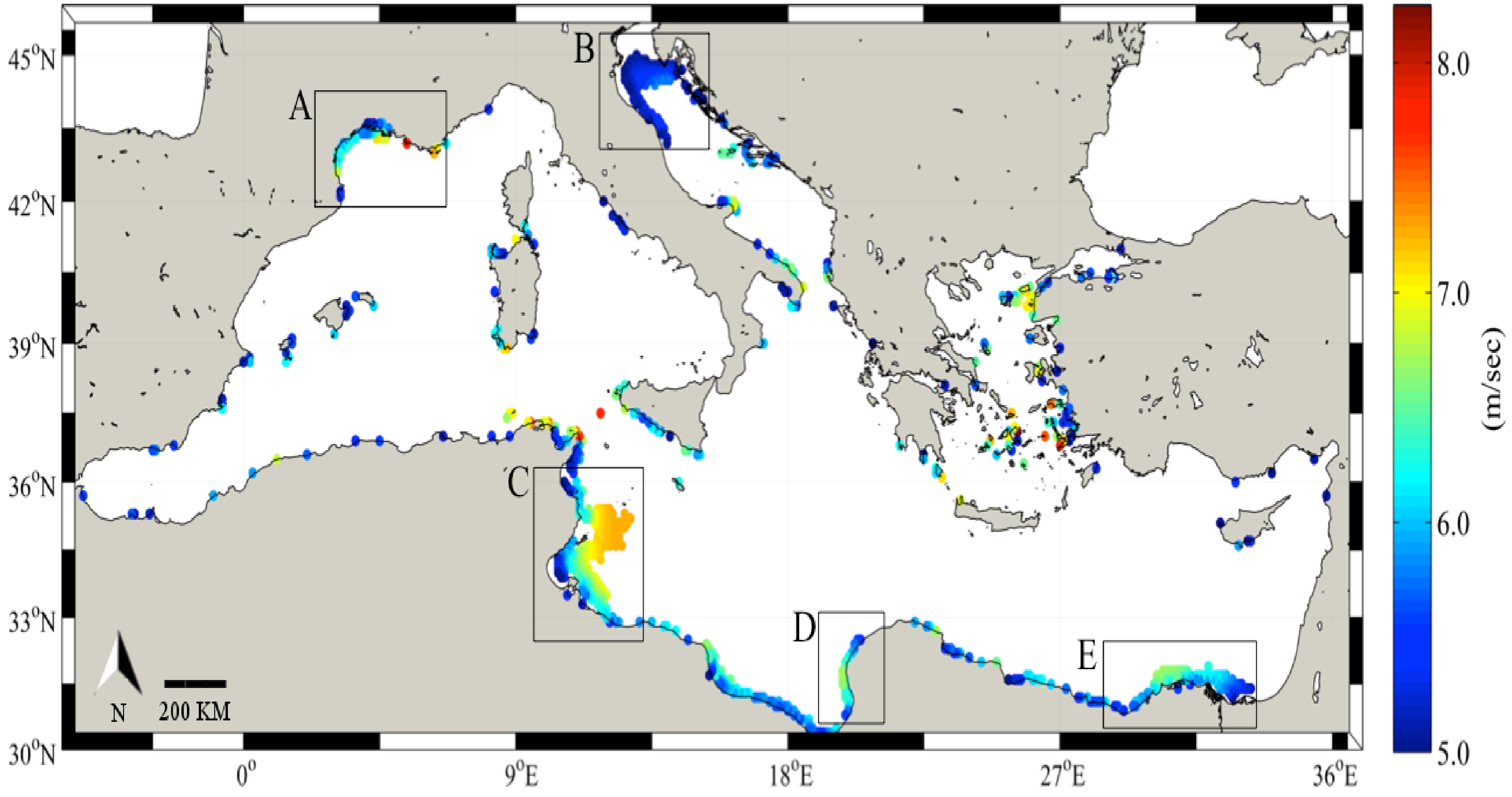
1.2 Offshore wind farm potential in the Mediterranean

In order to investigate the likely impacts to marine habitats caused by OWFs, it is imperative that candidate offshore wind farm sites are identified to enable site-focused analyses. For effective OWF site selection, many parameters should be considered, such as distance from shore, bottom morphology and type of sediments, and electrical grid infrastructure availability; however the most important criteria are typically wind resource availability and bottom depth (Soukissian et al., 2017).

To identify potential OWF areas throughout the Mediterranean basin, suitable wind speeds and bottom depths were identified using a combination of wind model outputs and bathymetry data. Since the current fixed-bottom wind turbine technology (monopile, gravity-based, jacket and tripod foundations) is limited to water depths up to 50 m, the depth range considered was 20–50 m, and the lower threshold for the mean annual wind speed (measured at 80 m above mean sea level) was set to 5 ms^{-1} in accordance with European Environment Agency (EEA) recommendations (EEA, 2009). Wind speed outputs covering a 10-year period (1995–2004) were obtained from the Eta-Skiron model which is a modified version of the non-hydrostatic workstation Eta model (Papadopoulos et al., 2011, 2002; Papadopoulos and Katsafados, 2009), and combined with bathymetry data from the General Bathymetry Chart of the Oceans global relief (Papadopoulos et al., 2011; Uppala et al., 2005). In accordance with the above parameters, potential wind energy sites (model grid points 0.10° × 0.10°) were identified, and regions with high densities of such point locations were highlighted as offshore wind energy hotspots. The Eta-Skiron mesoscale meteorological model is used for the dynamical downscaling of the European Centre for Medium-Range Weather Forecasts ECMWF Era-40 reanalysis data (Uppala et al., 2005) and the ECMWF operational forecasts, with a fine spatial (0.10° × 0.10°) and temporal resolution (3 h) for analysing the Mediterranean Sea.

Many Mediterranean coastlines seem poorly suited to OWF development. Several large areas have high concentrations of suitable wind speed and depth locations (e.g. exploitable potential), and have quantitatively been identified here as hotspots. These ‘hotspots’ include the coasts of the Gulf of Lyons, the Northern Adriatic Sea, the entire coastal area of the Gulfs of Hammamet and Gabès in Tunisia, off the Nile River Delta, and the Gulf of Sidra in Libya (Figure 1.1). They spatially cover the width and breadth of the Mediterranean Sea. Here, the potential effects on birds, marine mammals, fish, benthos and plankton throughout the Mediterranean are considered and, where possible, the possible impacts of OWFs within the specific hotspot regions.

Figure 1.1 All possible locations which satisfy offshore wind development requirements of 20–50 m depth and an annual wind speed higher than 5 ms^{-1} (at a height of 80 m above sea level). No colour indicates conditions not met. A: Gulf of Lion, B: North Adriatic Sea, C: Gulfs of Hammamet and Gabès, D: Gulf of Sidra. E: Nile Delta.



1.2 POTENTIAL IMPACTS TO MEDITERRANEAN MARINE ECOSYSTEMS

Biological effects resulting from the construction, and operation, of offshore wind farms were identified in a review of studies in Northern European Seas. Peer reviewed literature took precedence, and primary literature was obtained from several databases including CAB abstracts, Google Scholar, Web of Science, Science Direct, and Scopus. Relevant grey literature was also included in the compilation of information, and expert opinions were sought from several research institutes and industry experts (references herein). For clarity, impacts were separated via taxa (e.g., birds, marine mammals, fish, and benthos). The impacts to each taxon, alongside a brief description of the current status in the Mediterranean are provided, and where possible likely impacts to the Mediterranean marine environment are described.

1.2.1 BIRDS

Wind farms affect resident and migrating birds, through avoidance behaviours, habitat displacement, and possible collision mortality, but such impacts are difficult to monitor offshore (Desholm and Kahlert, 2005; Marques et al., 2014). Seabirds that use the marine environment for foraging or resting may be displaced by OWFs (Scott et al., 2014). The Mediterranean has a low diversity of seabirds, but these species tend to be long-lived with low fecundity, traits that often make species vulnerable to abrupt environmental change (Coll et al., 2010; Scott et al., 2014) (Table 1.1). Fortunately, most Mediterranean marine birds are listed as “least concern” on the IUCN red list, although the Audouin's gull (*Ichthyaetus audouinii*) is listed as “near threatened,” the Yelkouan shearwater (*Puffinus yelkouan)* as “vulnerable,” and the Balearic shearwater (*Puffinus mauretanicus*) as “critically endangered” (IUCN 2014). All 16 Mediterranean countries have made commitments to protect these species at a national level (UNEP-MAP, 1999). With the exception of shearwaters (Cooper et al., 2003), Mediterranean seabird population sizes appear to be increasing, particularly the yellow-legged gull (Bourgeois and Vidal, 2008; Coll et al., 2010; Thibault et al., 1996). These increases have been attributed to increased fish discards and improvements in coastal conservation (Donald et al., 2007; Oro, 1996; Oro and Ruiz, 1997), but changes to fishery discard practices following the reform of the Common Fisheries Policy may reverse this (Bicknell et al., 2013).

Studies of northern European seabird populations have developed vulnerability indices to indicate seabirds most likely to be affected by the presence of OWFs (Bradbury et al., 2014; Furness et al., 2013; Garthe and Hüppop, 2004). The North/Baltic Sea-based studies assessed 18 of the 32 Mediterranean seabirds. The Wind Farm Sensitivity Index obtained from Garthe and Hüppop (2004) (Table 1.1), is calculated an index based on 9 factors including flight manoeuvrability; flight altitude; percentage of time flying; nocturnal flight activity; sensitivity towards disturbance by ship and helicopter traffic; flexibility in habitat use; biogeographical population size; adult survival rate; and European threat and conservation status. Both the vulnerability indexes for collision impacts, and the vulnerability index for disturbance impacts (Table 1.1), are obtained from Furness et al., (2013). The Vulnerability Index for Collision Impacts is calculated by combining ranked values of population percentage, manoeuvrability, flying time (including nocturnal flights) and conservation importance. Disturbance impact index is calculated using ranked scores from disturbance by ship and helicopter traffic, habitat flexibility, and conservation importance. Notable exclusions to the list are the endemic species of the Mediterranean, which pose a greater conservation risk due to their small population sizes (Gallo-Orsi, 2003). Garthe and Hüppop (2004) identify the Black and Red-throated diver, the Sandwich tern, and the great Cormorant as the most sensitive of the Mediterranean seabirds within their index, and rated the Black-legged kittiwake, and the Black-headed gull as the least sensitive when all parameters were combined. Advancing this approach, Furness et al. (2013) separated the hazards due to collision risk and habitat distribution. They identified the lesser black-backed gull and the northern gannet as seabirds sensitive to collision risk, and both the red and black-necked divers as most susceptible to long-term habitat displacement (Table 1.1). The approach of identifying at risk species via vulnerability indices (see above) is useful for the planning stages of OWFs; however, it does not determine if introduction of OWFs will have a detectable change in seabird population trends (e.g. collision mortality impacting species fecundity). Focus should preferably be given to understanding any direct effects OWFs will have on foraging success, e.g., diving behaviour and prey characteristics, which in turn will impact reproductive success, juvenile survival, and population trends (Scott et al., 2014).

Threats to Mediterranean bird populations are also directed towards migratory species. Worldwide, migratory species are declining in greater numbers than resident populations (Wilcove and Wikelski, 2008), and the Mediterranean basin is a major transit route for Saharan-Eurasian migration, as evidenced by both the Mediterranean-Black Sea flyway and the Adriatic flyway (Boere and Stroud, 2006; Denac et al., 2010). Many long-distance bird migrants, e.g., raptors and storks, rely on land-lift via thermal upwelling for long-distance flight (Alerstam and Pettersson, 1977; Pennycuick, 1972) and avoid broad fronts such as the Mediterranean Sea and the Saharan desert (Alerstam and Pettersson, 1977), creating bottlenecks at narrow passages of the Mediterranean Sea (e.g. Gibraltar, the Straits of Sicily, Messina, and the Belen pass in Turkey) (Bijlsma, 1987).

High collision levels of migrating terrestrial birds (442 individuals in 14 months) at a well-lit observing platform during periods of bad weather and poor visibility (Hüppop et al., 2006), indicate that wind farms located near the coast, or in prominent migration bottlenecks, may pose a significant risk to migrating birds. In addition to collision risks, avoidance behaviour of birds may also impact flight path decisions. Turbine avoidance tactics apply to both resident seabirds, and long-distance migrants (Plonczkier and Simms, 2012; Vanermen et al., 2015). Changes to migratory routes are difficult to monitor and may have large, indirect effects on flight energetics (Hüppop et al., 2006).

Avoidance behaviour is possible at several scales, which are typically classified as (i) micro, (ii) meso, or (iii) macro strategies. Micro-avoidance is the behavioural response employed by birds to actively avoid rotating blades. Meso-avoidance is the avoidance of the whole rotor swept zone by species that fly at rotor height within the wind farm, and macro-avoidance indicates a change of flight path to totally avoid the presence of a wind farm (Scottish Government, 2014). Macro-avoidance has been shown in some migrating individuals: The common eider *Somateria mollissima,* for example, exhibited avoidance behaviours of a wind turbine resulting in an additional flight path increase of ca. 500 m (Masden et al., 2009). The long-term consequences of employing avoidance techniques remain unclear (Warwick-Evans, 2016). Effects in migrating birds will be highly dependent on the specific life histories of a species, expenditure of avoidance strategies, energy reserves, and weather conditions during migratory periods. Additionally, until migration routes across the Mediterranean Sea are better understood, developers face large difficulties in wind farm spatial planning in the region.

Table 1.1 Mediterranean seabird sensitivity assessments (Wind farm sensitivity index, vulnerability index for collision impacts, and the vulnerability index for disturbance impacts) highlighting most and least vulnerable species according to index "-" = Index not applied. List obtained by searching the BirdLife International database for species present in the Mediterranean (www.birdlife.org).

Common Name	**Species**	**Endemic**	**Wind Farm Sensitivity Index**	**Vulnerability Index for Collision Impacts**	**Vulnerability Index for Disturbance Impacts**
Cory's Shearwater (Mediterranean)	*Calonectris diomedea diomedea*	Y	-	-	-
Yelkouan Shearwater (Mediterranean)	*Puffinus yelkouan*	Y	-	-	-
Balearic Shearwater	*Puffinus mauretanicus*	Y	-	-	-
European Shag (Mediterranean)	*Phalacrocorax aristotelis desmarestii*	Y	-	150	14
Great Cormorant	*Phalacrocorax carbo*	N	23.3	-	-
Pygmy Cormorant	*Phalacrocorax pygmeus*	N	-	-	-
Audouin's gull	*Larus audouinii*	N	-	-	-
Little Gull	*Hydrcoleus minutus*	N	12.8	-	-
Lesser black-backed gull	*Larus fuscus*	N	13.8	960	3
Slender billed gull	*Larus genei*	N	-	-	-
Mediterranean gull	*Larus melanocephalus*	N	-	-	-
Black-headed gull	*Larus ridibundus*	N	7.5	-	-
Caspian gull	*Larus cachinnans*	N	-	-	-
Black legged kittiwake	*Rissa tridactyla*	N	7.5		-
Yellow legged gull	*Larus michahellis*	N	-	-	-
Great skua	*Catharacta skua*	N	-	320	3
Caspian tern	*Hydroprogne caspia*	N	-	-	-
Common tern	*Sterna hirundo*	N	15.0	229	8
Little tern	*Sterna albifrons*	N	-	212	10
Sandwich tern	*Sterna sandvicensis*	N	25.0	245	9
Lesser-crested tern	*Thalasseus bengalensis*	N	-	-	
Razorbill	*Alca torda*	N	15.8	32	14
Atlantic puffin	*Fratercula arctica*	N	15.0	27	10
European Storm petrel	*Hydrobates pelagicus*	Y	-	91	2
Northern gannet	*Morus bassanus*	N	-	-	-
Osprey	*Pandion haliaetus*	N	-	-	-
Eleanore's falcon	*Falco eleonorae*	N	-	-	-
Red throated diver	*Gavia stellata*	N	43.3	213	32
Black throated diver	*Gavia arctica*	N	44.0	240	32
Great crested grebe	*Podiceps cristatus*	N	19.3	84	8
Red-necked grebe	*Podiceps grisegena*	N	18.7	-	-

In addition to migration routes and bottlenecks, wetlands around the Mediterranean are also widely used by seabirds. They provide suitable stopover sites for long-distance migrants to feed, rest and molt (Kirby et al., 2008). Some of the main wetlands around the Mediterranean are located within close proximity of potential OWF hotspots, particularly the Po Delta in the Northern Adriatic Sea the Nile Delta, the Gabès Delta and the Camargue Delta in the Gulf of Lion (Figure 1.2). Due to the bathymetry of the Mediterranean, and the steep continental slope of most coastlines, deltas provide feasible sites for wind farm constructions. High densities of avian abundances in these regions means that OWF resource overlap will be a key factor in Mediterranean marine spatial planning in regard to OWFs.

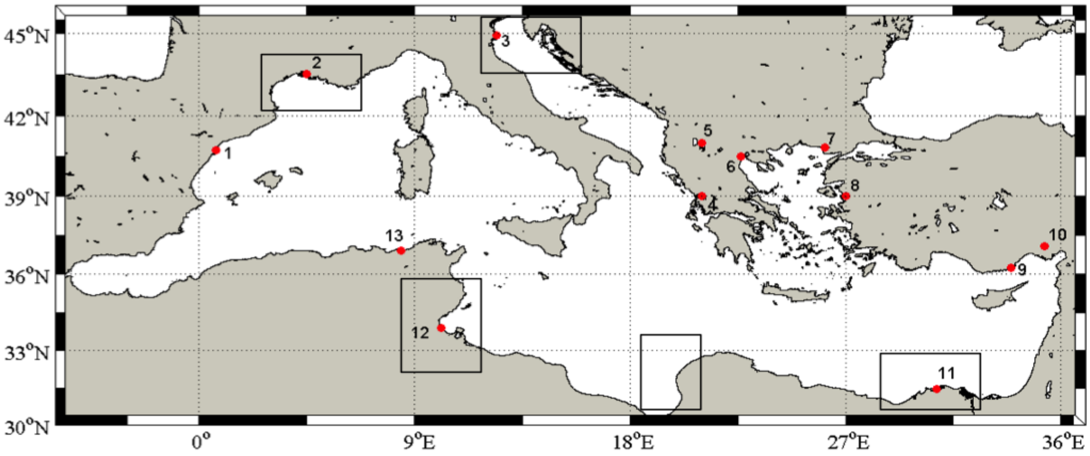


Figure 1.2 Main Mediterranean wetlands and overlapping OWF potential hotspot areas as indicated in figure 1.1 (black squares) (adapted from Jourdain et al., 2007). 1. Ebro Delta; 2. Camargue Delta; 3. Po Delta; 4. Amvrakikos Gulf; 5. Prespa Basin; 6. Aliakmonas Delta; 7. Evros Delta; 8. Gediz Delta; 9. Göksu Delta; 10. Seyhan Delta; 11. Nile Delta; 12. Gabès Delta; 13. El Kala.

There are several possible measures to reduce any of the potential impacts that wind farms will have on Mediterranean avian populations, e.g., shutdown orders and changes to the level of light being emitted by offshore structures to reduce phototaxis towards them (Poot et al., 2008; Saidur et al., 2011). However, it is essential that OWF construction be preceded by a thorough siting-process to avoid increased risk to sensitive areas (migration bottlenecks), and species.

1.2.2 MARINE MAMMALS

The Mediterranean Sea is home to both resident and visiting marine mammals, nearly all of which have shown a decrease in abundance in recent years, with the exception of visiting humpback whales whose numbers have appeared to increase (Coll et al., 2010; Frantzis et al., 2004). At a basin level, total population sizes are difficult to assess with several species being classified as “data deficient” by the IUCN red list (IUCN 2014) (Table 1.2). Nonetheless, certain regions have been identified as important habitats for marine mammals. Monitoring programs show a high percentage of fin whale sightings within the Ligurian Sea in comparison with other regions of the Mediterranean Sea (Notarbartolo-Di-Sciara et al., 2003). The Alborean Sea has been shown to be an important area for long-finned pilot whale populations (Cañadas and Sagarminaga, 2000), and there is also evidence that due to the east-west basin migration of Sperm whales, the Strait of Sicily and the Strait of Messina, are critical areas which enable inter-basin migration (Frantzis et al., 2011). Despite multi-year surveys, long-finned pilot whale populations are apparently low in abundance in other regions of the western Mediterranean Sea, with few sightings recorded over around the central Mediterranean Sea, the Balearic Sea, the Provençal Basin, and the Ligurian Sea (see Verborgh et al., 2016 for a review).

In regard to OWF development and site overlap, several species of marine mammals frequently use the coastal marine environment earmarked for potential developments including the critically endangered Mediterranean monk seal, the common Bottlenose dolphin, and visiting Humpback whales (Bearzi et al., 2009; Dendrinos et al., 2008; Frantzis et al., 2004). An assessment of the total number of species shows that the Gulf of Lion OWF hotspot displays the highest densities of resident marine mammals and as such can be considered as the most sensitive in regard to OWF development (if it is assumed that individual species are sensitive). The Gulfs of Hammamet and

Gabès, the Gulf of Sidra, and the Nile Delta hotspots appear to support low populations of resident marine mammals (Figure 1.3).

Table 1.2 List of resident Mediterranean marine mammals, their current population trends, IUCN status and important habitats noted within the Mediterranean.

Species	Mediterranean Population trend (IUCN)	IUCN Conservation Status	Important regions	Source
Tursiops truncatus	Declining	Vulnerable	Coastal waters until the continental shelf	(Bearzi et al., 2009)
Ziphius cavirostris	Unknown	Data Deficient	Offshore regions	(IUCN, 2012)
Balaenoptera physalus	Declining	Vulnerable	Corso-Ligurian Basin	(Notarbartolo-di-Sciara et al., 2008)
Globicephala melas	Unknown	Data Deficient	Alboran Sea	(Cañadas and Sagarminaga, 2000)
Grampus griseus	Unknown	Data Deficient	Continental shelf waters	(Bearzi et al., 2011)
Delphinus delphis	Declining	Endangered	Alboran Sea, Gulf of Vera, Sicily channel, South-eastern Tyrrhenian Sea, Aegean Sea, Eastern-Ionian Sea	(Bearzi et al., 2003)
Physter macrocephalus	Declining	Endangered	Continental shelf waters, offshore regions	(Reeves and Notarbartolo di Sciara, 2006)
Stenella coeruleoalba	Declining	Vulnerable	Alboran Sea, Ligurian Sea	(Forcada et al., 1994)
Monachus monachus	Declining	Critically endangered	Archipelago of Madeira, Area of Cabo Blanco, Ionian Sea, Aegean Sea	(Dendrinos et al., 2008)

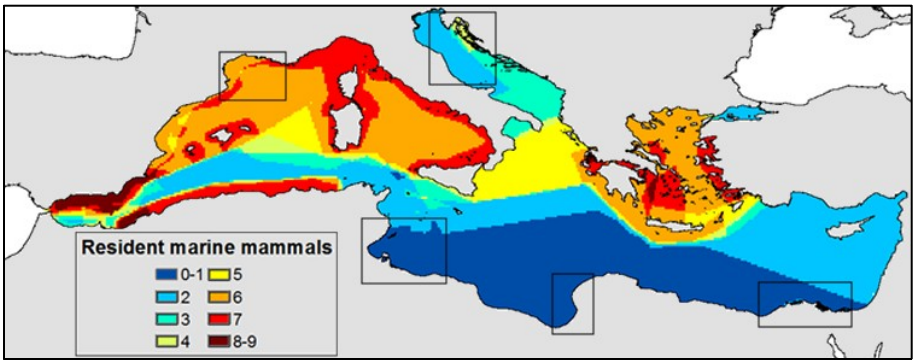


Figure 1.3 Species number of all resident marine mammals ranges in the Mediterranean (*n* = 9) with overlay of OWF hotspots as defined by figure 1.1 (Black squares) (Image adapted from Coll et al. 2010).

Through monitoring programs and generalised impact assessments for marine noise, noise from pile driving during construction has been identified as the most important adverse effect of OWF's for marine mammals (Bergström et al., 2014). Marine mammals are heavily reliant on sound for foraging, orientation and communication and are thus susceptible to negative effects (i.e. sound masking, temporary hearing threshold shift, or in more extreme cases permanent threshold shift) of sound generation (Madsen et al., 2006). Depending on the hearing range of a species, pile driven construction has the ability to produce sustained hearing impairment, although for most species, hearing thresholds and detection levels of noise from wind turbines are as yet undetermined (Bergström et al., 2014). A study measuring the propagation of sound during the construction phase of an offshore site in the NE of Scotland indicated Bottlenose dolphins would suffer auditory injury within a 100 m range of the site and that they exhibit behavioural disturbance up to 50 km from pile driving activity (Bailey et al., 2010a). With the use of passive acoustic listening devices, acoustic monitoring during the construction and operational phases of the Nysted wind farm indicated a possible change in habitat use by the harbour porpoise (*Phocoena phocoena)*, and a reduction of echolocation activity (Carstensen et al., 2006). Furthermore, a long-term study at the same wind farm (10 years) also showed a decline from baseline levels of echolocation signals (Teilmann and Carstensen, 2012).

In contrast, a similar study at the Dutch wind farm, Egmond aan Zee, measured significantly higher acoustic activity inside the farm in comparison with a control site (Scheidat et al., 2011), and this effect was mirrored in a study of harbour seal (*Phoca vitulina*) foraging which indicated an increase in habitat utilization (tag duration: 25–161 days) at two operational wind farms (Alpha Ventus and Sheringham Shoal) (Russell et al., 2014). The repeated grid-like movements indicated for the first time, successful foraging behaviour by an apex predator within an OWF, indicating evidence of acclimatization. The apparent differences between probable habitat uses may be due to local-scale ecological differences e.g. local population habituation of wind farms, inter-species differences, or differences in construction type of wind farms (Scheidat et al., 2011). Due to critical population levels of the Mediterranean monk seal (*Monachus monachus*) in the Mediterranean (estimated as >500 mature individuals, www.iucnredlist.org), and the equivocal evidence from the literature, the observed increases in seal foraging behaviour around wind farms should be cautiously interpreted with regards to the Mediterranean monk seal (Dendrinos et al., 2008; Russell et al., 2014).

In regard to the impacts of noise levels in the Mediterranean, the semi-enclosed Mediterranean also suffers from some of the highest volume of shipping routes in the world (Abdulla, 2008) (Figure 1.4). Increased motorized vessel shipping during the operational phase of wind farms also increases noise levels to the area, and impacts marine mammals (Madsen et al., 2006). Generally speaking, underwater noise from wind farms is influenced by water depth, wind speed, turbine type, wind farm size, and substratum type (Bailey et al., 2010b). High levels of existing background noise from maritime traffic in the Mediterranean, heightens the risk of cumulative effects which can mask the communicative abilities of marine mammals (David, 2006). By assessing the spatial density of traffic routes from 2013, it is clear that the OWF hotspots of the Gulf of Lion, the North Adriatic Sea, and the Nile Delta show an already high density of vessels within the area (up to 140 m vessels km^{-2} day^{-1}); thus, high levels of background noise can be expected in these regions. The Gulf of Hammamet and Gabès, and the Gulf of Sidra suggest much lower levels of background noise stress. The behaviour of noise propagation in the marine environment, and differing audiograms of marine mammal species mean that defining thresholds in which ambient noise impacts marine mammals is difficult, however it is clear that the use of underwater noise propagation models by policy makers will be required to help understand the combined influence of OWF construction, operation, and maintenance shipping, with current levels of background noise at site-specific locations.

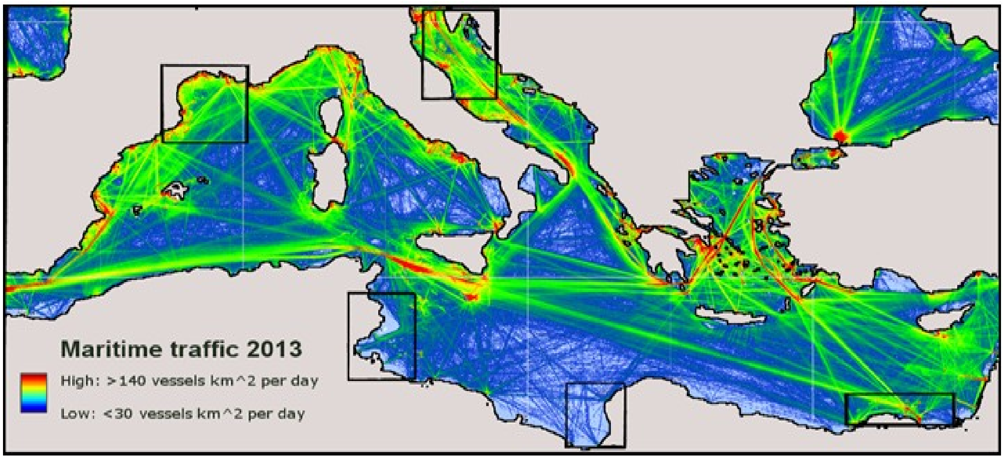


Figure 1.4 Combined density of 2013 maritime routes of all commercial vessels obtained via AIS vessel monitoring with overlay of OWF hotspots (black squares) (source: www.marinetraffic.com).

1.2.3 FISH

Throughout the basin, many Mediterranean coastal communities depend on fishing-related activities, particularly artisanal fishing (Tzanatos et al., 2006). Of the 513 species, and 6 subspecies of fish in the Mediterranean, 8% are currently classified as threatened by the IUCN (Malak et al., 2011), and there has been an alarming decline of Mediterranean fish stocks over the last two decades with around a 40% decrease in biomass (Vasilakopoulos et al., 2014).

The principal impacts to fish populations caused by wind farms are noise, electro-magnetic fields, and novel habitat gain (Gill and Taylor, 2001; Wahlberg and Westerberg, 2005; Wilhelmsson and Langhamer, 2014). Recent studies have shown that the noise generated by pile-driving during the construction phase of OWF farms can generate acute stress responses in juvenile fish species (Debusschere et al., 2016). Although the responses were recorded as acute, it is possible that repeated and prolonged exposure in the wild may lead to a decrease in fitness (e.g. fecundity). During the operational stages of OWFs, evidence indicates some fish permanently avoid wind turbines at a limited range of up to 4 m under high wind speeds (13 ms^{-1}), and that their ability to communicate and utilize orientation signals is often masked (Bailey et al., 2010b). Additionally, increased background noise and seabed vibration from operational OWFs and associated marine traffic also influences fish detection distances, which affects both foraging and communication behaviours (Sigray and Andersson, 2011; Wahlberg and Westerberg, 2005) (Figure 1.4). Greater numbers of experimental studies on individual fish species are needed before the impact of anthropogenic noise on fish can be effectively considered in environmental impact assessments of offshore wind farms (Radford et al., 2014).

Electromagnetic fields (EMF) occur around intra-turbine, array-to-transformer and transformer-to-shore cables. The electro-sensitivity of many marine species is unknown, and there is a dearth of peer-reviewed information regarding the effects of electro-magnetic fields. Elasmobranchs are thought to be especially sensitive, due to their electro-sensory organs (Tricas and Gill, 2011). Several shark and ray species react to wind farm cables, most commonly with avoidance behaviour, as is the case with the small-spotted catshark (*Scyliorhinus canicula*) (Gill and Taylor, 2001), but whether this has any affect at the population level is unknown. Magnetic fields could influence geomagnetic patterns used by some migratory marine species for navigation (Öhman et al., 2007), and reports also show that electro-magnetic fields from OWFs may affect fish migration. Gill *et al.* (2009) identified eight migratory fish species sensitive to electromagnetic fields, including the European eel *Anguilla anguilla*, the Atlantic salmon *Salmo salar* and the Yellowfin tuna *Thunnus albacares*. There is limited *in-situ* data describing the impact of EMF on fish, but of those studies, several indicated reduced swimming speeds (Öhman et al., 2007; Westerberg and Lagenfelt, 2008).

A direct influence to Mediterranean fish populations from the presence of OWFs is the addition of novel, vertical habitat, in an area previously void of hard substratum. Supporters of marine renewable energy developments have often cited the potential for artificial structures to function as fish aggregation devices (FADs) similar to artificial reefs (Ashley et al., 2014). Through colonization of the novel, hard substrata by macrofauna and algal species, juveniles, and small herbivorous fish species are attracted to the artificial substratum, which in turn attracts individuals of higher trophic levels (Figure 1.5).

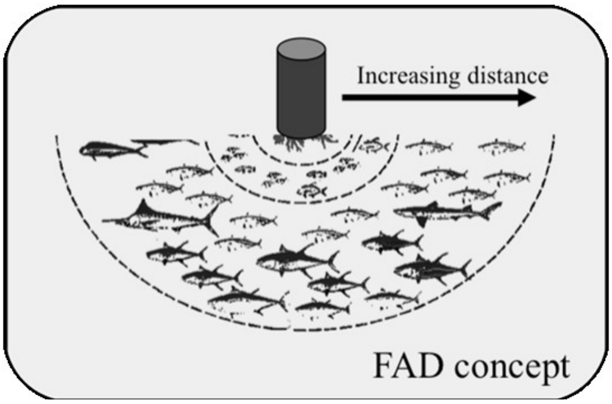


Figure 1.5 Schematic diagram of the FAD concept. Not to scale. Produced by author.

Several studies have found greater abundances of certain fish within OWFs than in comparison to surrounding areas (i.e. Atlantic cod *Gadus morhua*, pouting *Trisopterus luscus,* and several species of gobies), causing possible spill-over effects to the adjacent areas (Krone et al., 2013; Reubens et al., 2013; Wilhelmsson et al., 2006a). Potentially the new habitat provides increased foraging for both primary and secondary food resources, and protection grounds from currents to reduce energy use in juveniles or smaller species (Wilson and Elliott, 2009).

Offshore structures have been shown to be some of the most productive habitats for fish in the marine environment (Claisse et al., 2014) however, there is considerable debate between ecologists as to whether changes in species biomass within an OWF area will be due to production of new biomass from increased food availability, or attraction of existing individuals from outside the system (Reubens et al., 2014a). Stomach content analysis and energy profiling have shown that OWFs are suitable feeding grounds for both Atlantic cod (Gadus) and pouting species (Trisopterus) (De Troch et al., 2013; Reubens et al., 2014b). Juvenile recruitment of Atlantic cod has also been observed at wind farms in the Belgian part of the North Sea (Reubens et al., 2014b). However changes in prey densities may be masked by increased predation rates (Bergström et al., 2013; Russell et al., 2014), and will potentially strengthen predator avoidance behaviours such as diel migration (Reubens et al., 2014b), making the disentanglement of attraction-production dynamics difficult and likely only possible with extensive, long-term (>10 year), data sets (Gill and Taylor, 2001).

It is difficult to state the effects OWF implementation will have on fish communities based on the findings of northern European studies as the majority of existing monitoring programs focus on species that are not generally present in Mediterranean waters (e.g., Atlantic cod), and may have differences in life histories and habitat use (Bergström et al., 2013; Reubens et al., 2013; Reubens et al., 2014a). However, the possibility for creating *de facto* marine protected areas (MPAs) due to fishing restrictions imposed within OWFs is an interesting aspect in the developments of OWFs in the Mediterranean Sea, which has been successfully utilized at offshore platforms in the North Sea (Duineveld et al., 2007). Monitoring of fishing activities in UK wind farms indicates a decrease in both trawling and static fishing at the sites largely due to impracticality (Ashley, 2014; Ashley et al., 2014; Inger et al., 2009). Fishermen are less willing to drag their trawling gear within turbines as they risk entanglement. IN addition, there is a potential to monitor fishing activity of static and recreational fishermen by using fixed cameras to aid in fishing restriction compliance. It is clear that well protected MPAs in the Mediterranean result in significantly higher biomass than those with no or minimal protection (Wilhelmsson et al., 2006a), although many Mediterranean MPAs lack adequate protection (Montefalcone et al., 2009). Enforcement of fishing restrictions in the Mediterranean is a difficult issue, but the benefit associated with the introduction of fixed structures is that fishing regulations may be easier to apply.

1.2.4 BENTHIC COMMUNITIES

The Mediterranean harbours many important benthic habitats including vermetid reefs, coralligenic concentrations, shallow sublittoral rock, seamounts, deep-sea coral reefs, and abyssal plains (Danovaro et al., 2010; Mo et al., 2012). The shallow sub-littoral sediment is a particularly valuable habitat for the Mediterranean benthos, as it is the preferred habitat of the endemic seagrass *Posidonia oceanica*. Posidonia beds are listed as a priority natural habitat under Annex 1 of the EC Habitats Directive (92/43/EEC on the Conservation of Natural Habitats and of Wild Fauna and Flora), due to their endemism, high productivity, and provision of ecosystem services (Jackson et al., 2015; Vassallo et al., 2013). Favourable substratum conditions for OWF construction throughout Europe is typically soft sediment areas, the same habitat most suitable for *Posidonia oceanica*. Thus as both a habitat, and a species, it is at significant risk from several direct physical pressures that include increased sedimentation during construction, and changes in hydrographic regimes during operation of OWFs (Vanhellemont and Ruddick, 2014). Any plans for OWFs in the Mediterranean Sea will have to be carefully designed around the distribution of *Posidonia oceanica* to ensure the correct conservation practices for this priority species.

Additionally, changes in hydrographic regimes during operation of OWFs can cause small-scale shifts in macro-benthic assemblages (Coates et al., 2014; De Backer et al., 2014). Indeed, studies have shown negative correlations between the distance from a turbine, and sediment grain sizes, in the Belgian part of the North Sea (Coates et al., 2014). In the same study, changes in organic matter and a shift in species assemblages were also noted. Most likely due to scouring at the base of the turbine and re-suspension of sediments, the closer to the turbine the soft sediment community samples were taken, the greater the increase in macrobenthic density and diversity, and the larger the grain size (Coates et al., 2014).

During the operational phase of an offshore wind farm, the addition of hard substratum to the area has a large effect on the benthic environment. Recruitment and colonization of artificial habitats provided by turbine foundations, increases the structural complexity and productivity of an environment previously low in *in-fauna* diversity and density (Birklund and Petersen, 2004; Kerckhof et al., 2009, 2010, 2010; Langhamer, 2012; Lindeboom et al., 2011; Maar et al., 2009; Shi et al., 2012; Van Hoey et al., 2004; Wilhelmsson and Malm, 2008), meaning that benthic ecosystem dynamics are modified, and a new local ecosystem is created.

Research at an offshore research platform in the German Bight indicated that 35 times more macro-zoobenthos biomass was associated with the additional hard substratum than the equivalent area of soft benthic sediment (Krone et al., 2013). Although the increase in macro-zoobenthos biomass may increase carbon through-flow within the benthic ecosystem, in many cases, species assemblages associated with artificial structures differ from the environment they replaced, and the long-term effect of change in ecosystem structure and functioning associated with OWF developments is unknown.

Species assemblage is influenced by many parameters including material and texture of offshore structures, larval supply, oceanographic conditions, temperature, salinity, and water depth (Knights et al., 2012; Langhamer, 2012). The number of defining parameters that vary spatially (and temporally) in influencing colonization of offshore artificial structures (e.g. larval supply), highlights the need for extensive area-specific research, long-tem *in-situ* experiments, and ecosystem impact modelling to fully understand regional implications of OWFs. To date only one study has assessed the changes to ecosystem structure and functioning due an increase in hard substrata biomass caused by OWF installation (Raoux et al., 2017). It is important to understand whether OWF installation substantiates functional changes at an ecosystem level, as opposed to impacts to individual species. Failure to address ecosystem impacts in a holistic way could impede efforts to reduce any negative impacts (i.e. decreases in biodiversity) that OWF’s could have on an already strained marine ecosystem.

With regard to the Mediterranean, work into epibenthic colonization has focused on concrete artificial reefs (Fabi, 2002; Fabi and Fiorentini, 1994; Sinis et al., 2000), or anthropogenic structures (Airoldi et al., 2015; Airoldi and Bulleri, 2011; Bulleri and Airoldi, 2005; Ferrario et al., 2016; Ido and Shimrit, 2015; Ordóñez et al., 2013; Perkol-Finkel et al., 2012). Only two studies have investigated an offshore steel structure in the Mediterranean (Goren, 1980; Kocak et al., 1999) (Figure 1.6). Dominating speciesof epibenthic assemblages varied depending on the location and duration of the monitoring program, which ranged from 11 months to 20 years. Most studies note early colonization by hydrozoans, bryozoans, and serpulidae (Fabi, 2002; Goren, 1980; Kocak et al., 1999; Moreno et al., 1994; Relini et al., 2000). In several studies, this was succeeded by the establishment and dominance of the commercially farmed *Mytilus galloprovincialis* (Airoldi and Bulleri, 2011; Fabi, 2002; Moreno et al., 1994; Relini et al., 2000), however several artificial structures showed no such dominance (Badalamenti, 2002; Goren, 1980; Ponti et al., 2015; Relini et al., 2000, 1994). Multiple drivers (food availability, environmental factors) affect regional community structure with the most significant being larval supply, which is largely determined by small-scale hydrodynamics and settlement cues (Pineda et al., 2010). Temporal changes in larval supply may impact species turn-over rates and β-diversity through time, as resident species may be lost or replaced, meaning that long-term monitoring of the benthic community structure on artificial substrates is essential to identify shifts in composition through time. The only long-term data set on a concrete artificial reef (20 years) reported five distinct phases of species assemblage: dominance of pioneer species, mussel dominance, mussel regression, mussel absence, and finally dominance of bryozoan bio-constructions (Nicoletti et al., 2007). Additionally, differences in the material used for offshore structures may have a significant effect on community composition. The surface roughness of a material has been shown to influence macrobenthic community composition (Anderson and Underwood, 1994; Berntsson et al., 2000; Cacabelos et al., 2016); the two Mediterranean offshore steel structures were both dominated by bivalves after 52 and 70 months (Relini et al., 1994, 2000). Due to regional differences in larval supply which impacts benthic colonization communities, regional pilot studies and detailed hydrodynamic analyses are essential for understanding the structure of benthic communities expected to colonize OWFs.

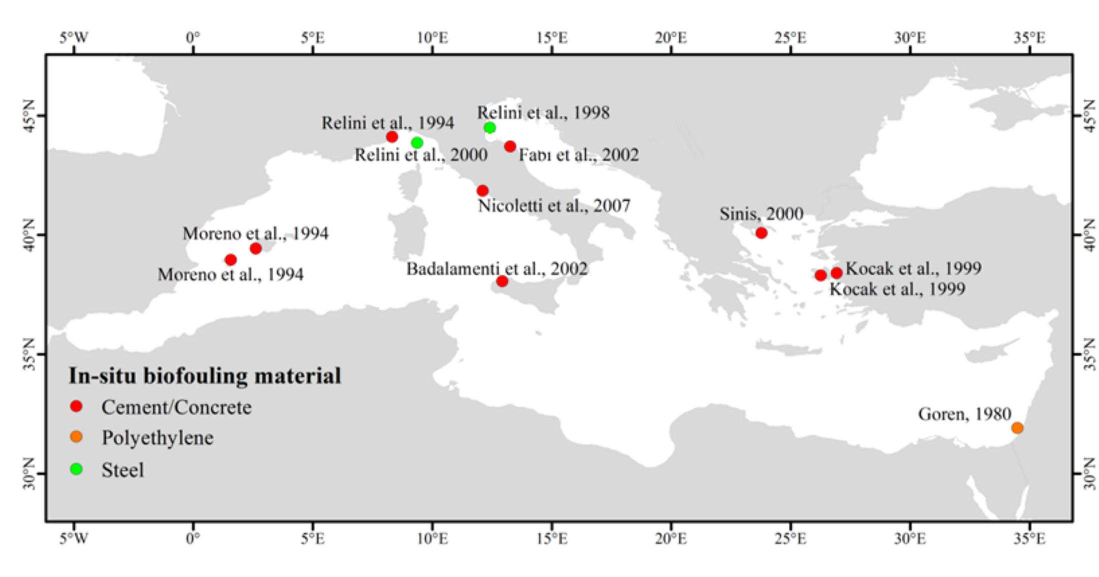


Figure 1.6 Map showing the spatial distribution of *in-situ*, shallow water artificial reef bio-fouling studies on different materials in the Mediterranean. Not including bio-fouling studies on coastal defences (i.e. breakwaters) or marinas.

Finally, the susceptibility of the Mediterranean Sea to non-indigenous species due to the connection with the Red Sea via the Suez canal (Galil et al., 2014), and the colonization of artificial substrata in the Mediterranean by alien species (Airoldi and Bulleri, 2011, Badalamenti, 2002; Çinar, 2006) mean that wind farms may also act as benthic “stepping stones” (Adams et al., 2014). The presence of available hard substrata from wind turbines for the recruitment and settlement of larvae has the potential to extend larval connectivity across biogeographic boundaries (Adams et al., 2014). Wind turbines could facilitate range extension of alien species within the Mediterranean marine environment, which in turn may potentially reduce the β-diversity of the basin by encouraging the replacement of local species over time (Adams et al., 2014; Langhamer, 2012). Considering the importance of this issue for the Mediterranean, identifying areas where benthic connectivity is likely to be high particularly in regions identified as offshore wind farm hotspots (Figure 1.1), is a priority concern.

1.5 SCOPE OF THESIS

With the identification of the five Mediterranean offshore wind farm hotspots (the Gulf of Lion, the North Adriatic Sea, the Gulfs of Hammamet and Gabès, the Gulf of Sidra, and the Nile Delta) (Figure 1.1), there is an ever-growing need to assess the biological costs and benefits of OWFs in the region. It is clear from the literature review that there are many uncertainties surrounding the impacts that building offshore wind turbines in the Mediterranean will have. As one of the most abrupt alterations to the Mediterranean ecosystem is likely to be the addition of hard substratum to a dominantly soft substrata environment, the scope of the thesis will thus focus on the likely impact that offshore wind farm installation will have on benthic communities in the region.

One of the areas identified as an OWF 'hotspot' by the feasibility study due to its relatively high wind speeds and shallow depths, is the Northern Adriatic. Using the wider area (Adriatic/Northern Ionian), as a case study for likely OWF development, the work will address several key questions:

1. What will be the likely community structure of macrobenthic organisms and demersal fish at wind turbines in the Adriatic – Ionian region?
2. Considering the likelihood of offshore structures operating as stepping stones for benthic communities, is it possible to identify areas of high or low connectivity for benthic communities, to help inform marine spatial planners with regards to the licensing of offshore marine renewable energy development?
3. Finally, what are the potential ecosystem-level impacts of building offshore wind farms in this region of the Mediterranean?

The work will take a holistic approach in addressing these questions. Chapters 2 and 3 will use results obtained from an *in-situ* pilot project designed to address the habitat use of simulated OWF pillars by macrobenthic organisms and demersal fish species. Chapter 4 will use a coupled bio-physical model to track the simulated release of benthic larvae from the coastal regions of the Adriatic and Northern Ionian to identify areas of both high and low connectivity. Chapter 5 will use ecosystem-modelling software to describe the likely long-term ecosystem impacts of offshore wind farm construction in the Adriatic. Chapter 6 will present the main conclusions from the study, and consider whether the aims of the thesis have been adequately addressed. The work is intended to provide valuable horizon-scanning recommendations to ecologists and marine spatial planners to best ensure that, where possible, disruptions to the marine environment from the installation of offshore wind farms are limited.

CHAPTER 2

INVESTIGATING THE LIKELIHOOD OF OFFSHORE WIND TURBINES OPERATING AS ARTIFICIAL REEFS FOR BENTHIC INVERTEBRATES IN THE MEDITERRANEAN

Contributors: Laura Bray, Antonella Pancucci-Papadopoulou, Emanuela Voutsinas, , Jason Hall Spencer

Contributions: I designed the experiment with help from AP. I identified all the biofouling organisms and fish species with the help of AP, and EV. I performed the statistical analyses and wrote the manuscript; JMH-S supervised the thesis.

2.1 INTRODUCTION

With the identification of Mediterranean offshore wind energy 'hotspots' (see Chapter 1), it is important to understand the impact that large-scale renewable energy developments will have on the marine environment. As aforementioned, one of the most significant impacts associated with the construction of renewable marine energy devices for benthic marine communities is the addition of novel artificial substratum to soft substrata environments (Inger et al., 2009) (Chapter 1). Artificial structures in the form of offshore wind turbines will certainly be colonized by benthic organisms. Whether or not they will differ from natural benthic community compositions remains to be seen, however, evidence generally indicates that artificial and natural communities will differ (Wilhelmsson and Malm, 2008). In addition, colonization of the artificial substrata may facilitate the creation of an artificial reef-like environment, which if successful (i.e. ability to support increased biomass and diversity levels of higher trophic levels) has the potential to benefit both environmental and commercial interests (Wilhelmsson et al., 2006). Possible attraction of mobile predators towards enhanced feeding grounds, or the provision of shelter by artificial structures may increase biodiversity and thus health of the ecosystem (Westerberg et al., 2013).

In response to the uncertainty surrounding the biological impacts of OWF construction in the Mediterranean, a pilot project was designed as port of the European FP7 program CoCoNET (Towards COast to COast NETworks of marine protected areas, coupled with sea-based wind energy potential) (Grant no. 287844). The pilot project was designed to monitor, and estimate, the types of benthic communities that would inhabit a renewable energy device foundation, in an area earmarked for renewable energy development. Operating at the spatial scale of a pilot project, it aimed to interpret whether an artificial reef-like environment could be created by Mediterranean OWFs, by investigating the potential colonization of sessile organisms and their associated mobile fauna. This chapter describes the methodology used for the pilot project, and examines the succession of epibenthic organisms on different types of artificial surfaces used for offshore marine renewable energy devices.

2.2 METHODS

2.2.1 STUDY SITE OF MEDITERRANEAN PILOT SCHEME

The area selected for the pilot project was the Diapontia Islands in the North Ionian Sea (Figure 2.1) (39.7666, 19.4950). The region is relatively unstudied and pre-sampling cruises were used to characterize the general area. Hydrography, habitat distributions, granulometry of sediment, and bio-chemical properties of the water column were measured to identify the possibility of use as a study area. These campaigns took place in March 2013, July 2013, and November 2013. The area was selected for its suitable water depths and relatively high wind speeds; furthermore it has been subject to the interest of offshore energy developers in the past because of these characteristics (Spiropoulou et al., 2015).

General habitat maps were obtained by divers conducting visual censuses at locations within the sampling area throughout the pre-sampling campaigns. At shallow depths (0 – 10 m), the area is characterized by rocky outcrops that support infra-littoral algal communities and encrusting algae (e.g. *Cystoseira* sp., coralligenous sp. *Dictyopteris* sp.,). At deeper depths (20 m - 30 m), the area is characterized by widespread *Posidonia oceanica* meadows extending up to depths of 35 m, extensive areas of coarse sands and fine gravels mixed by the waves, and small rocky outcrops (Figure 2.2).

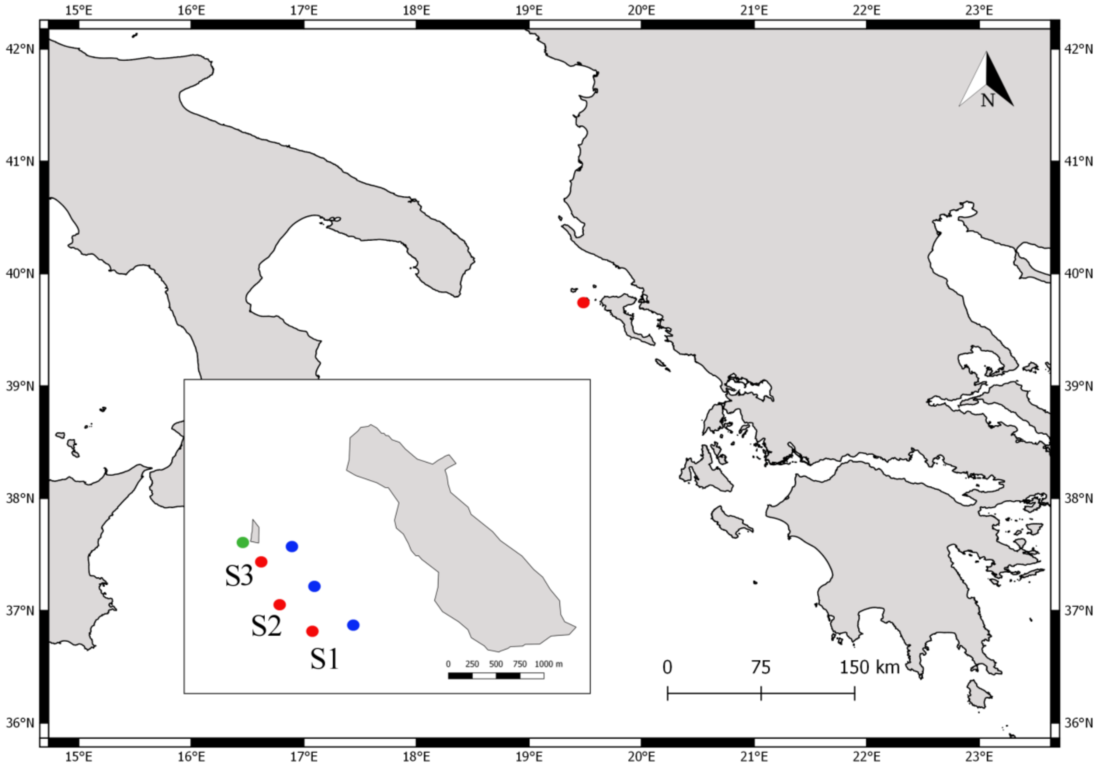


Figure 2.1 Location of *in-situ* experimental units in red (S1, S2 and S3) along the coast of Mathraki Island (Diapontia Islands) in the vicinity of Corfu, Greece. For reference, the location of the natural rocky outcrop in green is included, and three bare substratum control sites in blue (Chapter 3).

A portable CTD (Conductivity, Temperature, Depths) unit (Seabird Electronics SBE19), providing *in-situ* depth profiles of temperature and salinity in the water column was used at each location of the *in-situ* experimental units. In March and July of 2013, the water column was homogenized (Figure 2.3), aside from a small salinity decrease in the upper 2-5 m because of possible local contributions from rainwater run-off. The average water temperature ranged from 15.5° C in winter (March) and 22.1° C in summer (July 2014). Salinities ranged from 38.75 to 39. Lower salinities (38.4 - 38.5) were occasionally observed in the upper part of the water column.

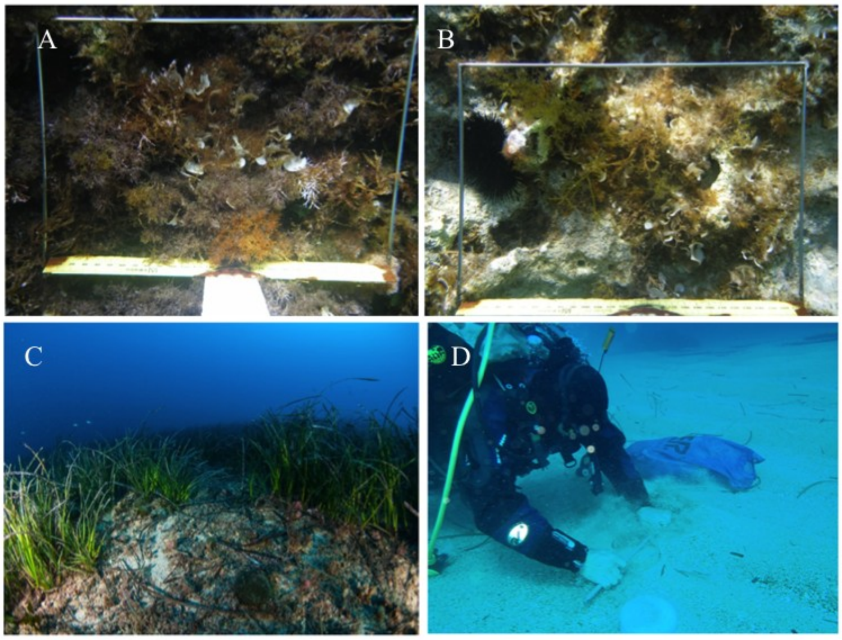


Figure 2.2 A) Photo transect of vertical hard substrata at 15 m depth covered in *Padina pavonica, Halopteris filicina,* and *Haloptilon virgatum.* Photograph taken by Emanuela Voutsinas. B) Photo transect of grazed facies at 5 m depth with encrusting algae, *Cystoseira* sp. and sea urchins. Photograph taken by Emanuella Voutsinas. C) *P. oceanica* meadows, and biocenecis of vermitids at 20 m depth. Photograph taken by Yiannis Issaris, D) Soft substrata location of experimental units at 20 m depth (C1, Figure 2.1). Photograph taken by Laura Bray.

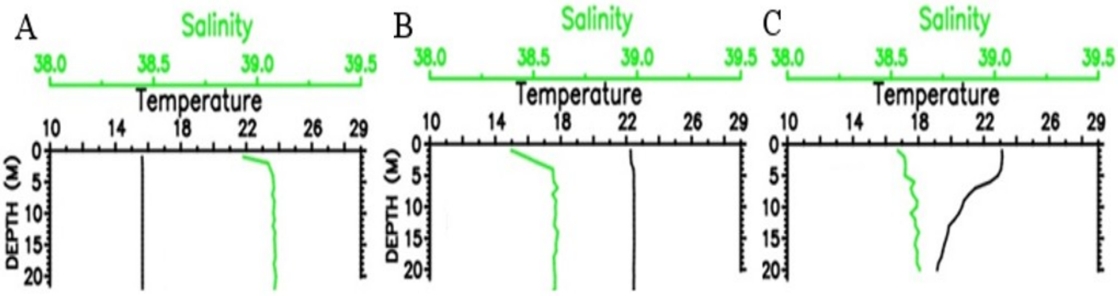


Figure 2.3 Indicative hydrographic (CTD) profiles of temperature in degrees Celsius and salinity during the sampling surveys (39.766 N, 19.495 E) of A = March, B = July, C = November 2013.

For dissolved oxygen (DO) and nutrients, seawater samples were collected at fixed depths (surface, 10, 20, 50 m) during the March 2013 survey using a messenger-activated Niskin bottle (5 litres). Results were averaged to produce a value for the whole water column (Table 2.1). Seawater samples for nutrients analysis were collected in 100 mL polyethylene bottles, “aged” with HCl 10%, and then frozen (-20 °C) until required for analysis (nitrate, nitrite and silicate) in the laboratory (BRAN+LUEBBE III autoanalyser). DO measurements were performed on board, immediately after sampling, using the Winkler method (Carritt and Carpenter, 1960). The area is classified as oligiotrophic for nitrates and ammonium salts, and mesotrophic for phosphates (Table 2.1).

Table 2.1 Average values of dissolved oxygen and nutrients at all locations (Figure 2.1) within the study site

Parameter	Average	Standard deviation
Dissolved Oxygen	7.08mg / L	0.20
Phosphates	0.02 ĩmol / L	0.01
Nitrate	0.15 ĩmol / L	0.08
Nitrite	0.06 ĩmol / L	0.02
Silicates	0.91 ĩmol / L	0.27
Ammonium	0.20 ĩmol / L	0.10

Additionally, during November 2013 and March 2014, a 300-Khz Acoustic Doppler Current Profiler (ADCP) was used to obtain snapshot surveys of the average water column current flow data, and to determine whether there is a prevailing current regime in the area. As sound (pings) transmitted from the ADCP bounces off particles in the water column and is re-received by the ADCP, the shift in frequency (Doppler Effect) can be interpolated to provide an estimation of water currents throughout the water column. Multiple measurements can provide an idea of circulation patterns within an area. Figure 2.4 shows a snapshot representation of the circulation in those two months. In November 2013, the circulation is generally westward with a branch flowing to the north between the islands of Erikousa and Othonoi. The flow seems to decelerate to the west of Erikousa, when it gets over the shallow area between the three Diapontia islands. Typically current speeds are between 10 and 20 cm sec^{-1}. In

March 2014, the flow is to the north, with some divergence to the east. Speeds are again between 10 and 20 cm sec^{-1}.

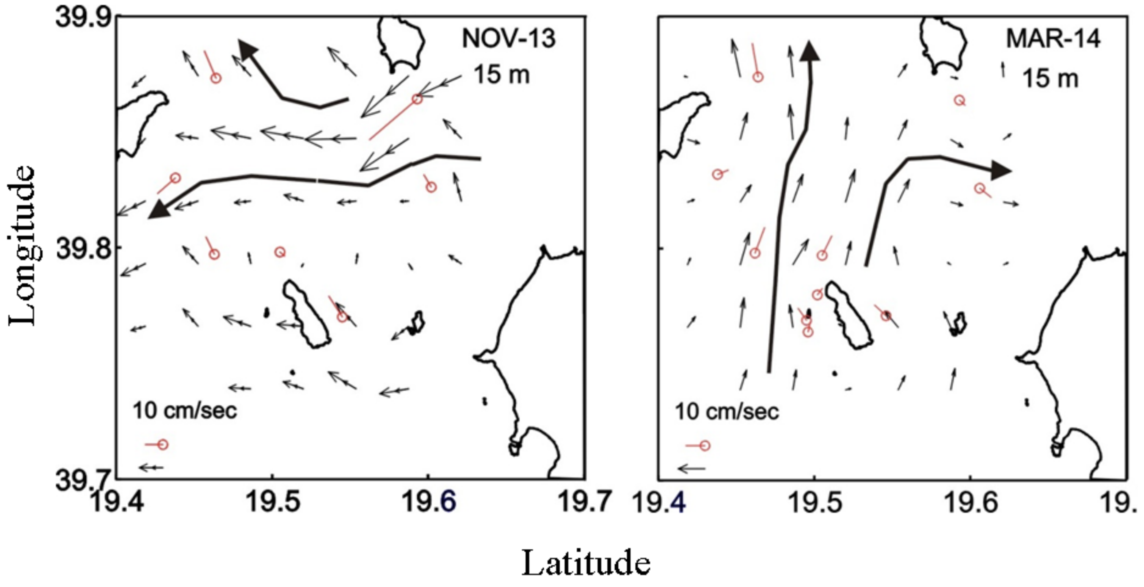


Figure 2.4 Flow fields at 15 meters during November 2013 (left) and March 2014 (right). Black arrows are the optimally estimated currents via objective interpolation based on direct current measurements indicated by the red sticks. In the lower left of each panel are the scaling references for a westerly flow at a speed of 10 cm/sec. Black thick lines with arrows at the end show a schematic structure of the circulation. Green circle indicates location of experimental set-up.

2.2.2 EXPERIMENTAL SETUP

To simulate the presence of an offshore wind turbine, vertical stanchions were deployed at three sites in the area. Locations of soft sediment substratum with depths of approximately 20 m were chosen for artificial structure deployment. The stanchions (S1, S2, and S3) were deployed in a line, 2 km parallel to Mathraki Island, with each stanchion being approximately 200 m apart (Figure 2.1). A nearby area of natural rocky outcrop (R) at 20 m depth, surrounded by soft substrata was also selected as a comparison site (see below). The natural rocky outcrop was located approximately 200m away from the nearest artificial unit (Figure 2.1). The units were deployed via winch and assisting divers from R/V Philia (Hellenic Centre for Marine Research) during March 2014.

Several factors were taken into consideration during construction, chiefly: the material, orientation, substrata type, and exposure period. A cylindrical polyethylene water storage tank (height 205 cm, diameter 95 cm), was encased in a PVC coated, galvanized steel, wire mesh. Holes were drilled at the base of the units, and steel rods inserted in a perpendicular design, and terminally welded onto steel stabilizing panels (50 cm x 50 cm) (Figure 2.5). Removable panels (n = 192) of stainless steel or concrete, two commonly used materials in marine renewable energy devices and their foundations, were cut to a size of 17 cm x 20 cm and used as artificial settlement surfaces. The panels were attached with cable ties to the wire mesh into subdivisions around the unit. These subdivisions comprised of 4 different orientations (N, S, E, and W), and two different heights with respect to the seabed (0 - 0.5 m (low), and 1.5 m - 2 m (high)). Within each subdivision, the positions of steel or concrete panels in relation to each other were randomly assigned. Within each unit, 300 kg of steel mooring chains were used as ballast weight.

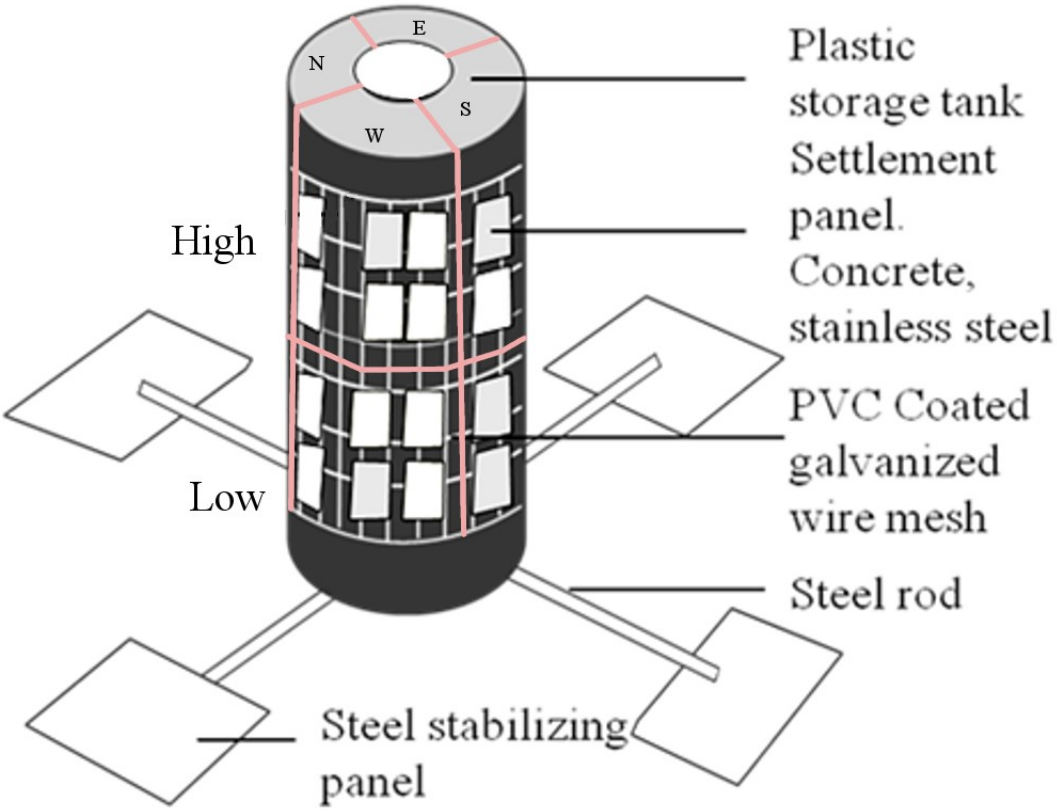


Figure 2.5 Schematic diagram of one of the stanchions that hold the artificial substrata panels. Total height is 205 cm and diameter of plastic tank 95 cm. Pink lines indicate how the plates were divided into areas (High/Low, North/South/East/West).

Pre-immersion, both steel and concrete panels were kept in fresh water renewed daily, for 3 days to remove any lixiviated substances that could interfere with settlement (Guy-Haim et al., 2015). Four replicates for each material (steel and concrete), orientation (N, S, E, and W), and for each height above the sediment (high and low) were attached to each experimental unit. A replicate panel was removed every sampling period to examine the species that had colonised the panel in the laboratory. To accommodate for four sampling campaigns; a total of 192 panels were deployed. Initially the pilot project was intended to run for a period of 18 months, with samples being recovered after 3 months (July 2014), 6 months (September 2014), 12 months (March 2015) and 19 months (October 2015). Unfortunately, destruction of the units before March 2015 by either high wind speeds or human interferences meant the premature discontinuation of sampling and monitoring of impacts (Figure 2.6). Nevertheless the experiment provided information regarding the initial 6-month settlement (March – September) patterns of bio-fouling organisms on the artificial surfaces.

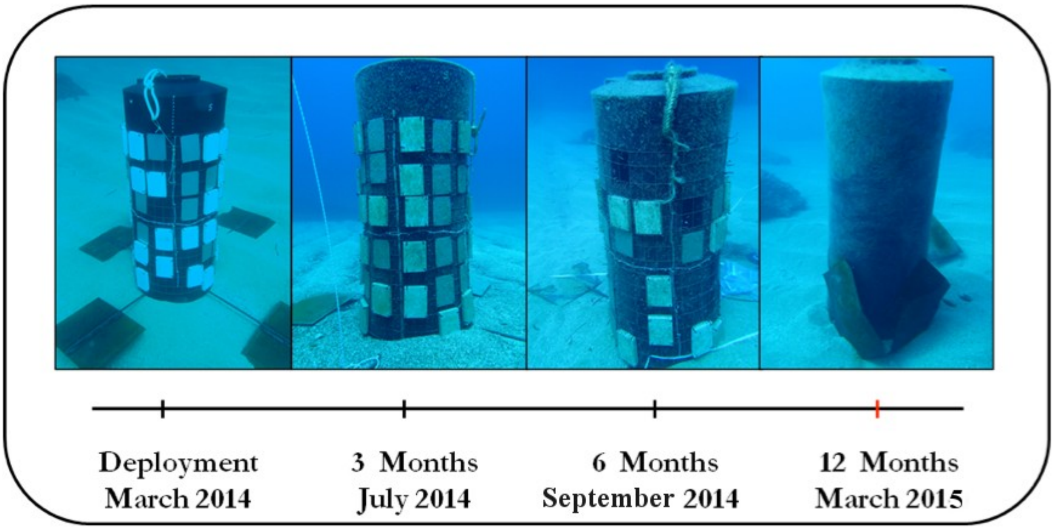


Figure 2.6 Photographic timeline of *in-situ* artificial units including March 2015 where the destruction of the units was discovered.

To determine the colonization patterns of the artificial substratum, the removable panels were carefully transferred *in-situ* to a polyethylene bag and taken to the surface. Photographs were taken of delicate species to aid taxonomic identification. Bio-fouling organisms were scraped from the surface, stored in alcohol, and identified to the highest possible taxonomical level in the laboratory (Zoobenthic laboratory, Hellenic Centre for Marine Research). After identification, species were grouped into polychaetes, crustacean, molluscs, and remaining taxa (e.g. bryozoans, tunicates), and the wet weight of each taxa calculated (College-B B154, Mettler Toledo). Bivalves, tubiculous polychaetes, and gastropods were weighed with their shells/tubes. Colonial organisms i.e. hydroids, bryozoans, colonial tunicates, were not included in the abundance counts due to difficulties in separating individuals, however they were included in the analysis of biomass.

To provide a comparison to a natural hard substrata environment, a nearby rocky outcrop (R) was selected for sampling (Figure 2.7, Figure 2.8). During March 2014 (the deployment date of the artificial substrata units), three similar areas at the rocky reef site were identified for replicates. All sites were between 17 – 22 m depths, and were characterized as westerly facing vertical rock wall adjacent to an area of at least 80 m of soft substrata. At each replicate site, a quadrat of 17 x 20 cm was sampled via MANOSS - a manually operated suction sampler for hard bottom benthos (Chatzigeorgiou et al., 2013). Epibenthic fauna was carefully scraped from the hard substrata and collected in a mesh net of 63 μm with the help of one-way valves and an airlift (Figure 2.8). Considering the area is not subject to benthic trawling by fisherman, and is isolated from any tourist dive locations, it is plausible that the area is not subject to destructive activities and that the sample collected may be assumed to have a relatively stable community structure. After the initial sampling at each site, three replicate areas of 1 m^2 were subsequently cleared of epibenthic fauna and flora using a chisel and a hammer. All attempts were made to ensure that there was a complete removal of epibenthic organisms and that only hard substratum was left, however the author feels that due to the high complexity of the substrata this was not possible, and that small encrusting organisms (bryozoans etc.) are likely to have remained. The corners of the area perimeter were marked with yellow cord with small floaters attached to enable the rediscovery of the plots (Figure 2.8). From the cleared plots subsequent samples were obtained, each time from a new location within the cleared area to ensure no overlapping of sample area occurred. Every time plates were removed from the artificial units i.e. July 2014 (after 3 months) and September 2014 (after 6 months), corresponding samples were taken from the cleared plots. Samples were taken to the surface and passed through a 0.25 cm sieve to remove large rock debris parts, stored in alcohol, and identified to the highest possible taxonomical level.

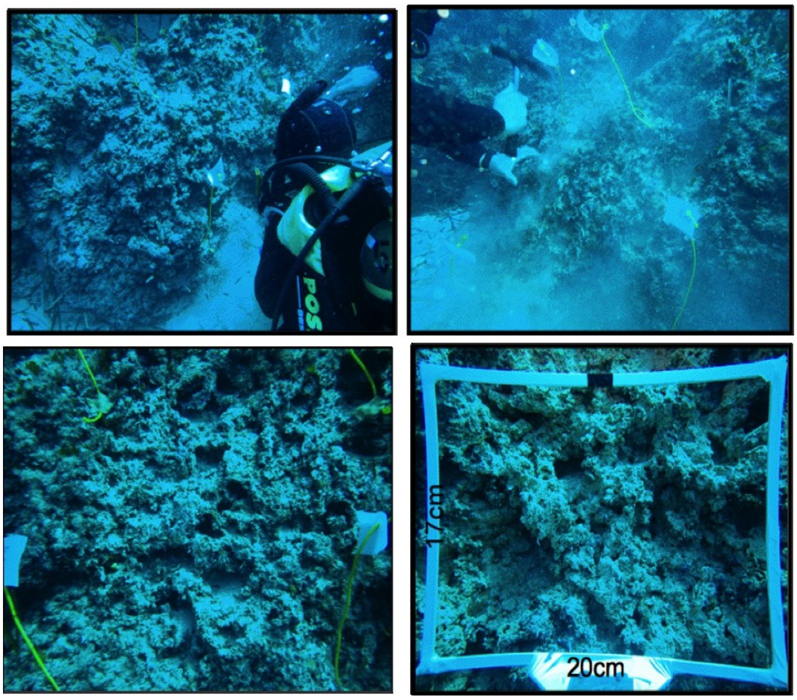


Figure 2.7 Photographs of Rock wall. Top left shows one of three replicate plots pre – clearance. Top right shows diver in the process of clearing plot with a hammer and chisel. Bottom left photograph indicates the cleared 1 m x 1 m plot complete with aides for site re-identification. Bottom right indicates size of surface scraped and taken for comparison with artificial substrata panels. Photographs taken by Laura Bray.

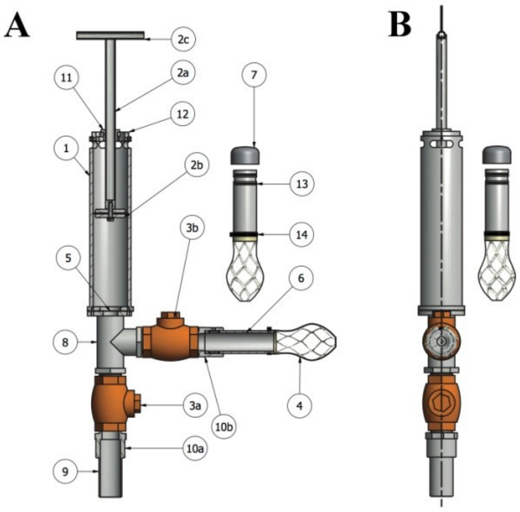


Figure 2.8 Schematic drawing of the sampler. A: side view; B: top view. Numbers indicate the following parts: (1) barrel (length 38 cm, inner diameter 7.5 cm; volume 1.7 l), with openings near the rear end to facilitate water flow; (2) plunger; (3) one-way valves (4) collection sock (5)

filter mesh; (6) connector tube (length 15 cm, inner diameter 4.5 cm); (7) cap of connector tube, equipped with thumbscrew to secure the cap onto the connector; (8) T-shaped connector; (9) nozzle (10) adapters (PVC connections) between T-shaped connector (11) plunger guide (12) rear-end cap (13) two O-rings (14) cable binder. Figure and description sourced from (Chatzigeorgiou et al., 2013).

Statistical analysis

To compare changes in benthic invertebrate abundance with time at the artificial units a linear model was used (nlme package). After establishing it unnecessary for the artificial unit location to be treated as a random factor the fixed model factors included were: (1) Site (S1, S2, S3), (2) Material (Steel, Concrete), (3) Orientation (North, East, South, West), (4) Height (High, Low), (5) Time (3 months, 6 months). After determining the need to consider an autocorrelation structure, a first-order autoregressive (AR(1)) correlation structure was used to account for the repeated measures on the same unit. Data were square root transformed to account for non-normality of the data.

In addition, the diversity (Species richness (Margalef), Pielous evenness (‘), Shannon index (H’), and Simpson index (S)), the number of non indigenous species (NIS), and the total number of individuals in both the natural substrate samples and the artificial substrate samples were compared using a T-test (IBM SPSS statistics 24.0). Data were square root transformed to abide by normal distribution requirements for statistical analysis.

2.3 RESULTS

In total, 11,607 macrobenthic individuals were identified from 95 species, the most common of which were calanoid copepod species, the tainaid *Chondrochelia savignyi*i (Krøyer, 1842), and the serpulid *Spirobranchus polytrema* (Philippi, 1844). Thirteen non-indigenous species (NIS) were identified, of which 4 species (*Hydroides elegans, Hydroides operculatus, Elasmopus pectenicrus,* and *Crepidula fornicata*) are listed within the top 100 worst Mediterranean invasive species (Streftaris and Zenetos, 2006). No statistical difference was noted regarding the abundance of NIS found on artificial substratum vs. the natural reef (t = 0.8671, *p* = 0.377). Likewise no statistical difference was noted in the numbers individuals that were present on plates (18.40 m^{-2} ± 253.75 SD) when compared to the rocky substrata (13.86 m^{-2} ± 82.04) (Figure 2.10) (t = 1.0013, *p* = 0.356). A corresponding difference was apparent for the macroinvertebrate total biomass, with higher biomass on natural rock (3.45 g m^{-2} ± 4.16) compared to the artificial sites (3.04 g m^{-2} ± 6.66), however, as with the abundance, the differences were not noted to be statistically significant (t = 0.85651, *p* = 0.471).

Ecological indicators

Species diversity was higher at the rocky reef site in comparison to the artificial substratum for several ecological indicators including species number (t = -6.4749, p = > 0.001), species richness (t = -7.296, p = >0.001) and Shannon index values (t = -2.9702, p = 0.003). Other species diversity measurements e.g. total number of individuals (t = -0.564, p = 0.577), Pielou's evenness (t = 1.259, p = 0.213), and Simpson's index (t = -0.291, p = 0.776) showed no significant differences of mean values between the two groups (Table 2.2).

Table 2.2 Average of ecological indicators ± standard deviation for the samples obtained from the artificial substratum vs. the rocky reef. S = Total species, N = Total individuals, d = Species richness (Margalef), J = Pielous evenness, H = Shannon index, 1 - λ = Simpson index. Significantly different values are highlighted in bold.

	S	N	**d**	J	**H**	1 - λ
Rocks	**24.78 ± 7.29**	195.22 ± 43.95	**4.49 ± 1.19**	0.56 ± 0.08	**1.79 ± 0.37**	0.66 ± 0.12
Artificial	**12.01 ± 5.22**	173.50 ± 115.63	**2.18 ± 0.85**	0.62 ± 0.12	**1.44 ± 0.32**	0.65 ± 0.11

Macrobenthic community composition

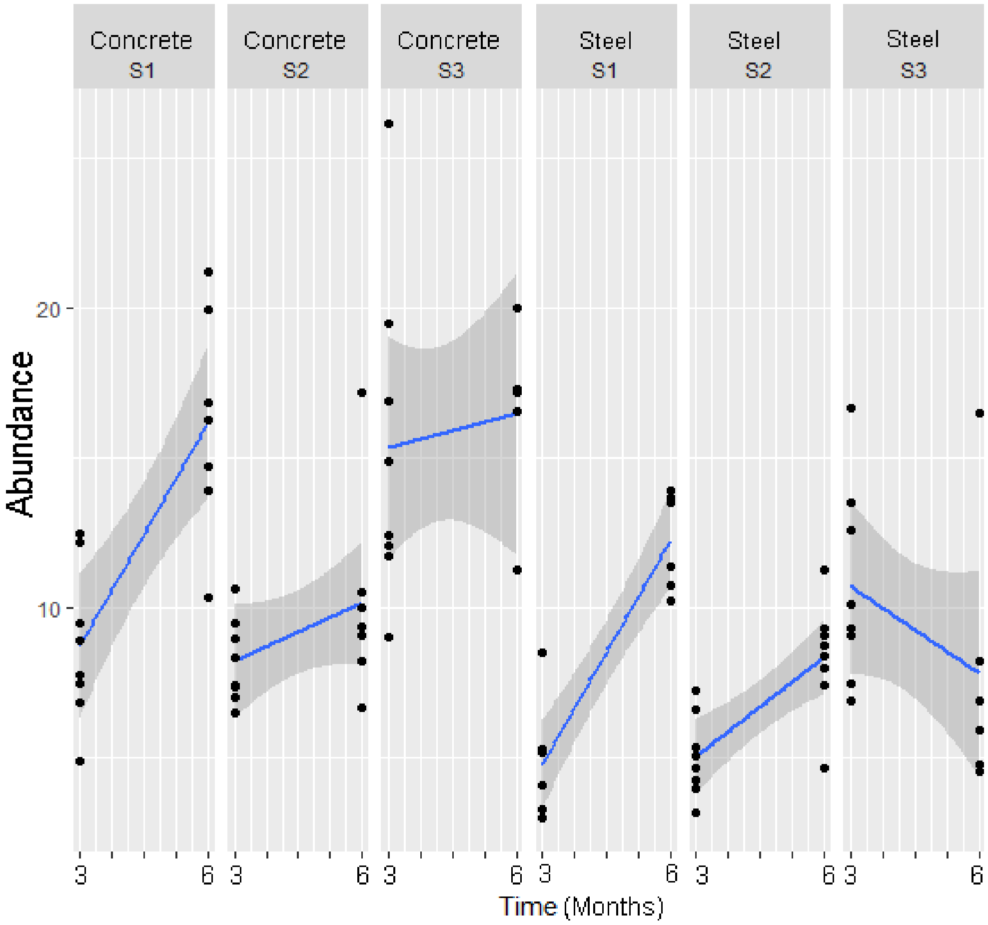
The number of macro invertebrate changed over time, in response to the material used on the artificial units, and depending on the location of the artificial units ($p = > 0.001$) (Table 2.3). Neither the height of the removable plates, nor their orientation influenced the number of macro invertebrate individuals.

Table 2.3 Results from linear model for abundance of macro-invertebrates. Statistically significant (p-values > 0.05) are highlighted in bold. DF = Degrees of Freedom

	DF	F-value	p-value
(Intercept)	**1**	**63.870**	**<.0001**
Material	**1**	**43.947**	**<.0001**
Time	**1**	**4.379**	**0.041**
Site	**2**	**16.290**	**<.0001**
Orientation	3	1.311	0.281
Height	1	2.233	0.141
Material:Time	1	0.504	0.481
Material:Orientation	3	0.304	0.822
Site:Orientation	1	0.498	0.485
Site:Material	**1**	**56.210**	**<.0001**
Site:Height	2	1.547	0.221
Site:Time	**2**	**12.670**	**<.0001**
Time:Orientation	3	0.915	0.440
Material:Height	1	2.657	0.109
Time:Height	1	0.157	0.694
Orientation:Height	3	0.914	0.441
Material:Time:Orientation	3	2.610	0.061
Material:Time:Height	3	0.785	0.38
Site:Material:Time	2	1.332	0.271
Site:Material:Height	2	0.021	0.979
Site:Time:Orientation	2	4.346	0.020
Site:Time:Height	2	0.436	0.649
Material:Orientation:Height	3	1.237	0.306
Time:Orientation:Height	3	0.470	0.704
Site:Material:Time:Orientation	6	2.212	0.063
Material:Time:Orientation:Height	6	1.030	0.387

Colonization was higher on concrete plates in comparison to those constructed with steel for each unit location. For both steel and concrete plates, the unit S3 (the unit closest to the natural rock wall) had a higher numbers of individuals after the first three months (Figure 2.9). The unit S1 had the increased greatest rate of increase from three to six months for both materials. The abundances of macro-invertebrates only decreased with time (from an average of 17.707 ± 23.294 to 12.821 ± 24.448) on the steel plates obtained from S3.

Figure 2.9 Regression plots of temporally auto-correlated abundance per site, per material and per time period (3 and 6 months).



2.4 Discussion

The statistically significant biodiversity values (species richness, species number and Shannon index) all indicated that samples obtained from the natural rock wall were higher in comparison to the artificial substrates. Thus corresponding with a large body of evidence that indicates that although artificial substrates support hard bottom benthic communities (as indicated by statistical similarity of the number of species in both habitats), they are not analogues of a natural rocky habitat, and that the expansion of artificial substrates may contribute towards regional diversity loss (Fauvelot et al., 2009). In addition, artificial substrates often host higher numbers of non-indigenous species in comparison to natural habitats likely due to competitive advantages over native species (Airoldi et al., 2015); however this was not the case for the present results. No significant differences of non-indigenous species observed between the natural and artificial habitats is good for developers and marine spatial planners who wish to counteract the spread of non-indigenous species in the region, however considering that other studies have consistently found differences between in natural and artificial habitat in the Ionian and Adriatic basins (Bulleri *et al*., 2005; Perkol-Finkel *et al*., 2006; Airoldi et al., 2015), it is possible that the scale of the pilot project (6 months) was not sufficient enough to determine whether a significant difference would exist in time.

Regarding the colonization of marine renewable energy device in the region, the results indicated that not all the tested variables had an effect on the community composition of the macrobenthic communities. Neither the direction that the substratum faced in the water column, or the height above the seabed, impacted the abundance or biomass of the macrobenthic communities. Local and small-scale hydrodynamics play a role in influencing larval settlement on a substrata (Pineda et al., 2010), and the Strait of Otranto has a complex hydrological pattern. It is an area of water mass exchange between the Adriatic and the Ionian seas, with current patterns varying on a variety of timescales, ranging from tidal to inter-annual (Ursella et al., 2013). The complex coastal topography of the Diapontia islands, variable prevailing winds (Michelato and Kovacevic, 1991), and the lack of a dominant circulation feature in the area suggest that current regimes in the area may have regularly changed throughout the sampling period (Figure 2.4). Therefore the orientation of the artificial substratum in the water column appears to not significantly impact the abundance or biomass of macroinvertebrate settlers in the region. In regards to the height above the seabed, no significant impacts to the macroinvertebrate abundance may be due to the size of the stanchions not being high enough to produce evidence of any vertical zonation (Andersson et al., 2009).

The material used however (whether it was concrete or steel), significantly influenced the number of macroinvertebrate individuals. Similar to Azevedo et al., (2006), more individuals were present on concrete panels in comparison to steel counterparts. Small-scale variations in roughness are highly influential to community composition (Cacabelos et al., 2016), however this is often species dependent, with many colonizing species preferring either smoother or rougher surfaces (Andersson et al., 2009). Additionally the high alkalinity of the boundary layer at a concrete substrata favours the settlement of some macrobenthic organisms (e.g. the hydroid *Laomedea* spp., and the tunicate *Ascidiella* spp. (Andersson et al., 2009; Petersen and Malm, 2006), however, due to the pre-soak procedure of the methodology it is not clear whether the alkalinity of the boundary layer has had an influence on the results presents here.

At most sites, the number of individuals increased with time even after temporal autocorrelation, and the number of individuals on steel and concrete plates increased at a similar rate with each other at each site. Only stanchion S3 had a minimal increase (concrete plates) or a marginally lower number of individuals (steel plates) after 6 months in comparison to replicates taken after 3 months of colonization (after temporal auto-correlation), potentially indicating a more stable community composition (Knowlton, 2004). In addition the concrete plates of S3 were consistently higher in numbers of individuals in comparison to the other stanchions. This may be due to its relative proximity to the natural rock outcrop, and thus simulation cues from the existing communities coupled with a higher larval supply from the source community (Bowden et al., 2001; terHorst and Dudgeon, 2009).

The results indicate that the artificial units are not surrogates for natural hard substrate environments (due to lower levels of biodiversity), however development of epibenthic communities is supported, whilst being dependant on the type of material used. In addition the present results indicate the importance of OWF siting within a region, most likely due to differences in larval supply. It is clear that for effective marine spatial planning, developers should thoroughly consider hydrodynamic regimes and connectivity networks in the region before construction of offshore wind farms commences.

CHAPTER 3

INVESTIGATING THE LIKELIHOOD OF OFFSHORE WIND TURBINES OPERATING AS ARTIFICIAL REEFS FOR LOCAL FISH COMMUNITIES IN THE MEDITERRANEAN

Contributors: Laura Bray, Antonella Pancucci-Papadopoulou, Yiannis Issaris, Jason Hall Spencer

Contributions: I designed the experiment with help from AP and YI. I identified all the fish species with the help of YI. I performed the statistical analyses and wrote the manuscript; JMH-S supervised the thesis.

3.1 INTRODUCTION

Artisanal fishing and large-scale commercial fisheries are socially and economically important in Mediterranean coastal communities (Papaconstantinou & Farrugio, 2000) yet overfishing is a serious problem throughout the region despite the best intentions of the European Common Fisheries Policy (Vasilakopoulos et al., 2014). Nevertheless, there are examples worldwide where the decline of fishing stocks has been reversed by good regional management practices (Tsikliras et al., 2015); one option put forwards as a solution to declining fish stocks is the use of artificial reefs to manage aquatic resources as they can provide fish habitat and increasing local biomass (Fabi et al., 2015).

Many artificial reefs have been deployed in the Mediterranean Sea, most of them constructed from concrete units (Fabi et al., 2011). Evidence as to whether these artificial reefs achieve their objective is inconclusive (Scarcella et al., 2015). It appears that the success of artificial reefs depends largely on location, construction material and management. The Food and Agriculture Organization has produced a set of guidelines for the use of artificial reefs in the Mediterranean (Fabi et al., 2015); there is no mention on the possibility of offshore energy structures serving as artificial reefs.

Offshore energy structures such as oil and gas platforms create similar habitat to offshore wind farms and can provide no-trawling areas which may provide a sanctuary for vulnerable marine species (Langhamer, 2012). Of the few Mediterranean studies at existing offshore gas station structures, some show no effects on fish (Castriota et al., 2011; Fabi, et al., 2002), with others indicating increases in fish abundance, biomass and diversity (Consoli, et al., 2013; Fabi et al., 2004; Scarcella et al., 2011). Whether offshore marine renewable energy devices can provide habitats that enhance fish populations remains to be seen. To investigate the likelihood of an OWF operating as an artificial reef, an analysis into the impacts a simulated OWF will have on local fish communities is presented here. Using the same artificial stanchions as described at the beginning of the chapter (Section 2.3.1), underwater visual transects were conducted in the areas surrounding the artificial vertical relief structures, and compared to control sites. The results are presented and discussed, and an analysis is provided addressing the effectiveness of OWFs operating as artificial reefs in the region.

3.2 METHODS

The *in-situ* setup consisted of 4 fixed transects (25m in length) attached to each artificial unit in a cruciform pattern, which enabled measurements of benthic fish spatial aggregations. The permanently attached transect line allows acclimatization and reduces the ‘tape effect’ seen in several non-herbivorous fish species (Dickens et al., 2011). Fractional underwater visual transects were used to allow the censuring of cryptobenthic species at different distances from the artificial units, a method already proved useful when studying fish assemblages around gas platforms in the Mediterranean (Andaloro et al., 2011), albeit with the possibility of diver presence influencing fish presence (Titus et al., 2015). Surveys were done by two divers, along a transect divided into three set distances from the artificial units (at 0 - 5 m distance, 5 - 15 m distance, and 15 - 25 m distance from the units) (n = 12). The location of individuals in each set distance, the species, and the estimated length, were recorded for each transect. Transects were recorded on video (Nikon COOLPIX AW120 Waterproof Digital Camera) in order to assist accuracy in species recording via post-hoc analysis of footage (Figure 3.1).

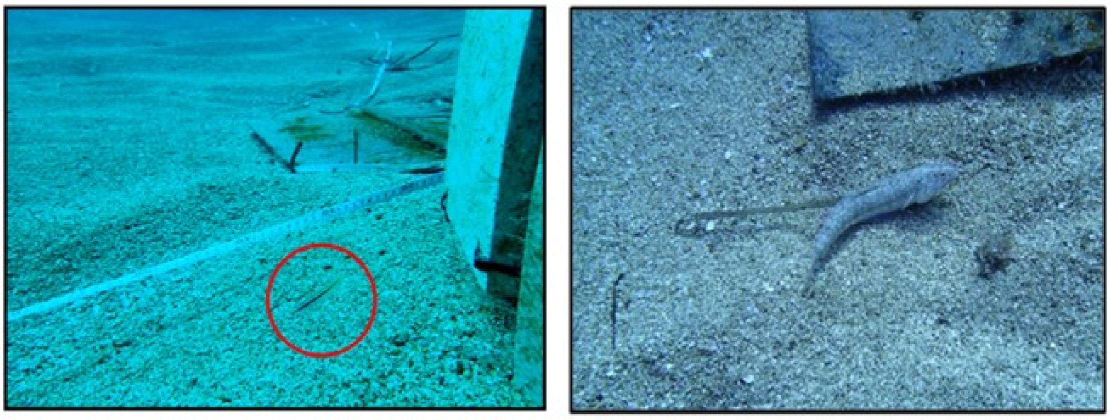


Figure 3.1 Screenshots of videos taken during visual transects of Bucchich's goby *Gobius bucchichi* (left) and the striped goby *Gobius vittatus* (right) in the near vicinity of a simulation unit. The fixed tape is visible in the left frame.

To provide a comparison between areas with vertical relief structures and areas without, three control sites of bare substrata areas at similar depths, and void of any hard substratum or Posidonia beds were also sampled (n = 12) (Figure 2.1). Additionally 'reference' conditions for natural vertical relief structures were investigated. Transects were conducted at a nearby rock wall (Figure 2.1, Figure 2.7) following the same methodology (n = 8). The two same divers completed all the 25 m transect, to minimize bias. Sampling was done between 03 - 15 July 2014, and all transects were taken within the same time period (12:00 - 15:00).

3.2.1 Statistical analysis

To determine whether habitat impacts the abundance of fish individuals, a one-factor ANOVA was performed. The three levels of the habitat factor were the artificial unit, bare substrate, and the natural rocky outcrop. Data were square root transformed and Tukey's test was used for post-hoc comparisons. To investigate differences in the levels of biodiversity between the three habitats, the Shannon index (*H'*) was applied (Spellerberg and Fedor, 2003), and compared with a one-factor ANOVA. Index values were not transformed.

In addition, a two-way ANOVA was used to compare changes in fish abundance at increasing distances from the artificial units. After establishing that the unit location should not be treated as a random factor, (by using a linear mixed-effects model (nlme package)), the fixed factors considered were: (1) Unit (S1, S2, and S3) and (2) Distance from unit (0-5 m, 5-15 m, 15 – 25 m). Data were calculated as individuals per 10m^3 and were arc-sin transformed to account for non-normality of the data. Tukey's test was used for post-hoc comparisons.

3.3 RESULTS

In total, individuals from 22 species were encountered during the conducted visual transects (Figure 3.2), including the invasive species *Siganus luridus,* which originally originates from the Red Sea, and has been present in the Mediterranean Sea since the opening of the Suez canal (Galil, 2009). A comparison of the number of fish individuals at each habitat type (bare substrate, artificial unit, and natural rock wall) indicated that habitat type had a significant influence on the variance of the samples (F = 4.793, *p* = 0.013). The natural rock wall had significantly higher abundance than both the bare substrata control site (*p* = 0.015), and the artificial units site (*p* = 0.01). No statistical differences were observed between the artificial units and the control site (*p* = 0.975). (Figure 3.3).

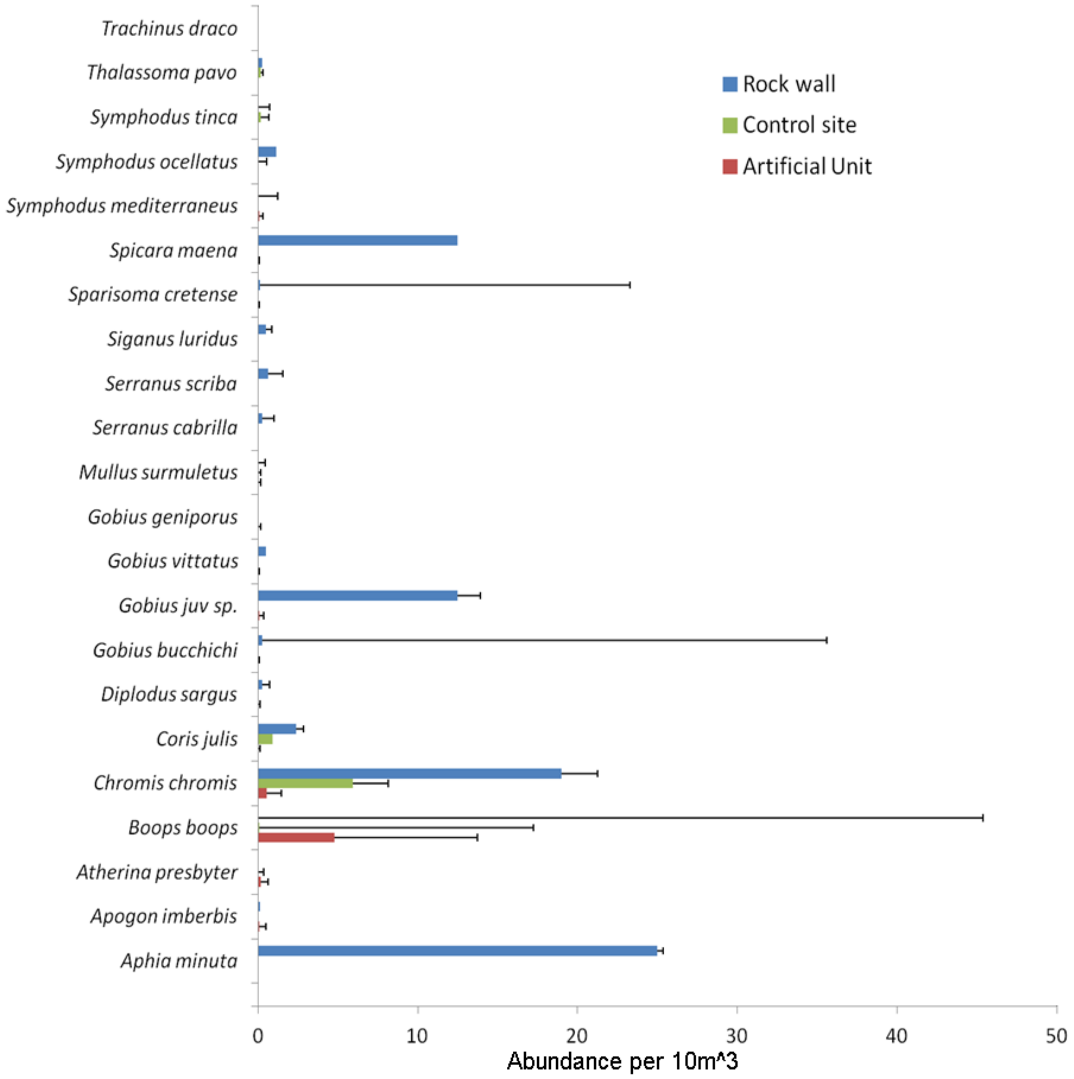


Figure 3.2 Total list of species encountered during visual transects and averaged means/10m^3 at each site with the inclusion of the standard deviation for each site.

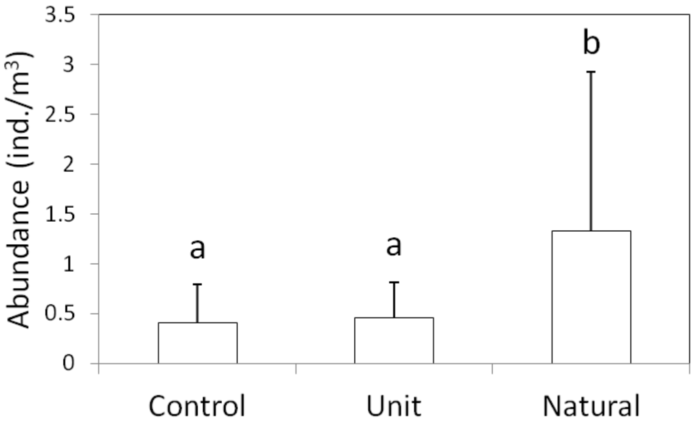


Figure 3.3 Mean (± SD) abundance of total fish species at each habitat (Artificial unit, with the inclusion of 1 standard deviation. Letters (a,b) indicate significantly different groups.

To investigate any differences in the levels of biodiversity at the sites (units, control site and natural reef) the Shannon-Wiener index was applied. Similar levels of diversity were noted between the artificial units (*H'* = 0.89) and the control site (*H'* = 1.02), and a higher diversity at the natural site (*H '*= 1.68), however the values were not significantly different (F = 1.641, p = 0.156).

To investigate any potential aggregation effects caused by the artificial unit to the spatial distribution of fish, comparison of the medians between the varying distances from the units was made. The results indicate aggregation of fish individuals around the unit compared to those measured at a distance further than 15 m. The one-factor ANOVA indicated that distance from the artificial unit significantly influenced the number of fish individuals (F = 3.17, *p* = < 0.001), and post-hoc testing indicated that only the furthest measured distance (15 - 25m) was significantly different to the closest (0 - 5m) (*p* = 0.037). The distances 5 – 15m, and 15 – 25m showed no significant differences (*p* = 0.340), as did the number of fish measured around the unit compared to a distance of 5 – 15 m (*p* = 0.376) (Figure 3.4).

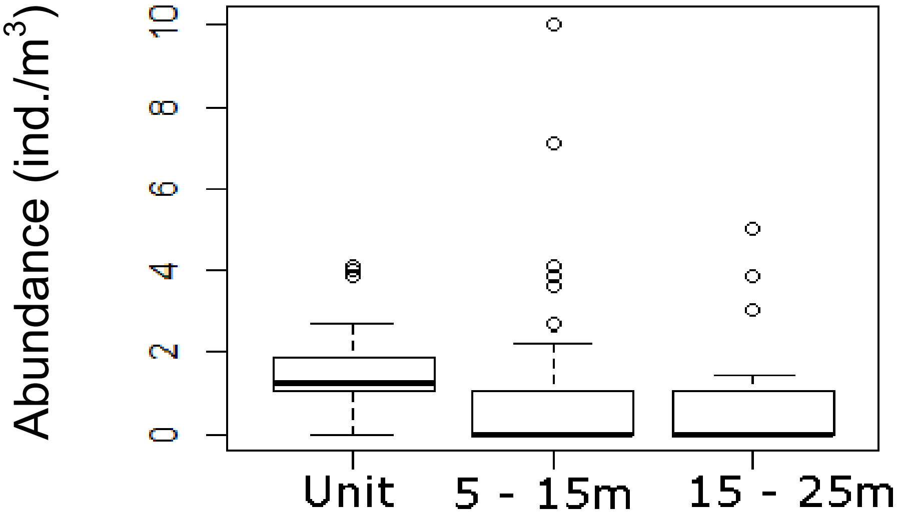


Figure 3.4 Average abundance of total fish species at increasing distance from the artificial unit with the inclusion of 1 standard deviation.

2.3.3 DISCUSSION

Surprisingly few studies have been conducted in the Mediterranean to assess the role that offshore installations have on motile fauna (Gomiero et al., 2013). The varied results from impact assessment studies at offshore gas stations indicate both a statistical increase in fish abundance at some sites (Consoli et al., 2013; Fabi et al., 2004; Scarcella et al., 2011), and no observed differences at others (Castriota et al., 2011; Fabi et al., 2002). The lack of operational marine renewable energy devices in the Mediterranean makes empirical evidence on their marine impacts unattainable.

The results presented in this chapter indicate that the pilot project had a limited impact on fish communities in the region. No statistical differences were observed between the number of fish present at the artificial unit, and those present in the control areas of bare substrates or the species diversity, thus contradicting several prominent field experiments which have demonstrated the reef effect of offshore wind farms on pelago-benthic and demersal fish communities in northern European seas (Wilhelmsson and Malm 2008, Maar et al 2009, Andersson and Ohman 2010, Leonhard et al 2011, Reubens et al., 2014b). This may be due to the scale of the pilot study not accurately describe the conditions of offshore structures despite it being of a similar design to other field experiments (Andersson et al., 2009; Wilhelmsson et al., 2006b). In addition, when compared to the natural vertical rock wall, both the control sites and the artificial units were statistically lower in both fish abundance than the natural rock wall, with the architectural complexity of the natural rocky outcrop evidently contributing to the increase in motile species (Diamant et al., 1986).

However, despite no indication of biomass production increases at the unit installations when compared to a control site, there is evidence of fish aggregation around the units. More individuals were present around the unit then at a distance of over 15 m from the site, suggesting that “attraction” as opposed to “production” is the dominant process in force (attraction/production debate, Chapter 1).

The findings indicate that offshore structures at the site may not operate as successful artificial reefs, and that just the construction of offshore structures, particularly in a region which suffers from high levels of over fishing, may not be a viable option for rectifying ecological damage or reversing stock depletion, as has been suggested in Northern European case studies (Bearzi et al., 2008). It is likely that regional management practices will play a much more prominent role in fish production than the addition of habitat to the region. The likelihood of fishing restrictions around offshore structures being adopted should be an important consideration for ecologists and marine spatial planners. Potential benefits from restricted fishing zones may bolster any weakly correlated positive impacts to fish assemblages from offshore wind farms (Guidetti and Sala, 2007).

3.4 ARTIFICIAL REEF POTENTIAL

Understanding the impacts that offshore structures, and in particular offshore wind farms, will have on the marine environment is becoming an ever-important consideration for marine spatial planners in the Mediterranean. The Marine Strategy Framework Directive considers maintaining biodiversity, and seafloor integrity, as essential aspects in the drive to reach good environmental status (GES), in all European Seas by 2020, and that offshore man-made structures will directly impact environmental status (Directive 2008/56/EC).

For effective use in policy planning stages, it is essential to recognize the scope and limitations of pilot projects. This is particularly true for impacts revealed at large spatial scales and those long term temporal periods e.g. the influence on marine mammal foraging behaviours (Russell et al., 2014). The OWF pilot project conceived by the CoCoNET project is the first of its kind in the Mediterranean, and has provided interesting results regarding the initial settlement of macrobenthic communities. Nevertheless several caveats are attached to the work. Despite evidence from the literature of their limited impacts to benthic communities (Chapter 1), investigating the role of noise, electro-magnetic fields, and changes in hydrology due to rotor down draft are outside the scope of this study and the likely impacts have not been investigated here. Additionally, an important consideration of the pilot project is the duration of the experimental period. A similar OWF pilot project for offshore relief structures in the Swedish part of the North Sea, had a continuous sampling duration of 5 months, with only a limited survey being performed after 12 months (Andersson et al., 2009). Due to a lack of large individuals and an absence of several taxa (i.e. chlorophyte green algae) expected to colonise artificial substratum in the area (Airoldi et al., 2015), it is likely that six months is not an adequate time frame to reach the final stage of ecological succession, however the 6-month duration provides an idea of initial settlement compositions.

Despite the limited time frame, the results indicate that the natural habitat appears to support a higher macro-invertebrate biodiversity, and number of individuals, than the artificial stanchions after six months. Furthermore, of the three artificial stanchions, one appears to respond differently in terms of the number and diversity of macrofauna communities. The apparent high natural variability highlights the importance of small scale siting within a wind farm.

Regarding the benthic fish communities, no increase of abundance are apparent with the placement of the artificial structure in an area of soft substrata, and the artificial reef potential appears limited. The number of defining parameters involved in influencing the spatial and temporal colonization of offshore artificial structures highlights the need for extensive area-specific research alongside holistic environmental impact assessments to fully understand regional implications of offshore wind farms.

CHAPTER 4

IDENTIFYING LEVELS OF CONNECTIVITY FOR BENTHIC COMMUNITIES IN THE ADRIATIC

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Contributions: I designed the experiment and methodology with some input from KT. DK produced graphs from the model outputs. I performed the statistical analyses. I wrote the manuscript with the help of DK. JMH-S supervised the thesis.

4.1 INTRODUCTION

Offshore structures such as oil rigs and wind farms can act as ‘stepping stones’ for benthic communities to expand their ranges across bio-geographic boundaries (Adams et al., 2014). These structures quickly become colonised by fouling organisms (Chapter 2), and over a period of years, can develop diverse assemblages of sessile organisms (Bergström et al., 2014). Primarily this is due to the larvae of most benthic marine organisms being transported via currents, providing a source for recruitment and benthic colonisation. For species with a 24 hour pelagic phase their larvae can travel around 1 km, species that have long pelagic phases the larvae can travel hundreds of km (Shanks, 2009). This dispersal mechanism is particularly important for sessile macro-invertebrates (Grantham et al., 2003) and the strength of connectivity between populations may help determine their ecological success (Melià et al., 2016; Treml et al., 2012). Paradoxically, assessments of the level of larval connectivity offshore structures have with other marine regions are not routinely applied.

Although marine renewable energy developments have not yet begun in the Mediterranean, the Adriatic is being considered for large scale wind farm developments as the region is windy and the sea bed is shallow and well suited to offshore construction (Bray et al., 2016). This chapter considers potential larval connectivity of benthic macro-invertebrates in the region. This knowledge may help in predicting the types of communities that will colonize OWF’s (Joschko et al., 2008; Wilhelmsson and Malm, 2008), and help assess whether they will encourage the spread of non-indigenous species (Bianchi, 2007), both of which are important aspects for the consideration of marine managers.

A relatively limited number studies have empirically measured the dispersal of marine larvae over large geographic scales (Cowen et al., 2006; Jones et al., 2009; Weersing and Toonen, 2009;). In-direct methods include the use of genetic markers, geochemical markers, tagging devices, and bio-physical dispersal models - all of which have various arguments in favour of and against (e.g. cost, accuracy, issues of scale) (Calò et al., 2013). Bio-physical models are able to track simulated larvae individuals over large temporal and spatial scales (Andrello et al., 2014) although there are major assumptions used with most hydrodynamic-based models, the most significant being the assumed passive nature of the individual larvae particles (Metaxas and Saunders, 2009).

In the Mediterranean, few studies focus on the connectivity and dispersal of marine species (Calò et al., 2013), and this paucity of information is an obstacle for policy makers in the region (Andrello et al., 2015; Muñoz et al., 2017). Those connectivity studies that use virtual particle trajectory methods tend to focus on the establishment and evaluation of marine protected areas (Andrello et al., 2013; Di Franco et al., 2015; Pujolar et al., 2013). Other approaches include the homogenous release of larvae particles throughout the whole Mediterranean (Dubois et al., 2016; Rossi et al., 2014), or the release from specific coastal sites at a regional level (Carlson et al., 2016; Melià et al., 2016; Schiavina et al., 2014; Schunter et al., 2011). Many such studies are tailored to determine connectivity of fish, and macroinvertebrate larvae trajectories are seldom modelled in the Mediterranean (Guizien et al., 2014; Padrón and Guizien, 2015; Schiavina et al., 2014).

In this chapter, a release of larvae from benthic populations along the coasts of the Adriatic Sea using coupled bio-physical models is simulated. Furthermore, the effect of pelagic larval duration (e.g. simulation duration) on dispersal is investigated. A homogenous larval production was assumed and evenly distributed Lagrangian particles for a range of pelagic larval durations (4, 8, 16, 20 days) were tracked to cover regionally common invertebrate taxa such as barnacles and gastropods (Villamor et al., 2014), rather than utilizing a particular target species (Rossi et al., 2014). The likely spread of larvae from benthic populations that originate from the major Adriatic ports was also tracked, as several studies show an increase in the abundance of non-indigenous species in or around Adriatic ports and marinas (David et al., 2007; Iveša et al., 2015; Pecarevic et al., 2013). The potential spread of non-indigenous species through corridors of artificial surfaces (Airoldi et al., 2015) is a critical aspect of marine connectivity studies. Essentially, the aim of this chapter is to identify areas of high vs. low connectivity to enable decision makers to effectively site offshore marine renewable energy developments, either with the aim of increasing the range of native species (high connectivity), or decreasing the spread of non-indigenous species in the area (low connectivity).

4.2 METHODS

The method was based on the Graph Theory approach used by Rossi et al. (2014) for identifying hydrodynamic provinces throughout the Mediterranean. Lagrangian particles were released from evenly distributed grid cells along the Adriatic coastline and then tracked for a range of known pelagic larval durations. Source and destination grid cells were compared to indicate regions of high and low connectivity.

4.2.1 STUDY AREA

The Adriatic Sea has a shallow northern section (average depth 40 m), a central section (average depth 140 m) and a southern section where troughs > 1200 m deep (Figure 4.1) channel deep water masses into the Eastern Mediterranean, particularly in late winter (Gačić et al., 2002; Malanotte-Rizzoli et al., 1997). The western coast is generally sandy, whereas the eastern side is predominantly rocky (Artegiani et al., 1997). The hydrography of the basin is influenced by several large rivers (Verri et al., 2014). The overall circulation is cyclonic, with three cyclonic sub-systems in the northern, middle and southern sections and a strong current flowing south along the coast of Italy from spring until autumn (Zavatarelli et al., 1998) (Figure 4.1).

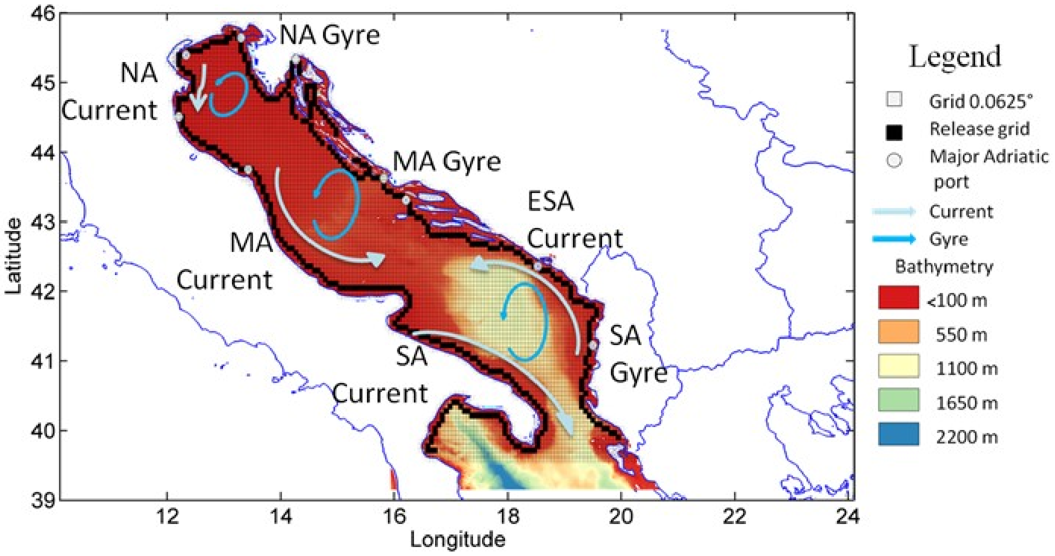

Figure 4.1 Adriatic larval connectivity matrix comprised of a 1/16th degree grid into which larval particles were released, showing locations of major ports (clockwise from left: Taranto, Ancona, Ravenna, Venice, Trieste, Rijeka, Sibenik, Split, Tivat, Durres), and major currents (NA = Northern Adriatic, MA= Mid-Adriatic, Sa= South Adriatic, ESA= Eastern South Adriatic. Bathymetry provided by www.emodnet.eu, hydrology adapted from (Artegiani et al., 1997) .

4.2.2 HYDRODYNAMIC GRID

Hydrodynamic model output data were obtained from the Mediterranean Monitoring and Forecasting Centre of the Copernicus Marine Environment Monitoring Service (http://marine.copernicus.eu), which has been running since 2000. The model is composed of an Ocean General Circulation Model (Tonani et al., 2013) and a coupled hydrodynamic-wave model with a horizontal grid resolution of 1/16˚ (ca. 6-7 km).

The following model description is sourced from the Copernicus website:

“The OGCM code is based on NEMO (Nucleus for European Modelling) version 3.4 (Madec et al 2008). The code is developed and maintained by the NEMO-consortium. The model solves the primitive equations in spherical coordinates. The Wave dynamic is solved by a Mediterranean implementation of the WaveWatch-III code. The background error correlation matrix is estimated from the temporal variability of parameters in a Reanalysis. Background error correlation matrices vary monthly for

each grid point in the discretized domain of the Mediterranean Sea. The assimilated data include: Sea Level Anomaly (a satellite product accounting for atmospheric pressure effect is used) from CLS SL-TAC, and vertical temperature and salinity profiles from Argo, XBT (eXpandable BathyThermograph) and Gliders. Objective Analyses-Sea Surface Temperature (OA-SST) fields from CNR-ISA OSI-TAC are used for the correction of surface heat fluxes with the relaxation constant of 40 W m-2 K-1. The analysis is done weekly, on Tuesday, for the previous 15 days. The assimilation cycle is daily (24hr) and is done in filter mode. 10-day forecast is produced every day. The forecast is initialized by a hindcast every day except Tuesday, when the analysis is used instead of the hindcast."

The Adriatic was subdivided into a 0.0625° x 0.0625° grid (each grid cell approx. 6.7 km^2) to match the resolution of the hydrodynamic model, providing 383 grid cells where larval particles were released from (release cell) (S1).

4.2.3 SIMULATED LARVAL TRANSPORT

Simulated larval particles were released from the centre of each of 383 grid cells along the Adriatic coastline, and trajectories were followed using the program ICHTHYOP (Lett et al., 2008). No behavioural parameters were assigned to the simulated larval particles thus assuming a passive trajectory. For the horizontal dispersion of the particles within the simulations, the default setting of ICTHYOP (10^-9 m^2/s^3) was used for TKE dissipation rate. Particle position was calculated every 2 hours, for four pelagic larval durations (4, 8, 16 and, 20 days). Consecutive release dates (n = 10) throughout June (starting from the 01/06 each year) were chosen to coincide with peak benthic macroinvertebrate spawning in the region (Villamor et al., 2014). Particles were released at the same time each day (00:00), and to account for inter-annual variability, the larval dispersal simulations were run for consecutive years covering the period 2011 - 2015 (n = 5). For each larval duration, a cumulative total of 3830 particles were released. A limited tidal range in the Adriatic Sea means atmospheric effects are the main forcing factors in the Adriatic Sea (Bolaños et al., 2014). With respect to this, particle releases were not factored around tidal stages as other larval dispersal models have done in more tide-dominant environments (Narváez et al., 2012).

4.2.4 POST SIMULATION ANALYSIS

Destination grid cells were calculated for each particle using MATLAB6.1, and both descriptive statistics and probability matrices were constructed from an amalgamation of all simulation years and release dates for each larval duration. Additionally a year-on-year analysis of the total distances that particles travelled was done to examine significant differences between years at specific locations. Due to the non-normal distribution of the data, non-parametric tests (e.g. Kruskall-Wallis and Mann-Whitney U Comparison) were used. To visualise the inter-annual differences of the larval trajectories a single simulation track from each year is presented which indicates particle position for 4, 8, 16 and 20 day durations. Locations of OWF's in early planning/concept stage as of April 2017 are included for reference.

Simulated larvae were considered to have self-replenished if, by the end of the simulation, particles remained in their original release grid. Probabilities of particle arrival were mapped for each grid cell and particle transport distances were calculated. To provide information on larval transport from industrialized regions (Figure 4.1), release grids located closest to the ten major Adriatic ports were selected and the particles released from these sites were presented separately.

The web-based network algorithm software *Infomap* was used to define network structure via the use of Graph Theory (Rosvall and Bergstrom, 2008) (www.mapequation.org). It enables the determination of where larval transport can be expected to flow quickly and easily between the grid cells of the Adriatic, by taking advantage of the duality between finding community structure in networks and minimizing the description length of a random walker's movements on a network (for a full review see Rossi et al., 2014). The communities identified by the *Infomap* software are presented visually along with the relative connection strength between each community.

In addition to community detection, *Infomap* also provides information on the importance of individual nodes via the use of its pageRank algorithm. PageRank (commonly used for ranking web pages) provides a nonlocal measure of centrality by defining the expected density of hypothetical random walkers on a node at stationary, within a weighted, directional, network (Lambiotte and Rosvall, 2012). PageRank for each cell is presented as a probability distribution with a numerical value between 0 and 1, i.e. a cell with a pageRank of 0.5 means that a random walker within the network would have a 50% chance of arriving at the given cell. Identifying the highest and lowest ranked nodes for each pelagic larval duration illustrates the most and least important grid cells within each network.

4.3 RESULTS

As expected, simulated increases in the duration of particle transport resulted in an increase in the distance travelled. Likewise, as dispersal duration increased, self-replenishment decreased. Overall levels of self-replenishment were very low, but were an order of magnitude higher at release grids close to Adriatic Ports (Table 4.1), likely due to the typical positioning of ports in enclosed bays. Dispersal distances increased from around 11 km for 4-day simulations, to 30 km for larvae that could survive for 20 days in the plankton. The greatest distance travelled by a particle during the 20 day simulation was 334 km (Table 4.1). The large Standard Deviations around each mean show that some particles remained close to the simulated release sites, whereas others travel far; this variability increased with dispersal duration.

Regarding inter-annual differences of the distances that the simulated particles travelled, the non-parametric (due to extreme outliers of the data) statistical test

Kruskal-Wallis test for equal medians was used to compare differences between years. All the pelagic larval duration simulations were significant in difference between years (For PLD4 H(2) = 856.82, *p* = 0.00; PLD8, H(2) = 661, *p* = 0.00; PLD16, H(2) = 480.91, *p* = 0.00; and PLD20, H(2) = 387, *p* < 0.01. Post hoc Mann-Whitney tests for yearly differences within each PLD showed most years are significantly different, with only 6 years not showing any significant differences (Table 4.2).

Table 4.1 Descriptive statistics for the range of particle trajectories (4-20 days) Avg. = Average, SD = Standard deviation, SR = self replenishment.

	4 days	**8 days**	**16 days**	**20 days**
Furthest distance (km)	88.7	205.5	308.3	334.7
Avg. distance (km) ± SD	11.0 ± 11.0	16.8 ± 17.3	25.7 ± 28.1	29.5 ± 34.0
Avg. Distance from ports (km) ± SD	7.6 ± 6.3	12.2 ± 12.4	20.3 ± 12.7	24.4 ± 30.9
Avg. SR(%) ± SD	0.01 ± 0.1	0.01 ± 0.00	0.01 ± 0.00	0.01 ± 0.00
Avg. SR at port sites (%) ± SD	0.11 ± 0.12	0.07 ± 0.06	0.04 ± 0.07	0.04 ± 0.07

Table 4.2 Matrix of Mann-Whitney test value (U), and probability (*p*), for each comparison of year-on-year particle distance for each dispersal duration. Highlighted in bold are yearly comparisons which show NO statistical differences. Values for Mann-Whitney test value (U) are shown to 3 s.f, and probability values (*p*), are shown to 2 significant figures.

		2012	2013	2014	2015
PLD4	2011	U = 7020000, *p* = 0.04	U = 5500000, *p* = 0.00	U = 6260000, *p* = 0.00	U = 5320000, *p* = 0.00
	2012		U= 5300000, *p* = 0.00	U = 6030000, *p* = 0.00	U = 5100000, *p* = 0.00
	2013			U = 6600000, *p* = 0.00	**U = 7190000, *p* = 1.00**
	2014				U = 6450000, *p* = 0.00
PLD8	2011	**U = 7220000, *p* = 0.93**	U = 5760000, *p* = 0.00	U = 6400000, *p* = 0.00	U = 5260000, *p* = 0.00
	2012		U = 5870000, *p* = 0.00	U = 6500000, *p* = 0.00	U = 5380000, *p* = 0.00
	2013			U = 6690000, *p* = 0.00	U = 6860000, *p* = 0.00
	2014				U = 6230000, *p* = 0.00
PLD16	2011	**U = 7070000, *p* = 0.18**	U = 6820000, *p* = 0.00	U = 6840000, *p* = 0.00	U = 5560000, *p* = 0.00
	2012		U = 6630000, *p* = 0.00	U = 6640000, *p* = 0.00	U = 5420000, *p* = 0.00
	2013			**U = 7260000, *p* = 1.00**	U = 6060000, *p* = 0.00
	2014				U= 5990000, *p* = 0.00
PLD20	2011	**U = 7 080 000, *p* = 0.26**	U = 6940000, *p* = 0.00	U = 7000000, *p* = 0.02	U = 5770000, *p* = 0.00
	2012		U = 6 750000, *p* = 0.00	U = 6800000, ***p* = 0.00**	U = 5620000, *p* = 0.00
	2013			**U = 7240000, *p* = 1.00**	U = 6100000, *p* = 0.00
	2014				U = 6030000, *p* = 0.00

4.3.1 PARTICLE TRANSPORT

In agreement with the statistical analysis of the year-on-year differences between the distances travelled by individual particles, the spatial depiction of the particle trajectories indicates high inter-annual variabilities (Figure 4.2). Larval sink locations (locations where particle tracks terminate) are not consistent, and although no clear inter-annual trends are apparent, an increased inter-connection between the east and west coasts after 2013 is noticeable. The model simulates particle transport from the central-eastern coastline to the west coast within the 41° – 44° latitudes for the years 2013, 2014, and 2015

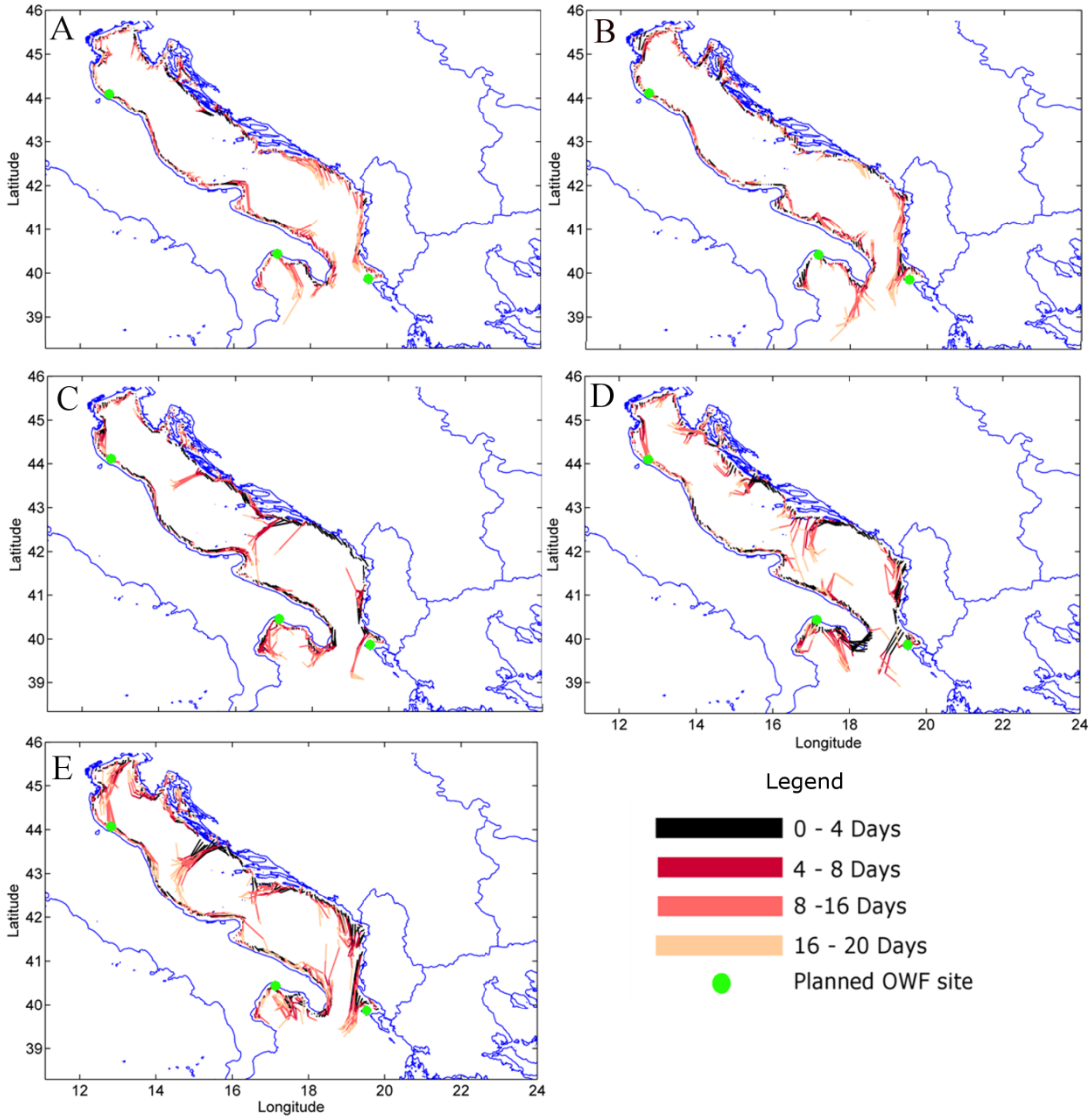

Figure 4.2 A single-track simulation to indicate source/sink information is presented with particle position taken from simulations of the 1st of June for each year. Panel A indicates the trajectory for the 1st of June 2011, B = 2012, C = 2013, D = 2014, and E = 2015. Positions shown for each time interval (0 - 4 day, 4 - 8 days, 8 - 16 days, and 16 - 20 days), and locations of offshore wind

farms currently in the early planning/concept stage in the region are also depicted (http://www.4coffshore.com).

After a simulation duration of four days, regions of grid cells with high concentrations of larval trajectory destination points include the region south of the river Durres delta, the port of Rijeka, the Kvarner Gulf (Croatia), the Gulf of Trieste, Gulf of Venice, and many locations along the Italian Adriatic Coast. Regions with grid cells of lower count densities include the Po river delta, and the offshore region of the Dalmatia coast. Similar results were found for 8, 16, and 20 day durations, with areas of low densities of larval trajectory destination points being mostly restricted to offshore regions such as the Bay of Kotor, the southern Region of Gulf of Trieste, and the Po river delta (Figure 4.3). A more detailed depiction in the form of a probability matrices is provided in the supplementary files of Bray et al., (2017).

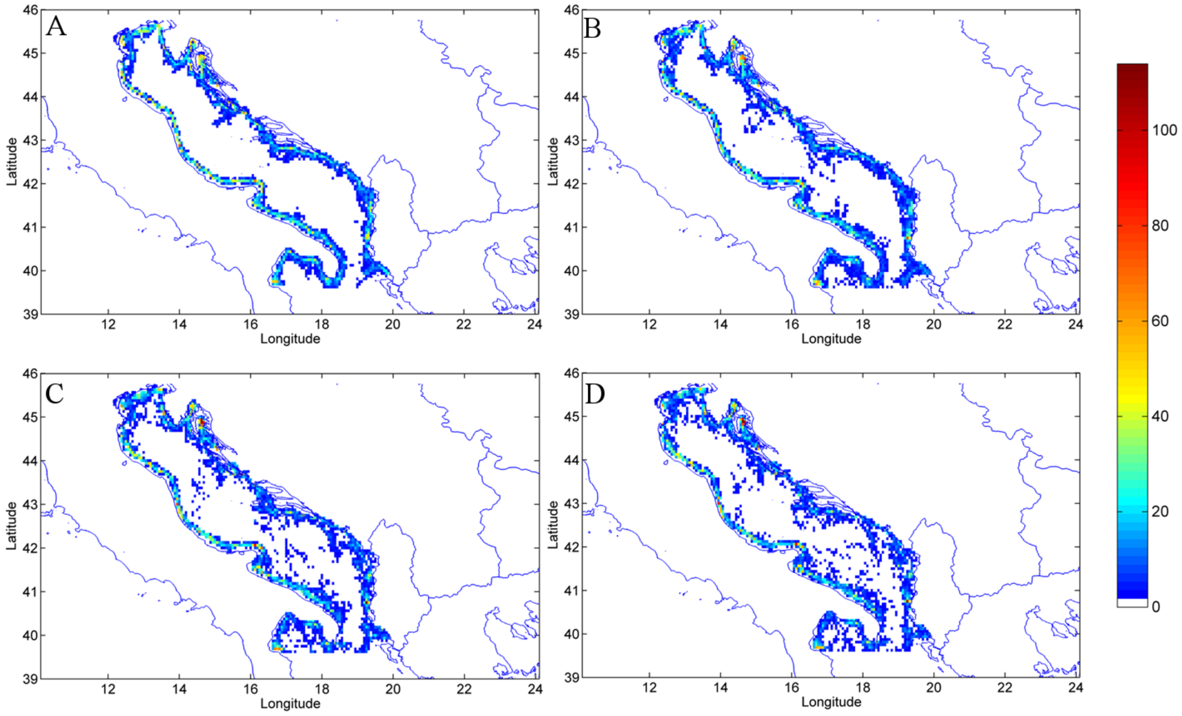

Figure 4.3 Grid count densities (i.e. number of particles in each grid at the end of simulation) of destination points of larval trajectories for A) four, B) eight, C) 16 and D) 20 day larval durations. Counts measured in absolute terms.

A high concentration of port destination cells was also located close to Split, the largest passenger port in the region, for all durations. For simulation durations of 8, 16 and 20 days, large sections of Albanian coast indicated high concentrations of larval trajectory destination points that were released from areas in close vicinity of ports (Figure 4.4).

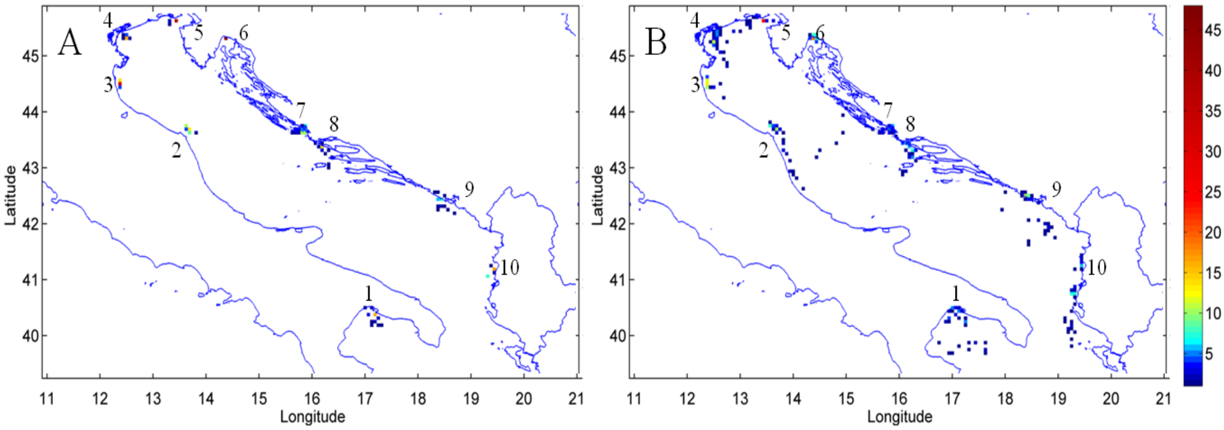

Figure 4.4 Grid count densities (i.e. number of particles in each grid at the end of simulation) of destination points of larval trajectories for each grid cell closest to each major Adriatic port for A) four and B) 20 day larval dispersal. Numbers indicate port locations: 1= Taranto, 2= Ancona, 3= Ravenna, 4= Venice, 5= Trieste, 6= Rijeka, 7= Sibenik, 8= Split, 9= Tivat, 10= Durres. Count densities are not defined by their release points. All PLD simulations produced similar patterns, albeit increasing dispersion with increasing larval duration so for convenience only the minimum and maximum larval durations are displayed Counts measured in absolute terms.

The grid cells within the network with relatively high self-replenishment included the Manfredonia Gulf, and the Adriatic coast of Italy. Regions of relatively lower cells of self-replenishment include the Po delta, south of the Gulf of Trieste, and the northern coast of Croatia (Figure 4.5).

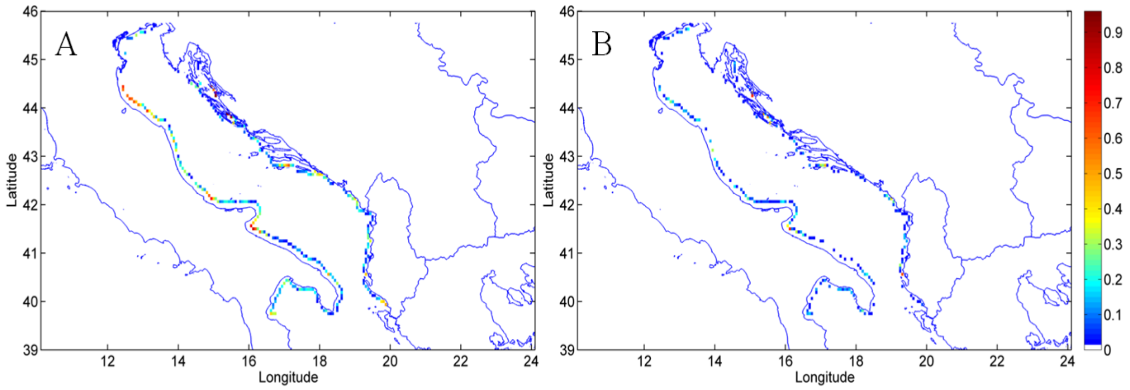

Figure 4.5 Percentage of self-replenishment for each release grid cell for A) four and B) 20 day larval durations. All PLD simulations produced similar patterns, albeit increasing dispersion with increasing larval duration so for convenience only the minimum, and maximum, larval durations are displayed. Increasing the larval duration means the self-replenishment of most release grids along the eastern coast of the Adriatic approaches 0%.

4.3.2 CLUSTERS AND NODE CENTRALITY WITHIN NETWORK

An increase of simulation duration resulted in fewer numbers of identified communities with the mapequation algorithm. *Infomap* clustering visualization (Figure 4.6), indicated that the four day larval duration, a transport network with 2022 nodes and 4883 links, was clustered into 76 modules with 110 inter-module links. The eight day larval duration, a network of 2362 nodes with 6462 links, was clustered into six modules with two inter-module links. Both the 16 (2650 nodes with 7484 links), and the 20 day simulation durations (2764 nodes with 7812 links) were clustered into four modules with one inter-module link. Figure 4.6 indicates the number of separate communities identified by the mapequation algorithm and the connection strength between them at the end of each simulation period.

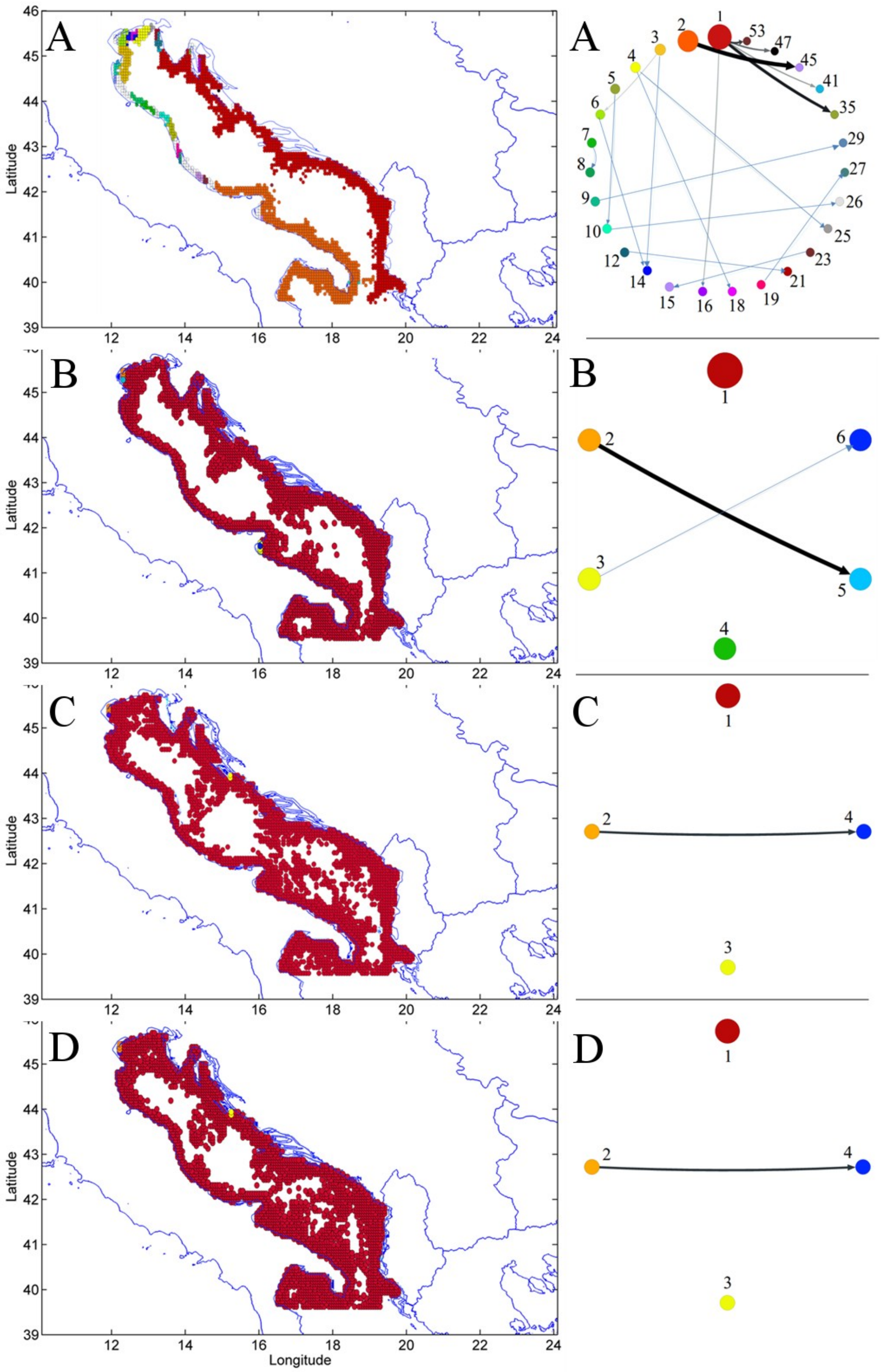

Figure 4.6 Community detection outputs from mapequation algorithm displayed spatially A) four B) eight, C) 16 and D) 20 day larval durations. Relative strength of connection, and thus thickness of arrows, between clusters is automatically calculated by the *Infomap* software and is

presented here as purely indicative. Numbers represent the communities identified by the *Infomap* software.

Mapping the PageRank for each grid cell indicated that for all PLD's the Adriatic coast of Italy contained some of the highest ranked grid cells, along with two locations within the Kvarner Gulf (Croatia), thus indicating these regions contained some of the most connected cells within the network. Regions with consistently lower ranked grid cells and thus less connected were the offshore basin regions (all durations), include the Po river delta (four day), and the Montenegro and Albanian coast (eight, 16, and 20 day durations) (Figure 4.7)

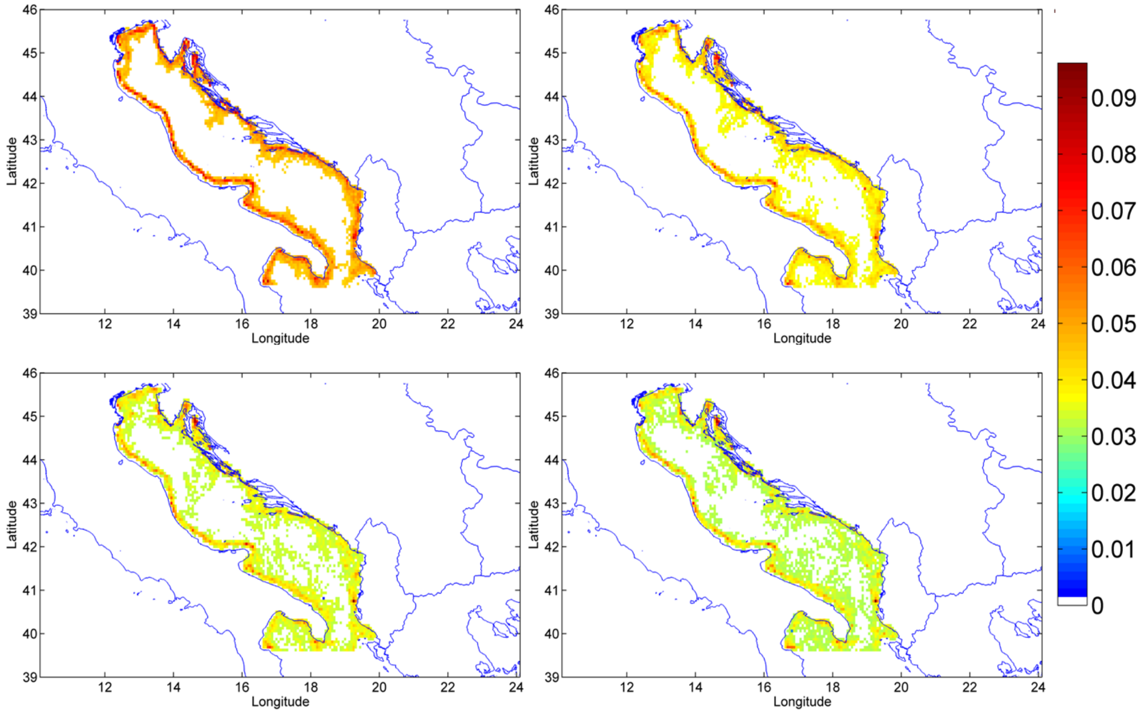

Figure 4.7 Spatial display of PageRank for each grid cell included within the network for A) four, B) eight, C) 16 and D) 20 day larval durations. No distinction made between grid cells given a PageRank value of zero and grid cells not assigned a Pagerank value. The scale indicates the probability distribution as numeric values ranging between 0 and 0.1.

4.4 DISCUSSION

The Adriatic region is a distinct marine sub-region assigned as a priority region for marine spatial planning (Bastari et al., 2016). High activity use often creates conflicts between economic development of the region, habitat protection, and preservation of biodiversity. The region already has a great deal of offshore activities (Manoukian et al., 2010) and there is scope for rapid development of offshore wind farms, particularly in the Northern Adriatic (Chapter 1). The Adriatic Sea is managed nationally with several of the six coastal states sharing the use of territorial waters; the co-ordination of marine management in this region is often fragmented. Optimised site selection is an important consideration for all offshore marine developments (Falconer et al., 2016, 2013), and due to the interconnected cross-boundary nature of marine systems, the approach presented here may prove useful in fostering basin-scale management of the biological impacts of offshore construction in the Adriatic Sea.

4.4.1 METHODOLOGICAL APPROACH

Three dimensional particle tracking models are useful for quantifying the dispersal of benthic invertebrate larvae (Metaxas and Saunders, 2009) and Graph Theory is an effective tool for exploring patterns of spatial connectivity (Treml et al., 2007). This approach has been widely used for the identification and evaluation of marine protected areas; however this is the first time it has been used as an aid for planning offshore construction. Nevertheless, there are several limitations associated with the approach which likely contribute to an overestimation of predicted dispersal distances (Shank et al., 2009). Real-world realization of the findings presented here requires additional information such as individual larval behaviour (Zhang et al., 2016), predator-prey interactions, environmental cues, and suitable substratum availability for settlement (Chan and Walker, 1998). The homogenous release of passive particles along the Adriatic coastline does not accurately reflect nature, as it can be assumed a more heterogeneous release of larvae will take place based on habitat type (e.g. rocky shore vs. soft substrate) but it does provide an insight into larval dispersal over large scales.

4.4.2 PARTICLE TRANSPORT

The distance larval particles were transported was shorter than other works which assessed dispersal distances in the region (Melià et al., 2016) as shorter pelagic larval durations were used. Some of the most prolific bio-foulers of the region (balanoid barnacles, serpulid worms, and ascidians) have short pelagic larval durations ranging from several hours to up to three weeks (Anil et al., 1995; Chan and Walker, 1998; Jacobs et al., 2006). The limited dispersal potential reflected within the 4, 8, 16, and 20 day simulations in comparison to the typical pelagic fish connectivity modelling of the Mediterranean (approximately 30 days) highlights the need for taxon-specific connectivity analyses as opposed to a one-model fits all approach.

Regarding the spatial dispersal of larval particles, there are several persistent larval sinks along the southern Italian shore, corroborating previous findings in the region (Dubois et al., 2016). The shelf area along the Western coast of Italy, consistently had high larval densities in the simulations, due to the hydrographic influence of the River Po (Orlic et al., 1992). During winter, the river output is confined to the northern basin but during the spring/summer spawning season, the Mid Western Adriatic current, and the South Western Adriatic currents, transverse the entire western coastline of Italy (Artegiani et al., 1997) (Figure 4.1). Offshore structures constructed along the southern Italian shores are likely to be much more exposed to higher levels of larval supply than other locations. Similarly, other regions that indicate relatively high self-replenishment and larval densities are found within the Kravner Gulf. The convoluted coastline of the Croatian archipelago clearly plays a large role in transportation of larval particles within the region.

Dispersal of simulated larvae that originate from the major ports of the Adriatic, congregate in high concentrations throughout the northern basin, largely due to the close proximity of the port of Ravenna, the port of Venice, and the port of Trieste (Figure 4.1). Multiple studies have shown higher abundances of alien species at several Adriatic ports (David et al., 2007; Iveša et al., 2015); likely due to direct transportation from fouling/ballast water or indirectly via the colonization of artificial substratum. The invasive barnacle *Amphibalanus improvises* has been recorded at the Rovinj port in Croatia (Pecarevic et al., 2013). Despite it’s fairly limited pelagic larval duration of 5 - 20 days (Anil et al., 1995), its high reproductive capacity and rapid establishment on both natural and artificial substratum has caused it to be classified as one of the worst invasive species in Europe (Vilà et al., 2009). The high levels of larval particles that originate from ports in the region may have implications for the Northern Adriatic hotspot identified in Chapter 1. The presence of alien macro-invertebrates (Zenetos et al., 2012), and high densities of shipping vectors (Figure 1.4), alongside the disproportionate advantage alien species often have in colonizing artificial substrata, means that offshore wind farms may create corridors for alien species invasions (Airoldi et al., 2015). Information regarding the likely destination of larval particles originating from ports and marinas in the Adriatic may assist marine spatial planners looking to reduce the spread of invasive non-indigenous species in the region; however in areas like the northern basin, high densities of existing ports and infrastructures may mean the colonization of alien species on offshore structures is unavoidable.

4.4.3 NODE CENTRALITY

The principal result from this chapter was the production of benthic invertebrate ‘connectivity’ map for the Adriatic. Grid cell centrality i.e. PageRank, is a good way of estimating how connected a cell is with the rest of the grid cells within the network. This measure can be important when spatially planning the position of offshore artificial structures. The potential for offshore structures to act as stepping stones by providing a suitable habitat for colonisation in areas outside of the typical range extension of a species is already documented (Adams et al., 2014), and can have both local and regional impacts on the maintenance of local biodiversity within marine ecosystems (Dafforn et al., 2015). On average, grid cells had low connectivity for all PLDs, particularly in offshore regions and the Po river delta; there were however, several regions of high importance within the network that included the Port of Rijeka, Italian Adriatic coast, and south of the river Durres. This information presented here will be important when deciding if offshore activities should be designed to increase, or decrease, benthic community connectivity. Of the connected grid cells the vast majority (>90 % of cells with pelagic duration more than four days) involved in the coastline-release network were part of one cluster, indicating that although connectivity of grid cells is relatively low, there is potential for interconnection throughout the whole Adriatic.

Connectedness of regions, particularly regions outside of marine protected areas, is an often-ignored aspect of marine spatial planning, but with the further development of offshore activities in the area and the likely impacts this expansion will have on marine biodiversity (Chapter 1), it should be an important consideration for regional marine spatial planners. The density of industrial ports in the Northern Adriatic indicates that the location of OWF in the Northern basis may influence the spread of invasive species in the region, suggesting more southerly locations appear more appropriate for development. The approach presented here is a pragmatic tool for identifying connectivity systems of benthic communities within a semi-closed system which can be expanded with *in–situ* data regarding the placement of offshore structures and habitat ranges of key benthic species.

CHAPTER 5

ECOSYSTEM-LEVEL IMPACTS OF A NOVEL HARD-SUBSTRUM HABITAT CREATED BY AN OFFSHORE WIND FARM IN THE ADRIATIC

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Contributions: I designed the experiment and methodology. I constructed and ran the model with some technical assistance from KT, performed the statistical analyses and wrote the manuscript. JMH-S supervised the thesis.

5.1 INTRODUCTION

The environmental ramifications of installing offshore wind farms are multi-faceted and impact many key species and functional groups (Chapter 1). Wind turbines, many of which have rotor tip heights of up to 120 m and diameters wider than 4 m, markedly alter on-site conditions. The installation of such large surface areas of artificial substratum provides new habitat available for utilization by many marine organisms (Inger et al., 2009). Additionally, it appears that, at least in some capacity, offshore wind turbines act as artificial reefs in the Mediterranean (Chapter 2). Wind turbines are typically colonized by benthic invertebrates and thus provide an additional food source in regions with often relatively low numbers of in-faunal benthic invertebrates (e.g. homogenous soft substrata areas) (Gray, 1974). It is presumed that the provision of additional feeding grounds for higher trophic levels, creates a type of artificial reef effect in regions where offshore wind farms are built (Langhamer, 2012).

Several post-construction studies suggest evidence of the artificial reef effect by noting an increase in numbers of certain fish species close to the turbine foundation (Reubens et al., 2014a; van Hal et al., 2017). Understanding the causes of such aggregation is complex, and may not only be due to the enhancement of food sources (Production vs. Aggregation debate, Chapter 1). In addition to providing new substratum in the water column, the presence of wind turbines also affects the air space overhead. Rotating blades and support structures directly influence the behaviours of seabirds in regions where OWF's have been installed (Harwood et al., 2017). Many species are at risk of collision with turbines or exhibit avoidance behaviour to some extent (Chapter 1, Table 1.1). Conversely, some studies suggests that several species appear to not be influenced by the presence of the wind farms, whilst some species even exhibit attraction behaviour towards the offshore structures (Dierschke et al., 2016). For instance, cormorants may use turbines, and supporting platforms, as outposts to extend hunting grounds out to sea (Krijgsveld, 2014).

Although it is necessary to quantify how individual species or functional groups will be influenced by offshore wind farm construction, it is also important to approach the issue holistically and infer what impacts will occur at an ecosystem level. To date, only one study has attempted to investigate the effects of the construction and operation of offshore wind farms with an integrated, ecosystem-based study (Raoux et al., 2017). The study used a mass-balance software program called Ecopath with Ecosim (EwE Version 6.5) to provide a snapshot of the environment before and after the hypothetical presence of an offshore wind farm. Raoux et al., (2017) used EwE to force the biomass of the functional groups within the ecosystem that are impacted by the presence of an offshore wind farm, in order to simulate the ecosystem impacts of OWF installation through time.

The present study seeks to adhere to the approach used by Raoux et al., (2017) for the Northern Adriatic; an area earmarked for OWF development (Chapter 1). Using ecological data collected from the region, this chapter aims to provide scenarios of the artificial reef effect that would occur with, and without, the presence of an offshore wind farm in the year 2020. In contrast to Raoux et al., (2017), only impacts considered a direct result of the novel habitat addition were parameterised i.e. the benthic invertebrate colonization of the substratum and the behavioural changes of marine birds in the vicinity due to the turbines. Despite evidence at some Northern European OWF sites, of fish and marine mammal aggregation (Reubens et al., 2014a; Russell et al., 2014), it is difficult to discern whether this increase is due to the aggregation of existing individuals from the wider region, or an increase in absolute production of biomass at the site due to increased food availability and foraging grounds (artificial reef effect). Additionally the pilot project results from Chapter 3, show no indication of fish aggregation at the simulated wind turbine foundations. Therefore, to avoid simulating false positive interactions, only the biomass increases of those species that likely wouldn’t be present without the availability of a hard substrate to colonise are considered (e.g. benthic macro-invertebrates) or those impacted by the presence of the rotating blades (seabirds). If sufficient prey resources are available within the EwE mode simulations, greater biomass of higher trophic levels (e.g. fish and marine mammals) are supported. The EwE software approach therefore simulates the production aspect of the artificial reef effect, as opposed to consideration of any aggregation from outside the wind farm region. In addition, individual species responses (i.e. due to sound and electromagnetic fields in the vicinity of the turbines) were not considered due to a dearth of knowledge regarding the physiological responses of fish/elasmobranchs during turbine operation (Öhman et al., 2007; Wahlberg and Westerberg, 2005).

In a first of its kind, this chapter also considers the changes in fishing effort inside the offshore wind farms. Typically fishing restrictions on mobile trawlers are enforced within offshore wind farm areas for safety reasons; however evidence from UK case studies also indicates a complete absence of static gear within many wind farms (Ashley, 2014). Thus, in addition to scenarios predicting ecosystem changes in response to the presence of wind turbines, a scenario is also presented indicating the impacts of reduced fishing effort at the site.

5.2 METHODS

5.2.1 STUDY AREA

The Northern Adriatic Sea is a shallow, semi-enclosed, heavily fished, sub-basin of the Adriatic Sea covering over 32,000 km^2 (Chapter 3, Figure 3.1). The trophic status of the region is largely defined by the discharge of fresh water from the Po River, the second largest river entering the Mediterranean Sea. This nutrient-rich input, along with strong seasonal changes in wind and current regimes contribute towards the eutrophic status is the region (Spillman et al., 2007). The high nutrient availability is reflected in the relatively high value of biomass used for the phytoplankton functional group in the model (13.2 t km^2 yr^{-1}) (Appendix A, Table A1).

5.2.2 ECOPATH DESCRIPTION

Ecopath is a multi-species, mass balanced model that provides a snapshot (typically per year) of the trophic flow within an ecosystem. In addition, Ecopath (through the Ecosim module) also enables a dynamic modelling capability i.e. how the ecosystem fluctuates through time under different scenarios. The parameterization of the model is based on two central equations that describe the (a) production rate and the (b) energy balance of each functional group (single species or group of species) defined by the user. In essence, the production rate of each functional group is described as the total catches + predation mortality + net migration + biomass accumulation + any other mortality (i.e. old age), as a function of biomass (eq. 1):

$$P = Y + M2 \text{ x } B + E + BA + M0 \text{ x } B \tag{1}$$

Where P = the production rate, Y = total fishery catch rate, M2 = the instantaneous predation rate, E = the net migration rate e.g. the emigration – immigration, BA = the biomass accumulation rate, M0 = the other mortality rate (e.g. old age), and B = the biomass of the functional group.

The second equation relates to the energy balance within each group, and requires that the consumption of the group is equal to the sum of the production rate + respiration rates + the total of unassimilated food (i.e. that which is directed to detritus). To fulfil these two equations, Ecopath requires data on the following parameters: the total biomass (t wet mass km^{-2}), the production to biomass ration (P/B) (yr^{-1}), the consumption to biomass ratio (Q/B) (yr^{-1}), and the Ecotrophic Efficiency (EE) of each functional group (provided as a fraction of 1) (Christensen and Walters, 2004). The EE is the fraction of the production that is used up in the system. It is dimensionless, difficult to measure directly, and entry into the model is optional. The model can calculate EE if values for the three other parameters are entered. Additionally, a diet matrix is required to define the relationships between prey and predators, to enable the calculation of energy flow between trophic levels and a quantification of the landings and discards of fishing fleets in the region. The version used for the present analysis was EwE 6.5 (www.ecopath.org). For full details on the equation balancing of the Ecopath software the reader is directed to Christiansen and Walker (2004).

5.2.3 NORTHERN ADRIATIC MODEL

The input data for the Northern Adriatic model is sourced from an existing Ecopath model. The model is based on data collected for the years 1996-1998 (Barausse et al., 2009), and is the most recently available Ecopath ecosystem data for the region. Considering the fact that the input data for the model was consolidated to describe the pre-1998 Northern Adriatic ecosystem, Ecosim was used to simulate the evolution of the system until the year 2020 (22 years). Fishing effort was assumed to remain the same through time. A static Ecopath was then exported from the simulation and used as the basis of the current study. The parameterizations of the 2020 model (Functional groups, B, P/B, Q/B, EE, fishing mortalities, and diet matrix) are included in Appendix A.

To infer the impacts to the marine ecosystem with the potential construction of an OWF at the site in the year 2020, three scenarios were conceived. The first scenario (S1) examined the evolution of the ecosystem if all parameters were to remain the same and no OWF construction occurred i.e. business-as-usual. The 2020 baseline model was run for another 10 years (until 2030), and a static Ecopath model was exported at the end of the simulation). The second scenario (S2) considered the artificial reef effect due to the installation of an OWF farm. Changes in the biomass of species or functional groups that typically colonize the artificial hard substratum of offshore wind turbines were obtained from post-construction surveys in the literature (see below). Additionally changes to seabird population sizes due to the presence of the OWF were considered.

Biomasses of the affected functional groups were forced into the 2020 Ecopath model, and ecosim used to simulate the 10-year evolution of the model. Likewise, the third scenario (S3) also considers the artificial reef effect, whilst in addition incorporating the likelihood of fishing restriction enforcement within the OWF site. Fishing effort from 2020 (hypothetical date of OWF construction in line with European objectives) until the end of the simulation was considered as 0 % for all fishing gears (Ashley, 2014) (Table 5.1). Ecological Network indices were then calculated using the in-built network analysis plug-in, and compared for each scenario output.

Table 5.1 Description of different scenarios used in methodology. It is assumed OWF installation will be in line with horizon 2020 predictions

Scenario 1 (S1) No change, Business as usual	**Scenario 2 (S2)** OWF installation	**Scenario 3 (S3)** OWF installation and fishing restriction enforcement
- No changes in fishing effort until 2030 - No forcing of biomass until 2030	- No changes in fishing effort until 2030 - Biomass forcing of certain functional groups in 2020 to simulate artificial reef effect	- Fishing effort at 100% until year 2020, fishing effort at 0% from 2020 - 2030 - Biomass forcing of certain functional groups in 2020 to simulate artificial reef effect

5.2.4 BIOMASS FORCING WITH EVIDENCE TO DATE FROM EXISTING OFFSHORE WIND FARM SYSTEMS.

Benthic invertebrates

To quantify the additional amount of benthic invertebrate biomass to be forced into the model simulation an extensive search of the literature was done. The biomass value is forced into the artificial reef scenarios (Scenario 2, Scenario 3), for the functional groups within the Ecopath model that are typically considered to colonize hard substrata (commercial and non-commercial bivalves, filter feeding invertebrates, and polychaetes).

Tethys, a knowledge database developed by the Pacific Northwest National Laboratory for the gathering, organising, and dissemination of information on the environmental effects of marine and wind energy development (https://tethys.pnnl.gov), was searched with multiple key words, e.g. post-construction, benthic invertebrates, benthos, biomass, foundation, impact assessment, colonizing organisms, impact, offshore wind, and near-field habitat. Of the 3528 available items within the data base, which include journal articles, reports, conference papers, presentations, book chapters, theses, magazine articles, workshop articles, websites and videos, a total of seven post-constriction studies at operational offshore wind farms containing data regarding fouling biomass estimates were identified (Table 5.2).

The dates of the post-construction surveys ranged from 12 months to up to 5 years after installation. Several of the values were recorded in Ash Free Dry Weight (ASFD), as opposed to wet weight (WW), the unit used for the EwE model inputs (Barausse et al., 2009) (Appendix A, Table A1). To transform ASFD to WW an average conversion factor 0.07 for the functional groups colonizing hard substratum was used (Filter feeding invertebrates = 0.025, Commercial and non-commercial bivalves = 0.058, Polychaetes = 0.16) (Ricciardi and Bourget, 1998). All the WW biomass values (calculated and measured) were plotted depending on how many months had elapsed after the construction date of the OWF when the survey was done (Figure 5.1).

Table 5.2 Post-construction surveys sourced from Tethys database containing quantitative data regarding the increase of biofouling organisms on offshore wind farm turbines [1] = (Vanagt and Faasse 2014), [2] = (Lindeboom et al. 2011), [3] = (Birklund and Petersen 2004), [4] = (Bouma and Lengkeek 2012), [5] = (Bunker 2004), [6] = (Emu Ltd. 2008), [7] = (Leonhard et al. 2006).

OWF site	Average biomass (g m^{-2})	Unit	Elapsed time (mo)	Reference	Wet weight (g m^{-2})
Prinses Amalia	441	ASFD	42	[1]	6300
Prinses Amalia	645	ASFD	66	[1]	9217
Egmond aan Zee	1257	ASFD	12	[2]	18219
Nysted	1040	ASFD	12	[3]	14861
Egmond aan Zee	450	ASFD	24	[4]	6431
Egmond aan Zee	1400	ASFD	31	[4]	20006
North Hoyle	1141	WW	12	[5]	1141
Kentish Flats	35007	WW	36	[6]	35007
Horns Rev	1050	WW	12	[7]	1050
Horns Rev	1700	WW	24	[7]	1700
Horns Rev	2650	WW	36	[7]	2650

A logarithmic trendline was fitted, and used to calculate a time series of total benthic invertebrate biomass increase per m^2 through time. As increases in biomass were reported as the wet weight (g) of colonizing species per square metre, additional calculations were done to identify the average increase in surface area. As no planning proposals for Northern Adriatic OWF's are available, the average values (diameter, number of turbines) of the OWF sites used to obtain the biomass of biofouling communities were used to define the increase of surface area that would occur (Table 5.3.

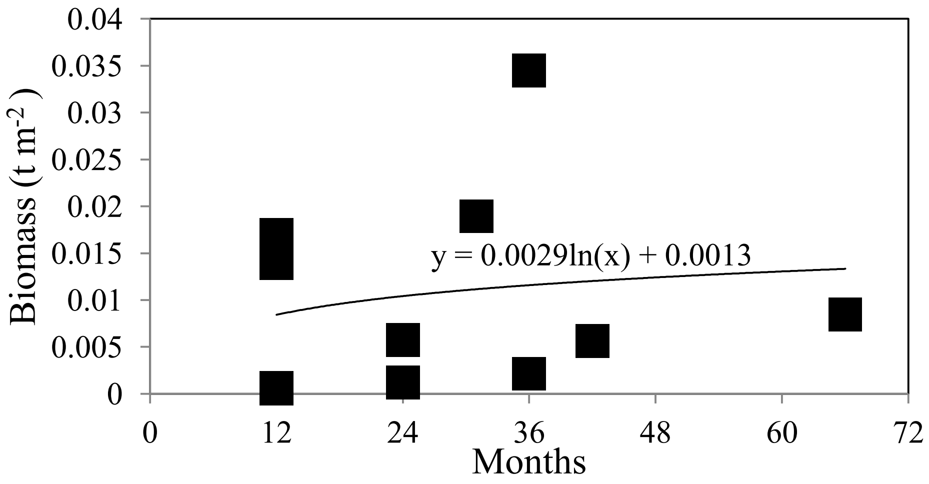

Figure 5.1 Wet weight (by month elapsed from OWF construction) of colonizing organisms (Table 4.2) on OWF's sourced from the literature (See in-text references), with logarithmic trendline displayed ($R^2 = 0.0275$).

The technical details (e.g. maximum height of turbine submerged in water) for each site were sourced (http://www.4coffshore.com), and the surface area calculated by assuming an even, cylindrical, design for monopiles and gravity base foundation types, and three cylinders for tripod foundations (Table 5.3).

Table 5.3 Dimensions of the wind farms used for the calculation of the biomass increase of benthic organisms. Due to differences in the foundation structure of the Nysted gravity base, surface area values were calculated separately. Source: http://www.4coffshore.com

OWF site	Max. water depth (m)	Average Diameter (m)	No. of turbines	Area of wind farm (km^2)	Type	Calculated total surface area (m^2)
North Hoyle	12	4	30	10	Monopile	150.80
Egmond aan Zee	18	4.6	36	24	Monopile	260.12
Alpha Ventus	30	2.7	12	4	Tripod	763.41
Horns Rev 1	11	4.2	80	21	Monopile	145.14
Nysted	9	4.2	72	26	Gravity base	415.16
Prinses Amalia	24	4	60	17	Monopile	301.59
Kentish Flats	5	4.5	30	10	Monopile	70.69
Average	13.86	4.3	59.7	18.86		229.09

To force the biomass for the four functional groups, the total biomass increase calculated from the literature review was multiplied by the average surface area increase (229 m^2) and divided equally by the four groups. As the EwE input data is also factored spatially (km^{-2}), the time series were divided by the average surface area wind farm area (229 m^2) (Table 5.3). Finally the baseline biomasses of each functional group from the 2020 ecopath model were added to each forcing time series (Table 5.4).

Table 5.4 Time series for forcing biomass inputs for scenario 2 and scenario 3 Ecosim models. Baseline value obtained from 2020 ecopath model output (Appendix A, Table A1).

	Filter feeding invertebrates (t km^{-2} yr^{-1})	Non-Commercial bivalves (t km^{-2} yr^{-1})	Commercial bivalves (t km^{-2} yr^{-1})	Polychaetes (t km^{-2} yr^{-1})
2020 Baseline value	**7.76**	**25.13**	**0.90**	**27.27**
2021	7.78	25.16	0.93	27.29
2022	7.79	25.16	0.93	27.30
2023	7.79	25.17	0.94	27.30
2024	7.79	25.17	0.94	27.31
2025	7.80	25.17	0.94	27.31
2026	7.780	25.17	0.94	27.31
2027	7.80	25.17	0.95	27.31
2028	7.80	25.17	0.95	27.31
2029	7.80	25.17	0.95	27.31
2030	7.80	25.18	0.95	27.31

Seabirds

Identifying the impacts of wind turbine blades on the presence of the pelagic marine bird populations was done by using a recent review of (Dierschke et al., 2016), which reviewed avoidance and attraction behaviour of seabirds in European waters. For the original Northern Adriatic model, the only avian species included in the seabird functional group are *Phalacrocorax carbo* (Great cormorant) and *Larus* *melanocephalus* (Mediterranean Gull) (Barausse et al., 2009). Of these two species, the cormorant is identified as being strongly attracted to OWF sites after a meta-analysis of post-construction surveys. The Mediterranean Gull, was not included in the meta-analysis review however other gull species (*Larus canus* and *Larus ridibundus*) were identified as weakly attracted (e.g. continued use of a marine area after the construction of the wind farm, but to a higher degree or at a higher abundance) and *Larus michaellis* was observed as being present within several wind farms without displaying any particular preference to the site (Dierschke et al., 2016). Using the attraction hypothesis, it is theorized that birds will primarily be attracted to the region as a feeding ground. It is plausible to assume that if there is an increase in prey availability (e.g. small pelagic fish) at the site, an increase of seabirds will occur within this ecosystem. Forcing a decrease in seabird biomass due to deterrence behaviours appears unnecessary within the proposed ecosystem model.

5.2.5 NETWORK ANALYSIS

The Network Analysis plug-in included in the EwE software calculated several indices for each output scenario which analyse whole system interactions and properties (Christensen et al., 2005). The indices applied to each output scenario within this study include the total system Throughput and Throughflow, the Transfer Efficiency, the Omnivory Index, Mixed Trophic Impact analysis, and the Keystoneness of individual functional groups. Presented in the results are brief explanations of each index, and a comparison of the results for the three scenarios, and included in each explanation are references to the original studies describing the parameterization of the index.

5.3 RESULTS

5.3.1 ECOPATH MODEL DESCRIPTIONS

Under the business-as-usual scenario (S1), the constructed Ecopath output model predicts that the total biomass of the system (including detritus) in the year 2030 will amount to 529.472 t km^{-2} (Table 5.5). Macroalgae, polychaetes, and non-commercial bivalves, make up 57 % of the total living biomass. This figure does not change for S2 and the hypothetical installation of an offshore wind farm. There is a slight increase in the forced biomass groups e.g. for filter feeding invertebrates and bivalves (Figure 5.2), but overall a limited differences in total biomass was observed (0.04%). The reduction in fishing pressure under S3 generates an increase in the percentage of top predators biomass e.g. sharks, rays, benthic piscivorous fish and an increase in the biomass of lower trophic levels e.g. macroinvertebrates, when compared to the business-as-usual scenario (Figure 5.2). When fishing effort is reduced (S3), the largest decrease in comparison to S1 is at the mid-trophic level. A large reduction in anchovy biomass (-10.3%) is likely due to increased competition from the previously heavily fished sardine populations, as they share the same prey sources e.g. plankton (Morote et al., 2010) (Appendix A, Table A3). Interestingly, no increase of seabird biomass was supported by the model, suggesting a limited artificial reef effect. The total biomass of the system increases to 533.77 t km^{-2} under S3.

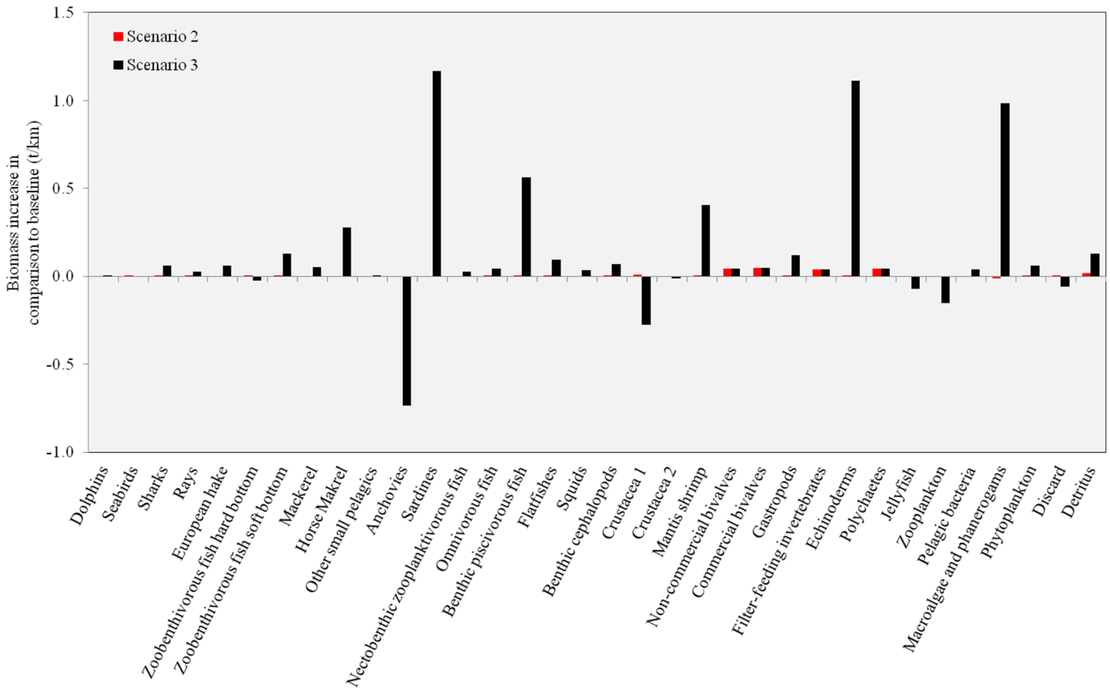

Figure 5.2 Absolute differences in biomass (t km^{1} year^{-1}) of each functional group for scenario 2, and scenario 3, in comparison with Scenario 1.

5.3.2 NETWORK ANALYSIS INDICES

TOTAL SYSTEM THROUGHFLOW AND TOTAL SYSTEM THROUGHPUT

Total system throughflow and total system throughput are closely related yet separate indices. Their common representation in the literature as TST values mean the two are often confused. Total system throughflow represents the total flow (of mass) into or out of each functional group, whilst total system throughput is the sum of all flows in the food web, thus providing an indication of the overall level of activity within the marine system (Fath et al., 2013). Flows are calculated (t m^{-2} $year^{-1}$) by combining the consumption, export, respiration and flows to detritus. For a detailed explanation the reader is referred to Ulanowicz (1986).

No noticeable differences in ecosystem activity were observed between the baseline scenario and the introduction of the hypothetical wind farm. The total system throughput of S1 and S2 is 10363.19 and 10363.28 t km^{-2} $year^{-1}$ respectively. For S3 the overall ecosystem increased by 0.18% to 10381.46 t km^{-2} $year^{-1}$ in comparison to S1. Differences in compartment throughflow were more apparent. In comparison to the baseline scenario, the S2 throughflow values for S2 functional groups varied less than 5 %, however the S3 indicates much more variation. The throughflow of the functional groups rays, benthic piscivorous fish, and sharks increased 228%, 127%, and 104% respectively (Table 5.6).

Table 5.5 Ecopath input values for 2030 Ecopath models. Scenarios 1 represents Business-as-usual model, Scenario 2 describes impacts of OWF installation, and Scenario 3 models impacts of OWF installation and fishing. B = Biomass, P = Production, Q = Consumption, and EE = Ecotrophic Efficiency.

	B (t/km²)			P/B (/year)			Q/B (/year)			EE			P/Q		
Group name	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Dolphins	0.006	0.006	0.008	0.054	0.054	0.031	14.000	14.000	13.986	0.278	0.278	0.000	0.004	0.004	0.002
Seabirds	0.012	0.011	0.010	4.610	4.610	4.657	69.340	69.340	69.467	0.000	0.000	0.000	0.066	0.066	0.067
Sharks	0.066	0.057	0.116	0.400	0.400	0.399	3.077	3.077	3.046	0.687	0.687	0.813	0.130	0.130	0.131
Rays	0.015	0.012	0.040	0.724	0.724	0.719	4.137	4.137	4.107	0.545	0.545	0.765	0.175	0.175	0.175
European hake	0.056	0.052	0.113	1.180	1.180	1.102	4.120	4.120	3.813	0.663	0.663	0.434	0.286	0.286	0.289
Zoobenthivorous fish hard bottom	0.229	0.190	0.164	1.901	1.901	1.909	6.410	6.410	6.433	0.611	0.623	0.587	0.297	0.297	0.297
Zoobenthivorous fish soft bottom	1.023	0.849	0.977	1.180	1.180	1.179	6.306	6.306	6.288	0.878	0.873	0.849	0.187	0.187	0.188
Mackerel	1.076	1.001	1.052	0.620	0.620	0.624	6.506	6.506	6.507	0.251	0.251	0.263	0.095	0.095	0.096
Horse Mackerel	1.035	0.963	1.241	0.570	0.570	0.572	5.941	5.941	5.927	0.600	0.595	0.558	0.096	0.096	0.096
Other small pelagics	0.356	0.330	0.332	1.889	1.889	1.873	8.294	8.294	8.280	0.851	0.840	0.702	0.228	0.228	0.226
Anchovies	7.531	7.131	6.398	1.100	1.100	1.098	12.440	12.440	12.464	0.930	0.930	0.930	0.088	0.088	0.088
Sardines	4.909	4.716	5.886	0.870	0.870	0.861	8.709	8.709	8.672	0.961	0.961	0.941	1.000	1.000	0.099
Nectobenthic zooplanktivorous fish	0.427	0.401	0.429	1.180	1.180	1.176	6.442	6.442	6.438	0.839	0.836	0.945	0.183	0.183	0.183
Omnivorous fish	0.133	0.115	0.159	1.624	1.624	1.615	15.040	15.040	14.941	0.965	0.961	0.926	0.108	0.108	0.108
Benthic piscivorous fish	0.218	0.201	0.761	0.850	0.850	0.512	3.304	3.304	1.982	0.699	0.699	0.916	0.257	0.257	0.258
Flatfishes	0.174	0.142	0.236	1.300	1.300	1.232	6.975	6.975	6.595	0.622	0.622	0.787	0.186	0.186	0.187
Squids	0.053	0.050	0.086	3.506	3.506	3.512	26.968	26.968	26.911	0.349	0.349	0.889	0.130	0.130	0.131
Benthic cephalopods	0.125	0.101	0.172	3.300	3.300	3.269	6.600	6.600	6.537	0.869	0.868	0.962	0.500	0.500	0.500
Crustacea 1	6.597	5.447	5.162	2.894	2.894	2.896	17.786	17.785	17.803	0.971	0.966	0.969	0.163	0.163	0.163
Crustacea 2	1.183	1.017	1.003	7.911	7.911	7.911	51.186	51.187	51.187	0.996	0.997	0.988	0.155	0.155	0.155
Mantis shrimp	0.102	0.082	0.487	1.500	1.500	0.462	4.560	4.559	1.412	0.892	0.892	0.301	0.329	0.329	0.327
Non-commercial bivalves	30.260	25.176	25.176	1.415	1.418	1.408	6.350	6.349	6.352	0.518	0.515	0.512	0.223	0.223	0.222
Commercial bivalves	1.067	0.948	0.948	1.415	1.446	0.772	6.350	6.336	6.500	0.784	0.470	0.013	0.223	0.228	0.119
Gastropods	9.535	7.917	8.031	1.735	1.735	1.734	9.717	9.717	9.712	0.907	0.903	0.899	0.179	0.179	0.179
Filter-feeding invertebrates	10.033	7.802	7.802	0.761	0.765	0.725	3.804	3.803	3.809	0.483	0.483	0.457	0.200	0.201	0.190
Echinoderms	10.965	8.949	10.059	0.803	0.803	0.800	2.514	2.514	2.508	0.429	0.426	0.333	0.319	0.319	0.319
Polychaetes	33.451	27.314	27.314	1.644	1.647	1.645	14.271	14.269	14.268	0.673	0.672	0.667	0.115	0.115	0.115
Jellyfish	1.141	1.025	0.952	14.813	14.813	14.797	44.439	44.439	44.436	0.150	0.150	0.146	0.333	0.333	0.333
Zooplankton	3.430	3.284	3.131	65.058	65.061	66.816	185.167	185.172	190.326	0.924	0.924	0.921	0.351	0.351	0.351
Pelagic bacteria	4.944	4.052	4.093	136.131	136.190	135.633	758.731	759.042	755.909	0.415	0.415	0.412	0.179	0.179	0.179
Macroalgae and phanerogams	41.422	42.420	43.414	1.700	1.700	1.681	-	-	-	0.193	0.193	0.188	-	-	-
Phytoplankton	12.486	13.166	13.226	175.228	175.226	174.836	-	-	-	0.403	0.403	0.402	-	-	-
Discard	0.061	0.058	0.000	74.285	74.331	74.391	-	-	-	0.290	0.500		-	-	-
Detritus	424.173	364.678	364.788	8.906	8.905	8.937	-	-	-	1.000	1.000	1.000	-	-	-
Sum	608.294	529.663	533.766	-	-	-	-	-	-	-	-	-	-	-	-

Omnivory index

The Omnivory index (OI) is a function of the variance of the trophic level of a consumer's prey groups. The closer the OI is to zero, the more specialized a predator is. The higher the value, the more trophic levels a predator feeds on. For the original paper referring to the parameterization of the omnivory index the reader is referred to Pauly et al., (1993). For S1, S2, and S3, the OI of anchovies and small pelagic were the only functional groups that were either zero, or close to zero, indicating that they fully specialized and was feeding on a single trophic level (Table 4.6).

Transfer Efficiency

Transfer efficiency describes the ratio between the sum of the exports plus the flow that is transferred from one trophic level to the next, and the throughput on the trophic level (Christensen and Walters, 2004). The scenarios are represented graphically with the use of a Lindeman spine, showing the aggregated flow of transfer through each trophic level (Lindeman, 1942). The Lindeman spines of S1 and S2 show similar patterns in transfer efficiency with increasing trophic level (Figure 5.3). With the reduction of fishing pressures (S3), slightly higher transfer efficiencies are apparent, which support an additional trophic level in comparison to the baseline scenario and the OWF installation alone (Figure 5.3C).

Mixed trophic impact

The mixed trophic impact (MTI) analysis is a method of indicating the combined direct and indirect trophic impact that occurs when the biomass of a functional group is infinitesimally increased. Impact values (either positive or negative) are presented as a matrix between the impacting group (predators) and impacted group (prey). The matrix values are calculated as the difference between a diet composition parameter (the fraction of the impacted group that contributes to the impacting group), and a competition parameter (the proportion of predation on the impacted group that is due to the impacting group operating as a predator). For additional information regarding the MTI aspect of the EwE plug-in, the reader is directed to (Ulanowicz and Puccia, 1990). The MTI matrices of S1 and S2 are almost identical, with no apparent differences between the two (Figure 5.4, Figure 5.5). The low trophic level group’s zooplankton, pelagic bacteria, and detritus all exert a strong influence on the system. Additionally three mid-level trophic groups (anchovies, horse mackerel, and large crustacean), negatively influence several of their predators, prey, and competitors. The MTI analysis of S3, indicates a similar trend, however the higher trophic level benthic piscivorous fish and squids also appear to exert strong influence on the ecosystem under this scenario (Figure 5.7).

Table 5.6 The throughflow and Omnivory indices of the functional groups. Note that throughflow is presented in % change from the baseline scenario; whist the OI of all scenarios (S1, S2, and S3) is presented.

Functional Group	**Throughflow (% Change from S1)**		**Omnivory index**		
	S2	S3	S1	S2	S3
Dolphins	0.00	32.50	0.07	0.07	0.16
Seabirds	0.00	-2.98	1.41	1.40	1.58
Sharks	0.00	104.02	0.33	0.32	0.49
Rays	0.00	228.03	0.29	0.29	0.25
European hake	0.00	99.53	0.10	0.10	0.13
Zoobenthivorous fish—hard bottom	0.25	-12.87	0.15	0.15	0.15
Zoobenthivorous fish—soft bottom	0.17	14.94	0.17	0.17	0.20
Mackerel	-0.05	5.07	0.22	0.22	0.24
Horse Mackerel	-0.03	28.50	0.24	0.24	0.36
Other small pelagics	-0.04	0.47	0.00	0.00	0.00
Anchovies	-0.03	-10.14	0.00	0.00	0.02
Sardines	-0.05	24.22	0.09	0.09	0.11
Nectobenthic zooplanktivorous fish	-0.04	6.89	0.11	0.11	0.12
Omnivorous fish	0.06	37.50	0.49	0.48	0.47
Benthic piscivorous fish	0.00	127.60	0.08	0.08	0.14
Flatfishes	0.30	57.75	0.14	0.14	0.18
Squids	0.00	70.69	0.08	0.08	0.13
Benthic cephalopods	0.15	68.32	0.21	0.21	0.34
Crustacea 1	0.16	-4.99	0.46	0.46	0.46
Crustacea 2	-0.06	-1.36	0.24	0.24	0.24
Mantis shrimp	0.54	85.68	0.09	0.09	0.15
Non-commercial bivalves	0.13	0.19	0.22	0.22	0.22
Commercial bivalves	4.76	7.56	0.22	0.22	0.22
Gastropods	0.07	1.46	0.25	0.25	0.26
Filter-feeding invertebrates	0.47	0.64	0.25	0.25	0.25
Echinoderms	0.04	12.14	0.32	0.32	0.32
Polychaetes	0.13	0.13	0.13	0.13	0.13
Jellyfish	-0.04	-7.16	0.29	0.29	0.29
Zooplankton	-0.03	-2.05	0.17	0.17	0.17
Pelagic bacteria	-0.03	0.59	-	-	-
Macroalgae and phanerogams	-0.01	1.14	-	-	-
Phytoplankton	0.00	0.22	-	-	-
Discard	0.14	-100	-	-	-
Detritus	0.00	0.34	0.35	0.35	0.37

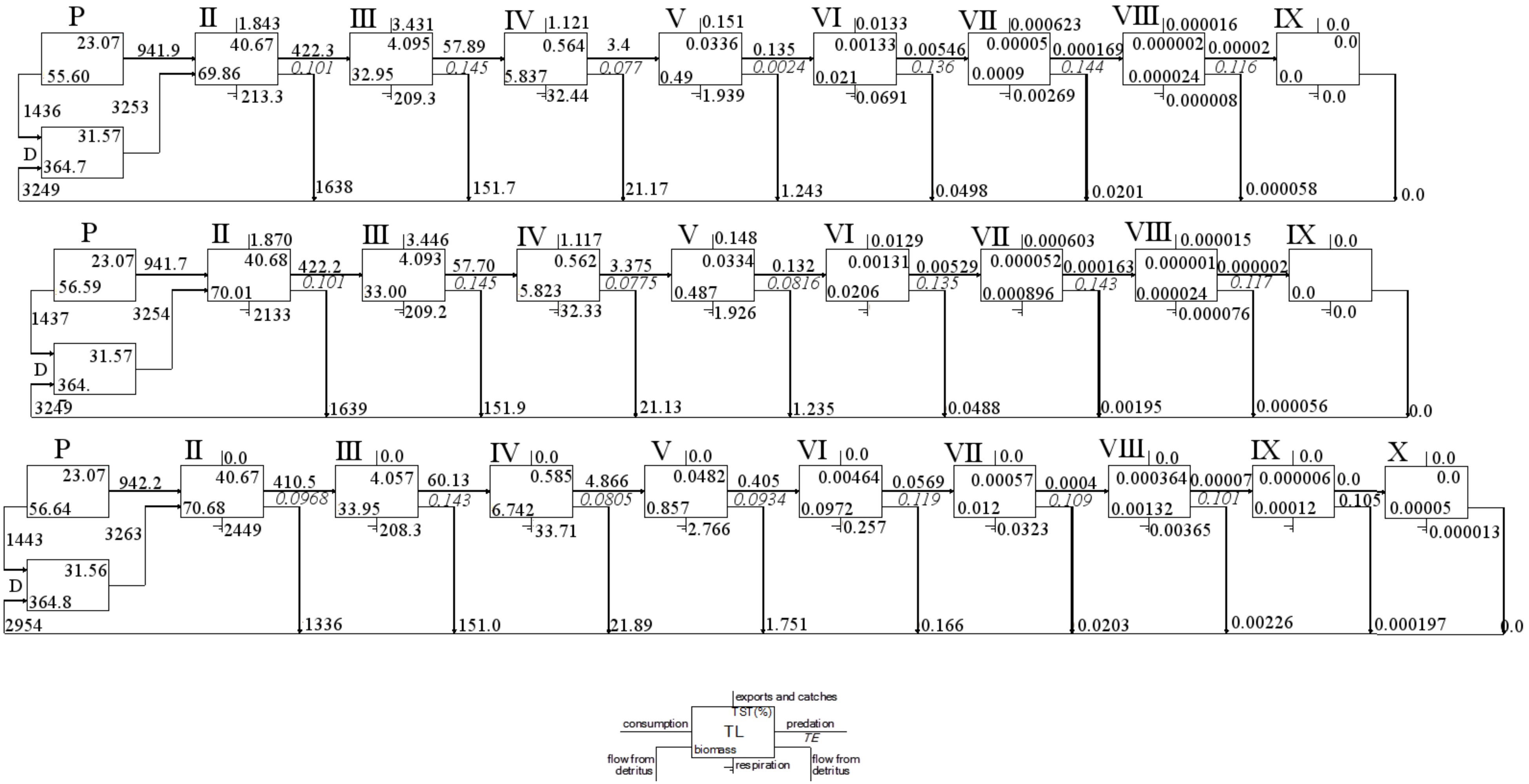

Figure 5.3 Lindeman spine diagrams of A = Scenario 1 (Business-as-usual), B= Scenario 2 (OWF installation), and C = Scenario 3 (OWF installation and fishing restrictions). TL = Trophic level, P = Producers, D = Detritus, TE = Transfer efficiency, TST = Total system throughput.

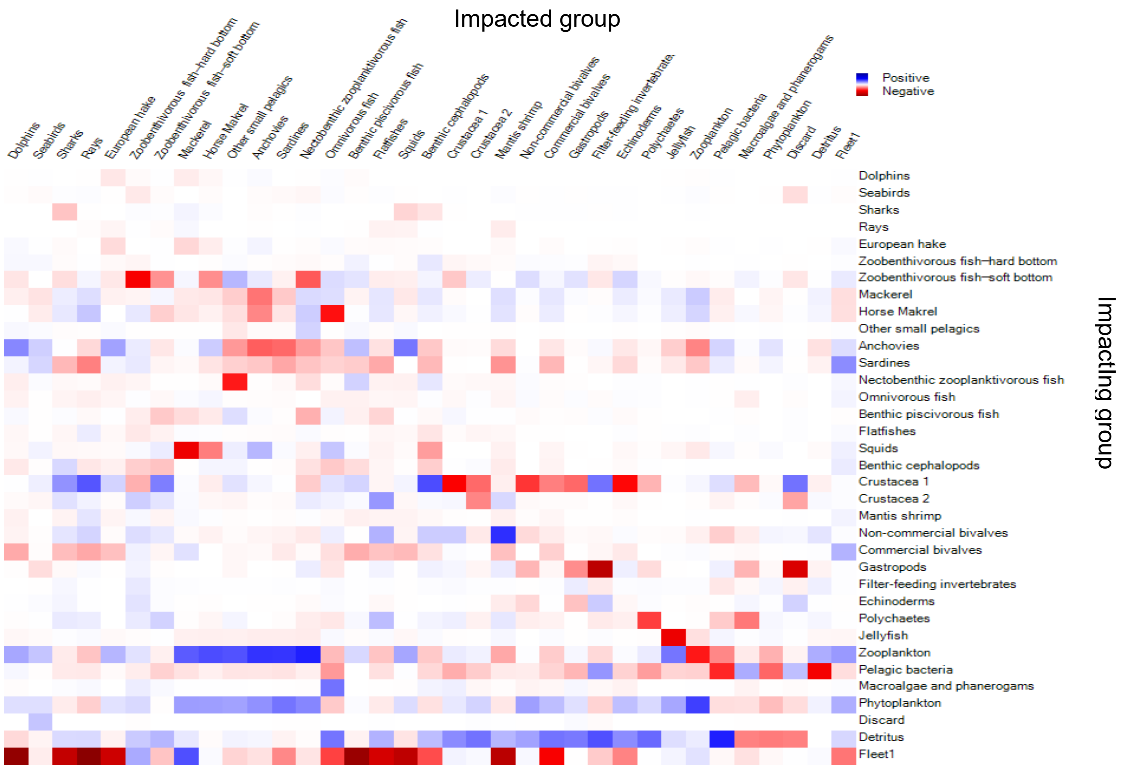

Figure 5.4 MTI analysis of Scenario 1 (Business-as-usual), note positive impacts are denoted in blue, negative in red.

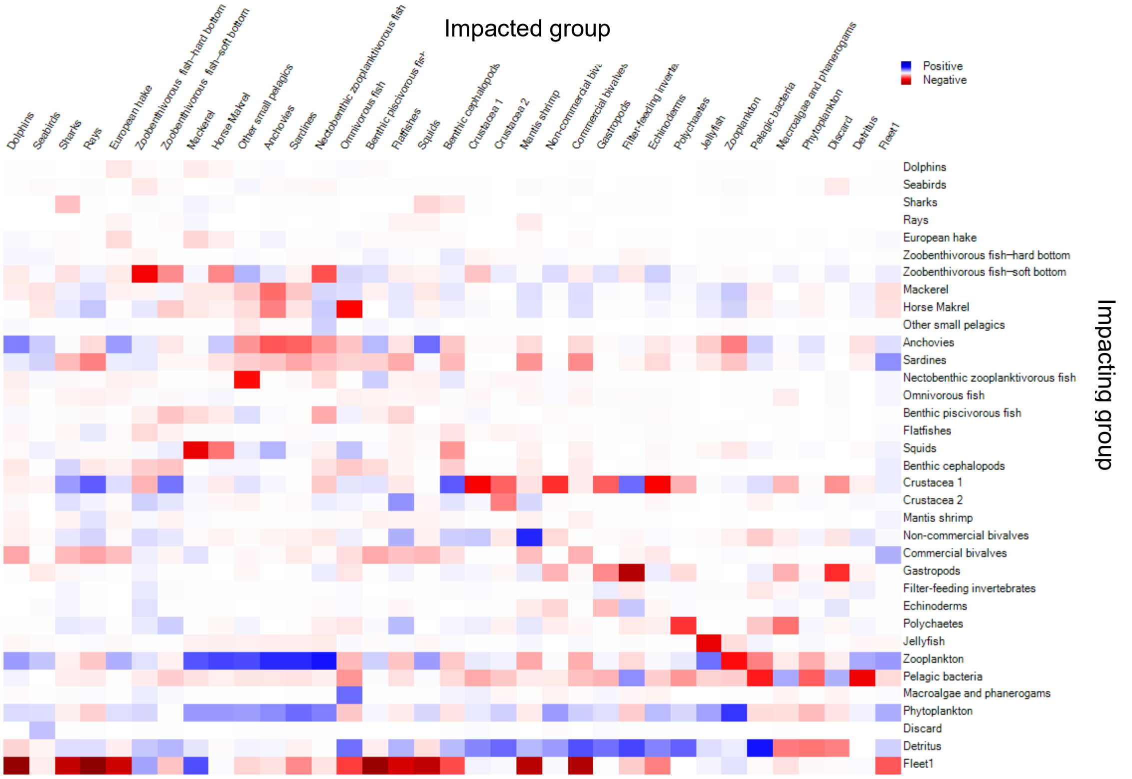

Figure 5.5 MTI analysis of Scenario 2 (OWF installation), note positive impacts are denoted in blue, negative in red.

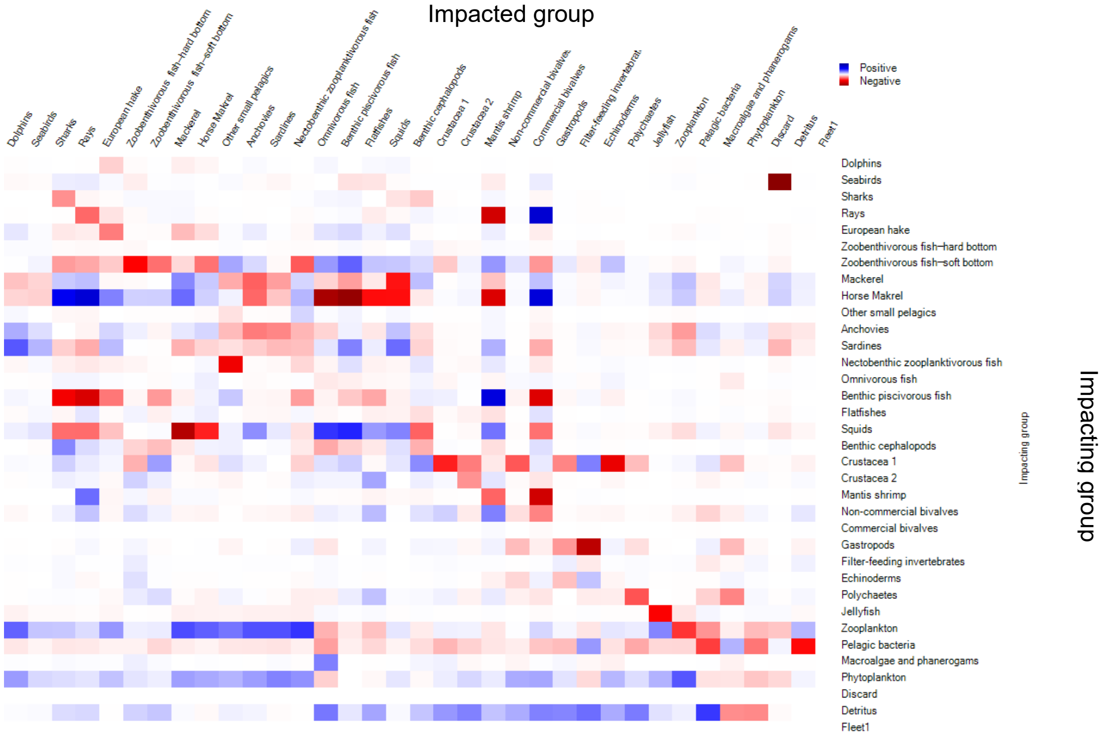

Figure 5.6 MTI analysis of Scenario 3 (OWF installation and fishing restrictions), note positive impacts are denoted in blue, negative in red.

To compare the mixed trophic analyses of each scenario, species and functional groups were grouped according to the trophic level assigned by the Ecopath software. The total sum of the differences between each trophic group was plotted for each scenario (Figure 5.7). There are limited differences between Scenario 1 and Scenario 2, however Scenario 3 differed to both Scenario 1 and Scenario 2. The mid level trophic level group (TL3 – mostly macroinvertebrates) had a more negative impact on the highest trophic level for Scenario 1 and 2 in comparison to Scenario 3 e.g. an infinitesimal increase in the biomass of macroinvertebrates, caused a more negative impact (decrease) in the biomass of the top predators. In addition, small biomass increases in both the top predators, and the primary producers for Scenario 1 and 2, more positively impacted the top predators than Scenario 3, suggesting an increase in within-group competition when fishing pressures are reduced, and an indirect release of primary producers from a bottleneck caused by proliferation of mid-level trophic levels under a heavily fished system.

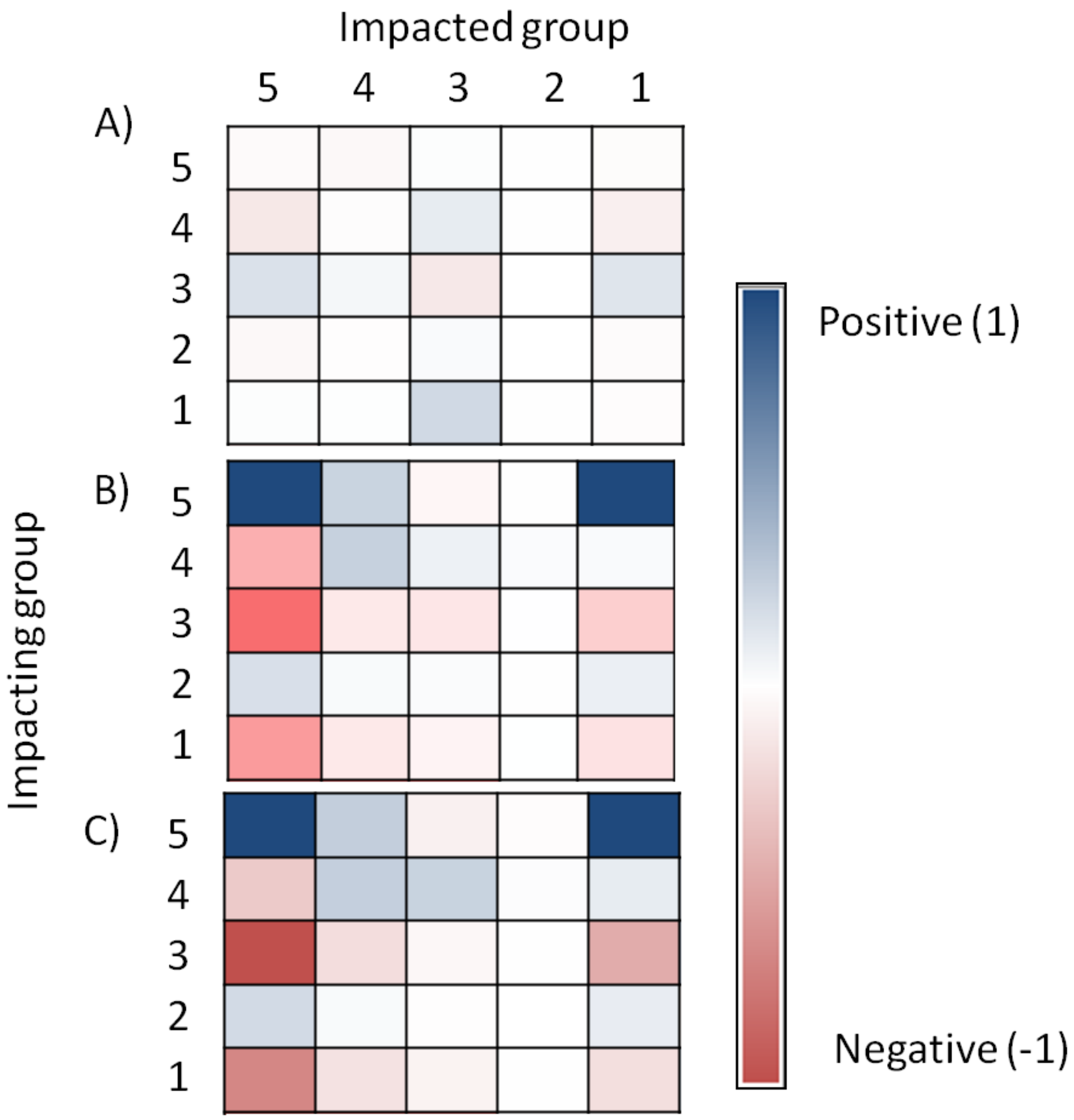

Figure 5.7 Differences in mixed trophic analyses for A) Scenario 1 minus the impacts of 2, B) Scenario 1 minus the impacts of Scenario 3, and C) Scenario 2 minus the impacts of Scenario 3. Numbers refer to trophic level. The lowest trophic level where TL ≤ 1 (TL1) included the functional groups Macroalgae and phanerograms, Phytoplankton, Detritus and Discards. Functional groups whose trophic level values ranged from 1 > TL ≤ 2 (TL2) included pelagic bacteria; Functional groups whose trophic level values ranged from 2 > TL ≤ 3 (TL3) included Polychaetes, Zooplankton, Gastropods, Non-commercial bivalves, Commercial bivalves, Crustacea 1 and 2, Omnivorous fish, Echinoderms, and Filter-feeding invertebrates. Functional

groups whose trophic level values ranged from 3 > TL ≤ 4 (TL4) included Jellyfish, Sardines, Anchovies, Other small pelagic, Nectobenthic zooplanktivorous fish, Zoobenthivorous fish—hard bottom, Mantis shrimp, Flatfishes, Zoobenthivorous fish—soft bottom, Mackerel and Horse Mackerel. The functional groups with the highest trophic levels (4 > TL ≤ 5) (TL5) were European hake, Seabirds, Squids, Rays, Dolphins, Benthic piscivorous fish and Benthic cephalopods.

Keystoneness

The keystoneness index (KS) is a measure of the extent to which a functional group disproportionately (based on its measure of biomass) influences the food-web structure of its community e.g. High KS values signify high influencing, low biomass functional groups (or species). The index used is based on a recent definition by Valls et al., (2015). For all scenarios the keystone index was highest the squid, and some of the lowest keystoneness values were reported for macroalgae and phanerogams, and filter feeding invertebrates (Table 5.7). For scenario 3, the keystoneness of high trophic predators, and especially dolphins, sharks, rays, horse mackerel, and benthic piscivorous fish increased significantly. Additionally the keystoneness of several non-commercial bivalves greatly decreased, likely due their increased abundance within the system. Regarding keystone species, scenario 3 appears to be structured, to a larger extent, by top-down regulation in comparison to the other scenarios.

Table 5.7 Keystoneness values per functional group, for scenario 1, and percentages changes in comparison to scenario 1 for scenario 2 and 3. Highest 10 % of values are shaded in blue and lowest 10 % of values are shaded in red for each scenario.

Functional group	Keystoneness index		
	Scenario 1	Scenario 2 % change	Scenario 3 % change
Dolphins	0.259	-0.39	100.39
Seabirds	0.166	6.02	236.14
Sharks	0.382	0.00	37.43
Rays	0.179	0.56	720.11
European hake	0.368	0.82	109.78
Zoobenthivorous fish—hard bottom	0.128	-2.34	-20.31
Zoobenthivorous fish—soft bottom	1.053	-0.28	10.45
Mackerel	0.705	0.43	44.11
Horse Mackerel	0.916	0.33	54.48
Other small pelagics	0.21	-0.48	50.48
Anchovies	0.692	0.43	-17.05
Sardines	0.669	0.30	17.94
Nectobenthic zooplanktivorous fish	0.896	0.45	20.42
Omnivorous fish	0.233	-6.01	44.64
Benthic piscivorous fish	0.641	0.16	118.10
Flatfishes	0.274	27.01	83.94
Squids	1.245	0.08	26.99
Benthic cephalopods	0.719	-2.09	32.13
Crustacea 1	0.931	-2.58	2.69
Crustacea 2	0.511	-0.59	3.91
Mantis shrimp	0.31	1.94	280.65
Non-commercial bivalves	0.12	-5.83	-8.33
Commercial bivalves	0.803	-0.50	-184.56
Gastropods	0.641	0.00	7.18
Filter-feeding invertebrates	-0.294	-1.70	-0.68
Echinoderms	-0.0736	-5.03	-107.98
Polychaetes	-0.208	0.96	-5.29
Jellyfish	0.111	1.80	96.40
Zooplankton	1.066	0.28	1.69
Pelagic bacteria	0.766	0.52	2.74
Macroalgae and phanerogams	-0.598	0.17	-1.51
Phytoplankton	0.259	-0.39	100.39

5.4 DISCUSSION

The artificial reef effect caused by the addition of new wind turbine substratum to a marine region is well noted in many scientific studies and environmental impact assessments (Chapter 1, however see Chapter 2 for equivocal evidence). What is less well known is the overall impact that this phenomenon described in Northern European case studies, could have at an ecosystem level in the Mediterranean. Using Ecopath with Ecosim to model the likely impacts of introducing offshore wind turbines to a system is a holistic approach only recently suggested by Raoux et al., (2017). The approach appears an effective method of applying ecological expertise to marine spatial planning questions, particularly if the models are paramatized by data that accurately represents the examined ecosystem. The Northern Adriatic is a marine region with many existing marine spatial pressures (Bastardie et al., 2017). Furthermore, with the identification of the area as an offshore wind farm hotspot (Chapter 1), and this pressure is likely to increase in the near future. Applying EwE in this manner is likely to become a useful instrument for policy decision makers in the future (Wilding et al., 2017).

The Ecological Network Analysis of the baseline scenario for the 2030 Ecopath models indicates that it is lower level trophic groups that predominantly structure the Northern Adriatic. The mixed trophic analysis indicates the exerting influence that zooplankton, pelagic bacteria, and detritus have on the system, and in agreement with the original model constructed by Barausse et al., (2009) for the period 1996-1998, the Ecotrophic efficiency values and MTI impacts of zooplankton in comparison with phytoplankton imply that zooplankton operates as bottleneck. The high EE value of zooplankton suggests it is almost completely exploited by the system, whilst the MTI analysis illustrates that an incremental decrease of zooplankton will have a strongly positive impact on phytoplanktonic species, suggesting that zooplankton limits the complete exploitation of the lowest trophic levels within the system. In parallel, lower mixed trophic impacts, biomass levels, and keystoneness values of the apex predators (e.g. sharks, dolphins, rays) of the system in the S3 scenario, indicate that the heavily depleted top predators of the system appear to play a limited role for the baseline scenario in structuring the ecosystem.

The addition of an artificial reef caused by the hypothetical installation of 50 offshore wind turbines appears to have a limited influence on the functioning of the ecosystem. When comparing the artificial reef scenario (S2) with that of the baseline, the only clear increases in functional group biomass where those that had been forced by the model (Commercial and non-commercial bivalves, filter feeding invertebrates and polychaetes). A small increase in detritus was also observed, presumably due to the increased throughflow of the commercial bivalves (5%). The model biomass forcing caused a diminutive (< 0.001%) increase in ecosystem activity, however, ecosystem functioning (Keystoneness values, transfer efficiencies, mixed trophic impacts, and omnivory index values), didn't change with the simulated introduction of an offshore wind farm.

These results are in contrast to the proposed hypotheses that offshore wind farms will operate as artificial reefs that aggregate apex predators to the area due to additional food sources (Langhamer, 2012). The aggregation of higher trophic levels (e.g. marine mammals, and piscivorous fish) has been observed in a few specific sites (Reubens et al., 2013; Russell et al., 2014), but the equivocal empirical evidence at many Northern European OWF case studies mean some uncertainty is associated with extrapolation of the results. In addition, the only other EwE approach to simulate ecosystem impacts of OWF installation indicated a positive response of piscivorous fish species, marine mammals, and seabirds, to the forced increase of colonizing macrobenthic biomass and fish aggregation around the wind turbines, however, it was at a level that didn't impact the structure and functioning of the ecosystem (Raoux et al., 2017).

The lack of any indications that increased food sources impact apex predators may be due to the time frame of the simulation period. Ecosim was only set to run for 10 years after the simulated introduction of the wind turbines, in contrast to Raoux et al., (2017), which simulated a 30 year evolution. The artificial reef may need time to attain the biological and ecological attributes needed to significantly influence ecosystem. This however, appears unlikely, the aggregation of pouting (*Trisopterus luscus*) at a wind turbine in the Thoringtonbank OWF in the North Sea was observed after less than one year after the turbine construction (Reubens et al., 2011). It remains is unclear whether the aggregation is due to an increase in productivity at the artificial reef site or the attraction of fish from within the region.

In areas where successful artificial reefs are introduced, the proportion of biomass increase due to either increased production rates within the system, or attraction/ redistribution of existing individuals from outside the modelled region, is still under debate (Chapter 1). Recent research indicates that although artificial reefs are, in general, very productive, only 4-5% of the production is new biomass (Smith et al., 2016). The present Ecopath model is parameterised to predict change within a closed system, and does not account for any emigration/immigration of biomass. Although the results imply no increase in the levels of higher trophic level production, it is possible that attraction from outside the system could occur; this however is not supported by results from the pilot project presented in Chapter 3.

Despite a negligible ecosystem impact with the introduction of the wind farm to the Northern Adriatic, the anticipated manner in which OWFs are to be managed (i.e. an almost total reduction in fishing) appears to significantly influence ecosystem functioning in the area. The Northern Adriatic is very heavily overfished, and has limited cross-national cooperation or marine protected area management (Severini, 2013). Offshore wind farms are likely to become *de-facto* MPAs even without specific management plans due to practical restraints associated with fishing inside wind farms and their supply stations (Ashley, 2014) (Chapter 1). Thus, it is important to consider the impacts that the reduction of fishing pressures will have in parallel with the introduction of an artificial reef to the area e.g. the increase in Keystoneness for higher trophic levels e.g. dolphins, seabirds, sharks, benthic piscivorous fish, and rays. By reducing fishing effort to 0%, the EwE model predicts a stronger response of the ecosystem in comparison to just the introduction of new substrata. The increase of apex predator keystoneness, transfer efficiencies, and changes to the mixed trophic impacts analysis, indicate that the system becomes more structured by top down predation as opposed to the availability of lower trophic levels (e.g. zooplankton).

This apex predator increase doesn't apply to the seabird functional group. The no-fishing scenario predicts no change in the biomass of seabirds within the system, likely due to the disappearance of discards as a food source when fishing efforts cease (Votier et al., 2004). This may be counteracted if attraction of fish individuals from outside the OWF area occurs; however declines of several population sizes of seabird species has been widely reported in OWF post-construction surveys (Leopold et al., 2010; Petersen et al., 1999; Welcker and Nehls, 2016). Such reductions are typically assumed to be as a result of avoidance behaviours employed by seabird individual (Desholm and Kahlert, 2005). These results indicate the importance of considering, in parallel to avoidance behaviour impacts, the reductions of prey availability (discards from fishing activities) as a possible rationale for seabird decline within OWF sites.

In addition, it is important to note that displacement of fisheries could occur as a result of total exclusion within the wind farm (Campbell et al., 2014), and that the overall costs and benefits of particular management actions (Knights et al., 2014) have not been considered here. Changes in fishing pressures, especially static gear fleets, outside the OWF perimeter (Campbell et al., 2014), may mitigate the increase in ecosystem activity, biodiversity, and biomass observed in the ecosystem when fishing pressures are reduced within the wind farm. When definite planning proposals for OWF sites are available for the study area, a detailed examination of regional fishing grounds, gear use, and fishing intensity should be completed to reduce the impact of fisheries displacement on the wider fish populations in the region.

By using the same holistic approach presented here to assess the medium-long term impact of installing an offshore wind farm, it would be possible for marine spatial planners to compare regions throughout the Mediterranean to identify areas which best respond to OWF installation. The prominent impacts caused by the type of fishing policy, in comparison to the effects generated from the presence of the offshore wind farm, highlights just how important it is for decision makers to consider site management plans in future environmental impact assessments of offshore wind farms.

CHAPTER 6

MAIN FINDINGS

The likely expansion of offshore renewable energy devices in the Mediterranean will bring with it the introduction of hard substrate to predominantly soft substrate areas. The colonisation of these new structures by benthic macro-invertebrate species will create new habitats for bentho-pelagic communities in the region (van Hal et al., 2017). Despite evidence from some existing offshore wind farm sites in Northern European seas, predicting the impacts that this new artificial reef-type habitat will have in the Mediterranean marine environment remains unclear. This thesis addresses some of this uncertainty by showing that:

1. Based on the results of the pilot project, it appears that those macro-benthic communities colonizning a wind turbine in the Adriatic – Ionian region will be influenced by the type of material used, and the location of the turbines. In addition, there was a minimal impact to overall fish abundance but some evidence, however, of aggregation around the structures.
2. Throughout the Adriatic, there are variations in the level of larval connectivity. Some areas are identified as being much more pivotal within the connectivity network than others, particularly after longer pelagic larval durations (20 days). Additionally, for pelagic larval durations longer than 4 days the number of separate communities detected is relatively few, implying that for benthic populations most of the Adriatic is connected.
3. The ecosystem-level impacts of building offshore wind farms in this region of the Mediterranean are relatively limited, however in comparison, the type of management plan i.e. denying fisheries access to the offshore wind farm has a much greater impact, and influences the functioning of the ecosystem.

This chapter provides an overall synthesis of the knowledge gained by the study, and considers how to apply the results within the context of future offshore wind farm development in the region.

6.1 ARTIFICIAL REEF POTENTIAL OF OFFSHORE WIND FARMS IN THE REGION

A commonly cited benefit regarding the introduction of marine renewable energy devices is the habitat creation potential of the new substratum (Inger et al., 2009). This thesis investigated whether an artificial reef effect would occur in the study area (Adriatic and Northern Ionian) and whether the marine ecosystem would benefit positively from this effect. The approach was divided into two components. Firstly, what type of species would colonize the new habitat, and secondly whether this would increase the aggregation of predators and higher trophic levels of predators within the ecosystem?

6.1.1 SUBSTRATA COLONIZATION

With regards to benthic communities, the most significant impact associated with the expansion of renewable marine energy devices, either as fixed gravity turbines or the foundations for floating devices, is the addition of novel artificial substrates to soft substrate environments (Inger *et al*., 2009). Relatively few studies determine the impacts of offshore wind farm (OWF) turbines and foundations on benthic communities (however see: Birklund & Petersen., 2004; Joschko *et al*., 2008; Wilhelmsson & Malm., 2008; Maar *et al*., 2009; Kerckhof *et al*., 2009; Lindeboom *et al*., 2011; Shi *et al*., 2012; Gutow *et al*., 2014; Mesel *et al*., 2015; Krone *et al*., 2017). The *in-situ* pilot project, which simulated colonization and habitat use of offshore wind turbines (Chapter 2, Chapter 3), was the first of its kind in the Mediterranean.

Throughout the study period the number of macroinvertebrate individuals increased, due to a rapid colonization of the bare surface provided by the colonization plates by recruits (Smith et al., 2014), however the colonization was also influences by the material used for the plates, and the location of the stanchions. However, the results from the pilot project indicate that not all variables measured had an impact on the community composition; neither the height of the plates, nor the direction they face in the water column (N, S, E and W) had an impact on the abundance of macroinvertebrate species. This information has important implications for developers concerned with habitat creation and range extension of regional benthic communities. It is clear that the design of wind turbines (i.e. the size of concrete foundations vs. steel diameter of stanchion) and the siting of the wind farms should be taken into consideration during the planning process. If the turbines are constructed in regions well connected to larval source regions, it is likely that population growth will occur (Pineda et al., 2010), thus implying that the selection of sites for wind turbines should be done in parallel with the establishment of benthic larval connectivity metrics.

It is important however, to consider that although the artificial substratum was indeed colonized by macro-invertebrates, when compared to natural hard substrata environments the number of individuals was fewer and the communities were of a lower diversity. It should be noted that due to the fairly limited time-scale of the pilot project (6 months) this difference may be due to a lag in colonization, however this difference in community composition is in accordance with several studies analysing differences between offshore renewable energy devices and natural habitats (Krone et al., 2013; Wilhelmsson and Malm, 2008).

Thus, it appears implausible to expect offshore wind farms to act as direct substitutes of natural hard substrate environments, but as no differences in the numbers of non-indigenous species were observed and the macroinvertebrate abundances increased with time, it seems there is a possibility for offshore wind farms in the area to operate as ‘stepping stones’ for benthic communities, albeit to a lesser extent than offshore hard substrate rocky outcrops.

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6.1.2 HIGHER TROPHIC LEVEL AGGREGATION

Over the past two decades artificial reefs in the Mediterranean have been met with varying degrees of success (Jensen et al., 2000). In many instances increases in the biomass of fish and high trophic level predators were evident (Charbonnelet al., 2002; Fabi et al., 2006) whilst in other locations no changes in the regional biomass of fish were observed (Barnabé, et al., 2000). Many factors influence the success of artificial reefs; however it is unclear when increased numbers of fish are observed whether this is due to an increase in production rate or whether the artificial reefs stimulate the aggregation of existing individuals to specific locations (Ruebens et al., 2014).

The results from the pilot project, although they noted a benthic colonization of macroinvertebrates that increased with time, didn't provide any indication of increases in fish abundance around the turbine simulations (Chapter 3). Analyses showed that the abundance of fish is similar at both the artificial units and the bare substrata control sites. Both the artificial units and the control sites harboured lower fish abundance and diversity when compared to a natural rocky outcrop, presumably due to lower lever of surface complexity and niche creation (St. Pierre and Kovalenko, 2014). The *in-situ* results indicate that artificial vertical relief sites in the region might not have the capacity to operate as successful artificial reefs.

In accordance with this finding, when the introduction of an offshore wind farm into the ecosystem was simulated i.e. an increase in the biomass of colonizing macroinvertebrate species, failed to predict any changes to the biomass of apex predators (Chapter 4). No clear changes in fish species biomass occurred and the hypothetical introduction of an offshore wind farm to the Adriatic hotspot had almost no impact on the functioning, structure or overall activity of the ecosystem.

These results contrast with several Northern European field studies at offshore renewable energy sites (De Troch et al., 2013; Langhamer, 2012; Reubens et al., 2013, 2011; Wilhelmsson and Malm, 2008). Considering that the *in-situ* results also indicated some aggregation of the fish around the artificial units, the increases in fish biomass in the Northern European field studies may be due to an aggregation of individuals at the site as opposed to the new hard substrate supporting higher rates of production.

6.2 SITE SELECTION AND SENSITIVITY

Knowing the importance that location has on influencing the macro-invertebrate communities that colonize an offshore wind turbine (Chapter 2), the identification of connectivity levels within a Mediterranean offshore wind farm hotspot (Chapter 1) is essential. The Adriatic ‘hotspot’ is an already heavily burdened area in terms of offshore use (Micheli et al., 2013; Zanuttigh et al., 2015). High wind speeds, and relatively shallow coastal depths mean that parts of the region are suitable for renewable energy development sites. In order to best aid site selection within the Adriatic, it is important to have a clear understanding of connectivity levels for benthic communities to identify possible larval source and destination regions. Since the seafloor of the Adriatic is predominantly sedimentary, new structural material (e.g. OWF foundations) may act as a conduit for the movement of benthic populations across the region (Bishop et al., 2017; Firth et al., 2016). Depending on the aims of a planning process, developers may wish to either increase the benthic community connectivity of a wind farm to encourage the development of an artificial reef at the site, and range expansion of native species (Langhamer, 2012), or decrease connectivity to reduce the likelihood of colonization by invasive species (Adams et al., 2014).

The construction of a connectivity map to identify high and low areas of macroinvertebrate connectivity is a novel approach for an OWF site selection process, which may prove vital to marine spatial planners. This thesis produces such a map for the Adriatic ‘hotspot’ (Chapter 3). The network analysis shows how the whole Puglia coast is well connected within the Adriatic network, and acts a sink for larvae from benthic populations in the Adriatic, making more likely that offshore wind turbines in this region will operate as stepping stones for benthic communities (Adams et al., 2014). In contrast, regions which are situated close to commercial ports and thus at higher risk of non-indigenous species presence (Seebens et al., 2016), may benefit from being less connected and more isolated from the connectivity network as to reduce the spread of these often ecologically harmful species (Simons et al., 2016); The Po river basin, and the Montenegro/Albanian coastline are some of the least connected sites in the Adriatic.

6.3 OFFSHORE WIND FARM ECOSYSTEM APPROACH AND MANAGEMENT PLANS

Along with other parameters (technical, social, economical etc), ecological targets should influence the decision making process for marine spatial planners in regards to the siting and managing offshore wind farms. The reduction of commercial fisheries at offshore wind farm sites is common in Northern European Seas (Adams et al., 2014). These reductions are often not due to the implementation of statutory instrumentation, but the reluctance of fisherman to endanger fishing apparatus on underground cables. Many offshore wind farm sites have become '*de facto'* marine protected areas. For the first time, this thesis considered the ecosystem level impacts of restricting fishing efforts at a potential offshore wind farm site in the Mediterranean (Chapter 5).

When an Ecological Network Analysis (ENA) was run that compared the Adriatic ecosystem with, and without, the enforcement of fishing restrictions at a simulated offshore wind farms, it showed that removing fishing activity from the area caused an increase in ecosystem activity, biodiversity, and biomass. In addition to these increases, the previously heavily fished apex predators of the Northern Adriatic played a larger role in structuring ecosystem. The results indicate the importance of considering the installation and operation of an offshore wind farm within the context of likely management schemes, and that the addition of novel hard substrate alone does not appear to generate the artificial reef effects noted by many existing offshore wind farms.

In conclusion, despite appearing to be a suitable area for offshore wind farm development, it appears that wind turbines in the Adriatic and Northern Ionian will only create effective artificial reefs if rigorous fishing restrictions are implemented and turbines are situated in areas well connected with larval source regions. In an attempt to combat the ever evident environmental impacts of fossil fuel use, offshore wind energy is a seemingly feasible renewable alternative, however without applying cumulative environmental impact assessments the influences on marine ecosystems cannot be feasible determined. To fully determine the context of environmental impacts from offshore wind farm installations in the Mediterranean, a holistic ecosystem approach, as is presented in this thesis, is essential.

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RELATED PUBLICATIONS

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Bray, L., Anastasopoulou, K; Hall-Spencer’ J. M; Reizopoulou, S. The potential role of offshore structures as artificial reefs in the Mediterranean – A pilot study. Proceedings of the 51st European Marine Biology Symposium. Journal of Marine Biological Association (In Review).

For copies of the aforementioned manuscripts which have been published at the time of thesis submission, the reader is directed to Appendix B. For a list of conferences, workshops, specialist training, and outreach programs attended by the author during the PhD study duration, the reader if directed to Appendix C.

APPENDIX A

Table A1. Parameters used for Adriatic model. All data is obtained from (Barausse et al., 2009). Reader is directed to the supplementary information of the study of Barausse et al., (2009) to obtain references for data presented. B = biomass, P = production, Q = comsumption, and EE = Eutrophic Efficieny.

Functional Group	B (t km^2 yr^{-1})	P/B (yr^{-1})	Q/B (yr^{-1})	EE	P/Q
Dolphins	0.006	0.054	14.0001	0.278	0.004
Seabirds	0.010	4.610	69.340	0.000	0.066
Sharks	0.056	0.400	3.077	0.894	0.130
Rays	0.012	0.724	4.137	0.870	0.175
European hake	0.052	1.180	4.120	0.937	0.286
Zoobenthivorous fish hard bottom	0.189	1.900	6.410	0.990	0.297
Zoobenthivorous fish soft bottom	0.847	1.180	6.306	0.998	0.187
Mackerel	1.001	0.620	6.506	0.265	0.095
Horse mackerel	0.962	0.570	5.941	0.593	0.096
Other small pelagics	0.330	1.889	8.294	0.851	0.228
Anchovies	7.134	1.100	12.44	0.910	0.088
Sardines	4.718	0.870	8.709	0.950	0.100
Nectobenthic zooplanktivorous fish	0.401	1.180	6.442	0.989	0.183
Omnivorous fish	0.115	1.624	15.040	0.996	0.108
Benthic piscivorous fish	0.201	0.850	3.304	0.970	0.257
Flatfishes	0.141	1.300	6.975	0.922	0.186
Squids	0.050	3.506	26.968	0.973	0.130
Benthic cephalopods	0.100	3.300	6.600	0.994	0.500
Crustacea 1	5.437	2.894	17.786	0.958	0.163
Crustacea 2	1.017	7.911	51.186	0.999	0.155
Mantis shrimp	0.081	1.500	4.560	0.897	0.329
Non-commercial bivalves	25.130	1.415	6.350	0.511	0.223
Commercial bivalves	0.902	1.415	6.350	0.876	0.223
Gastropods	7.911	1.735	9.717	0.898	0.179
Filter-feeding invertebrates	7.756	0.761	3.804	0.949	0.200
Echinoderms	8.939	0.803	2.514	0.433	0.319
Polychaetes	27.268	1.644	14.27	0.667	0.115
Jellyfish	1.025	14.813	44.439	0.150	0.333
Zooplankton	3.285	65.060	185.170	0.917	0.351
Pelagic bacteria	4.055	136.125	758.696	0.414	0.179
Macroalgae and phanerogams	42.435	1.700	-	0.200	-
Phytoplankton	13.165	175.230	-	0.403	-
Discard	0.058	-	-	0.970	-
Detritus	364.672	-	-	1.000	-

Table A2. Average annual landings and discards used for model. Reader is directed to the supplementary information of the study of Barausse et al., (2009) to obtain references for data presented.

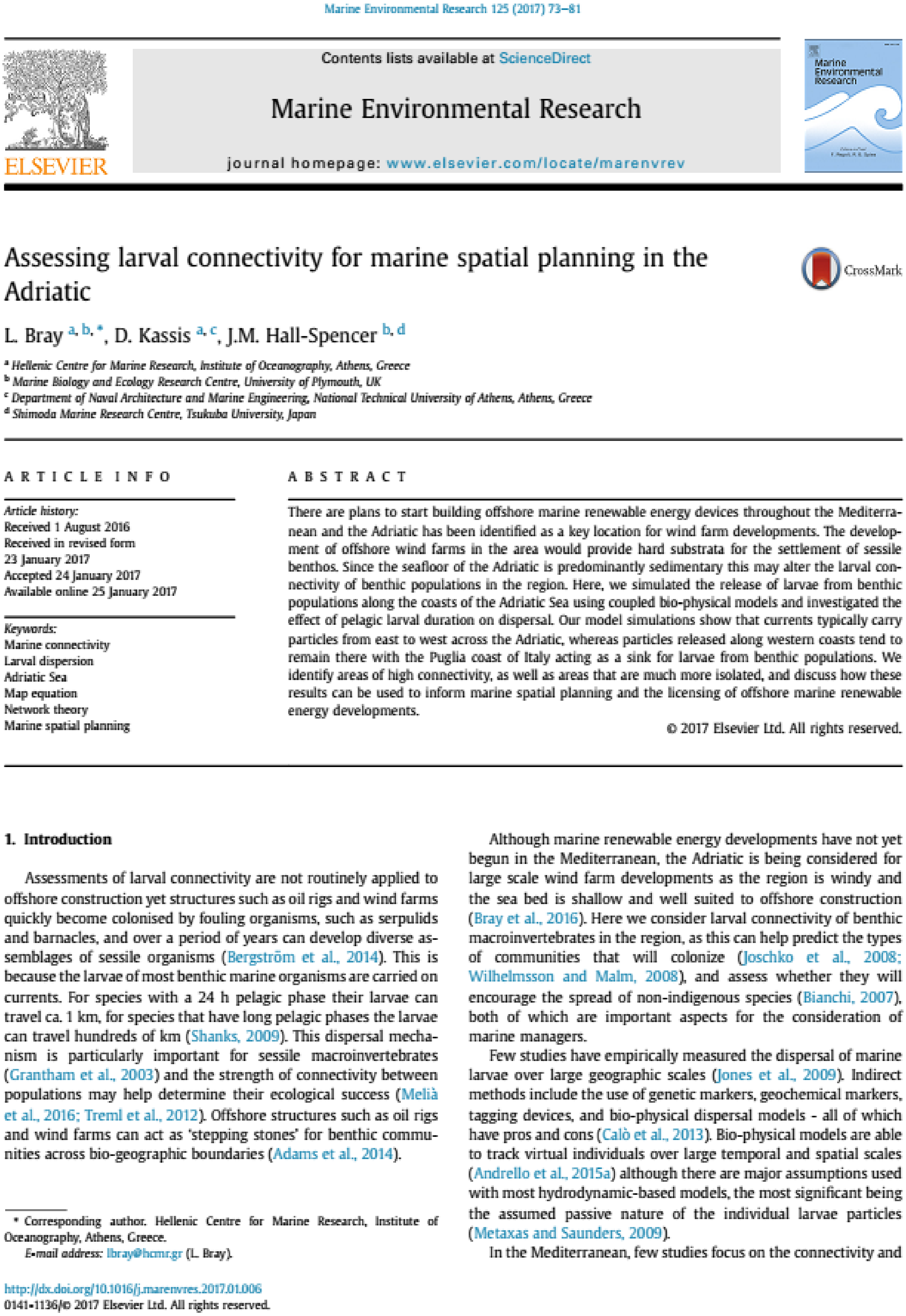
Functional group	Average landings ($t\ km^{-2}\ yr^{-1}$)	Average discards ($t\ km^{-2}\ yr^{-1}$)
Dolphins	-	0.0000942
Seabirds	-	-
Sharks	0.0136	-
Rays	0.00473	-
European hake	0.0353	-
Zoobenthivorous fish hard bottom	0.0162	0.00244
Zoobenthivorous fish soft bottom	0.167	0.172
Mackerel	0.0379	0.00113
Horse mackerel	0.0281	0.0856
Other small pelagics	0.0214	0.0814
Anchovies	0.711	0.2
Sardines	0.999	0.55
Nectobenthic zooplanktivorous fish	0.0729	0.0757
Omnivorous fish	0.0797	0.00174
Benthic piscivorous fish	0.0189	0.099
Flatfishes	0.0538	0.0479
Squids	0.0441	0.011
Benthic cephalopods	-	0.0109
Crustacea 1	0.178	0.384
Crustacea 2	0.068	0.00000523
Mantis shrimp	0.0957	0.00766
Non-commercial bivalves	-	0.0594
Commercial bivalves	0.553	0.0463
Gastropods	0.0155	0.309
Filter-feeding invertebrates	0.0000852	0.325
Echinoderms	-	0.819
Polychaetes	-	0.015
Jellyfish	-	0.0000586
Zooplankton	-	-
Pelagic bacteria	-	-
Macroalgae and phanerogams	-	0.992
Phytoplankton	-	-
Discard	-	-
Detritus	-	-

Table A3 Diet composition matrix of the Ecopath model for the year 2020.

	Prey/Predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
1	Dolphins																														
2	Seabirds																														
3	Sharks			0.01		> 0.01										> 0.01															
4	Rays			> 0.01	> 0.01	> 0.01										> 0.01															
5	European hake	0.02		> 0.01	0.015	0.013			> 0.01	> 0.01						> 0.01		> 0.01													
6	Zoobenthivorous Fish—hard bottom	0.02	0.013	> 0.01	0.015	> 0.01	> 0.01	0.023	> 0.01	> 0.01	> 0.01			> 0.01		0.031	0.012	> 0.01	0.05	> 0.01											
7	Zoobenthivorous Fish—soft bottom	0.002	0.013	> 0.01	0.015	> 0.01	> 0.01	0.028	> 0.01	0.02	> 0.01			> 0.01		0.158	0.018	> 0.01	0.145	> 0.01		0.04									
8	Mackerel	0.066		0.018	0.015	0.058				> 0.01						0.015		0.061													
9	Horse mackerel	0.064	0.004	0.017	0.015	0.056		0.016								0.031		0.059													
10	Other small pelagics	0.022	0.009	> 0.01	0.015	0.019	0.002	> 0.01	0.014	0.01	> 0.01			0.086		0.013	> 0.01	0.02													
11	Anchovies	0.465	0.155	0.127	0.015	0.41	0.04	0.015	0.408	0.382	> 0.01			0.022		0.329	0.04	0.486													
12	Sardines	0.313	0.103	0.084	0.015	0.274	0.026	0.032	0.13	0.1	> 0.01			0.016		0.181	0.03	0.256													
13	Nectobenthic zooplanktivorous fish	0.009	0.013	> 0.01	0.015	> 0.01	> 0.01	0.024	> 0.01	0.01	> 0.01			> 0.01		0.105	> 0.01	> 0.01	0.052												
14	Omnivorous fish		0.004	> 0.01	0.015	> 0.01	> 0.01	> 0.01	> 0.01	0.014	> 0.01			> 0.01		> 0.01	> 0.01	> 0.01	0.019												
15	Benthic piscivorous fish	0.01	0.004	> 0.01	0.015	> 0.01	> 0.01	> 0.01	> 0.01	> 0.01	> 0.01					> 0.01		> 0.01	> 0.01												
16	Flatfishes		0.007	> 0.01	0.015	> 0.01	> 0.01	> 0.01	> 0.01	> 0.01	> 0.01			> 0.01		0.013	> 0.01	> 0.01	> 0.01												
17	Squids	0.005	0.002	0.027	0.015		> 0.01	> 0.01	> 0.01	> 0.01						> 0.01		0.01	> 0.01												
18	Benthic cephalopods	0.003		0.111	0.015	> 0.01	> 0.01	> 0.01		0.005						> 0.01	0.01	0.035	0.027			> 0.01									
19	Crustacea 1			0.377	0.015	0.15	0.321	0.621	0.04	0.048	> 0.01			0.08	0.194	0.091	0.251	0.048	0.566	0.091		0.309									
20	Crustacea 2			0.071	0.015	> 0.01	0.153	0.063	0.01		> 0.01			0.071	0.017		0.227	> 0.01	0.048	0.047	0.045	0.09				> 0.01					
21	Mantis shrimp			0.00025	0.015	> 0.01	> 0.01	> 0.01								> 0.01	> 0.01														
22	Non-commercial bivalves			0.021	0.015		0.088	0.013									0.131		0.036	0.142		0.367			0.031		0.066				
23	Commercial bivalves			> 0.01	0.015		> 0.01	> 0.01									> 0.01		> 0.01	0.004		0.012			> 0.01		> 0.01				
24	Gastropods			> 0.01	0.015		0.083	> 0.01			> 0.01			0.042			0.063		0.025	0.079		0.107			0.032	> 0.01	0.068				
25	Filter-feeding invertebrates			0.014			0.036								> 0.01		> 0.01		> 0.01						0.032			> 0.01			
26	Echinoderms			0.016	> 0.01		0.06	> 0.01									> 0.01			0.022							> 0.01				
27	Polychaetes			0.05	> 0.01		0.147	0.119		0.01	0.01			> 0.01			0.173	0.01		0.109		0.07			0.033		> 0.01	0.04			
28	Jellyfish																												0.05		
29	Zooplankton			0.039	0.061		> 0.01	0.022	0.382	0.371	0.972	0.99	0.925	0.67	0.07	> 0.01	0.022	> 0.01	> 0.01		0.047		0.06	0.06		> 0.01	0.099		0.75	0.02	
30	Pelagic bacteria																				0.192		0.22	0.22	0.109	0.491	0.14	0.1		0.19	
31	Macroalgae and phanerogams						0.038						0.01		0.25					0.054					0.033		> 0.01	0.012			
32	Phytoplankton											0.01	0.065		0.02					0.022	0.121		0.22	0.22	0.109	> 0.01	0.14	0.12	0.05	0.6	0.15
33	Discard		0.1																0.01	0.012	> 0.01				0.012		0.01	> 0.01			
34	Detritus							0.014							0.445					0.415	0.59		0.5	0.5	0.606	0.497	0.463	0.717	0.15	0.19	0.85
	Import		0.573																												

APPENDIX B

Published manusripts at time of thesis submission



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Assessing larval connectivity for marine spatial planning in the Adriatic

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ABSTRACT

There are plans to start building offshore marine renewable energy devices throughout the Mediterranean and the Adriatic has been identified as a key location for wind farm developments. The development of offshore wind farms in the area would provide hard substrata for the settlement of sessile benthos. Since the seafloor of the Adriatic is predominantly sedimentary this may alter the larval connectivity of benthic populations in the region. Here, we simulated the release of larvae from benthic populations along the coasts of the Adriatic Sea using coupled bio-physical models and investigated the effect of pelagic larval duration on dispersal. Our model simulations show that currents typically carry particles from east to west across the Adriatic, whereas particles released along western coasts tend to remain there with the Puglia coast of Italy acting as a sink for larvae from benthic populations. We identify areas of high connectivity, as well as areas that are much more isolated, and discuss how these results can be used to inform marine spatial planning and the licensing of offshore marine renewable energy developments.

1. Introduction

Assessments of larval connectivity are not routinely applied to offshore construction yet structures such as oil rigs and wind farms quickly become colonised by fouling organisms, such as serpulids and barnacles, and over a period of years can develop diverse assemblages of sessile organisms (Bergström et al., 2014). This is because the larvae of most benthic marine organisms are carried on currents. For species with a 24 h pelagic phase their larvae can travel ca. 1 km, for species that have long pelagic phases the larvae can travel hundreds of km (Shanks, 2009). This dispersal mechanism is particularly important for sessile macroinvertebrates (Grantham et al., 2003) and the strength of connectivity between populations may help determine their ecological success (Melià et al., 2016; Treml et al., 2012). Offshore structures such as oil rigs and wind farms can act as 'stepping stones' for benthic communities across bio-geographic boundaries (Adams et al., 2014).

Although marine renewable energy developments have not yet begun in the Mediterranean, the Adriatic is being considered for large scale wind farm developments as the region is windy and the sea bed is shallow and well suited to offshore construction (Bray et al., 2016). Here we consider larval connectivity of benthic macroinvertebrates in the region, as this can help predict the types of communities that will colonize (Joschko et al., 2008; Wilhelmsson and Malm, 2008), and assess whether they will encourage the spread of non-indigenous species (Bianchi, 2007), both of which are important aspects for the consideration of marine managers.

Few studies have empirically measured the dispersal of marine larvae over large geographic scales (Jones et al., 2009). Indirect methods include the use of genetic markers, geochemical markers, tagging devices, and bio-physical dispersal models - all of which have pros and cons (Calò et al., 2013). Bio-physical models are able to track virtual individuals over large temporal and spatial scales (Andrello et al., 2015a) although there are major assumptions used with most hydrodynamic-based models, the most significant being the assumed passive nature of the individual larvae particles (Metaxas and Saunders, 2009).

In the Mediterranean, few studies focus on the connectivity and

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dispersal of marine species (Calò et al., 2013) and this paucity of information is an obstacle for policy makers in the region (Andrello et al., 2015b). Those connectivity studies that use virtual particle trajectory methods tend to focus on the establishment and evaluation of marine protected areas (Andrello et al., 2013; Di Franco et al., 2015; Pujolar et al., 2013). Other approaches include the homogenous release of larvae particles throughout the whole Mediterranean (Dubois et al., 2016; Rossi et al., 2014), or the release from specific coastal sites at a regional level (Carlson et al., 2016; Melià et al., 2016; Schiavina et al., 2014; Schunter et al., 2011). Many such studies are tailored to determine connectivity of fish and macroinvertebrate larvae trajectories are seldom modelled in the Mediterranean (Guizien et al., 2014; Padrón and Guizien, 2015; Schiavina et al., 2014).

In the present study, we simulated a release of larvae from benthic populations along the coasts of the Adriatic Sea using coupled bio-physical models and investigated the effect of pelagic larval duration (e.g. simulation duration) on dispersal. We assumed a homogenous larval production and tracked evenly distributed Lagrangian particles for a range of pelagic larval durations (4, 8, 16, 20 days) to cover regionally common invertebrate taxa such as barnacles and gastropods (Villamor et al., 2014), rather than utilizing a particular target species (Rossi et al., 2014). We also tracked the likely spread of larvae from benthic populations that originate from the major Adriatic ports, as several studies show an increase in the abundance of non-indigenous species in or around Adriatic ports and marinas (David et al., 2007; Ivesa et al., 2015; Pecarevic et al., 2013), and the potential spread of non-indigenous species through corridors of artificial surfaces (Airoldi et al., 2015) is a critical aspect of marine connectivity studies. Essentially, our aim was to identify areas of high vs. low connectivity to inform marine spatial planning and the licensing of offshore marine renewable energy developments.

2. Methods

Our method was based on the Graph Theory approach used by Rossi et al. (2014) for identifying hydrodynamic provinces throughout the Mediterranean. We modelled the release of Lagrangian particles in evenly distributed grid cells along the Adriatic coastline and then tracked these particles for a range of known pelagic larval durations. Source and destination grid cells were compared to indicate regions of high and low connectivity.

2.1. Study area

The Adriatic Sea has a shallow northern section (average depth 40 m), a central section (average depth 140 m) and a southern section where troughs > 1200 m deep (Fig. 1) channel deep water masses into the Eastern Mediterranean, particularly in late winter (Gačić et al., 2002; Malanotte-Rizzoli et al., 1997). The western coast is generally sandy whereas the eastern side is predominantly rocky (Artegiani et al., 1997) and the hydrography of the basin is influenced by several large rivers (Verri et al., 2014). The circulation is cyclonic overall, with three cyclonic sub-systems in the northern, middle and southern sections and a strong current flowing south along the coast of Italy from spring until autumn (Zavatarelli et al., 1998).

2.2. Hydrodynamic grid

Hydrodynamic model output data were obtained from the Mediterranean Monitoring and Forecasting Centre of the Copernicus Marine Environment Monitoring Service (http://marine.copernicus.eu) which has been running since 2000. The model is composed of an Ocean General Circulation Model (Tonani et al., 2013) and a coupled hydrodynamic-wave model with a horizontal grid resolution of 1/16° (ca. 6–7 km). We subdivided the Adriatic into a 0.0625° × 0.0625° grid (each grid cell approx. 6.7 km^2) to match the resolution of the hydrodynamic model giving 383 release grid cells (S1).

2.3. Simulated larval particle transport

Particles were released from the centre of each of 383 grid cells along the Adriatic coastline and trajectories were followed using the program ICHTHYOP (Lett et al., 2008). No behavioural parameters were assigned to the simulated larval particles thus assuming a passive trajectory. Particle position was calculated every 2 h, for four pelagic larval durations (4, 8, 16 and, 20 days). We chose consecutive release dates (n = 10) throughout June (starting from the 01/06 each year) to coincide with peak benthic macroinvertebrate spawning in the region (Villamor et al., 2014). Particles were released at the same time each day (00:00), and to account for inter-annual variability, the larval dispersal simulations were run for consecutive years covering the period 2011–2015 (n = 5). For each larval duration, a cumulative total of 3830 particles were

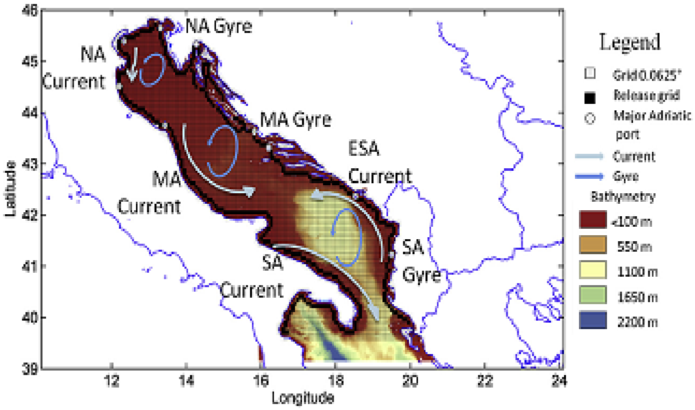

Fig. 1. Adriatic larval connectivity matrix comprised of a 1/16th degree grid into which larval particles were released, showing locations of major ports (clockwise from left: Taranto, Ancona, Ravenna, Venice, Trieste, Rijeka, Sibenik, Split, Tivat, Durres), and major currents NA = Northern Adriatic, MA = Mid-Adriatic, Sa = South Adriatic, ESA = Eastern South Adriatic. Bathymetry provided by www.emodnet.eu, hydrology adapted from Artegiani et al. (1997).

released. A limited tidal range in the Adriatic Sea means atmospheric effects are the main forcing factors in the Adriatic Sea (Bolaños et al., 2014). With respect to this, particle releases were not factored around tidal stages as other larval dispersal models have done in more tide-dominant environments (Narváez et al., 2012).

2.4. Post-simulation analysis

Destination grid cells were calculated for each particle using MATLAB6.1, and both descriptive statistics and probability matrices were constructed from an amalgamation of all simulation years and release dates for each larval duration. Additionally a year-on-year analysis of the total distances that particles travelled was done to examine significant differences between years. Due to the non-normal distribution of the data, non-parametric tests (e.g Kruksall-Wallis and Mann-Whitney U Comparison) were used. To visualise the inter-annual differences of the larval trajectories a single simulation track from each year is presented which indicates particle position for 4, 8, 16 and 20 day durations. Locations of OWF's in early planning/concept stage as of January 2017 are included for reference.

Simulated larvae were considered to have self-replenished if by the end of the simulation potential non-behaviour dispersal trajectories remained in their original release grid. Probabilities of particle arrival were mapped for each grid cell and particle transport distances were calculated. To provide information on larval transport from industrialized regions (Fig. 1), release grids located closest to the ten major Adriatic ports were selected and the particles released from these sites were presented separately.

We used *Infomap* to define network structure (Rosvall and Bergstrom, 2008) and it allowed us to examine cells within our grid of across the Adriatic Sea and determine where larval transport can be expected to flow quickly and easily between them, for details see Rossi et al. (2014). In addition to community detection, *Infomap* also provides information on the importance of individual nodes via the use of its pageRank algorithm. PageRank (commonly used for ranking web pages) provides a nonlocal measure of centrality by defining the expected density of random walkers on a node at stationarity, within a weighted, directional, network (Lambiotte and Rosvall, 2012). PageRank for each cell is presented as a probability distribution with a numerical value between 0 and 1, i.e. a cell with a pageRank of 0.5 means that a random walker within the network would have a 50% chance of arriving at the given cell. Identifying the highest and lowest ranked nodes for each pelagic larval duration illustrates the most and least important grid cells within each network.

3. Results

As expected, simulated increases in the duration of particle transport resulted in an increase in the distance travelled. Likewise, as dispersal duration increased, self-replenishment decreased. Overall levels of self-replenishment were very low, but were an order of magnitude higher at release grids close to Adriatic Ports (Table 1), likely due to the typical positioning of ports in enclosed bays. Dispersal distances increased from around 11 km for 4 day simulations, to 30 km for larvae that could survive for 20 days in the plankton and the greatest distance travelled by a particle during the 20 day simulation was 334 km (Table 1). The large Standard Deviations around each mean show that some particles remained close to the simulated release sites, whereas others travel far; this variability increased with dispersal duration.

Regarding inter-annual differences of the distances that the simulated particles travelled, the non-parametric (due to extreme outliers of the data) statistical test Kruskal-Wallis test for equal medians was used to compare differences between years. All the pelagic larval duration simulations expressed significant differences between years (For PLD4 $H(2) = 856.82$, $p = 0.00$; PLD8, $H(2) = 661$, $p = 0.00$; PLD16, $H(2) = 480.91$, $p = 0.00$; and PLD20, $H(2) = 387$, $p = 0.00$ (2 s.f.)). Post hoc Mann-Whitney tests for yearly differences within each PLD showed most years are significantly different, with only 6 years not showing any significant differences (Table 2).

3.1. Particle transport

In agreement with the statistical analysis of the year-on-year differences between distance travelled by individual particles, the spatial depiction of the particle trajectories indicates high inter-annual variabilities (Fig. 2). Larval sink locations (locations where particle tracks terminate) are not consistent, and although no clear inter-annual trends are apparent, an increased inter-connection between the east and west coasts after 2013 is noticeable. The model simulates particle transport from the central-eastern coastline to the west coast within the 41° – 44° latitudes for the years 2013, 2014 and 2015.

After a simulation duration of four days, regions of grid cells with high concentrations of larval trajectory destination points include the region South of the river Durres delta, the port of Rijeka, the Kvarner Gulf (Croatia), the Gulf of Trieste, Gulf of Venice, and many locations along the Italian Adriatic Coast. Regions with grid cells of lower count densities include the Po river delta, and the offshore region of the Dalmatia coast. Similar results were found for 8, 16, and 20 day durations, with areas of low densities of larval trajectory destination points being mostly restricted to offshore regions such as the Bay of Kotor, the Southern Region of Gulf of Trieste, and the Po river delta (Fig. 3). A more detailed depiction in the form of a probability matrices is provided in the Supplementary files (S2, S3, S4 and S5).

A high concentration of port destination cells was also located close to Split, the largest passenger port in the region, for all durations. For simulation durations of 8, 16 and 20 days, large sections of Albanian coast indicated high concentrations of larval trajectory destination points having being released from areas in close vicinity of ports (Fig. 4).

After 4 days, the grid cells within the network with relatively

Table 1
Descriptive statistics for particle trajectories (Avg. = Average, SD= Standard deviation, SR = self replenishment).

	4 days	8 days	16 days	20 days
Furthest distance (km)	88.7	205.5	308.3	334.7
Avg. distance (km) ± SD	11.0 ± 11.0	16.8 ± 17.3	25.7 ± 28.1	29.5 ± 34.0
Avg. Distance from ports (km) ± SD	7.6 ± 6.3	12.2 ± 12.4	20.3 ± 12.7	24.4 ± 30.9
Avg. SR(%) ± SD	0.01 ± 0.1	0.01 ± 0.00	0.01 ± 0.00	0.01 ± 0.00
Avg. SR at port sites (%) ± SD	0.11 ± 0.12	0.07 ± 0.06	0.04 ± 0.07	0.04 ± 0.07

Table 2
Matrix of Mann-Whitney test value (U), and probability (p), for each comparison of year-on-year particle distance for each dispersal duration. Highlighted in bold are yearly comparisons which show NO statistical differences. Values for Mann-Whitney test value (U) are shown to 3 s.f, and probability values (p), are shown to 2 s.f.

		2012	2013	2014	2015
PLD4	2011	U = 7020000, p = 0.04	U = 5500000, p = 0.00	U = 6260000, p = 0.00	U = 5320000, p = 0.00
	2012		U = 5300000, p = 0.00	U = 6030000, p = 0.00	U = 5100000, p = 0.00
	2013			U = 6600000, p = 0.00	**U = 7190000,** **p = 1.00**
	2014				U = 6450000, p = 0.00
PLD8	2011	**U = 7220000,** **p = 0.93**	U = 5760000, p = 0.00	U = 6400000, p = 0.00	U = 5260000, p = 0.00
	2012		U = 5870000, p = 0.00	U = 6500000, p = 0.00	U = 5380000, p = 0.00
	2013			U = 6690000, p = 0.00	U = 6860000, p = 0.00
	2014				U = 6230000, p = 0.00
PLD16	2011	**U = 7070000,** **p = 0.18**	U = 6820000, p = 0.00	U = 6840000, p = 0.00	U = 5560000, p = 0.00
	2012		U = 6630000, p = 0.00	U = 6640000, p = 0.00	U = 5420000, p = 0.00
	2013			**U = 7260000,** **p = 1.00**	U = 6060000, p = 0.00
	2014				U = 5990000, p = 0.00
PLD20	2011	**U = 7 080 000,** **p = 0.26**	U = 6940000, p = 0.00	U = 7000000, p = 0.02	U = 5770000, p = 0.00
	2012		U = 6 750000, p = 0.00	U = 6800000, **p = 0.00**	U = 5620000, p = 0.00
	2013			**U = 7240000,** **p = 1.00**	U = 6100000, p = 0.00
	2014				U = 6030000, p = 0.00

high self-replenishment included the Manfredonia Gulf, the Kvarner Gulf, the Adriatic coast of Italy, and south of the Po river delta. Regions of relatively lower cells of self-replenishment include the Po delta, the Gulf of Trieste, and the northern coast of Croatia. For durations of 8, 16 and 20 days the only relatively high self-replenishment regions were the Manfredonia Gulf, and the Kvarner Gulf, whilst the regions with very little self-replenishment include the Po delta, South of Gulf of Trieste, and most of the Kvarner Gulf (Fig. 5).

3.2. Clusters and node centrality within network

An increase of simulation duration resulted in fewer numbers of identified communities with the mapequation algorithm. *Infomap* clustering (Fig. 6) indicated that the four day larval duration, a transport network with 2022 nodes and 4883 links, was clustered into 76 modules with 110 inter-module links. The eight day larval duration, a network of 2362 nodes with 6462 links, was clustered into six modules with two inter-module links. Whilst both 16 (2650 nodes with 7484 links) and the 20 day duration (2764 nodes with 7812 links) were clustered into four modules with one inter-module link.

Mapping the PageRank for each grid cell indicated that for all PLD's the Adriatic coast of Italy contained some of the highest ranked grid cells along with two locations within the Kvarner Gulf (Croatia), thus indicating these regions contained some of the most connected cells within the network. Regions with consistently lower ranked grid cells and thus less connected were the offshore basin regions (all durations), the Po river delta (four day), and the Montenegro and Albanian coast (eight, 16, and 20 day durations) (Fig. 7).

4. Discussion

The Adriatic region is a distinct marine sub-region assigned as a priority region for marine spatial planning (Bastari et al., 2016). High activity use often creates conflicts between economic development of the region, habitat protection, and preservation of biodiversity. The region already has a great deal of offshore activities (Manoukian et al., 2010) and there is scope for rapid development of offshore wind farms, particularly in the Northern Adriatic (Bray et al., 2016). Here we explore how these developments will impact the marine benthic environment. The Adriatic Sea is managed nationally with several of the six coastal states sharing the use of territorial waters; the co-ordination of marine management in this region is often fragmented. Due to the interconnected cross-boundary nature of marine systems, the approach presented here may prove useful in fostering basin-scale management of the biological impacts of offshore construction in the Adriatic Sea.

4.1. Methodological approach

Three dimensional particle tracking models is useful for quantifying the dispersal of benthic invertebrate larvae (Metaxas and Saunders, 2009) and Graph Theory is an effective tool for exploring patterns of spatial connectivity (Treml et al., 2007). This approach has been widely used for the identification and evaluation of marine protected areas; however this is the first time it has been used as an aid for planning offshore construction. Nevertheless, there are several limitation associated with the approach. Real-world realization of the findings presented here requires additional information such as individual larval behaviour (Zhang

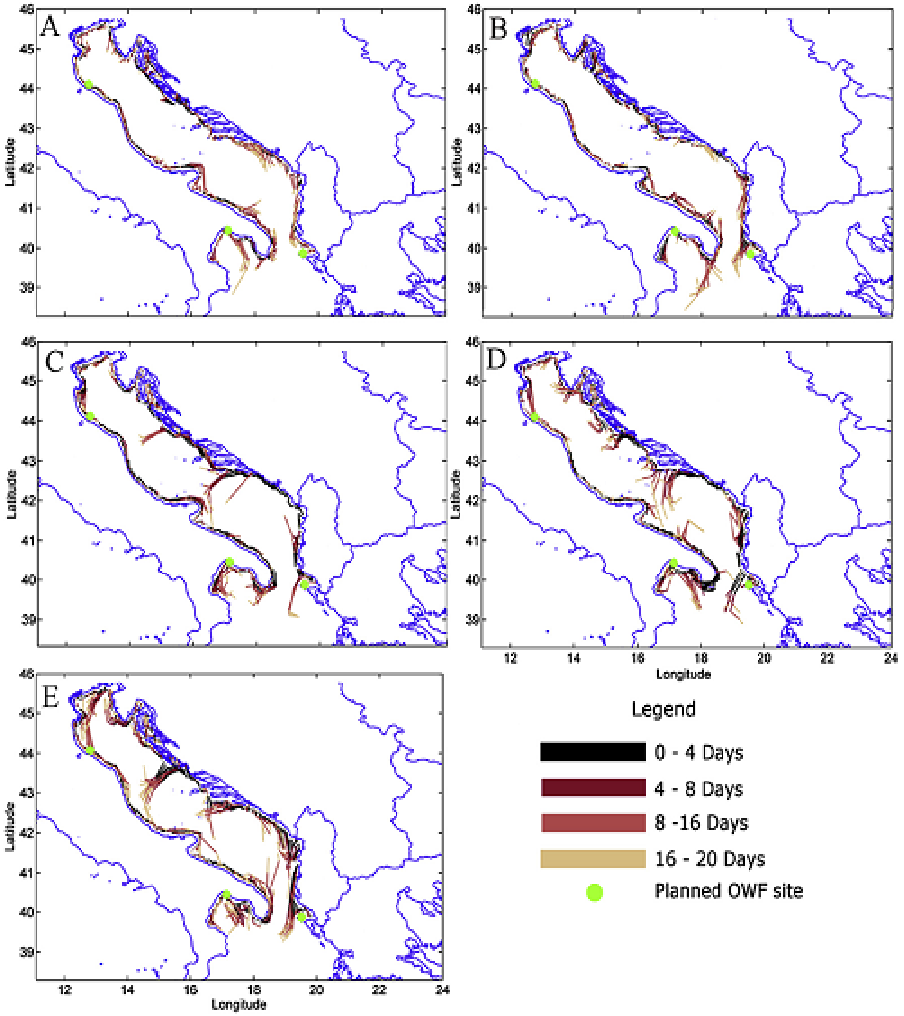

Fig. 2. A single track simulation to indicate source/sink information is presented with particle position taken from simulations of the 1st of June for each year. Panel A indicates the trajectory for the 1st of June 2011, B = 2012, C = 2013, D = 2014, and E = 2015. Positions shown for each time interval (0–4 day, 4–8 days, 8–16 days, and 16–20 days), and locations of offshore wind farms currently in the early planning/concept stage in the region are also depicted (http://www.4coffshore.com).

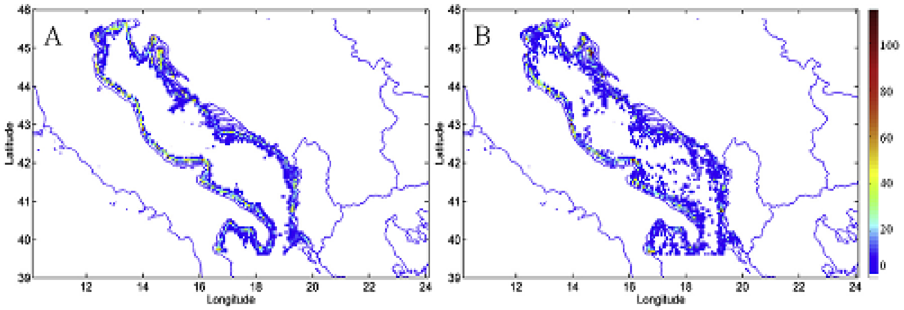

Fig. 3. Grid count densities of destination points of larval trajectories for A) four and B) 20 day larval durations. All PLD simulations produced similar patterns, albeit increasing dispersion with increasing larval duration so for convenience only the minimum and maximum larval durations are displayed. Counts measured in absolute terms.

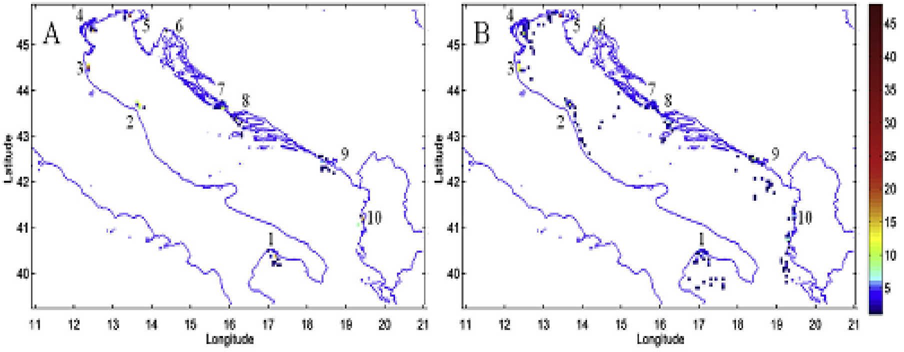

Fig. 4. Grid count densities of destination points of larval trajectories for each grid cell closest to each major Adriatic port for A) four and B) 20 day larval dispersal. Numbers indicate port locations: 1 = Taranto, 2 = Ancona, 3 = Ravenna, 4 = Venice, 5 = Trieste, 6 = Rijeka, 7 = Sibenik, 8 = Split, 9 = Tivat, 10 = Durres. Count densities are not defined by their release points. All PLD simulations produced similar patterns, albeit increasing dispersion with increasing larval duration so for convenience only the minimum and maximum larval durations are displayed Counts measured in absolute terms.

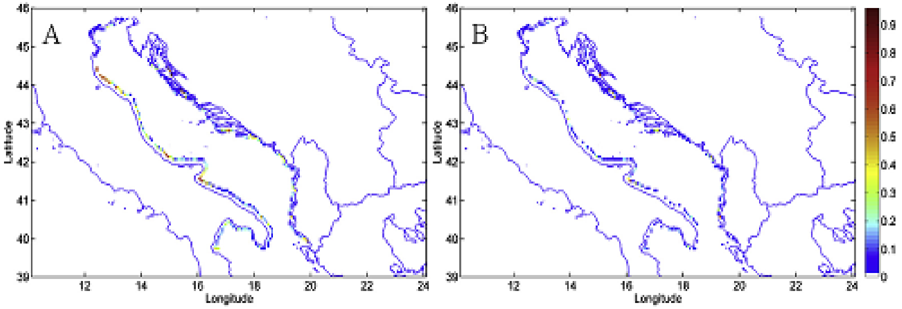

Fig. 5. Percentage of self-replenishment for each release grid cell for A) four and B) 20 day larval durations. All PLD simulations produced similar patterns, albeit increasing dispersion with increasing larval duration so for convenience only the minimum, and maximum, larval durations are displayed. Increasing the larval duration means the self-replenishment of most release grids along the Eastern coast of the Adriatic approaches 0%.

et al., 2016), predator-prey interactions, environmental cues, and suitable substratum availability for settlement (Chan and Walker, 1998). The homogenous release of passive particles along the Adriatic coastline does not accurately reflect nature but it does provide an insight into larval dispersal over large scales at an ecosystem level, and is a useful starting point for marine spatial planners.

4.2. Particle transport

The distance larval particles were transported was shorter than other works which assessed dispersal distances in the region (Melià et al., 2016) as they used longer pelagic larval durations. Some of the most prolific biofoulers of the region (balanoid barnacles, serpulid worms, and ascidians) have short pelagic larval durations ranging from several hours to up to three weeks (Anil et al., 1995; Chan and Walker, 1998; Jacobs et al., 2006). The limited dispersal potential reflected within the 4, 8, 16 and 20 day simulations in comparison to the typical pelagic fish connectivity modelling of the Mediterranean (approximately 30 days) highlights the need for taxon-specific connectivity analyses.

Regarding the spatial dispersal of larval particles, there are several persistent larval sinks along the southern Italian shore, corroborating previous findings in the region (Dubois et al., 2016). The shelf area along the Western coast of Italy, consistently had high larval densities in our simulations due to the hydrographic influence of the River Po (Orlic et al., 1992). During winter, the river output is confined to the Northern basin but during the spring/summer spawning season the Mid Western Adriatic current, and the South Western Adriatic currents, transverse the entire Western coastline of Italy (Artegiani et al., 1997) (Fig. 1). Offshore structures constructed along the southern Italian shores are likely to be much more exposed to larval settlement than other locations. Similarly, other regions that indicate relatively high self-replenishment and larval densities are found within the Kravner Gulf. The convoluted coastline of the Croatian archipelago clearly plays a large role in transportation of larval particles within the region.

Dispersal of simulated larvae that originate from the major ports of the Adriatic, congregate in high concentrations throughout the Northern basin, largely due to the close proximity of the port of Ravenna, the port of Venice, and the port of Trieste (Fig. 1). Multiple studies have shown higher abundances of alien species at several Adriatic ports (David et al., 2007; Iveša et al., 2015); likely due to direct transportation from fouling/ballast water or indirectly via the colonization of artificial substratum. The invasive barnacle *Amphibalanus improvises* has been recorded at the Rovinj port in Croatia (Pecarevic et al., 2013). Despite it's fairly limited pelagic larval duration of 5–20 days (Anil et al., 1995), its high reproductive capacity and rapid establishment on both natural and artificial substrates has caused it to be classified as one of the worst invasive species in Europe (Vilà et al., 2009). The high numbers of alien macroinvertebrates in the region (Zenetos et al., 2012), and the disproportionate advantage they often have in colonizing artificial substrata, means that offshore wind farms may create corridors for

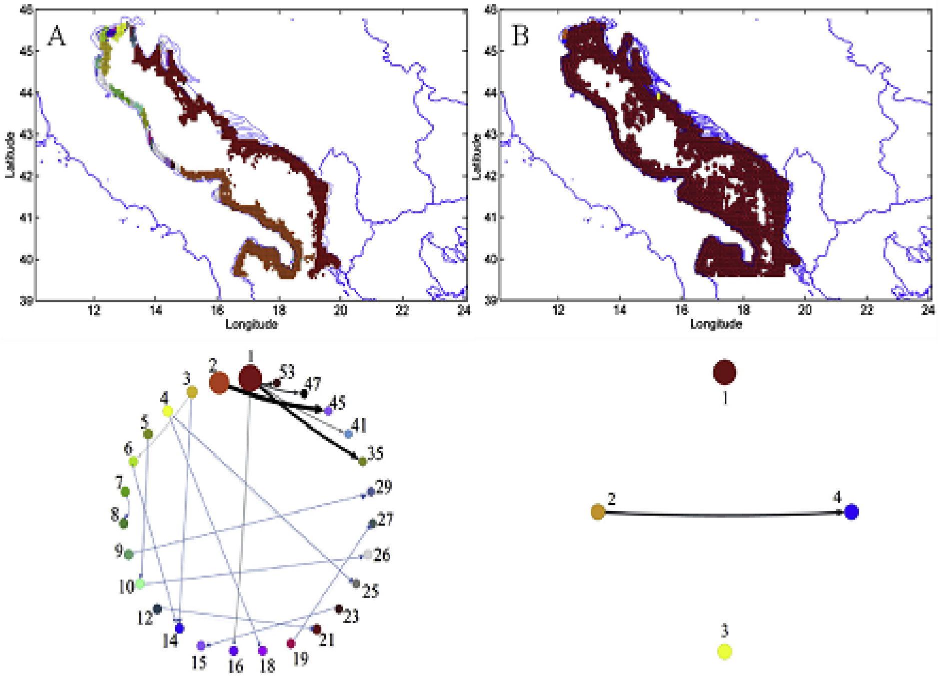

Fig. 6. Community outputs from mapequation algorithm displayed spatially A) four and B) 20 day larval durations. Relative strength of connection, and thus thickness of arrows, between clusters is automatically calculated by the *Infomap* software and is presented here as purely indicative. All PLD simulations produced similar patterns, albeit increasing dispersion with increasing larval duration, so for convenience only the minimum and maximum larval durations are displayed.

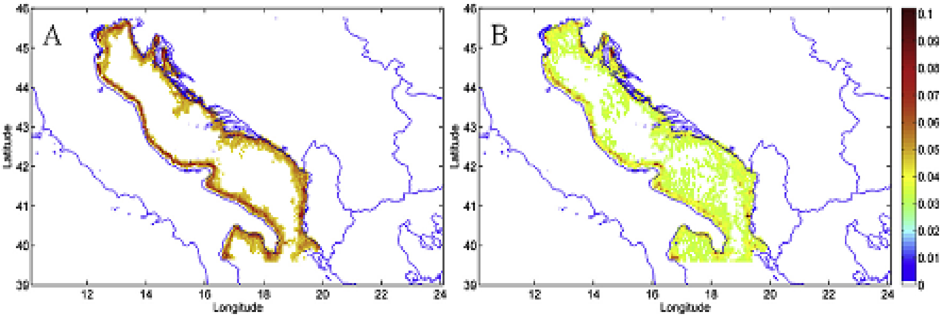

Fig. 7. Spatial display of PageRank for each grid cell included within the network for A) four and B) 20 day larval durations. No distinction made between grid cells given a PageRank value of zero and grid cells not assigned a Pagerank value. The scale indicates the probability distribution as numeric values ranging between 0 and 0.1. All PLD simulations produced similar patterns, albeit increasing dispersion with increasing larval duration, so for convenience only the minimum and maximum larval durations are displayed.

alien species invasions (Airoldi et al., 2015). Information regarding the likely destination of larval particles originating from ports and marinas in the Adriatic may assist marine spatial planners looking to reduce the spread of invasive non-indigenous species in the region; however in areas like the Northern basin, high densities of existing ports and infrastructures may mean the colonization of alien species on offshore structures is unavoidable.

4.3. Node centrality

Our principal result was the production of benthic invertebrate 'connectivity' map in the Adriatic. Grid cell centrality i.e. PageRank, is a good way of estimating how connected this cell is with the rest of the grid cells within the network. This measure can be important when spatially planning the position of offshore artificial structures. The potential for offshore structures to act as stepping stones by providing a suitable habitat for colonization in areas outside of the typical range extension of a species is already documented (Adams et al., 2014), and can have both local and regional impacts on the maintenance of local biodiversity within marine ecosystems (Dafforn et al., 2015). On average, grid cells had low connectivity for all PLDs, particularly in offshore regions and the Po river delta; there were however, several regions of high importance within the network which included the Port of Rijeka, Italian Adriatic coast, and South of the river Durres. This information presented here will be important when deciding if offshore activities should be designed to increase, or decrease, benthic community connectivity. Of the connected grid cells the vast majority (>90% of cells with pelagic duration more than four days) involved in the coastline-release network were part of one cluster, indicating that although connectivity of grid cells is relatively low, there is potential for interconnection throughout the whole Adriatic.

Connectedness of regions, particularly regions outside of marine protected areas, is an often-ignored aspect of marine spatial planning, but with the further development of offshore activities in the area and the likely impacts this expansion will have on marine biodiversity it should be an important consideration for regional marine spatial planners. The approach presented here is a pragmatic tool for identifying connectivity systems of benthic communities within a semi-closed system which can be expanded with in–situ data regarding the placement of offshore structures and habitat ranges of key benthic species. Identifying regions of relatively higher connectivity within the region i.e. the Italian Adriatic coast, south of the river Durres and port of Rijeka, is a useful starting point for providing information towards an intergraded management approach of the Adriatic Sea.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.marenvres.2017.01.006.

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Expected Effects of Offshore Wind Farms on Mediterranean Marine Life

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Abstract: Current climate policy and issues of energy security mean wind farms are being built at an increasing rate to meet energy demand. As wind farm development is very likely in the Mediterranean Sea, we provide an assessment of the offshore wind potential and identify expected biological effects of such developments in the region. We break new ground here by identifying potential offshore wind farm (OWF) "hotspots" in the Mediterranean. Using lessons learned in Northern Europe, and small-scale experiments in the Mediterranean, we identify sensitive species and habitats that will likely be influenced by OWFs in both these hotspot areas and at a basin level. This information will be valuable to guide policy governing OWF development and will inform the industry as and when environmental impact assessments are required for the Mediterranean Sea.

Keywords: renewable energy; Mediterranean Sea; ecological impacts; marine energy

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1. Introduction

The global demand for energy supply continues to increase rapidly [1]. Accelerated demographic and economic growth [2], modifications in energy usage as a result of climate change [3], and rising demands for rural electrification in many Middle East and North Africa (MENA) countries [4] have dramatically increased the energy demands of the Mediterranean region, a trend that is set to continue. Consequently, problems concerning the security of energy supply, and the impact of global warming and ocean acidification as a result of CO_2 emissions, have stimulated research and development into environmentally sustainable energy. This drive is reflected in the Horizons 2020 EU Renewables Directive (2009/28/EC), with member states required to obtain 20% of their energy consumption from renewable energy sources by 2020 [5]. Non-EU Mediterranean countries have also recognized the need to decrease reliance on hydrocarbons and most have adopted similar policies [4].

Europe is seeing a rapid expansion of the wind energy sector on land; however, higher mean winds speeds due to a reduction in offshore surface roughness [6], and comparatively lower visual and noise pollution than onshore wind farms [7], has led to a recent expansion of marine wind farms with further planned developments particularly within the North Sea and Baltic regions [8]. As of 1 July 2014, the EU had a combined offshore capacity of 7343 MW, with a further 30,000 MW expected by 2020 [5,9]. Currently, the Mediterranean Sea has no operational offshore wind farms (OWFs) yet this is expected to go ahead imminently [10].

The environmental effects of OWF construction in the Mediterranean are as yet unknown. The Mediterranean has particular characteristics including minimal tidal ranges, high levels of biodiversity and endemism [11], and a high potential for range extension of alien species [12]. The region is also exposed to a suite of coastal pressures including pollution, busy shipping lanes, eutrophication, urban development, habitat degradation, and overfishing [13]. The effects of existing OWFs may not be directly applicable to the Mediterranean, highlighting the need for site-specific analyses before the commencement of large scale offshore developments. The aim of this paper is to systematically assess the biological effects of existing OWFs in Northern European Seas and consider those in relation to the unique conditions of the Mediterranean basin, to horizon-scan for the potential environmental effects and solutions if construction goes ahead.

2. Methods

The most important technical criteria for the identification of a suitable OWF sites are wind resource availability and bottom depth. Evidently, for a rational candidate site identification, additional technical criteria should be also considered, such as distance from shore, bottom morphology and type of sediments, electrical grid infrastructure, *etc.*; however, the most important criteria are wind resource availability and bottom depth.

The wind speed threshold levels and the depth criteria were set in accordance with the EEA (2009) recommendations regarding the economic vitality and the distance for the minimum optical nuisance requirements of the OWFs, respectively. Since the current fixed-bottom wind turbine technology (monopile, gravity-based, jacket and tripod foundations) is limited to water depths up to 50 m, the depth range considered herewith is 20–50 m, and the lower threshold for the mean annual wind speed at 80 m above mean sea level was set to 5 ms^{-1} [6]. According to the above restrictions, using 10-year results (1995–2004) obtained from the Eta-Skiron model: [14–16] and the General Bathymetry Chart of the Oceans global relief [16,17], potential wind energy sites (model grid points) were identified, while regions with high densities of such point locations were highlighted as offshore wind energy hotspots. The Eta-Skiron mesoscale meteorological model is a modified version of the non-hydrostatic Eta model and is used for the dynamical downscaling of the ECMWF Era-40 reanalysis data [17] and the ECMWF operational forecasts, with a fine spatial (0.10° × 0.10°) and temporal resolution (3 h) [16]. An evaluation of the Eta-Skiron model performance as regards the wind power density estimation for the Mediterranean Sea is presented in Soukissian, Papadopoulos [18].

Biological effects resulting from the construction, and operation, of OWFs were identified in a review of studies in Northern European Seas. Peer reviewed literature took precedence and primary literature was obtained from several databases including CAB abstracts, Google Scholar, Web of Science, Science Direct, and Scopus. Relevant grey literature was also included in the compilation of information, and expert opinions were sought from several research institutes and industry experts (references herein). For clarity, impacts were separated via taxa (e.g., birds, marine mammals, fish, benthos and plankton). Detailed evidence obtained via the literature review is presented in a table format within the supplementary information (Table S1–S6); where available, specific evidence regarding the impacts at OWF hotspots will be highlighted.

3. Results and Discussion

Although many Mediterranean coastlines seem poorly suited to OWF development, some large areas have exploitable potential, including the coasts of the Gulf of Lyons, the North Adriatic Sea, the entire coastal area of the Gulfs of Hammamet and Gabès in Tunisia, off the Nile River Delta, and the Gulf of Sidra in Libya (Figure 1). The five hotspots spatially cover the width and breadth of the

Mediterranean Sea. Here, we consider the potential effects on birds, marine mammals, fish, benthos and plankton throughout the Mediterranean and, where available, the possible impacts of OWFs within the specific hotspot regions.

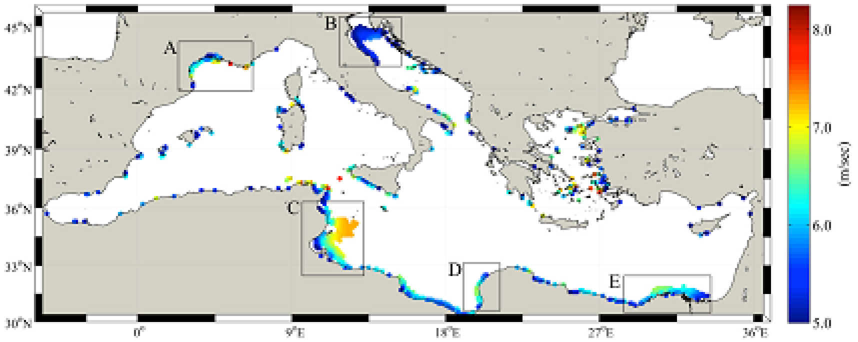

Figure 1. Locations which satisfy offshore wind development requirements of 20–50 m depth and an annual wind speed >5 ms^{-1} at a height of 80 m above sea level. A: Gulf of Lion, B: North Adriatic Sea, C: Gulfs of Hammamet and Gabès, D: Gulf of Sidra, E: Nile Delta.

3.1. Potential Effects of Mediterranean Offshore Wind Farms on Birds

Wind farms affect resident and migrating birds, through avoidance behaviors, habitat displacement, and collision mortality, but such impacts are difficult to monitor offshore [19]. Seabirds that use the marine environment for foraging or resting may be displaced by OWFs [20]. The Mediterranean has a low diversity of seabirds, but these species tend to be long-lived with low fecundity, traits that often make species vulnerable to abrupt environmental change [11,20] (Table 1). Fortunately, most Mediterranean marine birds are listed as "least concern" on the IUCN red list, although the Audouin's gull is listed as "near threatened," the Yelkouan shearwater as "vulnerable," and the Balearic shearwater as "critically endangered" [21]. All 16 Mediterranean countries have made commitments to protect these species at a national level [22]. With the exception of shearwaters [23], Mediterranean seabird populations appear to be increasing, particularly the yellow-legged gull [11,24,25]. These increases have been attributed to fish discards and improvements in coastal conservation [26–28], but changes to fishery discard practices following the reform of the Common Fisheries Policy may reverse this [29].

Studies of Northern European seabird populations have developed vulnerability indices to indicate seabirds likely to be affected by the presence of OWFs [30–32]. Using parameters that are heavily weighted on the risks of collision mortality (flight altitude, flight manoeuvrability, percentage of flight time, nocturnal flight altitude, disturbance by wind farm structures, ship and helicopter traffic, and habitat specialization), the North/Baltic Sea-based studies assessed 18 of the 29 Mediterranean seabirds. Notable exclusions to the list are the endemic species of the Mediterranean, which pose a greater conservation risk due to their small population sizes [33]. Garthe and Hüppop [13] identify the Black and Red-throated diver, the Sandwich tern, and the great Cormorant as the most sensitive of the Mediterranean seabirds within their index, and rated the Black-legged kittiwake, and the Black-headed gull as the least sensitive when all parameters were combined. Advancing this approach, Furness *et al.* [31] separated the hazards due to collision risk and habitat displacement. They identified the lesser black-backed gull and the Northern gannet as marine species sensitive to collision risk, and both the red and black-necked divers as most susceptible to long-term habitat displacement. These approaches lack any evaluation of species-specific OWF avoidance behavior and thus have large caveats attached to their findings. The approach of identifying at risk species via vulnerability indices is useful for the planning stages of OWFs; however, it does not determine if construction will have a detectable change in seabird population trends. Focus should preferably be given to understanding any direct effects OWFs will have on foraging success, e.g., diving behavior and prey characteristics, which in turn will impact reproductive success, juvenile survival and population trends [20].

Table 1. Mediterranean seabird sensitivity assessments highlighting most and least vulnerable species according to index. Y = Yes, N = No, Red = 10% most vulnerable Mediterranean species within index, Blue = 10% least vulnerable species within index, "-" = Index not applied.

Common Name	Species	Endemic	Listed under Barcelona Convention	Wind Farm Sensitivity Index [13]	Vulnerability Index for Collision Impacts [30]	Vulnerability Index for Disturbance Impacts [30]
Cory's Shearwater (Mediterranean)	*Calonectris diomedea diomedea*	Y	Y	-	-	-
Yelkouan Shearwater (Mediterranean)	*Puffinus yelkouan*	Y	N	-	-	-
Balearic Shearwater	*Puffinus mauretanicus*	Y	N	-	-	-
European Shag (Mediterranean)	*Phalacrocorax aristotelis desmarestii*	Y	Y	-	150	14
Great Comorant	*Phalacrocorax carbo*	N	N	23.3	-	-
Pygmy Comorant	*Phalacrocorax pygmeus*	N	N	-	-	-
Audouin's gull	*Larus audouinii*	N	Y	-	-	-
Little Gull	*Hydrcoleus minutus*	N	N	12.8	-	-
Lesser black-backed gull	*Larus fuscus*	N	N	13.8	960	3
Slender billed gull	*Larus genei*	N	Y	-	-	-
Mediterranean gull	*Larus melanocephalus*	N	Y	-	-	-
Black-headed gull	*Larus ridibundus*	N	N	7.5	-	-
Caspian gull	*Larus cachinnans*	N	N	-	-	-
Black legged kittiwake	*Rissa tridactyla*	N	N	7.5		-
Yellow legged gull	*Larus michahellis*	N	N	-	-	-
Great skua	*Catharacta skua*	N	N	-	320	3
Caspian tern	*Hydroprogne caspia*	N	N	-	-	-
Common tern	*Sterna hirundo*	N	N	15.0	229	8
Little tern	*Sterna albifrons*	N	N	-	212	10
Sandwich tern	*Sterna sandvicensis*	N	N	25.0	245	9
Lesser-crested tern	*Thalasseus bengalensis*	N	N	-	-	
Razorbill	*Alca torda*	N	N	15.8	32	14
Atlantic puffin	*Fratercula arctica*	N	N	15.0	27	10
European Storm petrel	*Hydrobates pelagic melitensis*	Y	Y	-	91	2
Northern gannet	*Morus bassanus*	N	N	-	-	-
Osprey	*Pandion haliaetus*	N	Y	-	-	-
Eleanore's falcon	*Falco eleonorae*	N	Y	-	-	-
Red throated diver	*Gavia stellata*	N	N	43.3	213	32
Black throated diver	*Gavia arctica*	N	N	40.3	240	32
Great crested grebe	*Podicep scristatus*	N	N	19.3	84	8
Red-necked grebe	*Podiceps grisegena*	N	N	18.7	-	-
Eared grebe	*Podiceps nigricollis*	N	N	-	-	-

Threats to Mediterranean bird populations are also directed towards migratory species. Worldwide, migratory species are declining in greater numbers than resident populations [34], and the Mediterranean basin is a major transit route for Saharan-Eurasian migration, as evidenced by both the Mediterranean-Black Sea flyway and the Adriatic flyway [35,36]. Many long-distance bird migrants, e.g., raptors and storks, rely on land lift via thermal upwelling for long-distance flight [37,38] and avoid broad fronts such as the Mediterranean Sea and the Saharan desert [37], creating bottlenecks at narrow passages of the Mediterranean Sea, including Gibraltar, the Straits of Sicily, Messina (Italy) and the Belen pass (Turkey) [39]. Wetlands around the Mediterranean provide suitable stopover sites for long-distance migrants to feed, rest and molt [40]. Some of the main wetlands around the Mediterranean are located within close proximity of potential OWF hotspots, particularly the Po Delta in the Northern Adriatic Sea the Nile Delta, the Gabès Delta and the Camargue Delta in the Gulf of Lion (Figure 2). Due to the bathymetry of the Mediterranean, and the steep continental slope of most coastlines, deltas provide feasible sites for wind farm constructions. High densities of avian habitat use in these regions means that OWF resource overlap will be a key factor in Mediterranean marine spatial planning in regard to OWFs.

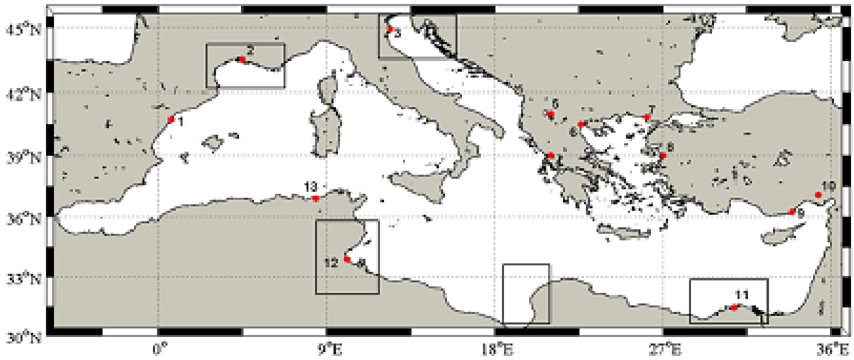

Figure 2. Main Mediterranean wetlands and overlapping OWF potential hotspot areas (adapted from [41]. 1. Ebro Delta; 2. Camargue Delta; 3. Po Delta; 4. Amvrakikos Gulf; 5. Prespa Basin; 6. Aliakmonas Delta; 7. Evros Delta; 8. Gediz Delta; 9. Göksu Delta; 10. Seyhan Delta; 11. Nile Delta; 12. Gabès Delta; 13. El Kala.

High collision levels of migrating terrestrial birds at well-lit observing platform during periods of bad weather and poor visibility [42] indicate that wind farms located near the coast, or prominent migration bottlenecks, may pose a significant risk to migrating birds. More recent evidence also shows alternative crossing options for some passerine species, including non-stop crossings over the Mediterranean Sea [43]. This indicates that species-specific migrations are not fixed either temporally or spatially, and individual route decisions are due to risk analysis of many parameters including energy reserves, weather conditions, and genetic disposition [44,45]. Until large-scale migration routes across the Mediterranean Sea are better understood, developers face large difficulties in wind farm spatial planning in the region. Obtaining this information is an essential task for potential OWF developers in the Mediterranean.

Aside from identifying crucial regions for migrating birds, one of the most poorly understood aspects about OWF effects on birds is avoidance behavior. Turbine avoidance tactics employed by a species may apply to both resident seabirds and long-distance migrants; however, any changes to migratory routes are extremely difficult to monitor and may have large, indirect effects [42]. Avoidance behavior is possible at several scales, which are typically classified into micro, meso, and macro strategies. Micro-avoidance is the behavioral response to actively avoid rotating blades. Meso-avoidance is classified as behavior whereby species that fly at rotor height within the wind farm and avoid the whole rotor swept zone and macro-avoidance being the behavioral alteration of a flight path due to the presence of a wind farm [46]. Macro-avoidance behavior strategies have been shown in some migrating individuals: The common eider *Somateria mollissima*, for example, exhibited avoidance behaviors of a wind turbine at a range of up to 500 m during the day [47]. The long-term consequences of employing avoidance techniques remain unclear. Among other parameters, real impacts to population trends of migrating birds will be highly dependent on the specific life histories of a species, expenditure of avoidance strategies, energy reserves, and weather conditions during migratory periods.

There are several possible measures to reduce the effects wind farms will have on Mediterranean avian populations, e.g., shutdown orders and changes to the phototaxis level of structures [48,49]. However, preventative initiatives are much more effective, *i.e.* ensuring planning and placement of OWFs are not in the vicinity of large population of species that have been identified as high risk within the Mediterranean. The sensitive species suggested here due to collision or habitat vulnerability include the lesser black-backed gull, the Northern gannet, and the red- and black-throated divers, while the endemic bird species and species whose populations are declining in recent decades (Shearwaters) are identified due to their conservation importance (Table 1). More understanding of the cumulative effects of all impacts, at all potential development sites, is needed. Until then, all future approaches in regard to OWF spatial planning in the Mediterranean should be of a cautionary nature.

3.2. Potential Effects of Mediterranean Marine Mammals

Marine mammals are often high profile, charismatic species and have the potential for high socio-economic value in their natural habitats [50]. It is therefore essential to understand the effects OWFs will have on Mediterranean populations. The Mediterranean Sea is home to both resident and visiting marine mammals, of which most are experiencing a decline in population trends, with the exception of visiting humpback whales whose numbers have appeared to increase [11,51]. At a basin level, total population numbers are difficult to assess with several species being classified as "data deficient" by the IUCN red list [20]. Nonetheless certain regions have been identified as important habitats for marine mammals. Monitoring programs show a high percentage of fin whale sightings within the Ligurian Sea in comparison with other regions of the Mediterranean Sea [52]. The Alborean Sea has been shown to be an important for long-finned pilot whale populations [53], and there is also evidence that due to the East-West basin migration of Sperm whales, either the Strait of Sicily, or the Strait of Messina are critical areas which enable migration [54].

In regard to OWF development, several resident marine mammals frequently use the coastal marine environment earmarked for potential developments including the critically endangered Mediterranean monk seal, the common Bottlenose dolphin, and visiting Humpback whales [51,55,56]. When assessing the combined species density of the resident marine mammals, the Gulf of Lion OWF hotspot displays the highest densities of resident marine mammals and as such can be considered as the most sensitive in regard to OWF development. The Gulfs of Hammamet and Gabès, the Gulf of Sidra, and the Nile Delta hotspots appear to support low populations of resident marine mammals (Figure 3).

The distribution of specific species of marine mammals is also of interest to developers. This is particularly true within the Northern Adriatic OWF hotspot. The Mediterranean monk frequently uses the coast of Croatia (Figure 1B, [57]), and the Bottlenose dolphins regularly sighted from the coast of Trieste and Kvarneric (Figure 1B, [55]). Other important areas for individual species include the the coast of Senigallia, and the Gulf of Gabes for the Humpback whale (Figure 1A,C) [51]. These regions will require particular attention during spatial planning stages of developers.

Through monitoring programs conducted in the Northern European seas, marine noise, and in particular pile driving during construction, has been identified as the biggest impact to marine mammals [58]. Increased motorized vessel shipping during the operational phase of wind farms also increases noise levels to the area, and so is also identified as an impact; however, this is not at a level expected to significantly affect marine mammals [59]. Depending on the hearing ranges of the species, pile driven construction has the ability to produce hearing impairment, although for most species, hearing reactions are as yet undetermined [60].

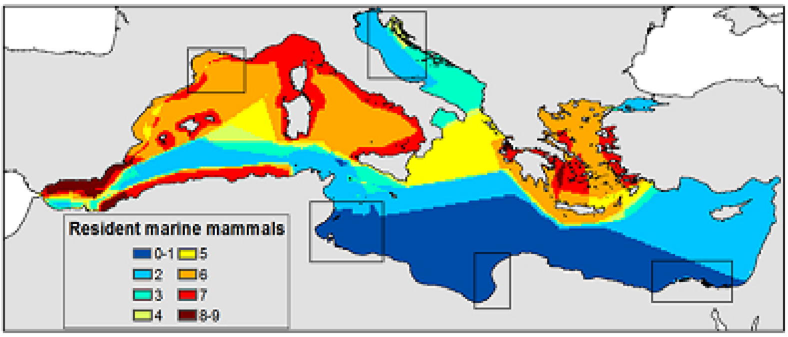

Figure 3. Species density of all resident marine mammals (n = 9) with overlay of OWF hotspots (Image adapted from Coll *et al.* [11]).

A study measuring the propagation of sound during the construction phase of an offshore site in the NE of Scotland implied Bottlenose dolphins would suffer auditory injury within a 100 m range of the site and behavior disturbances up to 50 km away [61]. With the use of T-POD porpoise detectors, acoustic monitoring during the construction and into the operational phase of the Nysted OWF indicated a possible change in habitat use by the harbor porpoise (*Phocoena phocoena*), with a reduction in the level of echolocation signals produced by the porpoises [62]. A long-term study (10 years) at the same wind farm also showed a decline from baseline levels of echolocation signals [63].

A similar study at the Dutch wind farm, Egmond aan Zee, after construction measured significantly higher acoustic activity inside the farm in comparison with a control site [64], and this trend was mirrored in a recent study of harbor seal (*Phoca vitulina*) foraging which suggested an increase in habitat utilization at two operational wind farms (Alpha Ventus and Sheringham Shoal) [65]. The repeated grid-like movements indicated for the first time, successful foraging behavior by an apex predator within an OWF. The apparent differences between probable habitat uses may be due to local-scale ecological differences, local population habituation of wind farms, or differences in construction type of wind farms [64]. Due to critical population levels in the Mediterranean, the observed increases in seal foraging behavior around wind farms may not be relevant to the Mediterranean monk seal [56,65].

In regard to the impacts of noise levels in the Mediterranean, the semi-enclosed Mediterranean also suffers from some of the highest volume of shipping routes in the world [66] (Figure 4). In general, noise from wind farms is influenced by water depth, wind speed, turbine type, wind farm size, and substratum type [67]; due to the high levels of existing background noise from maritime traffic in the Mediterranean, risks having a cumulative effect in masking communicative abilities of marine species [68]. When assessing the spatial density of traffic routes from 2013, the OWF hotspots of the Gulf of Lion, the North Adriatic Sea, and the Nile Delta show an already high density of vessels within the area (up to 140 m vessels$\cdot$ km$^{-2}\cdot$ day^{-1}); thus, high levels of background noise can be expected in these regions. The Gulf of Hammamet and Gabès, and the Gulf of Sidra suggest much lower levels of background noise stress. The use of underwater noise propagation models by policy makers will be required to understand the combined influence of OWF construction, operation and maintenance shipping with current levels of background noise at site-specific locations.

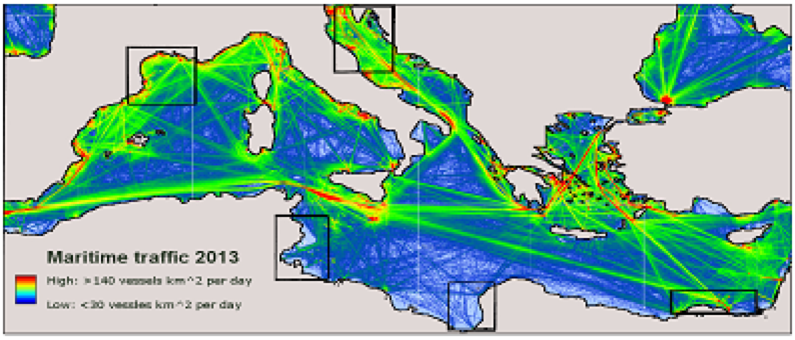

Figure 4. Combined density of 2013 maritime routes of all commercial vessels obtained via AIS vessel monitoring with overlay of OWF hotspots (source: www.marinetraffic.com).

3.3. *Potential Effects on Mediterranean Fish Communities*

Mediterranean coastal communities depend on fishing-related activities, particularly artisanal fishing, throughout the basin [69]. Of the 513 species and 6 subspecies of fish in the Mediterranean, 8% are currently classified as threatened by the IUCN [70], and there has been an alarming decline of Mediterranean fish stocks, with the largest declines in demersal fish stocks [71].

The principal impacts to fish populations caused by wind farms are noise, electro-magnetic fields, and novel habitat gain. Recent studies have shown that the noise generated by pile driving during the construction phase of OWF farms can generate acute stress responses in juvenile fish species. Although the responses were recorded as acute, it is possible that repeated and prolonged exposure in the wild may lead to a decrease in fitness [72]. During the operational stages of OWFs, evidence indicates fish permanently avoid wind turbines only at a range of up to 4 m under high wind speeds (13 ms^{-1}), and that their ability to communicate and utilize orientation signals is masked [67]. Increased background noise and seabed vibration from operational OWFs and associated marine traffic will also influence fish detection distances [73,74] (Figure 4). Greater numbers of experimental studies on individual fish species are needed before we can fully comprehend the impact of anthropogenic noise on fish [75].

Electromagnetic fields occur around intra-turbine, array-to-transformer and transformer-to-shore cables. The electro-sensitivity of many marine species is unknown, and there is a dearth of peer-reviewed information regarding the effects of electro-magnetic fields. Elasmobranchs are thought to be especially sensitive, due to their electro-sensory organs [76]. Several shark and ray species react to wind farm cables [77], but whether this has any affect at the population level is unknown. Magnetic fields could influence geomagnetic patterns used by some migratory marine species for navigation [78], and reports also show that electro-magnetic fields from OWFs may affect fish migration. Gill *et al.* [79] identified eight migratory fish species sensitive to electromagnetic fields, including the European eel *Anguilla anguilla*, the Atlantic salmon *Salmo salar* and the Yellowfin tuna *Thunnus albacares*. Limited *in situ* data are available about the actual effects on fish, which rely on magnetic fields for migration; however, several studies in the Baltic Sea have indicated that the swimming speed of European eels is reduced in the vicinity of underwater electric cables [78,80]. However, due to the limited availability of empirical studies, it is difficult to theorize the extent of the impacts that electro-magnetic fields have on Mediterranean marine species and their fecundity.

The most direct influence on fisheries due to the presence of OWFs is likely the addition of novel, vertical habitat to an area previously void of hard substratum. Numerous studies have found greater abundances in fish around OWFs than in surrounding areas, such as Atlantic cod *Gadus morhua*, pouting *Trisopterus luscus* and several species of gobies [81–83]. Currently, most empirically measured effects due to operational wind turbines are temporally limited, and not at the scale of OWFs [60]. Several studies have implied that the new habitat provides increased foraging for both primary and secondary food resources, and protection grounds from currents [84]. There is much discussion between ecologists over whether changes in species biomass will be due to attraction or production [85]. Stomach content analysis and energy profiling have shown that OWFs are suitable feeding grounds for both Atlantic cod and pouting species [86,87], indicating that there is extra biomass available at the sites. Juvenile recruitment of Atlantic cod has also been observed at wind farms in the Belgium part of the North Sea [87]. Changes in prey densities may also be masked by predation rates [65,88], and will potentially strengthen predator avoidance behaviors like diel migration, further complicating the relationship between attraction-production dynamics [77]. While the mechanics of fish abundance at OWFs is not yet fully understood and requires additional analysis, it is becoming increasingly obvious that any ecological benefits will only be attained if fishing practices are banned within the wind farms [89].

For Mediterranean fish species, it is difficult to state the effects based on the findings of Northern European studies as there is a limited availability of information, and the majority of existing monitoring programs focus on species that are not generally present in Mediterranean waters, e.g., Atlantic cod [82,85,88]. That being said, there is evidence that suggests a yield increase of fish populations at wind farms as opposed to the simple attraction model previously hypothesized [85,86]. This is of particular importance for the Mediterranean, as levels of fishing are unsustainable, and most fish stocks are in decline [71]. The possibility for creating *de facto* marine protected areas (MPAs) due to fishing restrictions imposed within OWFs is an interesting aspect in the developments of OWFs in the Mediterranean Sea. It is clear that well protected MPAs in the Mediterranean result in significantly higher biomass than those with no or minimal protection [90], although many Mediterranean MPAs lack adequate protection [91]. Enforcement of fishing restrictions in Mediterranean MPAs is a difficult issue, but the benefit associated with designating an area within a series of fixed structures is that fishing regulations may be easier to enforce. Benthic fishermen are less willing to drag their trawling gear within turbines as they risk entanglement, and there is a potential to monitor fishing activity of static and recreational fishermen by using fixed cameras. It is worth noting, however, that displacement of fishing effort is a serious concern for the management of reduced fishing effort regions [92]. The production of sound by many fish species for communication during spawning periods means it is also essential for policy makers to identify fish spawning grounds during environmental impact assessments, with the aim of restricting OWF construction in these noise-sensitive areas [93].

3.4. Potential Effects of Mediterranean Benthic Communities

The Mediterranean harbors many important benthic habitats including vermetid reefs, coralligenic concentrations, shallow sublittoral rock, seamounts, deep-sea coral reefs, and abyssal plains [94,95]. The shallow sub-littoral sediment is a particularly valuable habitat for the Mediterranean benthos, as it is the preferred habitat of the endemic seagrass *Posidonia oceanica*, listed as a priority natural habitat under Annex 1 of the EC Habitats Directive (2/43/EEC on the Conservation of Natural Habitats and of Wild Fauna and Flora), due to its endemism, high productivity and provision of ecosystem services [96,97]. Favorable substrate conditions for OWF construction throughout Europe is typically soft sediment areas, and the piling and drilling of foundations and monopile jackets constitutes the most direct impact to the marine environment. As they favor similar habitats required for the construction of OWFs, *Posidonia* beds are at risk from direct physical destruction, sedimentation, and changes in hydrographic regimes. Conversely, construction may prevent local trawling and therefore decrease the amount of destructive fishing methods that typically reduces *Posidonia* coverage [98]. Any plans for OWFs in the Mediterranean Sea will have to be carefully designed around the distribution of *Posidonia* to ensure the correct conservation practices for this endemic, priority species.

Other impacts to soft sediment communities from pre-construction states include changes in regional current regimes, causing shifts in macro-benthic assemblages on a localized scale [99,100]. Studies have shown negative correlations between distance from turbines and grain sizes in the vicinity of turbines (measured from a distance from turbine of 15 m to 200 m) in the Belgian part of the North Sea, which were also positively correlated with increases in organic matter and a shift in species assemblages. The closer to the turbine the soft sediment community samples were taken, the greater the increase in macrobenthic density and diversity [99].

Although changes in soft-sediment in-fauna are likely to be associated with wind farm construction, the most direct affect is the addition of hard substratum to the environment. Ecosystem shifts occur from changes in the existing soft sediment shifts and the addition of hard substratum in a habitat previously void of available settlement substratum. Recruitment and colonization of novel artificial habitats provided by turbine foundations increases the structural complexity and productivity of an environment previously low in in-fauna diversity and density [101–110].

Research at an offshore research platform in the German Bight indicated that 35 times more macro-zoobenthos biomass was associated with the additional hard substratum than the equivalent area of soft benthic sediment [81]. The increase in macro-zoobenthos biomass may appear beneficial in regard to productivity, yet in many cases species assemblages associated with artificial structures differ from the environment replaced and show lower levels of diversity than natural rock equivalents [81,110]. Long-term effects of ecosystem shifts are unknown, and species assemblages are influenced by many parameters including material and texture of offshore structures, larval supply, oceanographic conditions, temperature, salinity and water depth [111]. The number of defining parameters involved in influencing the spatial and temporal colonization of offshore artificial structures highlights the need for extensive area-specific research and long-tem *in situ* experiments to fully understand regional implications of OWFs.

With regard to the Mediterranean, the limited investigative work into epibenthic colonization has focused on concrete artificial reefs [112–114] or rock anthropogenic structures [115,116]. Most studies areas focus on the Northwestern region of the Mediterranean with the exception of two studies in Turkey and Israel [117,118]. Only two studies have investigated an offshore steel structure in the Mediterranean [117,118]. Dominating species of epibenthic assemblages varied depending on the location and duration of monitoring program, which ranged from 11 months to 20 years. Most studies cited the initial establishment of Hydozoa, Bryozoa and Serpulidae [112,117–120]. In several studies, initial colonization was followed with the establishment and dominance of the commercially farmed *Mytilus galloprovincialis* [112,115,119,120]. However, the establishment of mussel beds on artificial structures in the Mediterranean may be highly localized, as several artificial structures showed no such dominance [117,121–123] or highly variable results [124]. The only long-term data set on a concrete artificial reef (20 years) reported five distinct phases of species assemblage: dominance of pioneer species, mussel dominance, mussel regression, mussels absence, and finally dominance of bryozoan bio-constructions [125]. Differences in the material used for offshore structures may have a significant effect on climax community composition; the two offshore, steel study sites in the Mediterranean offshore steel structures were both dominated by bivalves after 52 and 70 months [119,121].

The susceptibility of the Mediterranean Sea to non-indigenous species [126] and the colonization of artificial substrata in the Mediterranean by alien species [122,127] mean that wind farms may also act as benthic "stepping stones" that facilitate range extension of alien species within the Mediterranean marine environment, which in turn may potentially reduce the biodiversity of the basin [111,128]. Due to the apparent locality factor of benthic colonization communites, small-scale pilot studies are essential for understanding whether windfarms will proliferate alien species at potential wind farm locations. The use of before-after, control-impact studies by policy makers is strongly recommended.

3.5. *Potential Effects of Mediterranean Planktonic Communities*

The oxygen rich, oligotrophic waters of the semi-enclosed Mediterranean produce a low nutrient availability that is, however, generally intensified along both a west-east and north-south direction [129]. There is, nevertheless, a high spatial heterogeneity in the distribution of plankton throughout the Mediterranean Sea due to complex hydrodynamic circulation patterns forming multiple gyres and upwelling systems [130]. Most marine invertebrate and fish species have a planktonic larval stage; therefore, it is crucial to understand any effects OWFs may have on planktonic communities. There has been much speculation about the impacts caused by OWFs in relation to plankton [6,107,110]. Evidently, at some scale any offshore construction will have an effect on local hydrographic regimes [131]; however, the extent that this will affect upwelling/downwelling episodes, and thus potentially phytoplankton blooms, is currently unknown. Analytical models indicate that OWFs affect surface gravity waves underneath rotor blades [132,133], and there is much speculation in the literature as to the impacts any hydrographic changes will have on plankton/nekton aggregations [19]. OWFs may also affect planktonic populations by providing hard substrata that facilitate planktonic connectivity through larval settlement during dispersal processes [134–136]. The presence of available hard substratum from wind turbines for the recruitment and settlement of pelagic larvae have the potential to extend passive larval connectivity across biogeographic boundaries [134]. It is noted that there is a scarcity of information in the literature regarding the impacts OWFs will have on planktonic communities.

4. Conclusions

As the likelihood of Mediterranean OWFs increases, there is an ever-growing need to assess the biological costs and benefits of OWFs in the region. The aims of this horizon-scanning review were two-fold: firstly, to identify areas likely to be considered for the development of OWFs within the region, and, secondly, to identify the biological impacts of these developments.

The five OWF hotspots are identified as the Gulf of Lion, the North Adriatic Sea, the Gulfs of Hammamet and Gabès, the Gulf of Sidra, and the Nile Delta. Understanding the species, habitats and taxa that are likely to be affected by the construction and operation of OWFs in these regions and the wider Mediterranean region will assist developers and policy makers with future spatial planning decisions regarding OWFs within the Mediterranean. Furthermore, the advancement and implementation of floating wind turbine technology may diminish many of the above mentioned effects, which, from this point of view, is a very promising perspective for the near future.

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Author Contributions: L.B. conceived the investigation; E.V. and T.S. analyzed model output data; L.B., S.D., S.R., C.A., and M.V.-L. performed and analyzed data from the literature review; L.B., M.J.A., and J.M.H.-S. interpreted the results. L.B., M.J.A., and J.M.H.-S. wrote the paper.

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Greening offshore wind with the Smart Wind Chart evaluation tool

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Abstract. The development of offshore wind farms (OWFs) and the establishment of marine protected areas (MPAs) comprise two main elements for the production of clean energy, and the simultaneous maintenance and protection of biodiversity in the Mediterranean and Black seas. Successful, efficient, and sustainable coupling of these two aspects presumes that the criteria for selecting suitable locations for the deployment of OWFs should not only include technical-engineering terms (e.g. high wind energy efficiency, bottom suitability, inland infrastructures) but also ecological–environmental considerations (e.g. the least possible impact on biodiversity, ecosystem functioning) and socio-economic aspects (e.g. effects on coastal and marine activities, development of marine spatial planning). In the context of the FP7 CoCoNet project, the integration between OWFs and MPAs is based on four main steps: (i) the identification of existing (networks of) MPAs focusing on the biodiversity distribution patterns and current legislation, (ii) the coupling of offshore wind potential within networks of MPAs, (iii) the evaluation of the knowledge gained up to date and the theoretical approaches at the two pilot sites of the Mediterranean and Black sea basins, and (iv) the development of the "Smart Wind Chart", a convenient and rational tool addressed to scientists and policy makers for the evaluation of maritime policy management schemes. The latter step comprises the core of this work.

1 Introduction

The exploitation of wind energy has long been recognized as a key element in the common EU energy policy in order to reduce the dependence on fossil fuel and emissions from carbon sources, to decouple energy costs from oil prices, and to ensure a secure energy supply. Although onshore wind energy generation is currently cheaper than offshore, finding new suitable onshore sites is becoming increasingly difficult in Europe. Offshore wind energy (OWE) is an attractive alternative solution to take advantage of marine winds, which tend to be higher, more frequent, and less variable than inland winds. Despite the tremendous development of offshore wind farms (OWFs) in the northern European countries, with more than half of the installed offshore capacity belonging to

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the UK, no full-scale OWFs have been developed yet in the Mediterranean and Black seas.

The development of OWFs in these two basins seems to be an appealing and environmentally friendly pathway. According to Gaudiosi and Bori (2010), the total wind energy production (offshore and onshore) could cover 10 % of electricity demand of the Mediterranean countries by 2030. On the other hand, the development of OWFs alongside other offshore (and coastal) human activities and the imperative need for the conservation of marine habitats seem to be, at first sight, heterogeneous (and often conflicting) aspects. In this regard, it is not straightforward or evident how to harmonize these aspects into a single holistic framework. The relevant arguments that have been raised concerning the impacts of OWFs on the local biotic and abiotic elements (EEA, 2009) necessitate the synthesis of these seemingly opposing aspects into a single holistic framework, where networks of marine protected areas (MPAs) will form the most important marine environmental conservation and protection units.

The EU-funded coordinated project "Towards COast to COast NETworks of Marine Protected Areas (from the shore to the high and deep sea), coupled with sea-based wind energy potential" (CoCoNet) is focused inter alia on the fulfilment of the synthesis of a holistic framework related to the protection and connection of the ecologically important areas in the Mediterranean and Black seas along with the exploitation of the climate-friendly offshore wind power (see also http://www.coconet-fp7.eu/). The CoCoNet project aims firstly to identify interconnected MPA networks within the Mediterranean and Black seas at the local, regional, and basin scale. According to Article 13 of the Marine Strategy Framework Directive, coherent, representative, and interconnected networks of MPAs should represent an integrated system of multiple protected areas designed to conserve regional biodiversity and ecosystem function. In this respect, the approaches considered for the evaluation of the degree of connectivity at the spatial scale are currents, propagules, beta diversity analysis on communities, and genetic diversity. The second main aim of the CoCoNet project is linked to the exploitation of OWE and deals with the integration of geotechnical, socioeconomic, and environmental aspects into a geospatial tool in order to evaluate the feasibility of the installation of OWFs in the examined basins. The identification of the "overall" favourable OWF locations is important in order to secure the financial viability and sustainability of the offshore project, as well as to minimize the negative impacts and maximize the positive effects of OWF installations in the marine environment as a whole. The major outputs from the CoCoNet project are (i) a set of guidelines for the design, management, and monitoring of networks of MPAs and the sustainable development of OWE along with marine conservation goals, and (ii) the Smart Wind Chart (SWC).

Aiming our attention at the second major output, SWC represents a robust and unbiased favourable site identification approach that should be implemented for the integration of all the available information in order to illustrate potential sites for deploying offshore wind installations and to join the networks of MPAs. The visualizing aiding tool for this approach is based on Geographic Information System (GIS) environment. During recent years, due to its flexibility and available features, this spatial analysis tool has been extensively used by renewable energy developers in the context of multi-criteria decision analysis for OWF site selection procedure (see, for example, Atici et al., 2015). Moreover, there are recent studies evaluating OWE development through GIS tools, but only at a national level; for instance, there are assessment studies for Denmark (Möller et al., 2012), Greece (Vagiona and Karanikolas, 2012), Portugal (Costa et al., 2006), and the UK (Cavazzi and Dutton, 2016). Although there are numerous studies proposing multicriteria approaches for wind farm siting (mainly onshore), an integrated approach combining the aforementioned viewpoints with evaluation procedures is still missing for offshore wind applications with reference to an extended spatial scale, such as a large basin.

In this work, the procedure for the implementation of the SWC is presented and applied in the Mediterranean Sea (MS), focused on the corresponding pilot area located in the northern Ionian Sea, in order to formalize the route leading to the identification of favourable zones for OWF development. The comparability of potential locations for developing offshore wind projects is based on quantifiable multi-parameter technical criteria, which are considered of most importance in the OWE industry, combined with environmental restrictions.

The structure of this work is the following: in Sect. 2, the methodology developed in the context of the SWC analysis is presented in more detail. In Sect. 3, the input data, including technical and environmental factors, and the pilot study area in the MS are described at length, and the results obtained after the implementation of the SWC are presented in the subsequent section. In Sect. 5, there is a discussion with reference to some important aspects related to potential misinterpretations of the SWC results and some generic knowledge/information gaps that were encountered at the basin scale. In the last section, some concluding remarks and suggestions are provided for further research.

2 Methodology for the development of the Smart Wind Chart

A first step towards the development of a structured methodology, from which optimal OWF sites are determined, is the identification of the main key actors that are involved in the development of OWE projects and are characterized by different, and often contradictory, priorities and requirements. Some of the included key actors include government bodies, policy makers, financing mechanisms, scientists, local communities, NGOs, and wind industry. Seen in this con-

Table 1. Ranking score and corresponding weightings of potential "go" areas.

	Offshore wind speed (10 m a.s.l.)	Water depth	Distance from shore	Proximity to very large or large ports	Electrical grid infrastructure	Bottom sediments
Categorization (rank)	> 6.9 m s^{-1} (5)	10–40 m (5)	10–20 km (5)	0–100 km (5)	> 400 kV (5)	Sand (5)
	6.3–6.9 m s^{-1} (4)	–	5–10 km (4)	100–200 km (4)	225–400 kV (4)	–
	5.7–6.3 m s^{-1} (3)	70–200 m (3)	20–100 km (3)	200–300 km (3)	36–225 kV (3)	Mud (3)
	4.9–5.7 m s^{-1} (2)	–	0–5 km (2)	300–500 km (2)	< 36 kV (2)	–
	4.1–4.9 m s^{-1} (1)	40–70 m (1)	> 100 km (1)	> 500 km (1)	Distribution grid (1)	Rock (1)
Weighting (%)	35	25	15	5	15	5

text, a holistic approach for OWF development requires the rational interweaving of geotechnical/engineering, socioeconomic, and environmental aspects. The development of the SWC moves towards this direction.

SWC is a flexible tool for the comparison and evaluation of the potential OWF locations according to some quantifiable multi-parameter eligibility criteria. However, the locations evaluated through this tool should not be considered as direct suggestions for future OWF development, but merely as favourable candidate areas that deserve further in-depth assessment in the context of detailed feasibility studies. Consequently, the SWC should be regarded as a marine planning tool rather than a decision-making platform.

There are two major steps for the implementation of the SWC: (i) preparatory actions and (ii) processing phase (see also Fig. 1). Preparatory actions include the assessment of the most important quantifiable factors (technical criteria), the factor rating table, and the identification of "no-go/restricted" areas, i.e. areas that are either excluded from further consideration or restricted under specific conditions, mainly due to environmental considerations. The primary technical factors consist of mean annual wind speed and bottom depth whilst the additional factors that were considered include distance to shore, proximity to ports, electrical grid infrastructure, and type of bottom sediments. These factors are categorized in rating tables, and rankings from 1 (least feasible) to 5 (most feasible) are provided for each category. Then, each factor is assigned a weight corresponding to its relative importance on the feasibility of an OWF development. Thus, the overall score for each location is a combination between the factor rating table and the relative weights assigned to each factor (see Table 1). A simple linearly weighted methodology was adopted in order to keep the methodology as straightforward as possible and be easily adoptable and flexible according to the different demands and requirements of the involved key actors. Moreover, future alteration of costs, diversity of local conditions, and any progress in offshore wind technology can be easily integrated in the developed methodology.

In this approach, the exclusion/restriction of an area is primarily based on environmental restrictions, namely national protected sites/MPAs and Natura 2000 sites, areas characterized by meadows of the seagrass *Posidonia oceanica*, fields of the algae *Phyllophora crispa*, biogenic habitats such as coralligenous, marl, and deep sea coral formations. Let us note that national protected areas/MPAs and Natura 2000 sites may belong to either restricted or no-go areas; they can be definitively characterized as no-go areas after detailed in situ assessment. On the other hand, areas where the rest of the environmental features are met are characterized as no-go areas; the sensitivity and the vital role (at biological/ecological level) of such features in maintaining the marine coastal equilibrium and preserving marine biodiversity over the long term impose the need to prohibit any (harmful) human marine activity in the corresponding areas. For example, the Mediterranean endemic seagrass species *Posidonia oceanica* is considered among the natural habitats requiring conservation under the EU's Habitats and Water Framework Directive, since it is among the few representatives of biological quality elements in the MS waters due to its recognized ecological indicator possibilities (Lopez y Royo et al., 2011), while *Phyllophora* beds supply benthic primary production and water oxygenation in the circalittoral zone, and provide breeding and feeding grounds, and nursery for diverse invertebrate and fish species (Salomidi et al., 2012). Other restrictions may be attributed to marine and maritime uses (e.g. military exercise areas, areas of fisheries and aquaculture, shipping lanes, oil and gas extraction areas). However, such considerations are the main object of marine spatial planning and refer to site-specific studies concerning OWF development (see Sect. 5 for more details).

In the processing phase, the two most important (technical) parameters have been firstly taken into consideration for the primary identification of the site suitability: mean annual wind speed and bottom depth. The former is evaluated for the grid points that are not under any environmental restriction, and then the corresponding value for the latter one is extracted. Regarding mean annual wind speed, the lowest threshold was set to 4.1 m s^{-1} at 10 m a.s.l. (meters above sea level). This rather low limit is justifiable since Eta-SKIRON underestimates (sometimes significantly) wind speed with respect to satellite data and buoy measurements (see Soukissian and Papadopoulos, 2015). Regarding bottom depth, three different water depth ranges were considered: (i) 0–40 m ("shallow waters"), (ii) 40–70 m ("intermediate"

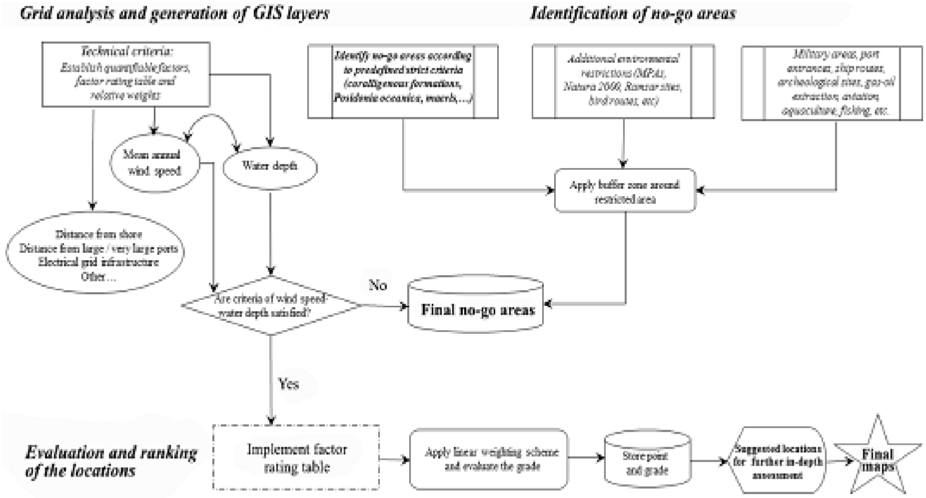

Figure 1. Flow chart for the development of the SWC.

or "transitional waters"), and (iii) 70–200 m ("deep waters"). Shallow and intermediate water depths refer to monopile, gravity-based, tripod, jacket, and tripile supporting structure, while deep waters refer to floating wind turbines technologies. Although fixed foundations are to date dominant in the offshore wind market, floating structures seem to be a viable alternative solution, especially for the Mediterranean waters, provided that the relevant floating technology reach the desired maturity level in the coming years.

If the combination of wind speed and bottom depth satisfies the adopted thresholds (i.e. mean annual wind speeds greater than 4.1 $\mathrm{m\,s^{-1}}$ at 10 m a.s.l. height and water depths smaller than 200 m), then the point (area) is characterized as "potentially go" area and is graded according to the examined factors and the relative weights, so that the final rankings of the locations are derived. At the end of the analysis, the potentially suitable sites worth being further assessed for OWF development are identified with the highest overall scores characterizing the most favourable sites. Let us note that any preliminary national spatial planning for OWFs is also included in the analysis. Such plans exist for France, Greece, Italy, and Spain; however, some of these plans are currently under revision.

3 Data and study area

3.1 Data sources

Wind data can be obtained from various sources that may use different measuring principles, devices, and configurations; these variations contribute to uncertainties in the long-term wind speed (and resource) assessment and should be properly considered (Soukissian and Papadopoulos, 2015). In this work, the results obtained from the Eta-SKIRON model (Papadopoulos et al., 2011) were used since they have the finest available spatial ($1/10^\circ \times 1/10^\circ$) and temporal (3 h) resolution. The simulation period is 15 years (1995–2009) with reference height at 10 m a.s.l., and the initial conditions were provided by the ERA-40 reanalysis data and the operational analyses of the European Centre for Medium-Range Weather Forecasts (ECMWF), through the Hellenic National Meteorological Service. The Eta-SKIRON model is based on the Eta-NCEP (National Centre for Environmental Prediction) model and was developed for operational purposes. Its unique capabilities make it appropriate for mesoscale simulations in regions with varying geographic characteristics.

The bathymetric information was obtained by the EMODnet Bathymetry portal (generated in February 2015) with grid size resolution $1/8 \times 1/8$ arcmin (see also http://www.emodnet-hydrography.eu/). This bathymetry has been produced from bathymetric survey data and aggregated bathymetry data sets collated from public and private organizations.

Distance from the shore is highly connected with socio-economic aspects; a short distance from the shore minimizes all the costs related to the technical infrastructure, installation and maintenance activities (i.e. capital and operating expenditures), but on the other hand it maximizes visual disturbance. The coastline from EEA was used for delimiting the specified distances from the shore (Table 1).

Similarly, regarding proximity to ports, as distances from ports shorten, the accessibility to the offshore wind project area is faster and more economical. These distances were derived from the World Port Index Database of the National

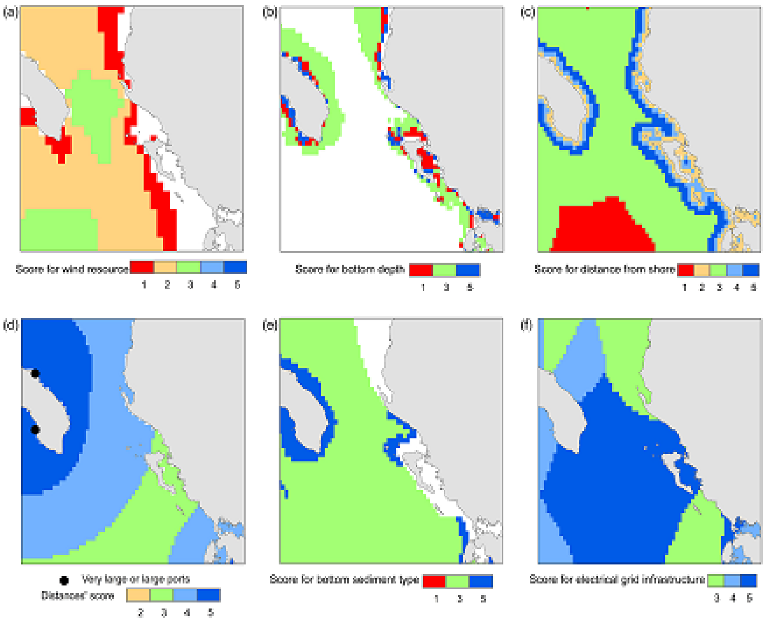

Figure 2. Results from the SWC as regards technical criteria in the Mediterranean pilot area: score for wind resource **(a)**, score for bottom depth **(b)**, score for distance from the shore **(c)**, score for proximity to ports **(d)**, score for bottom sediment type **(e)**, and score for electrical grid infrastructure **(f)**.

Geospatial Intelligence Agency (WPI 2015 pub. 150, http://msi.nga.mil/NGAPortal/MSI.portal). The ports included in the analysis are those whose controlling depth of the principal or deepest channel or the greatest depth alongside the wharf/pier is over 10 m.

Regarding bottom sediments, the existence of cobbles, boulders, dense or soft sand, etc., affects the foundation design and cable laying. The type of bottom sediments was derived via digitization of the Unconsolidated Bottom Surface Sediments of the International Bathymetric Chart of the Mediterranean (IBCM) of the Intergovernmental Oceanographic Commission, and the scale of the corresponding map is 1 : 1 000 000.

The assessment of the existing electrical grid infrastructure was based on the 2014 ENTSO-E Interconnected Network Grid Maps (see www.entsoe.eu). ENTSO-E's Interconnected Grid Map is one of the publications of this association. It displays the electricity transmission grid as of 31 December 2014, and the scale of the map is 1 : 4 000 000.

The data of all the environmental variables were derived from the "Mediterranean Sensitive Habitat" (MEDISEH) project (see http://mareaproject.net/). These data were partially based on (i) revision of historical and current data from the compilation of published and unpublished information, and (ii) habitat suitability modelling that was applied to fill spatial information gaps on the distribution of such species by predicting the locations that habitats are likely to be suitable for species to live (Giannoulaki et al., 2013). For the pilot site, the model results were based on a large number of personal observations that were collected related to the occurrence of *Posidonia oceanica*, and some point data sources for coralligenous and marl beds.

3.2 Description of study area

The Diapontia Islands, an island complex opposite to the north-western coasts of Corfu, Greece, were selected as a pilot area for OWF development in the context of the CoCoNet project and were used to develop the rationale of SWC. The main reason for this selection is that the Greek government (by the Ministry of Environment, Energy and Climate Change in 2011) has already preselected the area for potential OWF development; from this point of view, it comprises a realistic case to be studied in depth. The relevant activities are scheduled to take place in various phases. In the first phase (which is at the concept/early planning status), the foreseen capacity is 15 MW (three turbines each of 5 MW nominal power) (see www.4coffshore.com).

On the other hand, the selected area encompasses features that are very representative of the coastal areas of the MS. Specifically, Diapontia Islands do not belong to the few top-ranked areas according to the wind resource availability

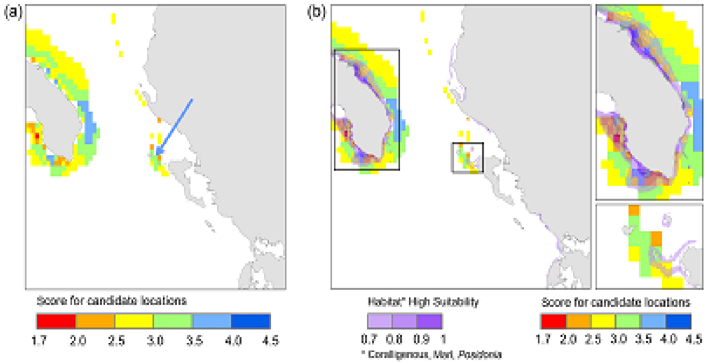

Figure 3. Overall score for candidate locations for OWF development in the Mediterranean pilot area **(a)**. Overall score combined with habitats (coralligenous formations, marls, *Posidonia oceanica* meadows) with high suitability in the Mediterranean pilot area **(b)**.

while the wind and wave climate is rather mild. Moreover, a Natura 2000 site is designated in the proximity of the examined area and, in the meantime, the wider area is well developed as regards tourism and fisheries, which comprise two of the most characteristic marine uses in the MS.

4 Results

Based on the methodology discussed in Sect. 2, the results for each examined parameter are firstly presented and described separately (Fig. 2a–f) for the pilot site in order to identify step by step favourable candidate locations for offshore wind exploitation, and then the final output is depicted (Fig. 3a) after combining the data sets via raster calculation method.

Specifically, the score as regards wind resource (Fig. 2a) is marginally appropriate (rank 1) for potential OWF development in the pilot site, while bottom depth (Fig. 2b) is ideal and characterized by the best score (rank 5). The score for distance from the shore (Fig. 2c) ranges between 2 and 4 in the wider examined pilot area while the corresponding score for distance from ports (Fig. 2d) is right before the top (rank 4). Finally, from Fig. 2e and f, it is evident that the bottom sediment type and the existing electrical grid infrastructure, respectively, are optimum for potential OWF development in this site (rank 5), since the existent electrical grid infrastructure seems to be adequate and the sea bed composition is sandy. The final scores as regards the examined technical parameters are depicted in Fig. 3a. The total technical suitability rating for the Othonoi site (one of the main islands of Diapontia Islands), shown with an arrow in the same figure, is between 3.0 and 3.5. Let us reiterate that the overall score of the candidate areas for OWF development are obtained by applying a linear weighting scheme. Comparing this rank to the other locations and according to the overall score scale, this area is considered "fair" as regards its favourability for OWF development.

Finally, superimposing the technical criteria with available data based on environmentally sensitive areas, we obtain the synthetic Fig. 3b. According to the obtained results, the likelihood for formation of *Posidonia oceanica* meadows, coralligenous formations, and marls is not very high in the study area based on the simulation models. However, in-depth assessment of these environmental aspects along with submarine archaeological findings and migratory bird routes are some of the main objectives of environmental impact assessment studies and require a systematic survey of any candidate area for OWF development. In conclusion, with a total relative score 3.0–3.5, it seems that Othonoi is a good choice in the northern part of the Ionian Sea. In this respect, the site has been rightly pre-selected from the Greek government as a site for potential OWF development.

5 Discussion

For the sake of completeness of this work, in this section some problems encountered during the development and implementation of the SWC are presented, along with some relevant remarks as regards the interpretation of the SWC results. The majority of the problems are related to the main knowledge/information gaps that were identified during the background data collection and assessment.

The first issue refers to data availability and quality. Specifically, regarding wind data the following (interconnected) issues are of most importance:

i. There is lack of high-resolution data at the basin scale. High-resolution wind data (usually obtained from numerical models or gridded satellite products) are necessary for the accurate estimation of the wind energy characteristics in nearshore/coastal areas (where the coastal morphology plays an important role in wind patterns) and the more detailed estimation of the relevant uncertainties.

ii. There is lack of offshore measurements at the usual operational turbine hub heights (i.e. within the range of 70–110 m a.s.l.). Measured wind data of the wind profile are necessary in order to estimate reliably the actual wind energy characteristics at hub heights. The wind profile can be accurately deduced only by utilizing measured wind data at various heights a.s.l., which, currently, may be obtained either by meteorological masts or lidar (light detection and ranging) measurements. Furthermore, such measurements are also necessary in order to evaluate less reliable data sources such as numerical models or satellite products. However, there is lack of lidar measurements in the Mediterranean and Black seas, and the installation, operation and maintenance of meteorological masts are expensive procedures, especially for offshore locations.

The data quality issue is also associated with the inherent uncertainties of wind speed data and, consequently, wind power density estimates, i.e. the most determinative parameter for SWC analysis. Before the implementation of the SWC, four different wind data sources (measurements from two buoy networks, satellite products, and results from two numerical weather prediction models) were assessed for the MS in order to be statistically analysed and evaluated. The analysis revealed that there were statistically significant deviations between the examined data sources. However, as it was recommended by Soukissian and Papadopoulos (2015), calibration schemes can be efficient in local spatial scales while it is very risky to apply them in large spatial scales, such as the examined basins. Eventually, the most representative data source with the highest spatiotemporal resolution was adopted. On the other hand, such uncertainties necessarily should be taken into consideration when site-specific studies are made for potential OWF development.

More detailed spatial information is vital regarding also marine environmental data. The more accurate mapping of important habitats at the critical bottom depths for OWF development, such as meadows of *Posidonia oceanica*, fields of *Phyllophora crispa*, coralligenous, and deep-water white coral formations, provide necessary background information for efficient OWF planning in wide areas. Moreover, bird migration routes over the same areas should be defined as accurately as possible and relevant quantitative information should be available.

However, the most important information gap seems to be the lack of a basin-wide marine spatial planning. Some European Mediterranean countries have coarse preliminary spatial plans and very few detailed plans at regional level tailor-made for OWE projects (e.g. at the Gulf of Lions in France). In addition, since OWF installations are expected to affect coastal zone activities, integrated coastal zone management (combined with relevant information on the socio-economic status of the area) is also required as a necessary complementary tool for marine spatial planning. In this regard, socio-economic aspects could be included in future SWC applications, even in extended sea areas.

Another aspect to call attention to, especially when interpreting the SWC results, is related to the appropriate consideration of the involved spatial scales. The spatial extent of the examined basin, the heterogeneity of the considered data sets, and the data spatial scales do not allow a detailed (in the spatial domain) and in-depth (as regards the involved parameters) analysis, let alone the lack of data, especially with respect to socio-economic aspects. Specifically, the geographical extent of the MS is of the order of several hundreds of kilometres, which assigns the magnitude of the spatial scale, and, in turn, the data involved. The next most important spatial scale is connected to the spatial resolution of wind data. The resolution of the numerical model we have used is roughly of the order of 10 km × 10 km (see also Sect. 3.1). Here, the underlying assumption is that the wind information contained in the area corresponding to the model grid point (i.e. 100 km^2) is homogeneous and representative of the entire "pixel". Although this assumption can be valid for morphologically homogeneous sea areas (e.g. offshore areas), there are cases in which it may be incorrect (e.g. for coastal areas). Moreover, the spatial resolution of bathymetric data is, in principle, not compatible with the wind data resolution, since it is roughly of the order of 200 m. Let us also mention that a large part of bathymetric data have been produced by interpolation methods and thus cannot be considered as fully accurate. In this respect, outputs of the SWC can only provide some preliminary suggestions for potential OWF development, mainly through indicative zones, in the examined basin. Since this spatial scale may lead to misjudgements as regards the development of offshore wind projects, a detailed local assessment of technical, socio-economic, and environmental features at the finest possible spatial scale is required for localized cases in order to reach final decisions and form strategies.

6 Conclusions

The identification of potential zones for offshore wind farm development is a very delicate procedure and should be based on sufficient and high-quality data regarding both biotic and abiotic elements of the marine environment. An integrated and interdisciplinary approach has been adopted integrating technical, and environmental criteria and data related to the offshore wind energy exploitation. The integration is implemented through a robust tool – the so-called Smart Wind Chart, aiming to maintain and secure the sustainable blue growth in the Mediterranean and Black seas through the support of offshore wind energy projects and marine habitat conservation. The application of the Smart Wind Chart in the northern Ionian Sea provided one of the most favourable candidate areas for offshore wind farm development in this part of the Hellenic seas, which, in this case, coincides with the suggestion of the Greek government. On the other hand, some knowledge and information gaps were evident during the implementation of the Smart Wind Chart while some additional aspects, related to the uncertainties of the wind data involved and the involved spatial scale of the data, were explicated in order to prevent misinterpretations of the outputs.

It is, however, important to note that the results obtained from the Smart Wind Chart cannot replace an in-depth environmental and socio-economic impact assessment study, which includes the corresponding mapping and monitoring of any candidate area and, in general, more detailed analysis with high-resolution data. Considering the future introduction of offshore wind farm installations in the Mediterranean Sea, the potential impacts should be closely linked to the environmental and ecological processes. Standardized monitoring protocols and a clear guidance on the assumptions and requirements of monitoring programmes are necessary, in agreement with the scientific community and regulatory bodies (Franco et al., 2015).

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APPENDIX C

Table D1. List of various activities throughout period of research studies

Event		*Location*	*Date*
Conference	11th Panhellenic Symposium of Oceanography & Fisheries	University of the Aegean, Mytilene, Lesvos Island	3 - 17 May, 2015
Conference	51st European Marine Biology Symposium	Rodos Palace Hotel, Rhodes Island.	26 - 30 September, 2016
Workshop meeting	CoCoNET Focus Workshop	Norfolk, UK	28 – 29 April 2014
Workshop meeting	CoCoNET Focus Workshop	Hellenic Centre for Marine Research, Anavyssos, Athens	9 – 10 June, 2014
Workshop meeting	CoCoNET Focus Workshop	Spanish Institute of Oceanography (EIO), Mallorca	6 - 7 October, 2014
Workshop meeting	CoCoNET Second synthetic workshop	Hellenic Centre for Marine Research, Anavyssos, Athens	25 - 26 May, 2015
Workshop meeting	CoCoNET Final synthetic workshop meeting	Università del Salento, Lecce, Italy	1 – 2 December, 2015
Specialist training	Macroinvertebrate taxonomy	Hellenic Centre for Marine Research, Anavyssos, Athens	1 – 30 September, 2014
Outreach	European Maritine Day	Pireaus, Greece	28 – 29 May, 2015