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1 **A review of methods and indicators used to evaluate the ecological** 2 **modifications generated by artificial structures on marine** 3 **ecosystems**

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16 **Abstract:**

17 The current development of human activities at sea (e.g. land reclamation, maritime activity
18 and marine renewable energy) is leading to a significant increase in the number of
19 infrastructures installed in marine settings. These artificial structures provide new hard-
20 bottom habitats for many marine organisms and can thus modify the structure and functioning
21 of coastal ecosystems. In order to better evaluate the nature of these modifications as well as
22 the potential benefits and/or impacts generated, it becomes essential to develop assessment
23 methods that can be applied to a wide variety of study sites from harbours to coastal offshore
24 environments. In this context, our study aims to review the different methods and indicators
25 available which are used to measure the modifications of biodiversity and ecological
26 functioning generated by such structures. Among the methods reviewed, we highlight some
27 that were developed specifically for artificial structures, and others intended for various
28 primary uses but which have been successfully transposed to artificial structures.
29 Nevertheless, we also point out the lack of reliable methods concerning some biological
30 ecosystem components impacted by artificial structures. In this context, we require the
31 adaptation or creation of brand-new indicators to achieve a better characterisation of the
32 ecological impacts generated by these structures. Overall, this study highlights a very high
33 number of existing methods, which provide stakeholders with useful tools to study the
34 impacts of artificial structures, and identifies the need to develop integrative indicators to
35 enhance the deployment of new artificial structures.

36

37 **Keywords:**

38 Artificial reef - Biodiversity- Ecosystem functioning - Ecological engineering – Coastal
39 management

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1. Introduction

Oceans worldwide are currently facing an increase in the deployment of artificial structures (AS), a phenomenon that has been called ‘ocean sprawl’ (Dafforn et al., 2015; Duarte et al., 2013; Heery et al., 2017; Todd et al., 2019). Ocean sprawl is likely to escalate in the near future, particularly due to the development of marine renewable energies to meet the growing energy demand as well as the need to improve coastal defences against rising sea level, extreme storms and flooding (Asif and Muneer, 2007). AS can be divided into two different types: 1) those commonly called artificial reefs which are designed and installed for a variety of reasons, but predominantly for their reef properties (e.g. ecosystem conservation/restoration, fish stocks enhancement, fisheries management, etc.; Jensen et al., 2000a; Vivier et al., 2021), and 2) those deployed for other primary infrastructural purposes associated with a range of maritime activities, such as oil platforms (Fabi et al., 2004; Love et al., 2021, 1999), breakwaters and seawalls (Airoidi et al., 2009; Bulleri and Chapman, 2010), pontoons (Connell, 2001), submarine cables (Taormina et al., 2020b, 2020a, 2018) or marine renewable energy facilities (Degraer et al., 2020; Langhamer, 2012).

The appearance of these AS in marine environments has led to various ecological effects on the host ecosystem (Airoidi et al., 2009; Dafforn et al., 2015). For instance, by mimicking the functions of a natural reef, AS can provide substrate and/or shelter for various organisms and may lead to changes in the structure and functioning of the ecosystems concerned (Bohnsack et al., 1991; Jensen et al., 2000a).

In response to this global increase in development, a new research field has arisen known as ‘ecological engineering’ (Odum, 1962). Briefly, ecological engineering focuses on the design of sustainable ecosystems that integrate human society with its natural environment for the benefit of both (Mitsch and Jørgensen, 2003; Odum, 1962). In our case, eco-engineering aims to provide stakeholders with options for the design and management of AS in the marine environment in order to support biodiversity whilst not compromising the function of the structure (O’Shaughnessy et al., 2020). However, to define the best solutions, it is necessary to efficiently evaluate the nature of modifications, potential benefits and/or any potential impacts generated by AS on marine ecosystems.

There is an important global challenge concerning the identification of simple methods to characterize the conditions of complex marine ecosystems (Borja, 2014). For this purpose,

1 ecological indicators, (*i.e.* selected variables, also called metrics) as well as indices (*i.e.*
2 combinations of multiple indicators in a single numerical value) have been extensively used
3 over the last two decades to monitor the condition of whole ecosystems (Rombouts et al.,
4 2013; Smit et al., 2021). The success of ecological indicators relies on their ability to reduce
5 complex information into simplified and easily interpretable metrics (Borja et al., 2013,
6 2015). The rise of the concept of ecological indicators has been mainly driven by legislation
7 such as the European Water Framework Directive (WFD) in 2000 and the Marine Strategy
8 Framework Directive (MSFD) in 2008. While numerous ecological indicators and indices are
9 available to investigate the impact of various anthropogenic activities on marine ecosystems,
10 assessment protocols have not been specifically defined to measure the ecological effects of
11 AS (with the exception of Lima et al., 2020).

12 In this context, our study aims to review the different methods and indicators available in
13 the literature used to measure the modifications generated by AS on marine ecosystems.
14 Specifically, we divide these methods and indicators into three categories according to their
15 objectives. Thus, we review the methods and indicators used to investigate 1) the
16 establishment of epibenthic communities on AS, 2) the colonization of AS by mobile
17 megafauna, and 3) the effects of AS on ecosystem functioning. Finally, to obtain a better
18 characterisation of ecological impacts of AS, we draw up a list of recommendations to address
19 the identified knowledge gaps.

20

21 **2. Establishment of epibenthic communities**

22 When submerged in seawater, AS are invariably colonized by a variety of epibenthic
23 organisms, forming the so-called “biofouling”. This colonization of a bare substratum by an
24 epibenthic community follows an ecological succession, *i.e.* a sequence of stepwise changes
25 in community composition until it eventually reaches a persistent stage, called the climax
26 (Clements, 1916; Connell and Slatyer, 1977). In marine environments, succession begins with
27 the early establishment of a biofilm (Cooksey and Wigglesworth-Cooksey, 1995), allowing
28 the subsequent development of pluricellular eukaryote colonisers. Firstly, these eukaryote
29 taxa are mainly composed of opportunistic species, which are then gradually replaced by
30 longer-lived morphologically complex species as the colonizing community evolves towards
31 its climax (Connell and Slatyer, 1977). A wide variety of abiotic and biotic factors affect the
32 epibenthic community composition, as well as the characteristics of the ecological succession
33 and the time needed to reach a climax state (Falace and Bressan, 2000). Biotic factors include

1 facilitation, competition and trophic interactions between organisms (Connell and Slatyer,
2 1977). On the other hand, abiotic factors can be split between local environmental conditions
3 (e.g. temperature, light, pH, salinity, currents; Bowden et al., 2006; Falace and Bressan, 2000;
4 Pérès and Picard, 1964) and artificial substratum properties (e.g. surface type, material,
5 texture, slope; Falace and Bressan, 2000).

6 Epibenthic communities are highly diverse and usually dominated by marine
7 invertebrates (e.g. Arthropoda, Mollusca, Tunicata, Bryozoa, Annelida, Cnidaria, Porifera),
8 and macroalgae (e.g. Chlorophyta, Rhodophyta, and Ochrophyta). These organisms can be
9 divided into two categories: 1) sessile taxa directly attached to the substratum, and 2) mobile
10 taxa holding onto sessile taxa (also called clinging taxa). When AS are installed on soft
11 muddy and sandy substrates, the development of epibenthic communities involves the
12 settlement of species previously absent, leading to an overall increase in local diversity (De
13 Mesel et al., 2015). Also, certain epibenthic organisms can create complex tri-dimensional
14 biogenic structures (e.g. kelps, corals, mussels) that further increase habitat heterogeneity
15 (Pickering and Whitmarsh, 1997). Conversely, in comparison with natural hard substrates,
16 epibenthic communities of AS usually display a higher density of non-indigenous species
17 (NIS; Airoidi et al., 2015; Mineur et al., 2012). For example, AS in the Adriatic Sea are
18 shown to display three times more non-indigenous ascidians than natural reefs (Airoidi et al.,
19 2015). In this way, AS may potentially facilitate the spread of NIS by creating new
20 connectivity routes via a stepping-stone process (Adams et al., 2014; Heery et al., 2017;
21 Mineur et al., 2012).

22

23 ***2.1 Ecological indicators***

24 Most studies of epibenthic communities associated with AS focus exclusively on their
25 biological structure, *i.e.* community properties in terms of composition of taxonomic entities
26 (Bohnsack and Sutherland, 1985; Bortone et al., 2000; Bortone and Kimmel, 1991).
27 Consequently, the data used in these studies are commonly based on a description of the
28 epibenthic community through an inventory of recorded taxa, along with associated
29 quantitative information [e.g. number of individuals, density, biomass or coverage (Bortone
30 and Kimmel, 1991)]. Such inventories are performed with different methods: some are non-
31 destructive, using *in-situ* visual censuses or underwater imagery, while others are destructive
32 with sampling of the epibenthic community and analysis in the laboratory (Bortone et al.,
33 2000). Based on the data obtained, univariate indices can be calculated to define community

1 characteristics in terms of diversity and evenness (e.g. species or taxonomic richness,
2 Shannon index H' , Pielou's index J' , etc.). In addition, standard similarity indices (e.g.
3 Sorensen; Bray-Curtis or Jaccard similarity indices) and multivariate community analyses
4 (Clarke, 1993) are commonly used to evaluate the similarity between different samples and
5 communities, for example to compare the epibenthic communities of artificial and natural
6 substrates (Bortone et al., 2000).

7 Instead of focusing on biological structure, a complementary novel approach is to
8 study the functional structure of communities. Functional structure is built on the individual
9 properties of organisms, also called biological traits of life (e.g. morphology, behaviour,
10 feeding strategy, etc.), rather than on their taxonomic identity. In the last decade, biological
11 trait analysis and functional diversity indices (e.g. functional richness, functional divergence,
12 functional evenness and Rao's quadratic entropy) have been extensively used to understand
13 the response of benthic communities to environmental gradients (Beauchard et al., 2017;
14 Lam-Gordillo et al., 2020; Martini et al., 2020; Mouillot et al., 2013). Nevertheless, these
15 approaches have been little used for epibenthic communities on artificial substrates (Firth et
16 al., 2016; Strain et al., 2021) and more broadly on natural hard substrates (Vinagre et al.,
17 2017b). Given the growing interest for using the biological traits approach in marine ecology
18 (Beauchard et al., 2017), we suggest that such methods need to be considered for development
19 of ecological indicators aiming to evaluate the ecological value of epibenthic communities
20 associated with AS.

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24 **2.2 Multimetric indices**

25 A fashionable approach to inform community characteristics, and more specifically
26 their quality status, is the use of ecological multimetric indices to aggregate diffuse biological
27 data into a single value (Ruaro et al., 2020). Such indices are usually developed to be simple
28 to derive, widely applicable and easy for the public to understand (Borja and Dauer, 2008). To
29 our knowledge, there are currently no ecological indices specifically designed to assess the
30 ecological quality of epibenthic communities associated with AS. More broadly, even
31 considering natural habitats, it appears that benthic communities associated with hard
32 substrates suffer from a lack of available ecological tools in contrast with soft-sediment
33 communities. Indeed, a wide diversity of methodologies has been developed and

1 intercalibrated for soft-sediment macroinvertebrate communities (e.g. AMBI by Borja et al.,
2 2000, BENTIX by Simboura and Zenetos, 2002, BQI by Rosenberg et al., 2004; BOPA by
3 Dauvin and Ruellet, 2007), mainly under the impetus of the European Water Framework
4 Directive (WFD) in the early 2000s. This disparity between soft and hard substrata can be
5 partly explained by the differences of sampling methods used to describe benthic
6 communities. Indeed, soft-sediment grab sampling is much easier to implement across a broad
7 depth range compared to costly and time-consuming methods implying the use of divers or
8 underwater imagery that are commonly employed for hard substrates.

9 Nevertheless, some ecological multimetric indices have been developed over the last
10 decade to describe epibenthic communities on natural hard substrates (see Table 1 for a
11 summary of their characteristics). Most of these indices are based exclusively on macroalgae
12 communities: CFR (Quality of Rocky Bottoms index in Spanish) by Juanes et al. (2008),
13 MarMAT (Marine Macroalgae Assessment Tool) by Neto et al. (2012), CCO (Cover,
14 Characteristic species, Opportunistic species) by Ar Gall et al. (2016), EEI-c (Ecological
15 Evaluation Index continuous formula) by Orfanidis et al. (2011) and QiSubMac (Quality
16 Index of Subtidal Macroalgae) by Ar Gall and Derrien-Courtel (2015). To our knowledge,
17 only one index has been developed exclusively for fauna: RMAT (Rocky shore
18 Macroinvertebrates Assessment Tool) by Vinagre et al. (2017a). This discrepancy can be
19 partly explained because macroalgae are usually dominant in clear shallow water and are
20 considered as good indicators because of their ability to adjust to various environmental
21 pressures (Benedetti-Cecchi et al., 2001; Gorostiaga and Díez, 1996; Soltan et al., 2001).
22 Nevertheless, spatial competition, facilitation and trophic interactions commonly occur
23 between fauna and flora on hard substrates (Ar Gall and Derrien-Courtel, 2015; Díez et al.,
24 2012), so it appears more appropriate to use indicators able to take into account both the
25 faunal and floral composition (Van Hoey et al., 2010). Under these circumstances, various
26 authors have proposed indicators that take into account not only the macroalgae but also the
27 macrofauna (Díez et al. (2012) with RICQI (Rocky Intertidal Community Quality Index;
28 Pinedo and Ballesteros (2019) with EGs-I (Indicators of Ecological Groups). While Pinedo
29 and Ballesteros (2019) obtained very similar results when comparing their index with and
30 without macrofauna, they nevertheless encouraged the simultaneous use of both groups for
31 the reasons discussed above.

32 Whether based on macroalgae or macrofauna, the ecological indices reviewed here
33 focus on the biological structure of communities. Indeed, some metrics are widely used in

1 community studies, such as species richness (for CFR, MarMAT, CCO, QiSubMac, RICQI;
2 Table 1) or the Hurlbert diversity index (for RMAT; Table 1). Also, some of these indicators
3 which rely on the relative occurrence of different ecological groups are based on the abilities
4 of taxa to tolerate environmental pressures (*i.e.* RMAT, EGs-I; Table 1). For this purpose,
5 taxa are divided into four groups according to their tolerance to stress: 1) sensitive taxa, 2)
6 stress-tolerant taxa, 3) indifferent taxa and 4) opportunistic taxa (Pinedo and Ballesteros,
7 2019). These indicators are in fact adapted from previous indicators developed for soft-
8 sediment communities, RMAT is based on the BENTIX indicator from Simboura and Zenetos
9 (2002); while EGs-I is an adaptation of the MEDDOC index from Pinedo et al. (2015). Such
10 indicators have been shown to be efficient to assess the ecological status of water bodies and
11 are mainly based on the tolerance of taxa to a particular pressure: for example, organic matter
12 enrichment (Pinto et al., 2009). As the classification of taxa within different ecological groups
13 is highly pressure-specific, it is important to bear in mind that such indices should not be used
14 unquestioningly in any given case-study. Most of the other multimetric indices (*i.e.* CFR;
15 MarMAT; CCO; EEI-c; QiSubMac; Table 1) make use of ratios between two ecological
16 groups *i.e.*: characteristic taxa (*i.e.* taxa linked to a particular biotic assemblage; Dauvin et al.,
17 2010) and opportunistic taxa (*i.e.* taxa that can quickly exploit new resources or ecological
18 niches as they become available; Dauvin et al., 2010). Conversely to ecological groups, where
19 the indices are based on the abilities of taxa to tolerate environmental pressures, characteristic
20 and opportunistic taxa are mainly defined by their growth strategies: characteristic taxa are
21 perennial species, while opportunistic taxa are annual species (Orfanidis et al., 2001). The
22 popularity of this ratio is essentially because it is assumed that, with an increase in any
23 environmental pressure, the density of the characteristic species will decrease, while the
24 density of opportunistic species will increase (Murray and Littler, 1978). In the context of AS,
25 metrics based on this ratio can be of particular interest since they can represent a proxy of the
26 ecological succession stage (Connell and Slatyer, 1977).

27 By focusing exclusively on biological structure, most of the indicators reviewed here
28 tend to neglect the functional structure of epibenthic communities. Indeed, none of the
29 indicators reviewed here rely on extensive biological trait analysis or functional diversity
30 indices. Instead, some of these indicators pay particular attention to certain traits. Thus, the
31 EEI-c, QiSubMac and RICQI indices make use of various metrics based on the morphology
32 of macroalgae *i.e.*: the relative coverage of different macroalgae morphology groups (EEI-c),
33 or the cover (RICQI) and density (QiSubMac) of structuring macroalgae. These latter are of
34 particular interest as they fulfil important ecosystem functions (e.g. habitat, nursery or feeding

1 functions; Christie et al., 2003; Eckman and Duggins, 1991; Edwards, 1980; Norderhaug et
2 al., 2002). RICQI focuses on another approach, using the coverage of herbivores and
3 suspensivores as metrics to characterize the feeding strategy of benthic fauna.

4 Among the indices reviewed here, the ALien biotic indEX (ALEX; Çinar and Bakir,
5 2014) stands out from the rest because it was developed to detect the impact of NIS. This
6 index is based on the relative abundance of four different biogeographic groups defined in
7 terms of species invasiveness: 1) native species (*i.e.* species naturally occurring in the region),
8 2) casual NIS (*i.e.* NIS with only one report in the region), 3) established NIS (*i.e.* NIS with
9 self-maintaining population) and 4) invasive NIS (*i.e.* NIS that have overcome biotic and
10 abiotic barriers and are able to expand their geographic range with significant impact on the
11 invaded habitats). While initially developed for soft-bottom macrofauna, ALEX has been
12 tested on epifaunal communities growing on AS associated with an Italian Mediterranean
13 harbour (Tempesti et al., 2020). In this study, Tempesti et al. (2020) point out that, despite the
14 high number of NIS present within the harbour, ALEX values are quite low and it can be
15 classified as having high or good environmental status. This result can be explained by the
16 fact that ALEX was designed for the Eastern Mediterranean Sea, which hosts highly degraded
17 assemblages with a higher coverage of NIS. Consequently, even though the biogeographic
18 groups chosen for ALEX seem appropriate, the index calculation may need recalibration as a
19 function of the considered bioregion.

20 The points highlighted above show the important need to develop an ecological
21 multimetric index aimed specifically at addressing the ecological quality of epibenthic
22 communities associated with AS. Such a tool should be able to allow stakeholders to identify
23 easily and robustly the performance of eco-engineered structures to enhance future AS
24 deployments. From our point of view, such a multimetric index should be composed of a set
25 of indicators with the following objectives: 1) to consider both flora and fauna
26 simultaneously, 2) to summarize the structural and functional diversity of the epibenthic
27 community; 3) to estimate the stage of the ecological succession, 4) assess the ability of AS to
28 host key-stone species and, 5) and their ability to prevent the establishment of non-indigenous
29 species.

30 Note that we focus here only on the macro-epibenthic community, *i.e.* that part of the
31 epibenthic community with organisms larger than 1 cm. Thus, biofilm are not considered here
32 and need dedicated methods (McManus et al., 2018; Riera et al., 2018) .

Table 1: Characteristics of nine multimetric indices aimed to evaluate the ecological quality of epibenthic communities of natural hard substrates. Crosses (X) indicate that at least one metric of the index (column) is related to the associated category (row).

Index	CFR	MarMAT	CCO	EEI-c	QiSubMac	Egs-I	RICQI	ALEX	RMAT
Authors	Guinda et al. 2014 / Juanes et al. 2008	Neto et al. 2012	Ar Gal et al. 2016	Orfanidis et al. 2011	Ar Gal et Derrien- Courtel 2015	Pinedo et al. 2019	Diez el al. 2012	Piazzì et al. 2015 / Tempesti et al. 2020	Vinagre et al. 2017
Target compartment	Macroalgae	Macroalgae	Macroalgae	Macroalgae	Macroalgae	Macroalgae & Macrofauna	Macroalgae & Macrofauna	Macroalgae & Macrofauna	Macrofauna
Area	Atlantic	Atlantic	Atlantic	Mediterranean Sea	Atlantic	Mediterranean Sea	Atlantic	Mediterranean Sea	Atlantic
Zone	Intertidal/Subtidal	Intertidal	Intertidal	Subtidal	Subtidal	Subtidal	Intertidal	Subtidal	Intertidal
Data	Coverage	Coverage	Coverage	Coverage	Density	Coverage	Coverage	Coverage	Density/Biomass
Number of metrics	3	7	3	5	14	4	8	4	4
Biological structure	X	X	X	X	X	X	X	X	X
Functional structure				X	X		X		
Non-indigenous taxa								X	
Structuring taxa					X		X		
Physiological status					X				
Ecological group									
Characteristic taxa	X	X	X	X	X				
Opportunistic taxa	X	X	X	X	X	X			X
Sensitive taxa					X	X	X		X
Tolerant taxa						X			X
Indifferent taxa						X			

3. Mobile megafauna aggregation

In addition to providing a hard substratum for epibenthic communities, AS also allow the aggregation of various species of mobile megafauna, especially teleost fish. Indeed, by increasing habitat complexity, AS provide shelters away from currents and against predators of mobile species. This refuge capacity depends mainly on properties of the structure such as its size, shape, construction material or degree of complexity (Charbonnel et al., 2002; Hackradt et al., 2011; Sherman et al., 2002). In addition, AS attract mobile taxa by increasing food availability. Many species are indeed shown to predate directly on the epibenthic community growing on AS (Bohnsack and Sutherland, 1985; Cresson et al., 2014a; Lindquist et al., 1994; Mavraki et al., 2021; Relini et al., 2002; Reubens et al., 2011; Taira et al., 2020). As a result, these consumers also represent a new food source for higher trophic levels inhabiting the AS (e.g. piscivorous benthic fish ; Cresson et al., 2019, 2014b). Regarding the role of AS for mobile fauna, there is a continuing debate concerning two dominant theories: 1) the “attraction hypothesis”, which assumes that AS only attract specimens from nearby ecological communities, without increasing the overall biomass production (Bohnsack, 1989) and 2) the “production hypothesis” which claims that AS increase the abundance and biomass of associated species (Pickering and Whitmarsh, 1997; Polovina and Sakai, 1989). In fact, the productive potential of an AS varies according to its characteristics (e.g. design, distance to natural reef, association with protected areas, etc.; Pickering and Whitmarsh, 1997) but also according to the characteristics of the species considered (e.g. substrate affinity, trophic level, etc.; Brickhill et al., 2005; Cresson, 2013). Thus, these two opposing theories appear to be not mutually exclusive, but rather represent the two extremes of the same *continuum* (Cresson, 2013).

3.1 Ecological indicators

A number of studies address the colonization of AS by megafauna by focusing on single-species dynamics, mainly involving species of important economic interest such as cephalopods (e.g. Mereu et al., 2018; Ulaş et al., 2011), decapods (e.g. Jensen et al., 2000b; Krone and Schröder, 2011) or fish (e.g. Reubens et al., 2013, 2011; Szedlmayer and Shipp, 1994). While such methods are important for resource and fisheries management, the use of more integrative multi-species approaches can provide a fuller picture of ecosystem quality. Thus, mobile megafauna communities associated with AS, especially fish assemblages, have been widely described (Bortone et al., 2000). Such community analyses rely on methods broadly similar to those described for epibenthic communities (see previous part ; Bortone et

1 al., 2000). Indeed, descriptions of megafauna communities are based on inventories of the
2 taxa inhabiting AS, using a quantitative evaluation (e.g. number of individuals, biomass, etc.)
3 performed with destructive (nets, traps, etc. ; Lima et al., 2020; Wu et al., 2019) or non-
4 destructive methods (underwater images, *in situ* count, etc.; Becker et al., 2017; Krone et al.,
5 2017; Mercader et al., 2017; Taormina et al., 2020a). Such data allow us to investigate the
6 biological structure of communities through generic univariate indices of diversity and
7 evenness (e.g. species or taxonomic richness, Shannon index H' , Pielou's index J' , etc.). The
8 biological structure of different communities can also be compared with each other using
9 standard similarity indices (e.g. Bray-Curtis or Jaccard similarity indices) and multivariate
10 community analysis (Clarke, 1993). Functional approaches such as biological trait analysis
11 and functional diversity indices have also been used in the last decade to study fish
12 assemblages associated with AS (Cresson et al., 2019; Dance et al., 2018; Koeck et al., 2014).
13 It is important to note that, among the wide range of traits commonly used, an emphasis has
14 been placed on trophic traits due to their capacity to summarize morphological, behavioural
15 and interspecific interactions and to reveal changes in ecosystem functioning (Cresson et al.,
16 2019; Mouillot et al., 2013).

17

18 **3.2 Multimetric indices**

19

20

21 A number of multimetric indices have been developed to evaluate the ecological
22 quality of natural ecosystems based on fish assemblages. Most of these indices were
23 developed for transitional waters (Souza and Vianna, 2020), such as the Estuarine Fish
24 Community Index (Harrison and Whitfield, 2004), AZTI's Fish Index (Uriarte and Borja,
25 2009) or the Estuarine Multimetric Fish Index (Delpech et al., 2010). For marine ecosystems,
26 the Marine Fish Community Index (MFCI) from Henriques et al. (2008) describes the
27 ecological status of fish assemblages from both rocky and soft bottoms (Table 2). Metrics
28 used for the calculation of MFCI are based on 1) fish diversity (*i.e.* species richness, number
29 of rare species) and 2) abundance (*i.e.* total abundance), as well as 3) proportion of species
30 according to their mobility (*i.e.* transient and resident species), 4) trophic groups (*i.e.*
31 omnivores, carnivores and herbivores), 5) commercial interest and 6) nursery function (*i.e.*
32 spawning species, juveniles). Also based on functional and structural properties, Teixeira-
33 Neves et al. (2016) created a preliminary fish-based multimetric index to assess the ecological
34 status of rocky reefs affected by thermal and urban influences (Table 2). The metrics used to

1 calculate this multimetric index are fairly similar to those used for MFCI (Henriques et al.,
 2 2008), except that the resilience capacity of fish species (*i.e.* its capacity to recover from
 3 changes in the environment, based on its life history characteristics, such as its fecundity and
 4 growth) is considered (*i.e.* low, medium and high resilience) rather than the nursery function.
 5 Finally, a multimetric index has been specifically developed to assess the ecological quality of
 6 fish assemblages associated with AS. Indeed, Lima et al. (2020) developed the Artificial Reef
 7 Multimetric Index (ARMI; Table 2), a multimetric index aimed at evaluating the performance
 8 of artificial reefs based on fish community analysis. This index includes 16 metrics divided
 9 into four categories: 1) AS - fish assemblage structure (*i.e.* number of species and individuals,
 10 total biomass, Shannon's index), 2) TL - trophic structure (*i.e.* mean trophic level), 3) VL-
 11 vulnerability (*i.e.* average value of extinction risk) and 4) EL- economic importance (*i.e.*
 12 average importance of fishing and/or tourism). Using spatiotemporal comparisons of
 13 computed ARMI, Lima et al. (2020) obtained higher ARMI scores within an artificial reef
 14 compared to a control area, suggesting a positive effect related to the fish assemblages.
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16 **Table 2:** Characteristics of three multimetric indices aimed at evaluating the ecological
 17 quality of marine fish communities. Crosses (X) indicate that at least one metric of the index
 18 (column) is related to the associated category (row). MFCI: Marine Fish Community Index
 19 and ARMI: Artificial Reef Multimetric Index.

Index	MFCI	Rock fish multimetric indices	ARMI
Authors	Henriques et al. 2008	Teixeira-Neves et al. 2016	Lima et al. 2020
Taxonomic diversity	X	X	X
Mobility	X	X	
Trophic groups	X	X	X
Commercial interest	X	X	X
Nursery function	X		
Vulnerability		X	X

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22 4. Ecosystem functioning

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24 Unsurprisingly, following the introduction of an AS, the formation of a new
 25 community affects the functioning of the ecosystem as a whole. As mentioned above, the
 26 growth of an epibenthic community can provide a prey resource for higher trophic levels,
 27 even up to apex predators (Cresson et al., 2014a; Raoux et al., 2017). Also, epibenthic
 28 communities are frequently dominated by suspension feeders and thus can act as biofilters.

1 Consequently, this leads to a reduction of suspended particulate matter concentration resulting
2 in a lower turbidity and increased light penetration (Degraer et al., 2020; Mavraki et al.,
3 2020), which can affect primary production (Slavik et al., 2019). In addition, areas
4 surrounding the AS are subject to enrichment in organic matter due to the deposition of faecal
5 pellets egested by the organisms (Davis et al., 1982; Maar et al., 2009). To improve our
6 understanding of these effects at the ecosystem scale, we need to describe the marine food-
7 web structure and functioning through holistic approaches. Among the most common
8 approaches, Stable Isotopic Analysis (SIA) and trophic modelling have been widely applied in
9 different contexts to construct and analyse trophic webs (McCormack et al., 2019; Navarro et
10 al., 2011).

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13 **4.1. Stable Isotopic Analysis**

14 The SIA method is based on measuring the isotope ratios of nitrogen ($^{15}\text{N}/^{14}\text{N}$, denoted
15 as $\delta^{15}\text{N}$) and carbon ($^{13}\text{C}/^{12}\text{C}$, denoted as $\delta^{13}\text{C}$) as intrinsic dietary tracers. Since $\delta^{15}\text{N}$ and $\delta^{13}\text{C}$
16 values are transferred from dietary sources to consumers in a predictable manner, these tracers
17 are used to describe the trophic position and feeding habits of organisms, and thus help us to
18 investigate trophic-web structure (Peterson, 1999; Peterson and Fry, 1987). To identify
19 fundamental trophic-web attributes, various isotopic indices have been developed (Table 3). A
20 first set of indices has been developed based on the position of organisms in 2-D space (*i.e.*
21 $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$) (Table 3; Jackson et al., 2011; Layman et al., 2007). Most common indices are
22 1) the convex hull area, or Total Area (TA) corresponding to the minimum convex polygon
23 containing all organisms, 2) the ranges of isotope ratios (Carbon Range CR and Nitrogen
24 Range NR) representing the ranges of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$, and 3) the mean Nearest Neighbour
25 Distance (NND), which is the average of the smallest Euclidean distance between all the
26 species taken two by two (Layman et al., 2007). Unfortunately, some metrics have been
27 shown to suffer from bias under certain circumstances, particularly because all species are
28 considered equal in the isotopic space (Hoeinghaus and Zeug, 2008). To include the
29 abundance of organisms, another set of indices called isotopic functional indices has been
30 recently developed (Table 3; Cucherousset and Villéger, 2015; Rigolet et al., 2015). These
31 indices are based on the transfer and adaptation of functional diversity indices, developed
32 initially for multiple biological traits (*i.e.* multi-dimensional; Mouillot et al., 2013), to isotopic

1 space. Isotopic functional indices thus aim to depict and quantify structural and functional
 2 characteristics of trophic-webs (Rigolet et al., 2015). These indices are: 1) Isotopic
 3 Divergence (IDiv) and 2) Isotopic Dispersion (IDis) which both provide information about
 4 isotopic richness (divergence of species from the community centre of gravity) as well as 3)
 5 Isotopic Evenness (IEve) and 4) Isotopic Uniqueness (IUni), both of which describe the
 6 regularity of species distribution within isotopic space (Cucherousset and Villéger, 2015).

7

8 **Table 3:** Set of structural and functional isotopic indices commonly used along with their
 9 acronyms.

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Type	Index	Acronym	Authors
Structural	Total Area	TA	
	Standard Ellipse Area	SEA	
	Centroid Distance	CD	
	Carbon Range	CR	Jackson <i>et al.</i> , 2011
	Nitrogen Range	NR	Layman <i>et al.</i> , 2007
	Nearest Neighbour Distance	NND	
	Standard Deviation of the NND	SDNND	
	Coefficient of Variation of the NND	CVNND	
Functionnal	Isotopic Divergence	IDiv	Cucherousset and Villéger, 2015
	Isotopic Dispersion	IDis	
	Isotopic Evenness	IEve	Rigolet <i>et al.</i> , 2015
	Isotopic Uniqueness	IUni	

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12 SIA is increasingly used to describe the structure of trophic webs of ecosystems
 13 associated with AS (Cresson et al., 2019, 2014a; Kang et al., 2008; Rezek et al., 2017; Zhang
 14 et al., 2021). While some authors have investigated the main trophic flows of AS ecosystems
 15 to inform the attraction vs. production debate (Cresson et al., 2019, 2014a), others have
 16 compared the trophic-web structure of communities associated with AS and those associated
 17 with a control natural habitat (Kang et al., 2008; Rezek et al., 2017; Zhang et al., 2021). In
 18 particular, Cresson et al. (2019) and Rezek et al. (2017) used isotopic functional indices (IDiv,
 19 IDis, IEve and IUni for the former authors, IDiv, IDis for the latter) that were previously
 20 defined to describe the trophic structures and temporal variations of communities associated
 21 with AS. All these studies tend to show that long-established AS ecosystems are able to
 22 support trophic structures and pathways similar to those of mature natural ecosystems. Such
 23 results demonstrate that SIA and associated isotopic indices are valuable methods to describe
 24 the functioning of ecosystems associated with AS. Furthermore, the use of SIA along with

1 classical diversity analysis provides complementary information for a better understanding of
2 the effects of AS on ecosystems (Rezek et al., 2017).

4 ***4.2. Trophic-web modelling***

5 Trophic-web modelling tools have been widely used to study ecosystem changes in
6 response to a wide range of perturbations such as fisheries (Corrales et al., 2017; Preciado et
7 al., 2019), invasive species (Baird et al., 2012), harbour extensions (Tecchio et al., 2016),
8 offshore wind farms (Pezy et al., 2020a; Raoux et al., 2019, 2017), eutrophication (Schückel
9 et al., 2015) and dredge spoil dumping (Pezy et al., 2018, 2017). Among the existing
10 modelling approaches, the Ecopath with Ecosim (EwE) software suite has been developed to
11 evaluate ecosystem-based management of fisheries (Christensen and Walters, 2004; Walters
12 et al., 1997) and has been intensively used over the last three decades (Christensen and
13 Walters, 2004). More specifically, Ecopath is designed to build a mass-balance snapshot of
14 the ecosystem functioning while Ecosim simulates the evolution of this ecosystem through
15 time. Since it is difficult to identify the holistic properties of ecosystems by direct
16 observation, Ecological Network Analysis (ENA) is commonly used to capture and assess
17 their complexity. ENA involves an integrative and holistic assessment of the marine
18 ecosystems status by analysing their food web structure, functioning and dynamics (Fath et
19 al., 2007). The output of ENA can be summarized in a large number of indices that describe
20 different aspects of the trophic-web (Table 4 ; de Jonge and Schückel, 2021; Heymans et al.,
21 2014; Rombouts et al., 2013; Safi et al., 2019). Among the most commonly used ENA
22 indices, we can cite: 1) Total system throughput (T) *i.e.* the sum of all flows occurring in the
23 system (Latham, 2006), 2) the System Omnivory Index (SOI) *i.e.* a measure of the trophic
24 specialisation predators and an indicator of the structure and complexity of a trophic network
25 (Libralato, 2008), 3) Finn's Cycling Index (FCI) *i.e.* the proportion of flows in the system
26 generated by recycling (Finn, 1980) and 4) Detritivory over Herbivory ratio (D/H) *i.e.* ratio of
27 detritus consumption compared to the consumption of primary producers (Latham, 2006).
28 Note that several ENA indices have been described as promising indicators of ecosystem
29 health status, and were thus considered to be incorporated into general management strategies
30 such as the OSPAR convention (Convention for the Protection of the Marine Environment of
31 the North-East Atlantic; Table 4; Niquil et al., 2014; Safi et al., 2019). The set of ENA indices
32 given in Table 4 was chosen as it allows us to characterise different trophic functioning
33 attributes, essential for a complete overview of induced changes (Safi et al., 2019, 2017).

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Table 4: Set of ENA indices commonly used along with their acronyms. Cross (X) indicates that the associated index has been either 1) proposed as food web indicator for the OSPAR Regional Sea Convention (Convention for the Protection of the Marine Environment of the North-East Atlantic; Safi et al., 2019) or 2) used in studies focusing on impacts of AS on trophic-web functioning (Raoux et al., 2017; Wu et al., 2019; Xu et al., 2019).

Index	Acronym	Safi et al. 2019	Raoux et al. 2017	Wang et al. 2019	Xu et al. 2019
Ascendency	A		X	X	
Average Mutual Information	AMI	X			
Connectance Index	CI	X		X	X
Detrivory over Herbivory ratio	D/H	X			
Finn's Cycling Index	FCI	X	X	X	X
Interaction Strength	IS	X			
Relative Redundancy	R/DC	X			
System Omnivory Index	SOI	X	X		X
Total System Throughput	T		X		X
Transfer Efficiency over trophic levels	TE	X	X	X	X

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While trophic-web modelling has been commonly applied over several decades predominantly in the field of fisheries management (Cury and Christensen, 2005; Pauly et al., 2000), it is more rarely used to study the effects of AS and has mainly focused on the evaluation of the impacts of offshore wind-farms (OWF; Pezy et al., 2020a, 2020b; Raoux et al., 2019, 2017; Wang et al., 2019). In this context, Raoux et al. (2019, 2017) carried out extensive studies to investigate the impacts of epibenthic community growth and fish aggregation caused by offshore wind farms on the trophic-web functioning in the Bay of Seine (English Channel, France). Using field data, these authors constructed an Ecopath model composed of 37 components or functional groups (from phytoplankton to seabirds) describing the ecosystem before OWF installation. Ecosim was then used to simulate the evolution of this ecosystem for 30 years after OWF installation, by increasing specific fish and benthic components associated with the foundations and scour protections. To investigate the impacts of OWF on ecosystem functioning, several ENA indices (Table 4) and Mean Trophic Level (MTL) were calculated for both trophic models (*i.e.* before and after OWF installation). Based on these indices, it appears that total ecosystem activity, recycling and ecosystem maturity increase after OWF installation. Also, the decrease in MTL suggests that

1 OWF installation causes a trophic-web reconstruction, mainly explained by a large increase in
2 filter feeder biomass. Finally, it appears that some higher trophic levels (*i.e.* piscivorous fish
3 and marine mammals) may respond positively to prey biomass aggregation. Other studies
4 used a similar approach to study the impact of an OWF (Jiangsu coast, China; Wang et al.,
5 2019) and an artificial reef installed for restoration purposes (Laizhou bay, China; Xu et al.,
6 2019). Contrary to Raoux et al. (2019, 2017), who simulated the impact of an AS on
7 ecosystem functioning, the two studies in China used biological field data sampled before and
8 after the AS installation to create Ecopath models. They also used a range of different ENA
9 indices to describe the ecosystem functioning, before and after installation of the AS (Table
10 4). In the same way as Raoux et al. (2017, 2019), Wang et al. (2019) and Xu et al. (2019)
11 were able to highlight that the ecosystem exhibits a higher level of maturity, activity and
12 recycling following the AS installation. Along the same lines, Salaun et al. (submitted), has
13 built an Ecopath model based on biological field data collected before and after the
14 implementation of an artificial reef (South Bay of Biscay, France). Their results are in
15 agreement with the studies mentioned above, indicating that the ecosystem will tend to
16 develop towards higher maturity following artificial reef implementation.

17 While the EwE models are easier to construct and use compared to most other
18 ecosystem modelling approaches, they can be criticized because these are based on a manual
19 balancing procedure without any evaluation of the uncertainty of input parameters (Nogues et
20 al., 2021). Thus, the ecological interpretation of single values of ENA indices mostly relies on
21 non-statistical comparisons with values obtained for ecosystems of the same type (Guesnet et
22 al., 2015). To consider the uncertainty for each input parameter and flow (ENA index values),
23 other mass balance modelling approaches, such as Linear Inverse Models (LIM), are more
24 suitable (Niquil et al., 2012). Thus, Nogues et al. (2021), recently transformed the Ecopath
25 models of Raoux et al. (2017) into LIMs to study the cumulative effects of OWF construction
26 and climate change. Their results are in accordance with the study of Raoux et al (2017)
27 because they show similar changes in ecosystem functioning and structure as well as a shift
28 from a benthic-pelagic system towards a more benthic system (Nogues et al., 2021).

29 Finally, it is noteworthy that SIA can be integrated into trophic-modelling to obtain a
30 better quantification of the uncertainties concerning the trophic flow of some species through
31 the use of isotopic mixing models (Parnell et al., 2010). In addition, SIA - and more
32 particularly the isotope ratios of nitrogen - can be used as a validation tool for determining the

1 accuracy of trophic levels computed by Ecopath models (Lassalle et al., 2014; Raoux et al.,
2 2020).

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5. Recommendations

Ecological indicators are increasingly used to investigate the impacts of diverse anthropogenic activities on marine ecosystems. This trend generates a wide range of new methods, which can lead to complex methodological choices. In our study, we review a large number of these methods. Among these approaches, some were developed specifically for AS (e.g. ARMI of Lima et al., 2020), while others were developed for other primary purposes but then successfully applied in the case of AS (e.g. isotopic indices and ENA indices). Nevertheless, we also highlight a lack of reliable methods concerning some compartments impacted by AS. In this context, it is necessary to adapt or create new indicators to obtain a better characterisation of the ecological impacts generated by AS.

5.1 A multimetric index for epibenthic communities

While the growth of epibenthic communities on AS is of great concern, especially in the field of eco-engineering, no multimetric index has so far been developed aimed specifically at assessing the ecological quality of such communities. A barrier to this development arises from the difficulty of defining “reference” conditions, *i.e.* a condition not impacted by the AS. Indeed, reference conditions have been widely used to calibrate and create various multimetric indices to evaluate the impact of an anthropogenic activity (Dauvin et al., 2010; Lavesque et al., 2009). Nevertheless, reference conditions are difficult to define in our case since AS generate the formation of new epibenthic communities previously absent, and do not arise from the perturbation of a pristine epibenthic community. Although we might consider the use of epibenthic communities associated with natural hard substrates as a reference, such communities are not always present in study sites dominated by soft substrates. Thus, one possibility is to develop a multimetric index which is not necessarily based on a comparison with a reference condition, but rather with other epibenthic communities (e.g. natural hard substratum, submerged blocks of harbours and seawall defences, various types of AS, etc.).

There is a great need to develop such an ecological multimetric index. Such a tool should allow stakeholders to identify easily and robustly the performance of eco-engineered structures in order to enhance future AS deployments. The multimetric indices described above for the epibenthic communities of natural substrates (Table 1) should lay the

1 foundations for such a tool. More specifically, in addition to classical community analyses
2 focusing on biological structure (*i.e.* diversity, evenness and similarity of communities), we
3 stress that complementary approaches should be employed focusing on biological traits. Also,
4 the ratio between the abundance of opportunistic and characteristics species (e.g. CFR;
5 MarMAT; CCO; EEI-c; QiSubMac; Table 1) is particularly well adapted in the case of
6 artificial substrates as it represents a proxy of the ecological succession stage. Finally, it
7 should be a priority to address the major concerns about the potential of AS to facilitate the
8 spread of NIS, by focusing on taxa invasiveness (e.g. the biotic index ALEX, Table 1).

9 To summarize, such a multimetric index should be composed of a set of indicators: 1)
10 to consider both flora and fauna simultaneously, 2) to summarize the structural and functional
11 diversity of the epibenthic community; 3) to estimate the stage of the ecological succession, 4)
12 to assess the ability of the AS to host key-stone species and, 5) to help prevent the
13 establishment of NIS. The calibration and calculation of such a multimetric index and
14 associated sub-indices should involve a dedicated investigation including a case study (see
15 Baux et al., 2020; Lavesque et al., 2009; Lima et al., 2020). Finally, the multimetric index
16 should be validated by testing its robustness on AS with different physical characteristics and
17 dates of submersion.

18 ***5.2 Potential improvement of ARMI***

19 The aggregation of mobile megafauna, especially fish, is historically the most
20 scrutinized impact associated with the presence of AS (Bohnsack and Sutherland, 1985).
21 Nevertheless, the ARMI multimetric index has been recently developed to evaluate the
22 megafauna colonization performance of an AS (Lima et al., 2020). Although ARMI is too
23 recent to have been tested on a wide diversity of sites, the proposed metrics would seem
24 appropriate since they combine the structural and functional aspects of fish assemblages.
25 However, we consider that ARMI can be significantly improved to address its primary
26 objective.

27 Firstly, a major drawback is the absence of a metric focusing on the nursery function
28 of AS. Indeed, the nursery function of AS has been extensively studied, in particular in the
29 field of eco-engineering (Bouchoucha et al., 2016; Mercader et al., 2019, 2017). The ability of
30 AS to facilitate reproduction and spawning of the megafauna plays an important role in the
31 “attraction vs. production” debate and should consequently be taken into account when
32 assessing the ecological effects and performance of these structures. Thus, we suggest that

1 ARMI should be improved by adding a set of metrics focusing on nursery functions. One
2 possibility is to add information about the presence of spawning species and juvenile
3 individuals, in the same way as the MFCI of Henriques et al. (2008).

4 Secondly, another potential improvement is to consider not only fish but also other
5 taxa of the mobile megafauna community, *i.e.* decapods, echinoderms and cephalopods.
6 Indeed, these taxa use AS in a similar way as fish, *i.e.* for shelter and feeding, and can thus
7 compete for the same resources. Considering the megafauna community as a whole would
8 thus provide a fuller picture of the impact of the AS on this compartment. Such minor
9 improvements would allow us to significantly transform the current version of ARMI into an
10 even more robust and sensitive method for evaluating the performance of AS in favouring the
11 aggregation of mobile megafauna.

12 ***5.3 Reliable but rigorous modelling approaches***

13 In order to investigate the functioning of marine ecosystems, in particular the structure
14 and dynamics of trophic webs, different methods have been developed such as stable isotopic
15 analysis and trophic modelling. Their applications in the case of AS has been a success, as
16 demonstrated, for instance, by the studies of Cresson et al. (2019, 2014b, 2014a) and Raoux et
17 al. (2019, 2017). This success can be mainly explained by the methods themselves, which are
18 by definition versatile and designed to characterize the potential modifications of various
19 ecosystems. Thus, their proper application depends on the model quality and the choice of
20 appropriate indices used to define ecosystem properties among the available array (Table 3
21 and 4).

22 Nevertheless, these approaches, and especially modelling, have the drawback of being
23 extremely exacting in terms of input data. Indeed, trophic models require a large amount of
24 biological data, such as the biomass and productivity of each compartment, but also accurate
25 characterizations of the different ecosystem fluxes. Unfortunately, such data are not always
26 available. For instance, biomass data are usually lacking for low trophic-level compartments
27 such as bacteria, zooplankton or meiofauna. The problem is similar concerning ecosystem
28 fluxes, which are only rarely measured *in-situ* by various methods such as isotopic and gut
29 contents analysis used to characterize predation or benthic chambers to quantify primary
30 benthic production. In either case, acquisition of these data would require a significant effort
31 that is sometimes difficult to implement. To overcome such gaps, it is possible to interpolate
32 the data of similar ecosystems and focus the interpretations of model outputs on

1 compartments for which data are available (Raoux, 2017), but this may lead to a reduction in
2 model robustness and quality. In this context, while ecosystem approaches are fashionable
3 and show many advantages for studying the impacts of AS, particular attention must be paid
4 to the quality and quantity of data already collected or possibly available. In the case where
5 the supply of such input data is impossible, it would be reasonable to fall back on less
6 integrative approaches, for instance by focusing on certain compartments of the ecosystem
7 (*i.e.* megafauna, epibenthic communities, etc.).

8 ***5.4 Toward social-ecological approaches***

9 In addition to their primary purpose (e.g. energy production, coastal defence, etc.),
10 secondary services provided by AS are numerous and closely related to their ecological
11 effects. For instance, AS can act as potential fishing and underwater tourism areas due to their
12 colonization by mobile megafauna (Belhassen et al., 2017; Lima et al., 2019). As another
13 example, epibenthic communities associated with these structures may act as biofilters due to
14 their filter-feeding activities (Degraer et al., 2020; Mavraki et al., 2020; Vozzo et al., 2021). A
15 large number of indicators have been developed to describe these different ecological services
16 (Hattam et al., 2015). According to Hattam et al. (2015), we should distinguish three different
17 types of indicator: 1) purely ecological indicators of ecosystem properties (e.g. population
18 characteristics, pollutant fluxes), 2) indicators of ecological processes which contribute to the
19 delivery of a service (e.g. provision of food resources or suitable habitats for key species), and
20 3) indicators of the human use of an ecosystem service (e.g. number of jobs, landings data).

21 Social-ecological approaches bringing together information from both human and
22 ecological subsystems (Österblom et al., 2013; Young et al., 2006) are thus increasingly being
23 used in the case of AS (Lima et al., 2019; Salaün et al., 2020). For instance, Haraldsson et al.
24 (2020) developed a social-ecological model to evaluate the effect of a future offshore wind-
25 farm on the marine ecosystem as well as the local human population. Their model highlighted
26 some indirect and unexpected changes in the social-ecological system. These authors
27 suggested the use of such a method to support decision-making processes in the development
28 of marine renewable energy.

29 Given that the purpose of our review is to focus on the ecological effects of AS, we do
30 not mention in detail the different methods and indicators available to describe socio-
31 economic impacts. Nevertheless, it appears fundamental to take account of social and
32 ecological systems which are closely interacting with each other so we can obtain the fullest

1 possible picture of the impacts of AS. Thus, we consider that the use of social-ecological
2 approaches should be favoured in the future to facilitate decision-making and discussions
3 between stakeholders.

4 5 **6. Conclusions**

6 In this study, we describe a large number of approaches aiming to investigate the
7 ecological impacts of AS. We thus provide stakeholders with a catalogue of useful tools to
8 characterize these ecological impacts, allowing us to draw lessons for future deployments. We
9 also give precise recommendations for the adaptation or creation of novel indicators that are
10 necessary to fill the identified methodological gaps.

11 To provide a comprehensive overview of the ecological modifications generated by
12 AS on marine ecosystems, the use of integrated assessments focusing on all of ecosystem
13 compartments must be prioritized by stakeholders. However, such integrated approach often
14 requires substantial fundings which are not always available, particularly for developing
15 countries. In such cases, the selection of a few strategic methods must be done. Selecting the
16 appropriate method will clearly depend on the expected results obtained from the ecological
17 studies carried out. For instance, if the objective is to investigate the performance of an eco-
18 engineered AS, the compartment targeted by the eco-design should be more finely examined.
19 For example, an AS eco-design based on surface features will impact epibenthic community
20 growth while an eco-design creating a variety of shelters will favour megafauna aggregation.
21 On the other hand, if the issue at stake is more concerned with the global impacts caused by
22 one or several AS, it would be reasonable to favour the ecosystem level approach. Since there
23 are as many indicators as there are types of data needed, the indicator chosen to investigate
24 the impacts of AS should necessarily be in keeping with the data at our disposal. In any case,
25 there is no universally applicable method as each case study will have different ecological and
26 socio-economical contexts or primary purpose. It will be the role of stakeholders and
27 scientists to work together to set the appropriate framework, allowing us to define suitable
28 objectives and target the choice of methodology.

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8

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