GECOMARS 2020

International workshop on ecosystem based management

4-5 February 2020, Marseille (France)

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PREFACE

GECOMARS : INTERNATIONAL WORKSHOP ON ECOSYSTEM BASED MANAGEMENT, MARSEILLE, FRANCE, 4-5 FEBRUARY 2020

The 'species-by-species', or 'stock-by-stock' approach in the case of fisheries, characterized the 20th century ecology. The ecosystem-based approach, which in the case of fisheries emerged at the end of the 20th century, represents the 'new frontier', the 21st century revolution in ecology.

According to the Marine Strategy Framework Directive, the ecosystem-based approach (EA) should enable us to understand and assess the functioning of marine and coastal ecosystems and their dependent services. As an integrated management process, this strategy promotes both conservation and sustainable use and provides benefits in a more equitable way via the social-ecosystem concept. First used for fisheries management, the EA has become a valuable tool for ecosystem services assessment and for marine protected areas (MPA) monitoring and governance, especially in the Mediterranean Sea but it could also be considered in the spatial planning and the management of other coastal areas. The perception that humans belong to ecosystems (socio-ecosystems) is a key feature of the EA and the catastrophic events due to anthropic pressures, in the global change context, remind us the price to pay.

The aim of organizing a Workshop on Ecosystem-based Management in Marseille was to create the opportunity to gather managers, stakeholders and scientists to discuss this crucial topic within the framework of the Integrated LIFE Marha project, headed by the French Office of Biodiversity and funded by the European Union.

Aix Marseille University, Pythéas and the Mediterranean Institute of Oceanography, the Marseille's city Tourism Office and Toulon Provence Metropole have also supported the organization of the event.

The upcoming challenge, in the current Global change, is to move forward to a suitable and carbon free management and use of our natural ecosystems. Only a concern at the Ecosystem scale will make it possible.

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ECOSYSTEM-BASED QUALITY INDICES: VALUABLE TOOLS FOR ENVIRONMENT MANAGEMENT

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ECOLOGICAL INDICATORS
ECOSYSTEMS
ECOSYSTEM-BASED APPROACH
MANAGEMENT
MEDITERRANEAN SEA

ABSTRACT. – Environmental issues have been addressed on the basis of three different approaches. (i) The earliest is the 'Human-centered' approach; it was characterized by the dichotomy between 'useful' species (for Man) and pests (competitors of humans). (ii) The species-centered approach was characteristic of the 20th century and remains the most common approach adopted in many countries and by several international agencies. It is based upon the notion of outstanding species, which are designated as deserving appropriate management, in contrast to 'ordinary' species. (iii) Finally, the 21st century ecosystem-based approach is the one that can best meet the challenges driven by global change and ensure the proper management of natural habitats. In contrast with indices based upon a species, or a group of species belonging to a given taxon, that may not detect a strong impact on the ecosystem, and even erroneously suggest a 'good ecological status', indices based on the functioning of the entire ecosystem, from primary producers to top predators, such as Ecosystem-Based Quality Indices (EBQIs), provide a realistic assessment of the ecological status. EBQIs have already been established for northwestern Mediterranean marine ecosystems: the *Posidonia oceanica* seagrass meadow, underwater marine caves and infralittoral reef macroalgal forests. They are currently being developed for coralligenous habitats, saltmarshes and circalittoral coastal detritic bottoms. The ecosystem-based approach can be applied to all types of ecosystem and it is important now to extend this approach to other ecosystems and regions. Ecosystem-based management and EBQIs are not incompatible with specific management measures based upon certain iconic species, which are also part of an ecosystem. The interest of ecosystem-based management is that it is not limited to the accumulation of specific management measures for iconic species, which can be mutually incompatible.

INTRODUCTION

To put things simply, we may consider that environmental issues have been addressed, successively over time or simultaneously, on the basis of three different approaches (Boudouresque *et al.* 2020). (i) The earliest is the 'human-centered' approach; this was characterized by the dichotomy between 'useful' species (for man) and pests (competitors of humans). The concept of ecosystem goods and services can be considered as a modern form of this approach (Balmford & Bond 2005, Pesche *et al.* 2013, Nordlund *et al.* 2016, Paoli *et al.* 2017). (ii) The species-centered approach (or species-by-species approach) was characteristic of the 20th century and remains the most common approach adopted in many countries and by several international agencies; it is supported by groups of experts working on a given taxon ('taxonomic lobbies'). It is based upon outstanding species (a fuzzy concept, including iconic species), which are designated as deserving appropriate management for a variety of reasons (attractive in appearance, rare, emblematic, threatened, etc.), in contrast to 'ordinary' species. (iii) Finally, the 21st century ecosystem-based approach, although still rarely used, is the one that can best embrace the challenges driven by global change and ensure the proper management of natural habitats (see below).

In order to assess the quality of the natural environment, the measurement of the physical-chemical parameters is necessary, but very insufficient and often meaningless: it is not the content in the water of a contaminant (*e.g.*, mercury) which is important *per se*, but its possible impact on individuals, populations or ecosystems (Alava *et al.* 2018, Outridge *et al.* 2018). It is for this reason that biological indicators, describing the state of environmental health on the basis of species, have been developed. In addition, the species integrate the characteristics of the environment over their entire lifespan (from a few months to several decades); their presence or absence is therefore easier to interpret, and less costly in terms of time and money, than physical-chemical measurements which are extraordinarily variable from one hour to the next, from one day to the next, etc. (*e.g.*, Pergent 1991, Casazza *et al.* 2002, Dauvin *et al.* 2010, Romero *et al.* 2015).

Ideally, a biological indicator should be (i) sufficiently sensitive to provide an early warning of change, (ii) based on species distributed over a broad geographical area, (iii) capable of providing a continuous assessment over a wide range of stress, (iv) relatively independent of sample size, (v) easy and cost-effective to measure, (vi) able to differentiate natural cycles or trends from those induced by humans, and (vii) relevant to ecologically significant phenomena (Noss 1990, Rombouts *et al.* 2013).

Here, we retrace the history of biological indicators, based on one or more taxa (taxon-based indicators), or on all of the taxa and their interactions in the framework of the ecosystem (ecosystem-based indicators) in the Mediterranean. Without questioning the usefulness of indicators based on a single taxon, which perfectly meet the objective assigned to them, we show the leap forward represented by indicators based on the ecosystem, in terms of management of natural environments and particularly coastal marine habitats.

THE AGE OF TAXON-BASED INDICATORS

In the framework of European Union (EU) directives, mainly the Habitats Directive of 1992 (HD: 92/43/ECC), the Water Framework Directive of 2000 (WFD: 2000/60/EC) and the Marine Strategy Framework Directive of 2008 (MSFD: 2008/56/EC), a number of biological indices have been developed. Their aim is to assess the water quality, at local scale or at the scale of large water bodies. Some of them are particularly efficient and are today widely used to monitor the water quality and to assess its required improvement at the scale of the EU coastline and that of some neighboring countries. They can be grouped into three main categories. (i) Indices based on morpho-functional groups of macroalgae; the EEI (Ecological Evaluation Index) compares opportunistic (r) *vs* K strategist groups; it has been developed and steadily improved by Orfanidis *et al.* (2001, 2003, 2011) and Simboura *et al.* (2005) (but see Iveša *et al.* 2009). (ii) The CARLIT index is based upon a dozen species and groups of species thriving in the upper infralittoral and in the midlittoral stages (stages *sensu* Pérès & Picard 1964), *e.g., Cystoseira* spp. (brown algae), *Ulva* sp. (green algae), articulated corallines (red algae) and the mussel *Mytilus galloprovincialis*. The very shallow or above sea level habitat of the taxa taken into account enables the exhaustive mapping of the coastline over hundreds and even thousands of kilometers (Ballesteros *et al.* 2007, Bermejo *et al*. 2013, Nikolić *et al.* 2013, Blanfuné *et al.* 2016, Torras *et al.* 2016, Blanfuné *et al.* 2017, De la Fuente *et al.* 2018). (iii) A number of biological indices are based on the seagrass *Posidonia oceanica* (Linnaeus) Delile, which plays a pivotal role in the functioning of Mediterranean coastal areas, and is sensitive to a wide range of human impacts (Molinier & Picard 1952, Boudouresque *et al*. 2006, 2009, 2012, Bonhomme *et al.* 2013, Giakoumi *et al.* 2015, Boudouresque *et al.* 2016). The BiPo index (Biotic Index based on *Posidonia oceanica*) combines four metrics: maximum depth of the meadow, declining or progressing trend of the depth limit, shoot density and leaf surface area at 15 m depth (Lopez y Royo *et al.* 2010). The PREI index (*Posidonia oceanica* Rapid Easy Index) includes the same metrics plus a fifth, the ratio between epiphyte biomass and leaf biomass (Gobert *et al.* 2009). The POMI index (*Posidonia oceanica* Multivariate Index) combines 11 metrics at the physiological level (*e.g.*, nitrogen, phosphorus and carbohydrate content in rhizomes), the individual level (*e.g.*, leaf surface area per shoot), the population level (shoot density, % of plagiotropic rhizomes, meadow cover) and the contamination level (nitrogen content of leaf epibionts and trace metals) (Romero *et al.* 2007). These indices are robust, with congruent results when compared with each over (Bennett *et al.* 2011, Lopez y Royo *et al.* 2011, Mascaró *et al.* 2012, Gerakaris *et al.* 2017). Other taxon-based indicators have been proposed: Conservation Index (CI) (Moreno *et al.* 2001), Substitution Index (SI) (Montefalcone *et al.* 2007a), Phase Shift Index (PSI) (Montefalcone *et al.* 2007b, Montefalcone 2009, Rigo *et al.* 2019). According to Boudouresque *et al.* (2012), CI, SI and PSI can be relevant to assess changes over time in *P. oceanica* seagrass meadows linked to global change (warming, anthropogenic impact, community shift, etc.). Finally, the content in phenolic compounds increases with stress and represents a generic indicator of different environmental stressors (Mannino & Micheli 2020).

The relevance of these biological indices has been validated by putting them in correlation with some of the anthropogenic pressures impacting the study area (*e.g.*, land area covered by urbanization, industrial and agricultural activities, particulate organic matter and nitrogen input), *via* pressure indices such as LUSI (Land Use Simplified Index) and HAPI (Human Activities and Pressure Index) (Flo *et al.* 2011, Bacci *et al.* 2013, Blanfuné *et al.* 2017). The MCAI (Multi-criteria Anchoring Index) measures the impact of anchoring on *Posidonia oceanica* meadows (Rouanet *et al.* 2013, Schohn *et al.* 2019). The relevance of these biological indices has also been validated through their ability to detect changes over time; these changes reflect the effectiveness of the EU water quality improvement policy (establishment of sewage treatment plants, reduction of air pollution, a major source of sea water contamination, etc.) (Blanfuné *et al.* 2017, De la Fuente *et al.* 2018, Shin *et al.* 2018).

THE AGE OF ECOSYSTEM-BASED INDICATORS

The MSFD is considered to be the environmental pillar of the Integrated Maritime Policy adopted in 2010 by the European Commission (IMP: 2010/477/EU). This directive established eleven criteria, to determine 'good environmental status' (GES): (i) biological diversity is maintained; (ii) introduced species are at levels that do not adversely alter the ecosystems; (iii) populations of all exploited fish and shellfish are safely within biological limits; (iv) all elements of the food webs are maintained at adequate levels to ensure the long-term abundance of the species; (v) human-induced eutrophication is at a minimum; (vi) sea-floor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded; (vii) permanent alteration of hydrographical conditions does not adversely affect marine ecosystems; (viii) concentrations of contaminants are at levels that do not give rise to pollution effects; (ix) contaminants in fish and other seafood do not exceed levels established by Community legislation; (x) properties and quantities of marine litter do not cause harm to the coastal and marine environment; and (xi) inputs of energy are at levels that do not adversely affect the marine environment.

The MSFD includes a major innovation: the ecosystem-based approach (Laffoley *et al.* 2004, Bryhn 2020); it appears in particular in criteria ii, iv, vi and vii. This approach is not to the taste of supporters of the species-by-species approach nor of the taxonomic lobbies, who seek to promote their beloved taxon (marine mammals, sea turtles, etc.), confuse habitat and ecosystem, and find it difficult to reflect at the level of the ecosystem. However, taxon-based indicators and ecosystem-based indicators are neither mutually exclusive, nor in opposition to each over, but simply complementary: they just do not measure the same thing (see below).

Fisheries managers were the pioneers of the ecosystem-based approach, under the names of EAF (Ecosystem Approach to Fisheries), EAFM (Ecosystem Approach to Fisheries Management) and EBFM (Ecosystem-Based Fishery Management) – Hereafter EBFM (Ward & Hegerl 2003, Pikitch *et al.* 2004, Turrell 2004, Rice 2005, Tudela & Short 2005). According to Turrell (2004), "*1. All aspects of the ocean are interrelated and should be treated as an integral whole. 2. In order to achieve a more rational management of resources and thus to improve the environment, States should adopt an integrated and co-ordinated approach to their development planning so as to ensure that development is compatible with the need to protect and improve environment for the benefit of their population*". But in fact, the real pioneers and promoters of the ecosystem-based approach were the authors of the Ecopath, the Ecopath with Ecosim and the Osmose models and their subsequent users (*e.g.*, Christensen & Pauly 1992, 1993, Polovina 1993, Opitz 1996, Walters *et al.* 1997, Bănaru *et al.* 2013, Coll *et al.* 2015, Piroddi *et al.* 2017, Bănaru *et al.* 2019, Hermosillo-Núñez 2020).

The taxon-based indicators do not provide information on the quality of the ecosystem, but on the quality of the water bodies: water transparency, nutrient and contaminant content, etc. It is, moreover, for this latter objective that they were designed. The quality of the ecosystem naturally depends on the quality of the water, but other parameters can be more important: habitat destruction, overfishing, biological invasions, etc. This obvious point was illustrated in a diagram, in a somewhat caricatural way, by Boudouresque *et al.* (2015) (Fig. 1).

MEDITERRANEAN ECOSYSTEM-BASED INDICATORS

The rationale governing the EBQIs (Ecosystem-Based Quality Indices) is based on (i) attempting to quantify and assess some compartments (*e.g.*, boxes 1 through 13 for the *Posidonia oceanica* ecosystem – EBQI/Pos; Fig. 2) of the conceptual model by means of a set of parameters, (ii) determining their relative weight and (iii) by using a

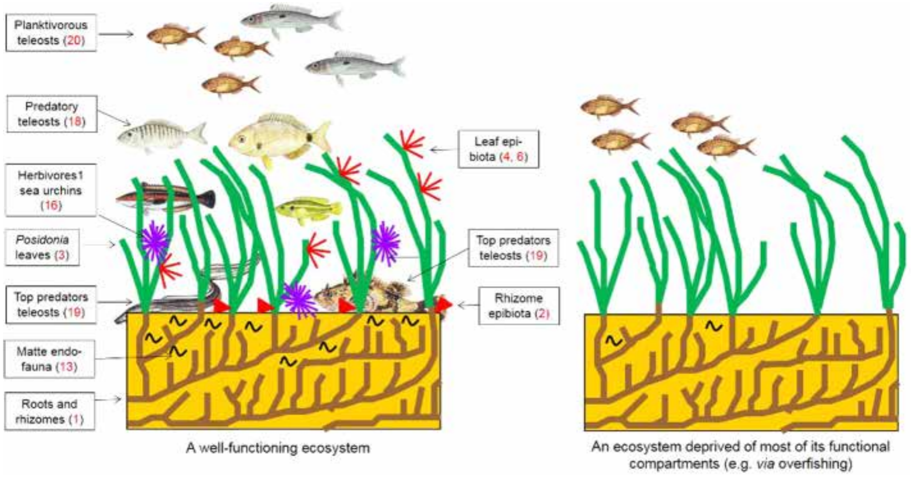

Fig. 1. – *Left.* A pristine *Posidonia oceanica* ecosystem, with species belonging to all functional compartments (for compartment numbers, see Fig. 2). *Right.* A *P. oceanica* meadow deprived of most of its functional compartments (*e.g.*, *via* overfishing), which could be considered as healthy on the basis of taxon-based indicators, based upon metrics such as shoot density and meadow coverage. From Boudouresque *et al.* (2015), modified and redrawn).

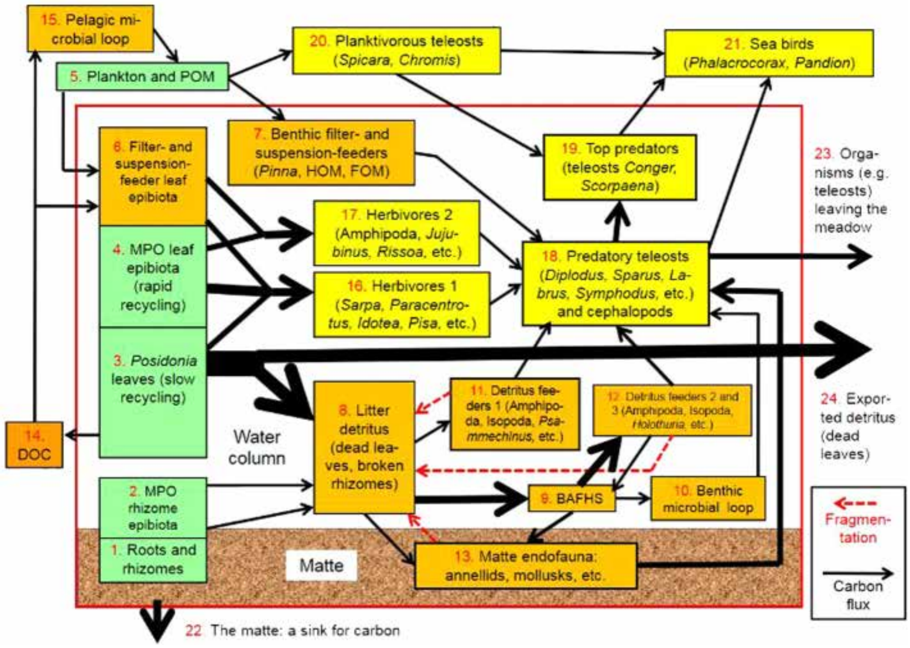

Fig. 2. – A conceptual model of the functioning of the *Posidonia oceanica* seagrass ecosystem. Functional compartments *(boxes)*: primary producers are in *green*; filter feeders, suspension feeders, litter, detritus feeders, dissolved organic carbon (DOC) and microbial loops are in *orange*; predators (including herbivores) are in *yellow*. POM: particulate organic carbon. BAFHS (bacteria, archaea, fungi and heterotrophic stramenopiles) are involved in the litter degradation. The width of the arrows roughly represents the volume of the carbon flow. The *P. oceanica* ecosystem properly speaking is included within the red rectangle. Boxes 1 through 13 correspond to the compartments *(boxes)* taken into account by the EBQI/Pos. From Boudouresque *et al.* (2015), adapted.

simple algorithm, calculating a rank for the ecosystem status within a given area, matching the five classes of the ecological status of the EU Water Framework Directive (WFD), from bad to high. In addition, (iv) it should be based on metrics that are easy to measure, species that do not require great taxonomic expertise for determination, and therefore able to be implemented routinely by managers; this implementation may require that a training course be organized for managers. EBQIs are a compromise between the completeness of the assessment and the need for an approach that is not too time-consuming (*e.g.*, less than one day for 4 divers per site for EBQI/Pos) (Ruitton *et al.* 2013, Personnic *et al.* 2014, Boudouresque *et al.* 2015, Ruitton *et al.* 2017).

The status of each functional compartment (box) is assessed by means of a semi-quantitative scale (4 through 0), from very good (4) to very bad (0). Calibration of the scale is based upon the available literature. Compartments are weighted according to their relative importance in the ecosystem functioning, from 5 (highest weighting) to 1 (lowest weighting). The grade for each compartment is determined by its status (0 through 4), multiplied by its weighting (1 through 5), and is therefore graded from 0 to 4 and 0 to 20 (depending upon the weighting of the considered compartment). The grades of all compartments are added up to give the final grade for the ecosystem status (EBQI) at a given site. For practical purposes, the EBQI is converted to a scale from 0 to 10 (*e.g.*, Table I for EBQI/Pos) (Personnic *et al.* 2014, Boudouresque *et al.* 2015, Ruitton *et al.* 2017). In the case of the EBQI/Pos, which considers 13 functional compartments:

$$EBQI = \left[\sum_{i=1}^{13} \left(W_i \times S_i \right) / \sum_{i=1}^{13} \left(W_i \times S_{max} \right) \right] \times 10$$

where: W_i is the weighting of the box i, S_i the status of the box i, S_{max} the highest possible grade (= 4) for a box and i is the number of the box (1 through 13).

For each box, each value of the box status and each site, a Confidence Index (CI) is proposed (Table II). The reason for the CI is (i) that data for one or several compartments may be missing or of poor quality at some sites, (ii) the reliability of available data may be different between boxes and sites, and (iii) it is worth drawing the attention of managers and scientists to those boxes (compartments) that are poorly known and which merit further field studies. The grade of each considered box is given by its CI (0 through 4) and by its weighting (1 through 5), and they are therefore graded from 0 through 20 (for EBQI/Pos). The grades of all considered boxes are added up, which gives the final grade for the CI at a given site. For practical purposes, the CI was converted to a scale from 0 to 4 (Personnic *et al.* 2014, Ruitton *et al.* 2017). In the case of the EBQI/Pos, which considers 13 functional compartments:

$$CI_{EBQI} = \left[\sum_{i=1}^{13} \left(W_i \times CI_i \right) / \sum_{i=1}^{13} \left(W_i \times CI_{max} \right) \right] \times 4$$

where W_i is the weighting of the box i, CI_i the Confidence Index of the box i, CI_{max} the highest possible Confidence Index (= 4) for a compartment (box) and i is the number of the box (1 through 13).

The first attempt to build an EBQI concerned the *Posidonia oceanica* ecosystem (EBQI/Pos) (Personnic *et al.* 2014, Boudouresque *et al.* 2015, Ruitton *et al.* 2017), one of the Mediterranean ecosystems for which the data available on the functioning and the different compartments are the most extensive (*e.g.*, Bell & Harmelin-Vivien 1983, Mazzella *et al.* 1992, Pergent *et al.* 1994, 1997, Boudour-

Table I. – Conservation status of the *Posidonia oceanica* ecosystem (EBQI/Pos) in Balearic Islands, Spanish and French Catalonia, West and East Provence, French Riviera and Corsica. For each compartment (see Fig. 2): the weighting (1 through 5) and the status grade (0 through 4) at the 17 studied localities. EBQI ranges from 0 to 10. SRDI: Specific Relative Diversity Index of fish. Ecological status classes: high (deep blue), good (light blue), moderate (green), poor (orange) and bad (red). From Boudouresque *et al.* (2015), adapted.

	Compartment	1	2	3-4	5	6	7	8	9	10	11	12	SRDI	13	EBQI.Pos
	Weight	3	5	4	2	2	2	2	5	5	5	3	3	1	
	Espardell (Balearic Islands)	4	4	3	3	3	3	3	3	1	1	1	3	2	6.4
	Sitges (Spanish Catalonia)	2	0	0	0	2	2	2	3.5	0	0	1	0	0	2.3
	Tossa de Mar (Spanish Catalonia)	2	3	4	0	2	2	4	3	2	0	2	3	1	5.6
L	Medes Islands (Spanish Catalonia)	2	3.5	4	3	2	2	2	2.5	4	4	3	4	2	7.9
O	Peyrefite Bay (French Catalonia)	2	3.5	2	4	2	2	2	2	3	1	1.5	4	0	5.8
C	Niolon (West Provence)	2	2.5	2	0	1.5	1	3	2	1	0	2	2	1	3.9
A	Prado Bay (West Provence)	2	2.5	2	0	2.5	2	3	2.5	3	1	1.5	3	2	5.3
L	Plateau des Chèvres (West Provence)	2	2.5	4	0	1.5	2	3	2.5	2	1	0.5	2	2	5.0
I	Saint-Cyr Bay (East Provence)	1	3	2	1	2	2	2	2	2	2	2	2	0.5	4.9
T	Gulf of Giens (East Provence)	3	4	2	2	2	1	3	1.5	1	0	1	1	0.5	4.3
I	Porquerolles North (East Provence)	3	2	3	2	2	0	1	1.5	1	1	2	2	1	4.3
E	Porquerolles South (East Provence)	3	4	4	3	3	3	2	2	2	2	3	3	1	6.9
S	Bagaud Pass (East Provence)	4	3	2	4	3	4	4	3	3	2	3	4	1	7.6
	Port-Cros South (East Provence)	4	4	4	4	3	4	3	3.5	4	4	3	4	1.5	9.3
	Villefranche Bay (French Riviera)	2	1.5	2	1	3	0	0	2	3	2	1.5	4	0	4.8
	Elbu Bay, Scàndula (Corsica)	4	3	1	4	3	2	2	2	2	1	1.5	3	4	5.7
	Valincu Gulf (Corsica)	4	3	2	2	2	2	3	2	2	2	2	2	1	5.4

Table II. – Criteria to assess the Confidence Index (CI) of the status of a compartment.

CI	Criteria
4	Field data available, recent and suitable with the recommended methods
3	Field data recent, partially completed with expert judgment
2	No quantitative field data but recent expert judgment
1	No quantitative field data, but non-recent expert judgment
0	No quantitative field data and no suitable expert judgment

esque *et al.* 2006, 2012, Deudero *et al.* 2014, Giakoumi *et al.* 2015, Ourgaud 2015, Ourgaud *et al.* 2015, Boudouresque *et al.* 2016).

Subsequently, the concept of EBQI was generalized and theorized (Ruitton *et al.*, 2013, 2017): (i) construction of a conceptual model of the ecosystem on the basis of existing literature; (ii) evaluation of carbon and/or nutrient flows between compartments, with weighting of their volume when possible (Fig. 3); (iii) choice of compartments that can be taken into account routinely (minimization of the sampling effort); (iv) weighting of these compartments (on a scale of 1 to 5) (Fig. 3); (v) assessment of the ecological status of the compartments taken into account (Fig. 4); (vi) calculation of the EBQI and CI for each locality. The decisions concerning all these steps are made through an expert meeting and a Delphi process (see Dalkey & Helmer 1963 for the Delphi process).

EBQIs were proposed for the Mediterranean undersea cave ecosystem, EBQI/Caves (Rastorgueff *et al.* 2015, Ruitton *et al.* 2017) (Fig. 5) and for shallow rocky reefs dominated by macroalgae, EBQI/Reefs (Ruitton *et al.* 2017, Thibaut *et al.* 2017) (Fig. 6). Undersea caves are remarkable infralittoral and circalittoral habitats widespread throughout the Mediterranean Sea (*e.g.*, Marseille area, Croatia) (Chevaldonné & Lejeusne 2003; Surić *et al.* 2010). They often originate from the marine flooding of karstic networks during the postglacial maximum transgression; they harbor specialized species, which are often Mediterranean endemics; some of them are regular bathyal and abyssal dwellers, which find in these caves environmental conditions similar to those of the deep sea (Harmelin *et al.* 1985, Vacelet *et al.* 1994, Bianchi *et al.* 1996, Janssen *et al.* 2013, Rastorgueff *et al.* 2015). Shallow rocky reefs dominated by macroalgae are also an infralittoral ecosystem. The ecosystem is characterized by *Cystoseira* (long-living brown algae – Stramenopiles; *Cystoseira sensu lato*) forests which can shift to barren grounds when herbivorous sea urchins (such as *Paracentrotus lividus* (Lamarck, 1816) proliferate; this proliferation is related to a disturbance of the ecosystem, such as overfishing of fish predators of sea urchins and organic pollution (Sala & Zabala 1996, Bonaviri *et al.* 2011, Boudouresque & Verlaque 2013, Ling *et al.* 2015, Thibaut *et al.* 2017).

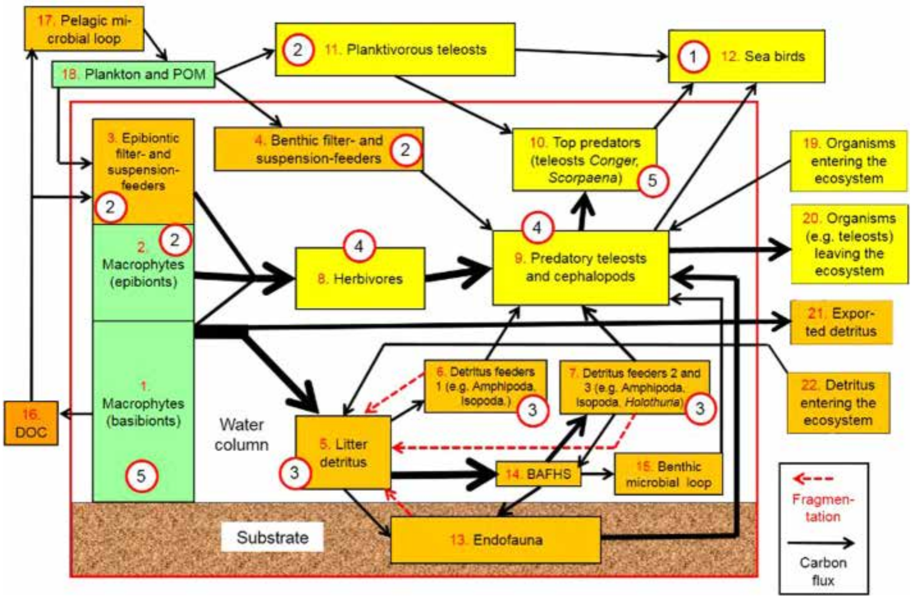

Fig. 3. – Conceptual model of a theoretical ecosystem (fictitious data). Compartments 1 through 12 are taken into account for designing the EBQI. Their weight (numbers in red circles) is established *via* expert judgement by means of a Delphi process. Primary producers are in green; filter feeders, suspension feeders, litter, detritus feeders, dissolved organic carbon (DOC) and microbial loops are in orange; predators (including herbivores) are in yellow. POM: Particulate Organic Matter). BAFHS: bacteria, archaea, fungi and heterotrophic stramenopiles involved in the litter degradation. The width of the arrows roughly represents the volume of the carbon flow. The ecosystem properly speaking is included within the red rectangle. From Ruitton *et al.* (2017), adapted and redrawn.

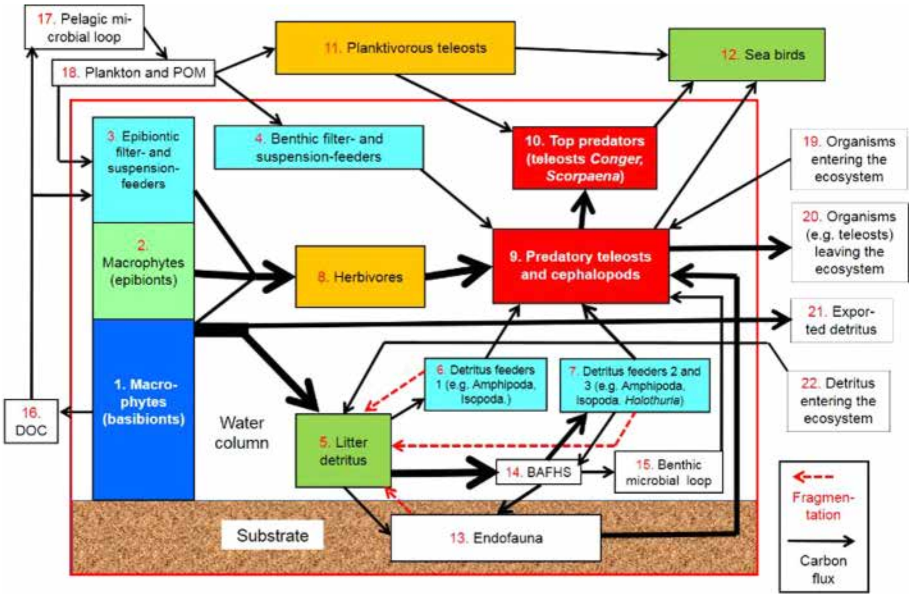

Fig. 4. – Conceptual model of a theoretical ecosystem (fictitious data). Assessment of the ecological status of the compartments taken into account: high (deep blue), good (light blue), moderate (green), poor (orange) and bad (red). From Ruitton *et al.* 2017, adapted and redrawn.

EBQIs are being developed for the coralligenous ecosystem (EBQI/Cor) (Ruitton *et al.* 2014, 2017), salt marshes and coastal lagoons (EBQI/sm) (Massinelli *et al.* 2017, Astruch *et al.* 2019a) and circalittoral coastal detrital sandy bottoms (EBQI/cd) (Astruch *et al.* 2019b). They are at the conceptual model development stage, choosing the compartments that can be used routinely and testing the metrics. The EBQI/Cor is even ready for publication (Ruitton *et al.* 2017).

Certain works, although they do not lead to the calculation of an EBQI, can clearly be referred to as exemplifying an ecosystem-based approach. For example, Bănaru *et al.* (2010) show in the Black Sea the switch from a complex top-down and bottom up functioning pattern of the coastal ecosystem (1965-1970) to a bottom-up pattern (2001-2005). The end-to-end model of the Gulf of Lions ecosystem (NW Mediterranean) is also referable to an ecosystem-based approach (Bănaru *et al.* 2019). They explicitly detail trophic flows between food web compartments, highlight the main primary producers and successive consumers as well as dominant and key species. Fisheries pressure on food web compartments and their direct and indirect effects may also be highlighted (Bănaru *et al.* 2013). These models allow depiction the combined effects of both climate and fisheries on the system functioning (Bănaru *et al.* 2019, Diaz *et al.* 2019). Astruch *et al.* (2019a) highlighted the need for an EBA within the saltmarshes of Hyères (Provence, France) as a way to make the management system more appropriate, evolving from a previous species-centered approach (*e.g.*,

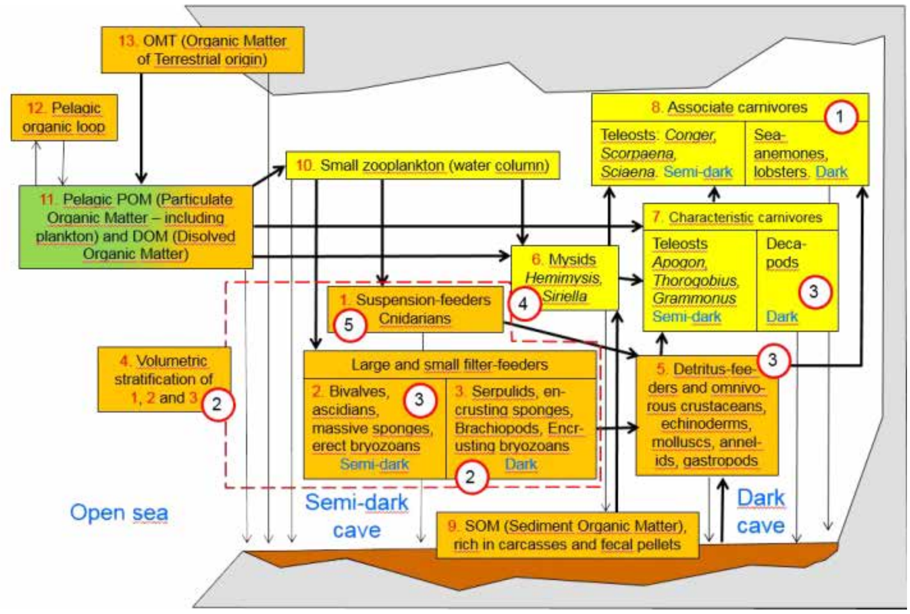

Fig. 5. – Conceptual model of structure and functioning of the Mediterranean undersea cave ecosystem. Compartments 1 through 8 are taken into account for designing the EBQI/Caves. Their weight is indicated by numbers in red circles. Primary producers are in green; filter feeders, suspension feeders, litter, detritus feeders, dissolved organic carbon (DOC) and microbial loops are in orange; predators (including herbivores) are in yellow. Distinction between semi-dark and dark caves appears in the bottom of concerned compartments. Arrows represent the intensity of the flux of organic matter between compartments (wide *vs* narrow arrows). From Rastorgueff *et al.* (2015) and Ruitton *et al.* (2017). Adapted and redrawn.

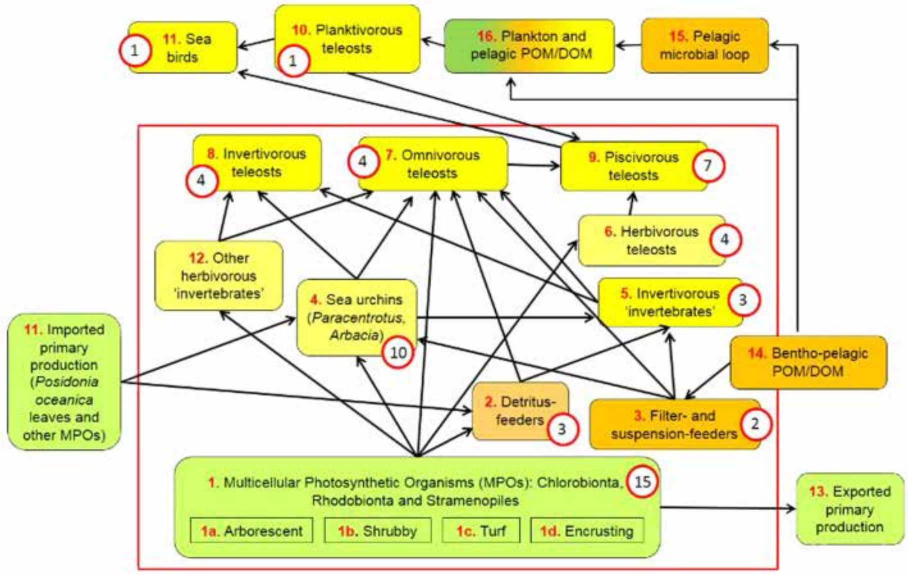

Fig. 6. – Conceptual model of structure and functioning of the Mediterranean shallow rocky reefs dominated by macroalgae. Compartments 1 through 10 are taken into account for designing the EBQI/Reefs. Their weight is indicated by numbers in red circles. Primary producers are in green; filter feeders, suspension feeders, litter, detritus feeders, dissolved organic carbon (DOC) and microbial loops are in orange; predators (including herbivores) are in yellow. The ecosystem properly speaking is included within the red rectangle. Arrows represent the flux of organic matter between compartments. From Thibaut *et al.* (2017) and Ruitton *et al.* (2017). Adapted and redrawn.

waterfowl enhancement, fight against so-called 'harmful species' such as the red fox *Vulpes vulpes*, etc.).

ECOSYSTEM-BASED INDICATORS: VALUABLE TOOLS FOR COASTAL MANAGEMENT

Environmental management is often perceived by managers and the general public as 'firefighter's work', which consists of putting out fires, running from one fire to another. A 'fire' is a species of which the abundance increases, or on the contrary decreases, a changing landscape, etc. This management approach is also deeply biased by taxonomic lobbies: a species of dolphin or bird, even if nothing indicates a decline, is the subject of multiple management programs, while a species of insect or macroalga, seriously threatened or even on the brink of extinction, is of little interest to NGOs and the general public (Boudouresque 2002, Thibaut *et al.* 2016, Verlaque *et al.* 2019). It is significant that (since 1992) 46 % of the European programs intended for the protection of the environment are dedicated to birds, against 26 % to mammals, 8 % to 'invertebrates' 8 % to flowering plants and 6 % to 'fish'; insects and macroalgae are not even mentioned (Tempier 2018, Mammides 2019). Members of a taxonomic lobby are not aware of this, because they are really attached to their preferred taxon, but management based on taxa is sometimes akin to a *millefeuille* (a French multi-layered cake): a layering of taxon-focused protection measures, the addition of measures to enhance

Table III. – Comparison between EBQI/Pos (from Personnic *et al.*, 2014) with taxon-based indices based mainly upon *Posidonia oceanica* (the organism itself) and aimed at establishing the ecological status of a seawater body. Ecological status: high (deep blue), good (light blue), moderate (green), poor (orange) and bad (red). From Personnic *et al.* (2014) and Boudouresque *et al.* (2015), adapted. EBQI/Pos ranges from 0 (lowest ecological status) to 10 (highest ecological status. PREI, POMI and BiPo indices, based upon distinct but similar metrics, range from 0 (lowest ecological status) to 1 (highest ecological status). * See text for the metrics of PREI, POMI and BiPo.

Locality (region)	EBQI/Pos	Taxon-based index	Type of taxon-based index	Reference
Port-Cros Island, south (East Provence)	**9.3**	**0.80**	PREI*	Gobert *et al.* (2009)
Medes Islands (Spanish Catalonia)	**7.9**	**0.75**	POMI*	Romero *et al.* (2007)
Elbu Bay, Scàndula (Corsica)	**5.7**	**0.80**	BiPo*	Lopez y Royo *et al.* (2010)
Tossa de Mar (Spanish Catalonia)	**5.6**	**0.68**	POMI	Romero *et al.* (2007)
Valincu Gulf (Corsica)	**5.4**	**0.39**	PREI	Gobert *et al.* (2009)
	5.4	**0.73**	BiPo	Lopez y Royo *et al.* (2010)
Prado Bay, Marseilles (West Provence)	**5.3**	**0.64**	PREI	Gobert *et al.* (2009)
Plateau des Chèvres, Marseille (West Provence)	**5.0**	**0.48**	PREI	Gobert *et al.* (2009)
Saint-Cyr Bay (East Provence)	**4.9**	**0.68**	PREI	Gobert *et al.* (2009)
Villefranche-sur-Mer Bay (French Riviera)	**4.8**	**0.28**	PREI	Gobert *et al.* (2009)
Gulf of Giens (East Provence)	**4.3**	**0.71**	PREI	Gobert *et al.* (2009)
Porquerolles North (East Provence)	**4.3**	**0.82**	PREI	Gobert et al. (2009)
Niolon, Côte Bleue (West Provence)	**3.9**	**0.47**	PREI	Gobert et al. (2009)
Sitges (Spanish Catalonia)	**2.3**	**0.24**	POMI	Romero *et al.* (2007)

a declining species or to hinder another species which is proliferating, regardless of the fact that the latter may be the predator of the former. The indices based on a taxon were not designed to manage taxa, but to assess the quality of a water body, and the effectiveness of measures intended to improve this quality, for example the establishment of sewage treatment plants. But for many stakeholders, if the indices based on *Posidonia oceanica* show positive results, it means that *P. oceanica* is fine, and if *P. oceanica*, an ecosystem engineer, is fine, the whole environment is fine. However, this is not always the case, as shown in Fig. 1: *P. oceanica* can thrive in a highly degraded ecosystem.

There is no significant correlation between taxon-based indices and ecosystem-based indices, as shown by the comparison between the EBQI/Pos and taxon-based indices (PREI, POMI, BiPo) for the same localities (Table III) (Personnic *et al.* 2014, Boudouresque *et al.* 2015). This is logical, since the two categories of indicators were designed to highlight different things, respectively the quality of the functioning of an ecosystem and the quality of a water body, but it is important to emphasize it again. The contrasting ranking of Porquerolles Island (north coast) from EBQI (poor) to PREI (high, first rank) (Table III), together with those of the Gulf of Giens, may be due to impacts other than the water quality, such as artisanal and recreational overfishing.

Ecosystem-based management and EBQIs are not incompatible with specific management measures based upon certain iconic species, which are also part of an ecosystem. The interest of ecosystem-based management is that it is not limited to the accumulation of specific management measures for iconic species, which can be mutually incompatible when, for example, an iconic species proliferates at the expense of other iconic species in the same habitat.

The spread of invasive species is considered as one of the most worrying environmental issues in the 21st century (Schmitz & Simberloff 1997, Canning-Clode 2015, Maxwell *et al.* 2016). The Mediterranean Sea is the region worldwide most severely hit by invasive species, with more than 800 non-indigenous species (Verlaque *et al.* 2015, Zenetos *et al.* 2017, Galil *et al.* 2018). Invasive species can deeply alter the food webs and the functioning of marine ecosystems (Vitousek *et al.* 1996, Boudouresque *et al.* 2005, Thomsen *et al.* 2016, Boudouresque *et al.* 2017a; David *et al.* 2017). Although an index accounting for invasive species has been proposed (ALEX – Alien Biotic Index) (Piazzi *et al.* 2015, 2018), food web approach (see *e.g.*, David *et al.* 2017) and EBQIs are the most effective tools for tracking their overall impact on the ecosystem.

Overfishing is also one of the major environmental pressures that affect marine ecosystems, *via* extirpation of target species, reducing the top predators compartment, reducing the mean trophic level (‘fishing down the food web’) and increasing the abundance of herbivorous sea urchins (Pauly *et al.* 1998, Sala *et al.* 1998, Pauly & Palomares 2005, Myers *et al.* 2007, Sala *et al.* 2012, Boudouresque & Verlaque 2013, Boudouresque *et al.* 2017a, Bryhn *et al.* 2020). Obviously, EBQIs tackle fishery pressure better than taxon-based indices in coastal areas. On a larger scale, offshore ecosystem functioning indices are proposed to highlight the state of ecosystems and relate it to the pressure of fisheries (Coll *et al.* 2016, www.indiseas.org).

The ecosystem-based approach can be applied to all types of ecosystems and it is important now to extend this approach to the pelagic ecosystems and their coupling with benthic ecosystems, infralittoral sandy bottoms, the beach-dune-*Posidonia oceanica banquette* ecosystem (see Boudouresque *et al.* 2017b, Otero *et al.* 2018), the deep sea and terrestrial ecosystems. Obviously, application perspectives of the EBA must reach areas away from the northwestern Mediterranean: eastern, central and southern Mediterranean, and worldwide coastal areas.

CONCLUSIONS

Ecosystem-based indices are the natural tools required for ecosystem-based management. They allow a comprehensive approach to the management of natural coastal areas, in particular Marine Protected Areas. They provide answers to different questions compared to taxon-based indices and are therefore not in opposition to them. It is obvious that the management of an ecosystem is much more complex than that of a single species or group of species. However, ecosystem-based management, and therefore ecosystem-based indices, represents the future. It will take time for stakeholders and the general public to understand that the complexity of the functioning of ecosystems can lead to responses which, at times, are counter-intuitive, but much more realistic and effective.

There is ever-increasing evidence of global change occurring. For example, in the eastern Mediterranean Sea, community-shift is dramatically altering ecosystem functions and services, leading to new ecosystems, most often less effective than the native ones. Monitoring and combating these major challenges must be undertaken at ecosystem scale, taking into account the whole functioning of the impacted ecosystems.

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THE *POSIDONIA OCEANICA* MATTE: A UNIQUE COASTAL CARBON SINK FOR CLIMATE CHANGE MITIGATION AND IMPLICATIONS FOR MANAGEMENT

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POSIDONIA OCEANICA MEADOW
CARBON SINK
MAPPING
SEISMIC REFLECTION
CARBON SEQUESTRATION
CARBON STOCK
CLIMATE CHANGE MITIGATION

ABSTRACT. – Seagrass meadows have long been recognized for their high ecological and economic value (ecosystem services). More recently, a potential role in climate regulation, due to their ability to fix and sequester carbon, has been the focus of intensive study. In the Mediterranean Sea, the matte, a specific structure built by the seagrass *Posidonia oceanica,* is of particular interest because it keeps buried for thousands of years massive amounts of carbon. Recent studies carried out along the Corsican coasts show a mean fixation of 1.62 Mg C ha^{-1} yr^{-1}, with a sequestration rate of between 27 and 30 %, a mean matte thickness of 210 cm and 711 Mg C ha^{-1} of organic carbon trapped in the matte. That is to say, a stock corresponding to 1,580 years of *P. oceanica* carbon sequestration, confirmed by radiocarbon analysis. An extrapolation to the Mediterranean basin (1.0 to 1.5 million hectares covered by *P. oceanica* meadow; mean matte thickness: 210 cm) shows that the total stock of organic carbon sequestered in the *P. oceanica* matte might be as much as 711 to 1,067 million Mg C. The conservation of the *P. oceanica* meadows thus constitutes an issue of major importance since any degradation of the matte, which has been built up over the past millennia, would very likely result in the release of considerable quantities of carbon. Rather than playing a major role in the attenuation of the impact of climate change (blue carbon sequestration), the *P. oceanica* meadow would then become a source of carbon that would be likely to amplify the greenhouse gas emissions. Management of *P. oceanica* meadows should take into account not only their role in carbon sequestration, but also the whole the full range of their ecosystem services, in relation with the functioning of the ecosystem.

INTRODUCTION

Seagrass meadows have long been recognized for their high ecological and economic value and associated ecosystem services (Costanza *et al*. 1997, Boudouresque *et al*. 2012, Vassallo *et al*. 2013, Picone *et al*. 2017). More recently, a potential role in climate regulation, due to their ability to fix and sequester carbon, has been the focus of intensive study (Nellemann *et al*. 2009, Fourqurean *et al*. 2012, Pergent *et al*. 2014). Carbon sequestration by seagrass is estimated at 15 % of total blue carbon, although seagrass cover represents only 17.7 to 61.0 million hectares at biosphere scale (Spalding *et al*. 2003, Kennedy & Björk 2009, UNEP-WCMC 2013).

Among the sixty-four species of seagrass (Guiry & Guiry 2020), *Posidonia oceanica* (Linnaeus) Delile, a Mediterranean endemic species, appears to be the most efficient in carbon storage; the *P. oceanica* meadow is the only ecosystem able to match peatlands and mangroves because it builds a unique structure: the matte. Made up of rhizomes and roots, with sediment that fills the interstices this highly resistant structure can reach several meters in height, and the organic matter it contains can persist for millennia (Mateo *et al*. 1997, 2006, Serrano *et al*. 2012, Boudouresque *et al.* 2016, Monnier *et al*. 2019a). In the light of the Paris Agreement, where major carbon sinks such as coastal vegetation are taken into account for the first time, and due to the exceptional extension of the *P. oceanica* meadows in Corsica (Valette-Sansevin *et al*. 2019), an extensive survey of these blue carbon sinks was performed to (i) inventory the main Blue Carbon Ecosystems (surface area and 3D extension – matte thickness), (ii) estimate the fixation and carbon sequestration rates, and (iii) assess the standing carbon stocks within the matte.

MATERIAL AND METHODS

The study was conducted in the Natura 2000 area "*Grand Herbier de la Côte Orientale*", a 100 km sandy coastline area along the eastern coast of Corsica (Fig. 1).

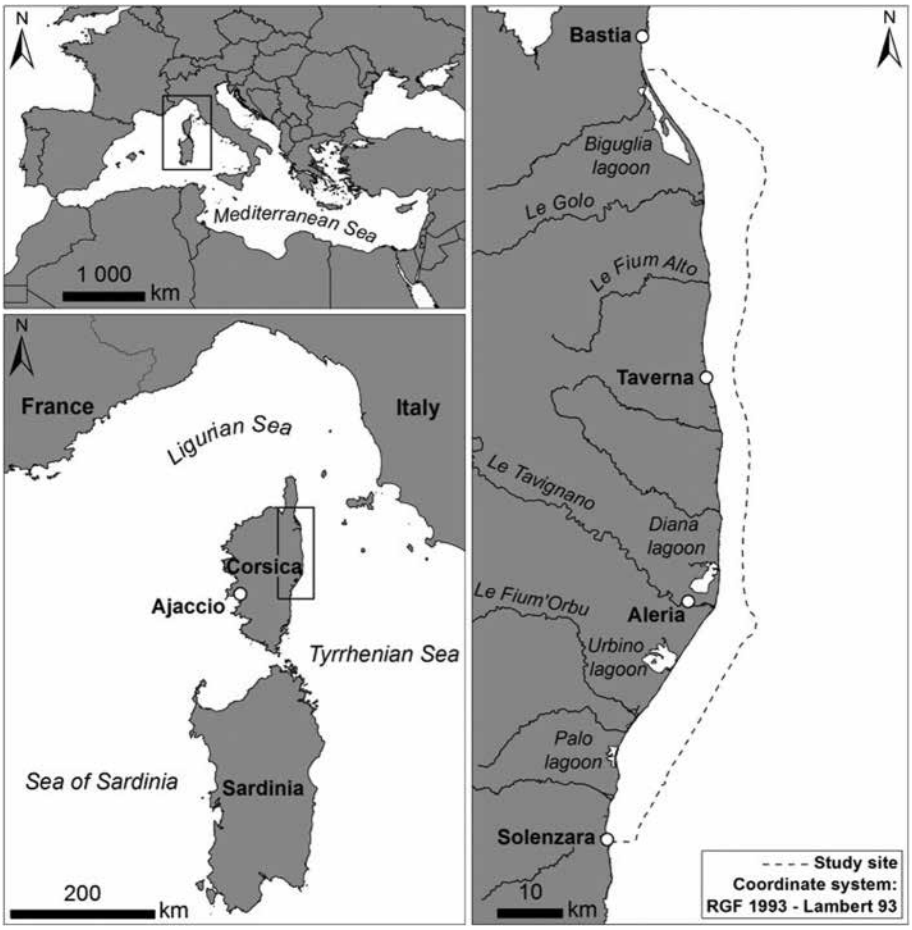

Fig. 1. – Location of the study site (Eastern continental shelf of Corsica Island, NW Mediterranean Sea).

The inventory of surface area covered by seagrass beds was performed, (i) for the shallow area (surface to –15 m), by remote sensing based on aerial images (BD ORTHO® of the IGN – French National Geographic Institute) with a 0.5 m resolution, (ii) for the deep area (–10 to –50 m), through several oceanographic surveys using exhaustive acoustic coverage (coupling a multibeam echosounder and a side-scan sonar), and (iii) for the other sectors, by a survey of existing data (Valette-Sansevin *et al*. 2019). Ground truthing data were acquired using a bathyscope for the shallow area (0 to –10 m) and for deeper areas, a Remotely Operated Vehicle (ROV), scuba diving inspections, underwater video camera recording (Pergent *et al*. 2017) and by collecting samples (Van Veen grab). After remote sensing following the method of Bonacorsi *et al*. (2013), data were integrated into a Geographic Information System (ArcGIS® 10.2.2., ESRI).

The assessment of *P. oceanica* matte thickness was carried out using the high-resolution seismic reflection method. Three devices with distinct emission frequencies were deployed for data collection: a Sparker (1 kHz) and two sediment profilers; the Manta EDO (2.5 kHz) and the Pesk Avel (3.5 kHz). In total, 510 seismic profiles were acquired corresponding to 3,095 km of data (Monnier *et al*. 2017, 2019b). Mapping of matte thicknesses is done after integrating the data into the ArcGIS® 10.2.2. software and using the ordinary kriging method. The map covered a range from –10 m (upper limit of data acquisition) to –40 m depth (lower limit of the *P. oceanica* meadow generally observed at this site).

Organic carbon fixation and sequestration were estimated at six sites (between –5 and –30 m) along three transects at the Natura 2000 site (Biguglia, Taverna and Urbino). Carbon fixation was measured through an assessment of primary production by the lepidochronology method (Pergent & Pergent-Martini 1991), and (ii) carbon sequestration was estimated on carbon fluxes (consumption by herbivores and detritivores, leaf litter exportation and seagrass tissues (sheaths, rhizomes and roots) buried in the *P. oceanica* matte; Pergent *et al*. 1997). Elemental analysis contributed to measurement of the total carbon content (%C) in the different tissues of *P. oceanica* after removal of epiphytes.

Estimation and characterization of organic carbon stocks was performed by collecting vertical cores in the matte with a Kullenberg gravity corer during the CARBONSINK oceanographic survey (2018). The matte samples were collected at three stations (–10 m, –20 m and –30 m) close to the same transects (Biguglia, Taverna and Urbino). Analysis of samples were undertaken after drying, sieving (< 2 mm) and separating sediment slices in different fractions (calcium carbonates, mineral and organic material). The total organic matter content (%TOM) and the elemental analysis (%C) were carried out on the fine

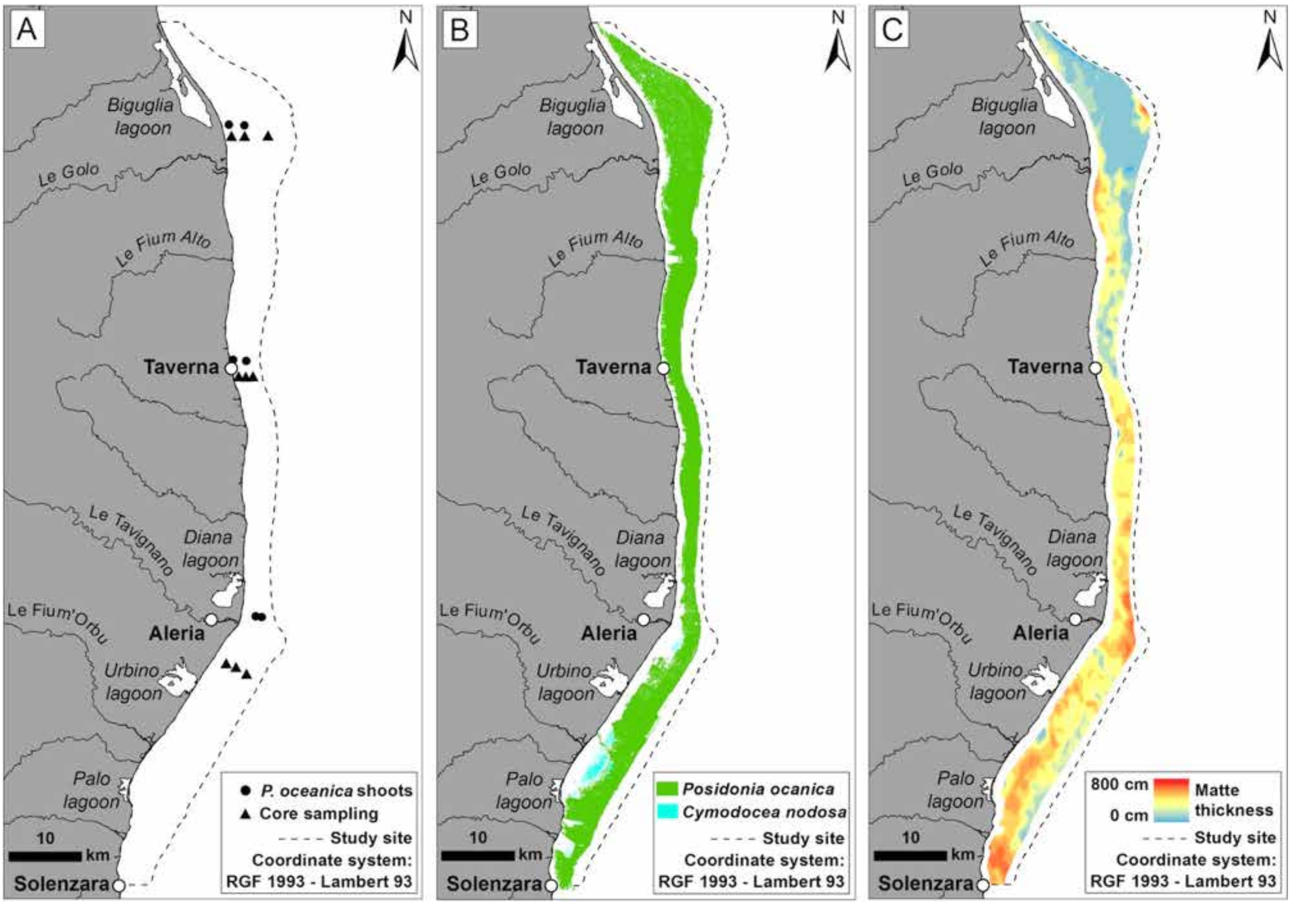

Fig. 2. – **A**: Location of *P. oceanica* transects and shoots and matte core sampling; **B**: Distribution of seagrass meadows (from Valette-Sansevin *et al*. 2019); **C**: Map of *P. oceanica* matte thickness estimated within the study site (from Monnier *et al*. 2017).

fraction and the coarse organic fraction following Monnier *et al*. (2019b). The standing organic matter and carbon stocks per unit area (cumulative stocks; kg C m^{-2} or Mg C ha^{-1}) were calculated according to Howard *et al*. (2014).

RESULTS

At the Natura 2000 site, seagrass meadows represent the most extensive habitat with a surface area of 20,425 ha for biocoenosis of *P. oceanica* meadows and 798 ha for association with *Cymodocea nodosa* (Ucria) Ascherson; respectively, more than 38 % and 47 % of the seagrass beds along the Corsican coastline (Fig. 2). The mean upper and lower limits of *P. oceanica* meadows range from –5 m to –40 m respectively. The distribution of *C. nodosa* beds is mainly located near the mouth of coastal rivers and locally beyond the lower limit of the *P. oceanica* meadows (Fig. 2).

The matte thicknesses of the *P. oceanica* meadow shows a high variability (Fig. 2). The mean thickness of the matte, estimated at 210 cm for the whole site, increases from north (160 cm) to south (270 cm). The thickness of the matte also seems to be greater near the coast with gentle slopes (between –10 and –25 m). Maximum thickness (up to 8 m) is recorded at the mouth of the main coastal rivers (Golo, Tavignano, Fium'Orbo and Travo).

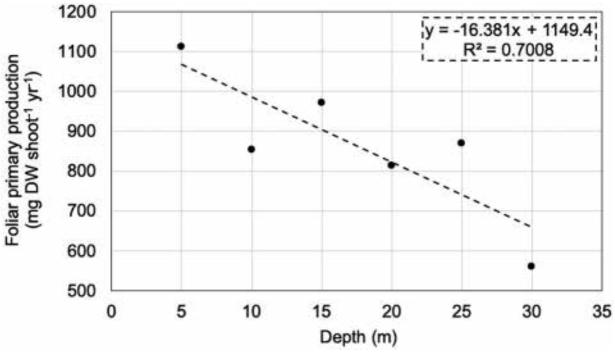

Fig. 3. – Foliar primary production of *P. oceanica* (blades) according to depth at the Natura 2000 site.

The primary production of blades decreases significantly with depth and varies between 1,112.0 mg DW $shoot^{-1}$ yr^{-1} (–5 m) and 560.8 mg DW $shoot^{-1}$ yr^{-1} (–30 m) (Fig. 3). Total carbon fixation by the *P. oceanica* meadow (blades, sheaths and rhizomes) also varies with depth, between 3.51 (–5 m) and 0.34 Mg C ha^{-1} yr^{-1} (–30 m) with an average of 1.62 Mg C ha^{-1} yr^{-1} (Table I). The total fixa-

Table I. – Carbon fixation by *P. oceanica* at the Natura 2000 site.

Sites	Density	Blades		Sheaths		Rhizomes		Total carbon fixation	
	shoot.m^{-2}	g DW.m^{-2}.yr^{-1}	g C.m^{-2}.yr^{-1}	g DW.m^{-2}.yr^{-1}	g C.m^{-2}.yr^{-1}	g DW.m^{-2}.yr^{-1}	g C.m^{-2}.yr^{-1}	kg DW.ha^{-1}.yr^{-1}	kg C.ha^{-1}.yr^{-1}
–5 m	550.5	612.1	249.9	178.9	71.9	69.1	28.9	8601.0	3507.5
–10 m	377.2	332.3	138.7	93.8	38.5	47.3	20.2	4734.3	1973.2
–15 m	300.6	295.5	122.0	65.3	27.7	39.0	16.6	3998.6	1662.8
–20 m	279.4	226.9	93.9	49.3	20.1	28.0	11.9	3041.1	1258.2
–25 m	204.2	177.4	73.0	39.1	15.9	18.6	7.9	2351.5	968.0
–30 m	106.1	59.5	24.0	14.9	6.2	9.1	3.8	834.6	342.9

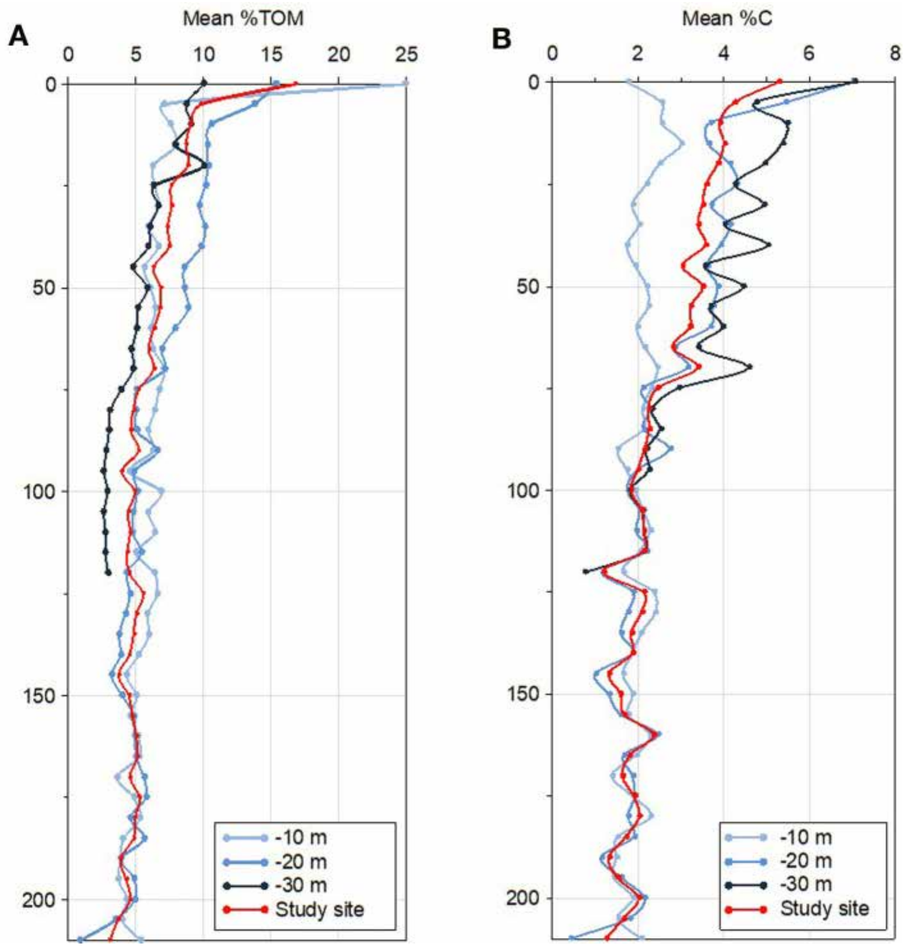

Fig. 4. – Changes in the main parameters (%TOM and %C) in) the *P. oceanica* cores collected at the Natura 2000 site. %TOM and %C are expressed as percentage of the total sample dry weight (%). Study site (red lines) correspond to the mean values for the different stations and depths.

tion at the Natura 2000 site corresponds to 33,063 Mg C yr^{-1}.

Primary production allocated to the sheaths, rhizomes and roots is estimated at between 0.1 (–30 m) and 1.0 (–5 m) Mg C ha^{-1} yr^{-1}, with an average of 0.45 Mg C ha^{-1} yr^{-1}. The total carbon sequestration at the Natura 2000 site represents 9,175 Mg C yr^{-1}, corresponding to a mean of 27.8 % of carbon fixation.

The matte cores collected at the study site are mainly constituted of seagrass debris (sheaths, rhizomes and roots) integrated in a dark brown sandy-muddy sedimentary matrix. The average length of the cores sampled at the

Table II. – Average stock of total organic matter and organic carbon of *P. oceanica* at the Natura 2000 site (SD: Standard Deviation).

Bathymetry	100 cm core		210 cm core	
	kg TOM m^{-2}	kg C m^{-2}	kg TOM m^{-2}	kg C m^{-2}
– 10 m	83.3	38.5	183.3	91.4
– 20 m	69.3	33.2	121.5	76.2
– 30 m	51.0	27.3	92.0	42.6
Study site (Mean ± SD)	68.5 ± 20.3	33.2 ± 9.2	133.8 ± 48.7	71.1 ± 28.9

study site is 215 cm, with a maximum value of 365 cm. The base of the matte (reflector) is thus frequently reached, contributing to calibration of the high-resolution seismic reflection data in complement to the matte wall heights measured in the 'intermattes' by scuba diving.

The standing stocks of total organic matter (kg TOM m^{-2}) and organic carbon (kg C m^{-2}) were standardized to 100 cm to allow comparison with literature, and 210 cm corresponding to the mean estimated matte thickness at the study site. Although different locations are compared, soil parameters of the cores change with water depth and seagrass soil depth. The vertical trends feature a slow decrease in %TOM and %C with soil depth and water depth (Fig. 4). The proportion of TOM and C in sediment significantly decreases through the top 100 cm of the soil and remains rather constant down to the core bottom (210 cm; Fig. 4). Thus, the mean TOM and C content decrease (respectively from 16.8 % to 4.0 % and from 5.3 % to 2.0 %) for the 0-100 cm section, and remain relatively constant in the lower section (100-210 cm) with 4.6 ± 0.6 %TOM and 1.8 ± 0.3 %C (mean ± SD). The characterization of samples shows that cores contained on average 68.5 ± 20.3 kg DW TOM m^{-2} and 33.2 ± 9.2 kg C m^{-2} in the first meter of soil (Table II). For the total core sequence (*i.e.*, 210 cm), the average amount of TOM and C stored are 133.8 ± 48.7 kg DW TOM m^{-2} and 71.1 ± 28.9 kg C m^{-2}, respectively (Table II). Whatever the soil depth considered, the TOM and C stock decrease with water depth (Table II).

DISCUSSION

The surface area covered by *P. oceanica* meadow in the Natura 2000 site is more extensive than anywhere else in the Mediterranean Sea, with more than 64 % of the seabed between 0 and 40 m depth and up to 206 ha km^{-1} of coastline covered, even in comparison with the Corsican coastline as a whole where this biocoenosis covers on average 61 % of the seabed (Valette-Sansevin *et al*. 2019, Table III). This exceptional coverage is linked to the particular topography of the eastern continental shelf of Corsica (very low slope: < 2 %; Pluquet 2006) and to reduced anthropogenic pressures (Cannac-Padovani 2014).

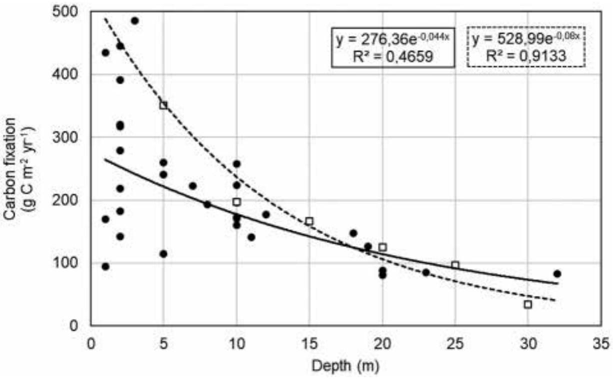

Fig. 5. – Carbon fixation (g C m^{-2} yr^{-1}) by *P. oceanica* meadows (blade, sheaths and rhizomes) in the Mediterranean Sea (circles, solid line) and in the study site (squares, dotted line). Data from Pergent-Martini *et al*. 1994, Pergent *et al*. 1997, Guidetti 2000, Dumay 2002, Vela 2006 and references therein.

Total carbon fixation recorded at the Natura 2000 site is comparable with other values recorded in the Mediterranean Sea, but is slightly higher, mainly between the sea surface and –20 m (Fig. 5). The fixation and carbon sequestration at this site correspond respectively to 6.5 % and 1.75 % of annual CO_2 release by anthropogenic activities on the island of Corsica (based on mean release rates by inhabitant in France in 2018, approximately 1.9 million Mg CO_2; Global Carbon Atlas 2020). At regional scale, the *P. oceanica* meadows as a whole (53,737 ha; Valette-Sansevin *et al*. 2019) contribute respectively to the fixation and carbon sequestration of 16.6 % and 4.6 % of CO_2 emissions on the island (Global Carbon Atlas 2020).

On the basis of (i) the mean carbon fixation corresponding to the integrative depth of –15 m (138.5 g C m^{-2} yr^{-1}), and (ii) the area covered by *P. oceanica* (1.0 to 1.5 million hectares; Topouzelis *et al*. 2018, Traganos *et al*. 2018, Valette-Sansevin *et al*. 2019), the total carbon fixation for the whole of the Mediterranean Sea should be between 1.39 and 2.08 million Mg C yr^{-1}, that is to say the equivalent of 5.08 to 7.62 million Mg equivalent CO_2 release.

The use of the high-resolution seismic reflection method has contributed to providing an accurate estimate of the

Table III. – Surface covered by the *Posidonia oceanica* meadow at the Natura 2000 site and on the main islands of the Western Mediterranean basin (* Valette-Sansevin *et al.* 2019, ** Ruiz *et al.* 2015, *** Ministero dell'Ambiente e della Tutela del Territorio 2001, **** Calvo *et al.* 2010, # GADM 2020, ## EMODnet, 2020).

	P. oceanica **surface area (ha)**	**Length of coastline (km) #**	***P. oceanica*** **surface area by length of coastline (ha.km^{-1})**	**Seabottom surface area in 0-40 m depth range (ha) ##**	***P. oceanica*** **surface area in 0-40 m depth range (%)**
Study site	20,425	99	206.3	31,769	64.3 %
Corsica*	53,735	1,177	45.7	88,509	60.7 %
Balearics**	63,316	1,435	44.1	112,022	56.5 %
Sardinia***	153,382	2,403	63.8	303,740	50.5 %
Sicily****	76,000	2,007	37.9	318,393	23.9 %

spatial extent of carbon stocks represented by *P. oceanica* matte at the study site. The edification of this structure is mainly due to the vertical growth of orthotropic rhizomes which avoids the burial (sediment inflows) of the aboveground living biomass, resulting in a relatively slow upward rise of the bottom (Boudouresque & Jeudy de Grissac 1983, Boudouresque *et al*. 1984). The thickness of the matte results from the balance between an accretion of material (sediment and debris) and decomposition and erosion processes linked to different factors (Mateo *et al*. 1997, 2006).

The seismic data highlight a high variability of the thickness of bioformations which could be explained by (i) the natural land-based inputs at the mouth of the coastal rivers, but also near the lagoon inlets ('flushing flow'), and (ii) the sediment dynamics related to ocean currents (coastal drift) at the underwater deltas (Monnier *et al*. 2017, 2019a). Although high-resolution seismic reflection data appears to be a reliable tool to determine the thickness of the sedimentary carbon stocks buried under the *P. oceanica* meadows (Lo Iacono *et al*. 2008, Tomasello *et al*. 2009), core sampling and the subsequent geochemical analysis remain essential as a basis for a precise quantification of carbon stocks.

The decrease in total organic matter (%TOM) and carbon (% C) with depth within the matte (Fig. 4) suggests that contrary to what is often claimed – that is to say that within the matte, sheaths, rhizomes and roots are rot-resistant (Molinier & Picard 1952, Boudouresque *et al.* 2012) – degradation does occur within the matte. This hypothesis has already been formulated by Boudouresque *et al.* (2019) in the Bay of Hyères (Provence, France).

Taking into account (i) the average thickness of the matte (210 cm), (ii) the average quantity of carbon measured (711.4 ± 289.4 Mg C ha^{-1}), and (iii) the average annual carbon sequestration rate (0.45 Mg C ha^{-1} yr^{-1}), the carbon stock present at the Natura 2000 site would correspond to the carbon sequestered over a period of approximately 1,580 years.

In the context of climate change, the carbon storage capacity of the *P. oceanica* matte over several thousand years is a major advantage (Boudouresque *et al*. 1980). Within the "*Grand Herbier de la Côte Orientale*" site, the average carbon stock in the first meter of sediment (33.2 ± 9.2 kg C m^{-2}) is generally comparable to values observed in the literature for similar heights of matte (28-237 kg C m^{-2}; Romero *et al*. 1994, Mateo *et al*. 1997, Serrano *et al*. 2012, 2014, 2016). Compared with other seagrass beds, the amount of carbon stored by *P. oceanica* is globally higher (Fourqurean *et al*. 2012, Lavery *et al*. 2013). The carbon stocks for the *P. australis* and *P. sinuosa* seagrasses are estimated as between 10.8-32.0 kg C m^{-2} and 1.8-6.6 kg C m^{-2}, respectively (Lavery *et al*. 2013, Serrano *et al*. 2014, 2016). Moreover, this value is similar or higher than those measured in several terrestrial ecosystems considered to be efficient in carbon storage such as peatlands (120 kg C m^{-2}; Warner *et al*., 1993), wetlands (13-73 kg C m^{-2}; Laffoley & Grimsditch 2009) and the boreal forests (9-34 kg C m^{-2}; Serrano *et al*. 2014).

The carbon stock present at the Natura 2000 site is estimated at 14.5 ± 5.9 million Mg C. For the entire coastline of Corsica, the value is estimated at 38.2 ± 15.6 million Mg C, the equivalent of 79 years of CO_2 emissions (based on mean release rates by inhabitant in France in 2018; Global Carbon Atlas 2020), much more than all of the cumulative emissions since the mid of 20th century.

At the scale of the Mediterranean basin, where the surface area covered by *P. oceanica* is estimated at between 1.0 and 1.5 million hectares (Topouzelis *et al*. 2018, Traganos *et al*. 2018, Valette-Sansevin *et al*. 2019), the total carbon stock present in *P. oceanica* matte is estimated as 711 to 1,067 million Mg C, the equivalent of 1 to 3 years of CO_2 emission by all Mediterranean countries (Global Carbon Atlas 2020). Even partial degradation of these mattes, and the concomitant release of this carbon into the environment, would have negative consequences for the patterns of change in temperature in an already worrying context of climate change.

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THE USE OF VERY HIGH-RESOLUTION IMAGES FOR STUDYING *POSIDONIA OCEANICA* REEFS

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SEAGRASS
UPPER LIMIT
DRONE
3D MODEL
DEAD *MATTE*
CARBON SINK
UAVS
ATOLL

ABSTRACT. – *Posidonia oceanica* is an endemic Mediterranean seagrass that forms wide and dense meadows from the surface up to about 40 m depth. This species can develop a biogenic structure called *matte*, a typical terraced formation built up by itself, consisting of intertwined rhizomes, roots and sediment, which may allow shoots to reach the sea surface forming reefs (*récif barrière*), considered natural monuments. *Posidonia oceanica* reefs are particularly exposed to the ongoing increase in temperature and in storm frequency and intensity due to climate change, with negative repercussions on their conservation. Much more attention must be paid to the implementation of monitoring tools able to detect early signs of reef regression. In this study, the distribution of *P. oceanica* reefs located along southern coasts of Sicily (Italy, Mediterranean Sea) was investigated. A remote sensing approach was used to assess reef extension, morphological features (*e.g.*, atolls), upper limit and occurrence of dead *matte*. In particular, very high-resolution drone images (~ 2 cm pixel size), combined with GPS field data, were used for 2D- and 3D-reconstruction of *P. oceanica* reefs. The 3D-model allowed to estimate bathymetrical distribution of *P. oceanica* reef enabling to achieve a more detailed and complete understanding of the *P. oceanica* reef architecture.

INTRODUCTION

The dominant and endemic Mediterranean seagrass *Posidonia oceanica* (Linnaeus) Delile is a long-lived species that forms extensive meadows and grows from nearly the water surface to depths up to 40 m (Mazzella & Buia 1986). It constitutes a "climax" community and its presence attests to a relatively stable environment (Francour *et al.* 1999, Montefalcone *et al.* 2008). *Posidonia oceanica* meadows play a pivotal role in numerous ecological and geomorphological key processes such as nutrient recycling, provision of food for fauna, shelter and nursery areas for many species, sequestration of carbon, stabilization of sediment, attenuation of waves through their canopy (Mazzella *et al.* 1992, Mateo *et al.* 1997, Buia *et al.* 2000, Hemminga & Duarte 2000, Vizzini 2009). *Posidonia oceanica* can grow on different substrates including sand, which is easily penetrable by the roots, rocks, in which crevices host the very sturdy roots, and *matte*, a unique terraced biogenic structure formed by the entanglement of roots, rhizomes and sediment (Jeudy de Grissac & Boudouresque 1985). *Posidonia oceanica* can adapt the direction of its growth (horizontally or vertically) to the rate of sediment deposition. Under sedimentation, rhizomes grow vertically (orthotropic growth) to avoid burying (Molinier & Picard 1952, Caye 1980, 1982, Boudouresque & Meinesz 1982) resulting in *matte* edification. In sheltered and shallow water this process can lead to the *matte* rise up, with the leaf tips over, close to the sea surface and subsequent formation of a reef-made barrier (*récif barrière sensu* Boudouresque & Meinesz 1982). Reefs can persist for century or millennia and are increasingly considered as "natural monuments" (Calvo 1987, Pergent *et al.* 2007, 2014, Bonacorsi *et al.* 2013, Boussard *et al.* 2019) and their census is still in progress (Rouanet *et al.* 2019). Both natural and anthropogenic disturbances may endanger *P. oceanica* reefs. These structures are particularly threatened by temperature rise (Tomasello *et al.* 2009, Pergent *et al.* 2014) and erosion caused by sea storms (Short & Neckles 1999, Boudouresque *et al.* 2012), which are expected to increase in intensity due to climate change (IPPC 2019). Furthermore, erosion leads to the exposure and then oxidation of the carbon accumulated within the *matte*, causing the shift of the reefs from sinks to sources of carbon (Boudouresque *et al.* 2016). The stability of these structures along the Mediterranean coasts has been evaluated showing different results, since they have been observed in regression (Boudouresque *et al.* 1975, Tomasello *et al.* 2009, Pergent *et al.* 2014), or in progression (Bonacorsi *et al.* 2013). To gain a better understanding of the actual state of *P. oceanica* reefs, much more attention should be paid to the implementation of monitoring tools able to detect the first signs of reef regression (Pergent *et al.* 2014). Improving our knowledge of these particular biogenic structures is of pivotal importance to check their status, given that the available maps used to estimate their changes do not constitute a reliable baseline because of their low accuracy (Bonacorsi *et al*. 2013). Recently, the use of very high-resolution images acquired by Unmanned Aerial Vehicles (UAVs) has increased markedly in the field of remote sensing of the environment, due to their advantages in terms of lightweight and low cost required allowing to work at an appropriate spatial and temporal scale needed to study ecologically relevant variables (Anderson & Gaston 2013). Over the past decade, UAVs have been widely used for terrestrial studies, consequently enhancing, as technological developments rapidly advance, their versatility and functionality so much that their use has opened new opportunities such as monitoring of sensitive marine habitats (Ventura *et al*. 2018). Currently, UAVs are able to define also the 3D-reconstruction of an object or scene through a high number of photographs taken from different points of view (Figueira *et al*. 2015). Very recently, they have also been tested for seabed mapping with particular reference to marine vegetation (Duffy *et al.* 2018, Ventura *et al.* 2018), showing a very promising potential. The aim of this study was to test the use of UAVs (drone) images for evaluating the distribution of *P. oceanica* reefs located along the Southern coast of Sicily (Italy, Mediterranean Sea). In this framework, we used photogrammetry obtained by UAVs to analyze reef features and their bathymetrical distribution.

MATERIALS AND METHODS

Study area: The study was carried out in June 2019 along the Southern coast of Sicily (Italy), within the site of European community importance "Fondali Di Capo San Marco-Sciacca" (cod. ITA040012) (Fig. 1). Straight beaches, with medium fine sands mostly quartzy, interspersed with cliffs that subtend, rather irregularly, several small pocket beaches, characterize the geomorphology of the site. The cliff behind, about 5 m high, is mainly composed of sandy clay deposits from the Upper Pliocene - Middle Pleistocene, and is dominated by a terrace of the Tyrrhenian Sea. At the foot of the cliff there are sand – gravelly deposits, very irregular, protected by little protruding promontories, but above all by cobbles (64-256 mm) and boulders (> 256 mm) widely present in the coastal area immediately in front; these deposits are to be linked to the erosion of the cliff, being different from those present in the neighboring areas in terms of granulometry, composition and color. Within the site, *P. oceanica* forms several reefs along the coast (Fig. 1). In particular, six main reefs are present almost continuously distributed along about 2 km, forming a lagoon-like environment with the co-occurrence of other two seagrasses, *Cymodocea nodosa* (Ucria) Ascherson and *Zostera noltei* Hornemann (Perzia *et al.* 2011). Rarely, *P. oceanica* grows at few meters far from the shoreline (about 3 m), forming a *récif frangeant* (Boudouresque & Meinesz 1982). In this study, the reef located in front of the

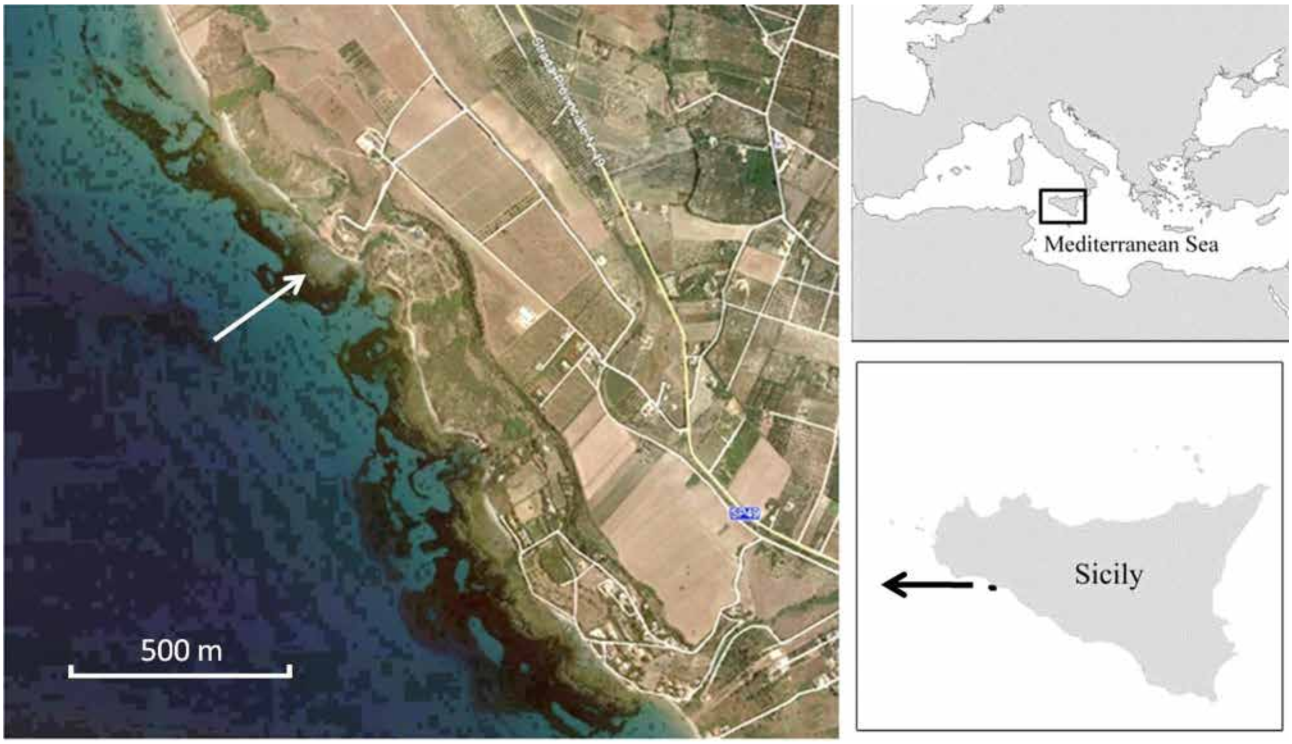

Fig. 1. – Study area with indication of the Maragani's reef.

location “Maragani”, was investigated (Fig. 1). The reef is settled mainly on rock resulting in processes of collapse and retreat due to causes connected to the same intrinsic instability of the cliff system (lithological and geomorphological) and to the subsurface hydraulic circulation typical of the coastal stretch. The material crushed by the cliff remained on site incorporated within the interior of the meadow. The reef attenuates wave energy (Fig. 2), determining the formation of a small lagoon-like area characterized by a very shallow depth (< 1 m), low hydrodynamism, seabed with dead *matte*, muddy sediment and rare and very scattered rock blocks. Within the lagoon, water exchange is guaranteed by tidal oscillations and / or breaking waves that pump water inside the lagoon from which they come out through small lateral channels within the reef (AT personal observation).

Data acquisition: Two types of aerial images acquisition were performed in the area: one flight was carried out at a height allowing a resolution enough to avoid photo-interpretation errors, while the other one was carried out at a higher resolution to estimate bathymetry.

In particular, the first type of acquisition was performed by using a DJI Mavic Pro UAV (743 g take-off weight) for assessing seagrass distribution. The drone was equipped with a 12 Mpx camera (CMOS sensor) with a focal length of 4.73 mm to collect 98 calibrated photos each covering an area about 40 × 30 m. The flight height was 30 m, which, as demonstrated by Casella *et al.* (2017), is suitable to depict shallow water characteristics, reaching a ground sample distance (GSD) of ~ 1 cm/pixel according to the following formula:

$$GSD_{cm/pix} = [(Sw_{mm} \times Fh_{m}) / (Fl_{mm} \times Iw_{pix})] \times 100$$

where GSD is the photo resolution on the ground, Sw is the sensor width, Fh is the flight height, Fl is the focal length of the camera, and Iw is the image width (Ventura *et al.* 2018). The flight time was 11 minutes. In this case, the overlap of the images was 60 %, while sidelap was ~ 30 % for a total ~ 1.6 ha of sea recorded.

For the second flight a DJI Mavic 2 UAV (~ 907 g take-off weight) was used for image acquisition to estimate bathymetry. The drone was equipped with a 20 Mpx camera (CMOS sensor) with a focal length of 10 mm to collect 102 calibrated images covering the same area as the former flight (1.6 ha) where a single photo covers an area of about 80 × 50 m. The flight height was 65 m (GSD = 1.52 cm/pix) and a flight time of 7 minutes. The overlap and sidelap of the images for the bathymetry esti-

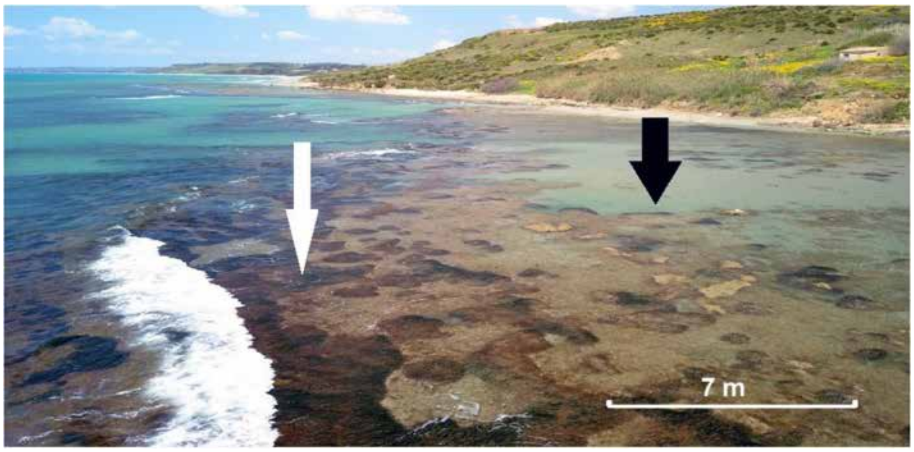

Fig. 2. – The waves break on the reef (white arrow). The black arrow points the small lagoon behind.

Fig. 3. – The flight plan with the sequence of image shot.

Fig. 4. – Target positioning with Topcon HiPer HR.

mation were ~ 75 % (Fig. 3). In both cases the angle of the camera was set at 90° (to collect nadiral images). Take-off and landing were controlled manually by an operator on the field through a remote control. A Global Navigation Satellite System (GNSS) Topcon HiPer HR (horizontal and vertical error 5 mm + 0.5 ppm and 10 mm + 0.8 ppm, respectively) was also used to detect 11 landmarks on the field (five underwater and six on land) as control points using quadrangular targets 50 x 50 cm size (Fig. 4). One control point acquired on the beach/sea interface was used to rescale all elevation of the other points. Further 16 control points, set as 0 hydrometric, were also chosen directly in the images in correspondence of *Posidonia* leaves that clearly emerged at different points at the time of flight. Both surveys were carried out on same morning characterized by calm sea conditions and clear sky.

Data processing:

2D mosaic

The realization of the 2D mosaic of the *P. oceanica* reef along Maragani coast involved the use of Image Composite Software (ICE 2.0), an advanced panoramic image stitcher (https://www.microsoft.com/en-us/research/product/computational-photography-applications/image-composite-editor/), which allowed to create 2D high-resolution scenarios from a set of overlapping photographs. More specifically, the images of the 30 m flight were imported into the software and a structured serpentine panorama was set, selecting the initial photo and the direction to follow, emulating the programmed flight plan.

Bathymetry

The aerial photographs were analyzed with an appropriate software (Pix4D – https://support.pix4d.com/hc/en-us) that uses advanced SfM (Structure for Motion) and multi-view stereo (MVS) algorithms to construct an ortho-photomosaic and a 3D point cloud from overlapping photographs. The software is capable of automatic identification of key points on all photos, bundle adjustment, point cloud densification, mesh building and texturing (Casella *et al*. 2017, Marre *et al.* 2019). In a first step, the photographs were aligned by means of SfM algorithms identifying image feature points and then the movement of those points throughout the image data set was monitored (~ 180,000 points). The software also calculated the relative camera positions at the moment of image acquisition and internal calibration parameters. Secondly, a dense point cloud was built obtaining ~ 40,000,000 points. Thirdly, the geometric details were built through the analysis of the pixel values operated by the algorithms. It is a sophisticated procedure based on an advanced computer vision solution that enables the creation of high-quality 3D-content from a series of overlapping images. Then, the mesh was textured with photographs. The SfM approach requires a set of points of known coordinates (ground control points) that measures the difference between true coordinates and its coordinates calculated from all photos, to compute pixel-to-earth transformations and to georeference the data point cloud. From the point cloud, the software generated an orthorectified photomosaic with a resolution of 1.49 cm/pix and a DEM with a resolution of 5.95 cm/pix. Finally, the orthophotos and DEMs were exported from Pix4D and imported into a GIS software (Quantum GIS). The distortion resulting from the air-water interaction in the final DEM was corrected through the Snell's law by multiplying the cells in the submerged areas by the refractive index of water (1.34), assuming a planar water

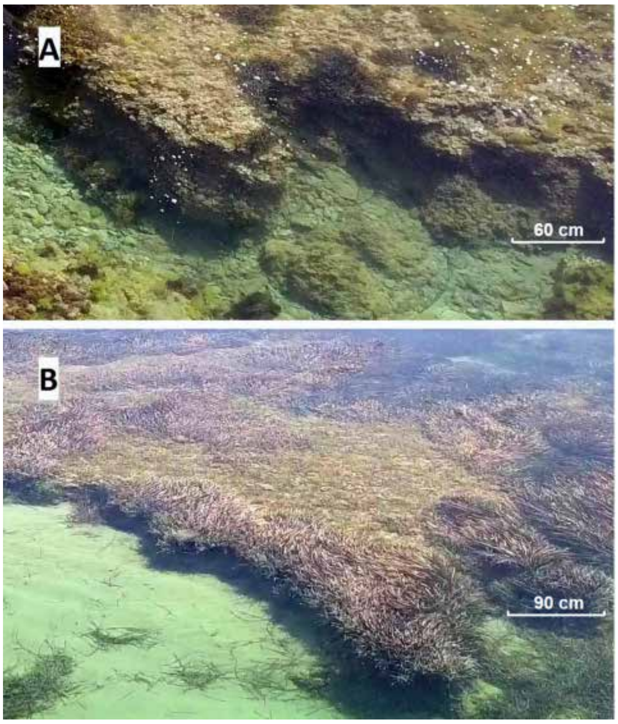

Fig. 5. – Aerial images of *matte* wall detected on the outer (**A**) and inner (**B**) side of the reef in a close-up perspective view obtained by drone flying at few meters above the sea.

surface unaffected by waves or surface rippling, and integrating these values into the original DEM (Woodget *et al.* 2015, Agrafiotis *et al.* 2020). A sub sample of DEM points was then regressed *vs* the control points set acquired in field by GPS.

3D Model

The last step was to construct a 3D model of the reef. In particular, the 2D photomosaic obtained with the 30 m flight was georeferenced using, as a base, the orthophoto derived from the flight performed at 65 m. The high-resolution georeferenced image was then re-projected directly on the 3D-surface (Rende *et al.* 2015). In this way, an excellent level of detail was reached still maintaining the ability to identify unequivocally *P. oceanica* meadow in a 3D vision. Then some meadow morphologies were digitalized in order to record upper limit bathymetrical position and *matte* elevation. The upper limit was traced on patches longer than 5 m and facing toward the lagoon, maintaining always a distance of 5 cm from the leaves, in order to measure the bathymetrical position of the seabed on which shoots were settled thus avoiding to erroneously record the top level of canopy. Moreover, other morphologies were also analyzed. In particular, dead *matte* walls previously noted during field activities (Fig. 5) were recognized on the photomosaic first and then their thickness was measured on the corresponding 3D model. These measures were finally compared with those recorded *in situ*.

RESULTS

Meadow distribution

Generated photomosaic of the reef, obtained by the flight at 30 m, showed very clearly the distribution of *P. oceanica* meadow (Fig. 6), since it was possible to recognize the leaves of the seagrass in the whole image (Fig. 7). The high level of transparency combined with shallow water allowed to detect important features of the sea bottom. In particular, on the right side of the area, proceeding from North to South, *P. oceanica* meadow is interspersed with an extensive sandy glade, among them it shows a wide band with an almost continuous coverage. Southernmost, the meadow deviates to east until reaching the coast. Inside the meadow several rocky blocks mainly in the southernmost part are present (Fig. 8). Proceeding towards the coast, the meadow becomes progressively more fragmented near its upper limit where the reef emerges with leaves up to the surface also forming

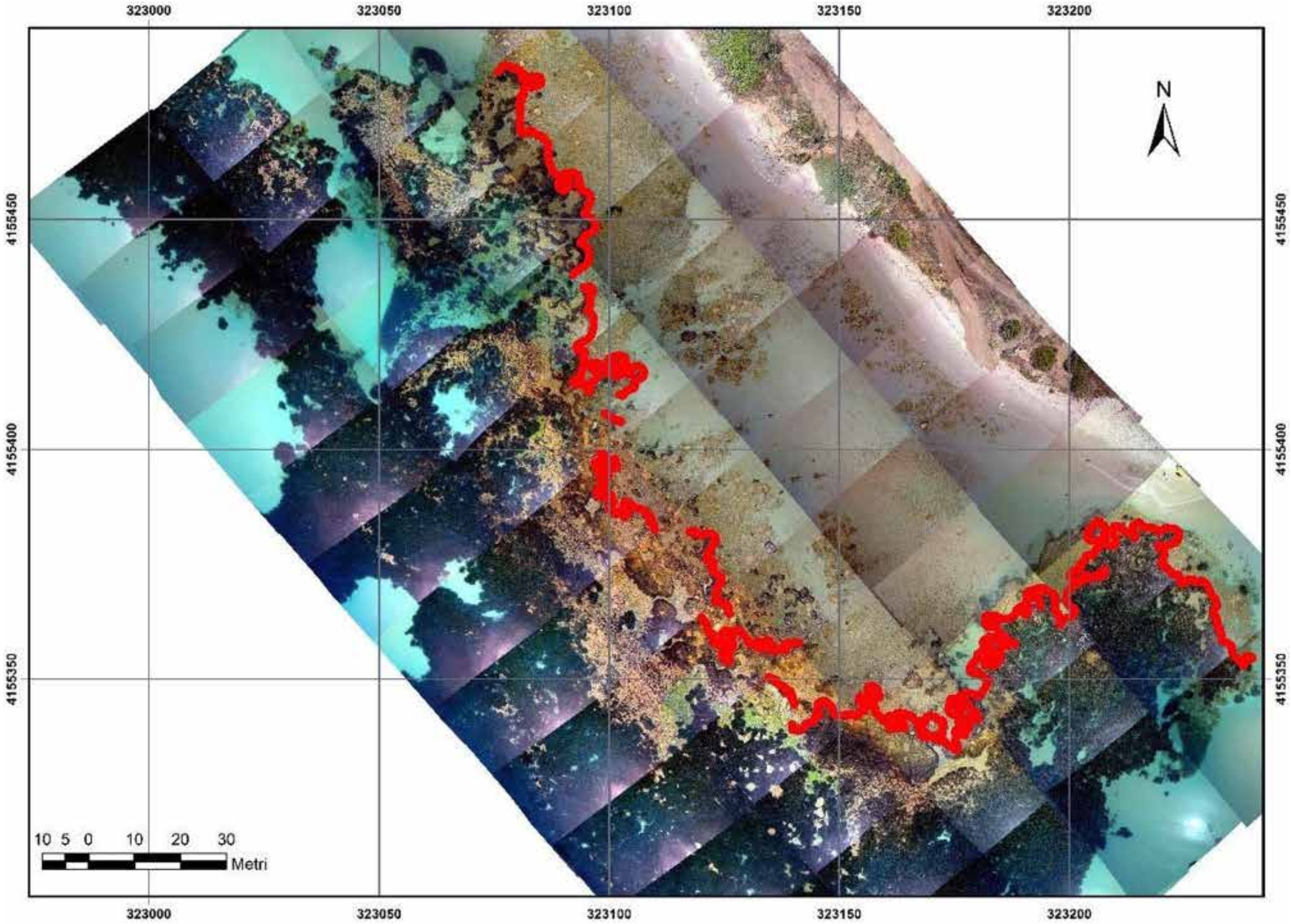

Fig. 6. – Reef photomosaic with *P. oceanica* upper limit over-imposed.

atoll-like patches more or less arched and surrounded by dead *matte* (Fig. 9). The atoll structures have an average diameter that oscillates between 1.27 m and 5.26 m, showing an irregular shape sometimes elongated with the presence of multiple arches on the perimeter. The inner dead *matte* showed variable sizes, resulting in an average thickness of the live atoll ring ranging from 0.24 m to 3.85 m. Patches composed predominantly by living shoots occurred densely grouped in the South-Eastern sector of the reef, very close to the coast (Fig. 10). Inside the reef small rocky blocks, grouped or alone mixed with sporadic patches of *P. oceanica* are also found.

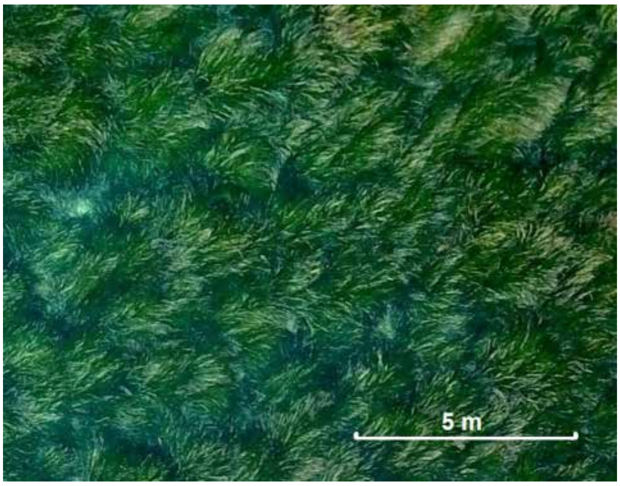

Fig. 7. – Continuous *P. oceanica* meadow. The leaves, more or less covered by epiphytes (lighter), can be distinguished.

Fig. 8. – Several emerging rocky blocks can be identified within or outside the meadow.

Bathymetric map

The DEM generated through the images acquired with flight height of 65 m allowed to obtain the bathymetric map of the area. Regression between control and estimated bathymetrical points extracted from DEM showed a linear agreement on average (Fig. 11). The map shows bathymetry ranging from 0 to 2.31 m (Fig. 13).

A rising seabed coincides with the reef platform, which has an average depth of –0.13 ± 1×10^{-5} m. Two depressions in the area behind the reef were detected. Particularly in the North, the depression has an average depth of –0.42 m, is wider and expands deeper towards the open sea, passing the reef through a channel 4 m wide and with a max depth of –0.75 m. In the South, the depression shows a very elongated shape following the inner side of the reef until it becomes parallel to the coast and then goes out from the lagoon through a small channel, 2 m wide and –0.45 m deep (Fig. 12). The depth distribution of the *P. oceanica* upper limit calculated by bathymetry via GIS showed an average value of –0.27 m.

3D reconstruction

By 3D reconstruction, performed matching DEM with photomosaic on the whole area, we could appreciate the 3D image of reef structure and the morphometric information associated (Fig. 13). 3D representation of reef platform showed further details concerning their morphology allowing better understanding of the extension of the reef, from leaf canopy up to landscape scale. This picture gives a real and immersive perception of reef architecture from sea surface to greater depth. For example, particular mor-

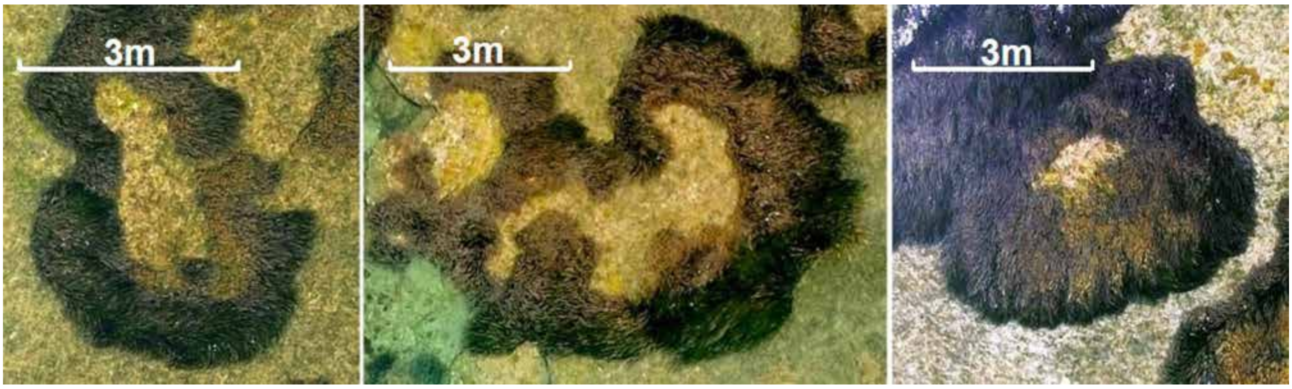

Fig. 9. – Examples of atolls with a multiple arched perimeter. Note how the empty central zone varies in size and shape.

phologies at the boundaries of the reef coincide with the wider channel entering inside the small lagoon reported before (cf. bathymetric map). Some of these boundaries corresponded to dead *matte* wall. Spatial profile from 3D model allowed to estimate *matte* wall thickness ranging from 0.21 to 0.82 m and average value equal to 0.59 m ± 0.20 S.D. (n = 10), while *matte* wall thickness at same point *in situ* ranged from 0.35 m to 1.00 m and

Fig. 10. – Patches of *Posidonia* composed mainly of living shoots.

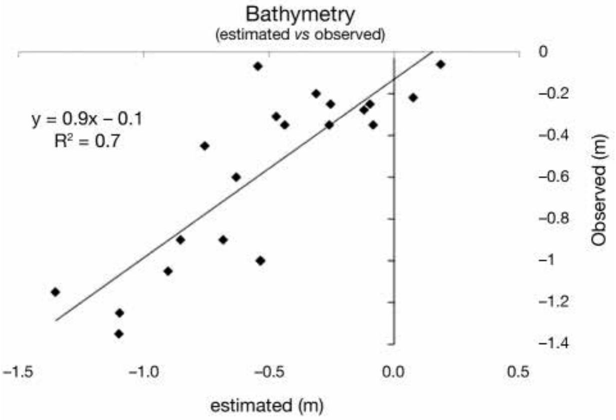

Fig. 11. – Regression between bathymetry recorded *in situ* and estimated through the model.

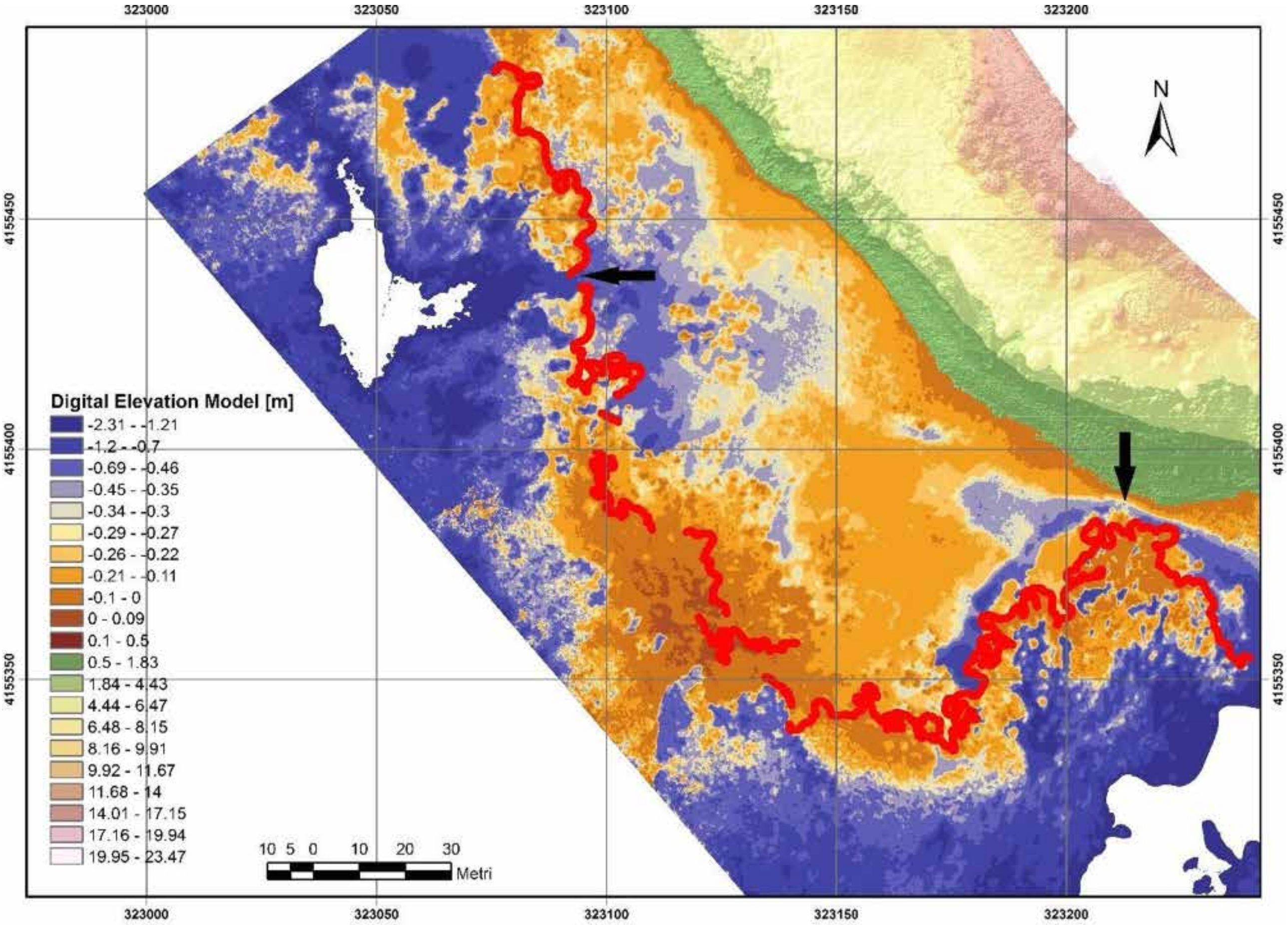

Fig. 12. – Digital elevation model of the area. Red lines and black arrows indicate *P. oceanica* upper limit and the two channels, respectively.

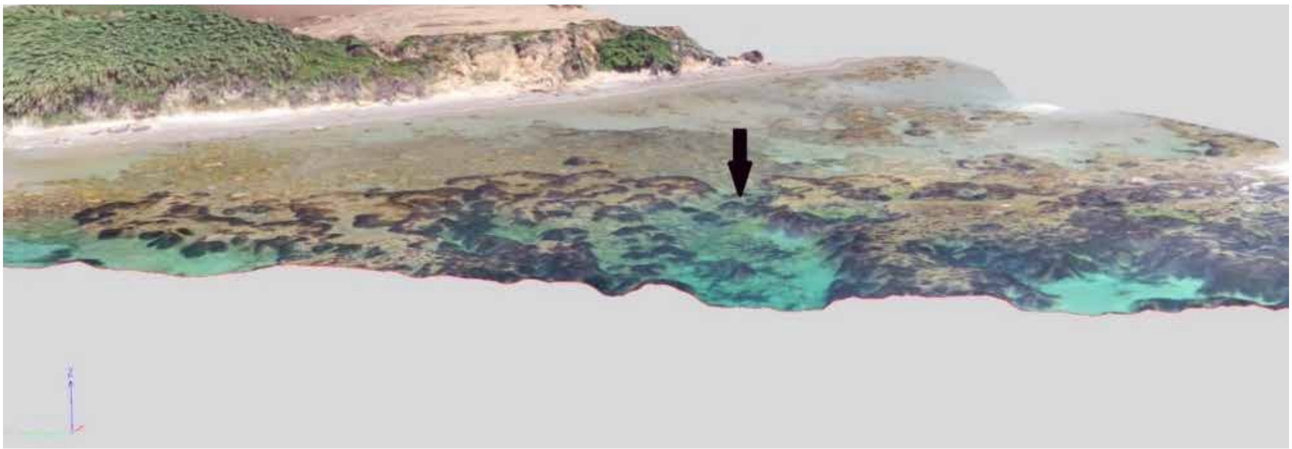

Fig. 13. – 3D-model of the *P. oceanica* reef; the arrow indicates the northern channel.

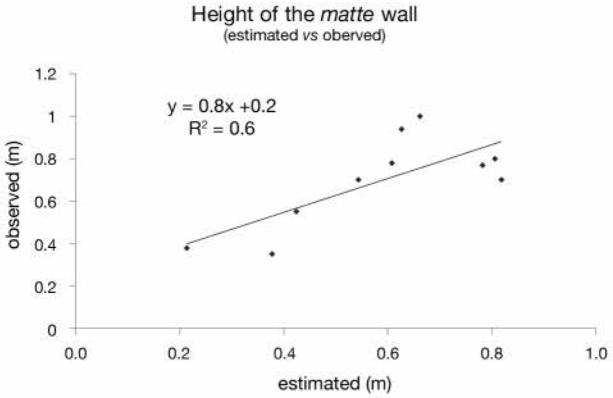

Fig. 14. – Regression between dead *matte* thickness recorded *in situ* and estimated via the 3D-model.

average value equal to 0.70 m ± 0.22 SD (Fig. 14). On the basis of these results the error of mean *matte* wall thickness estimate was –15.9 %.

DISCUSSION AND CONCLUSION

Our study showed that it is possible to analyze *Posidonia oceanica* reefs by using very high-resolution images acquired from drones. The approach used has also highlighted the potential of this method to obtain 3D representation of reefs. This was allowed because seabed was clearly visible through the water surface, and with further processing it was possible to quantify bathymetry via digital terrain models (DTMs) (Ventura *et al*. 2018). Traditionally, high-resolution bathymetry maps have been successfully obtained by gathering high-resolution seabed MBS data in shallow and deep waters (Di Maida *et al*. 2011). However, such device cannot be used in very shallow water (less than 50 cm) due to multiple acoustic reflections between the sea surface and the seabed resulting in a significant noise, heavily affecting the beam signal. Our results provide evidence that UAVs are a very simple and, at the same time, cheap solution able to represent not only seagrass 2D distribution, but also its 3D architecture at a small spatial scale in a very high heterogeneous environment, similarly to what has already been achieved in meadows at greater depths through passive (Rende *et al.* 2015, Ventura *et al.* 2018), or active remote sensing methodologies (Komatsu *et al.* 2003, Di Maida *et al.* 2011). The 2D seagrass distribution combined with DEM model gave, indeed, a very accurate bathymetrical distribution of *Posidonia* reefs, which in this specific case represent the upper limit of the species in the area. The natural upper limit of *P. oceanica* settled on sand or other soft bottoms and on *matte* can be predicted on the basis of physical parameters, which, in a large-scale study carried out along the Mediterranean coasts, was estimated to occur at depths greater than 3.2 m (Montefalcone *et al.* 2019). Only in presence of rocky substrates, the upper limit can be considerably shallower due to the ability of rhizomes to anchor tenaciously, thus allowing the plant to endure the hydrodynamic forces (Montefalcone *et al.* 2016). This is our case, since shoots are settled directly, or through a thin *matte* layer, on rocky outcrops or stones corroborating the statement emphasized by Calvo *et al.* (1995) and Badalamenti *et al.* (2015), that the distribution, settlement and development of *P. oceanica* meadows often coincide with the occurrence of this kind of substrate, because of the peculiar traits of the root system, which enhances the mechanical properties of the plants (Badalamenti *et al*. 2015, Balestri *et al.* 2015, Tomasello *et al.* 2018, Zenone *et al.* 2020).

The resolution of drone acquisition was so high that allowed the estimation of other features of the reef, such as the distribution of surrounding dead *matte* and its thickness, which in some points reached 1 m. To our knowledge, this is the first time that such data have been obtained through aerial images, giving the opportunity to estimate not only the distribution of the foliar canopy, but also of the dead hypogeal component represented by dead *matte*. Although these estimates were possible for only the emerging and visible portions of the *matte*, which sometimes can be much extended vertically below the bottom (Lo Iacono *et al.* 2008, Tomasello *et al.* 2009, Monnier *et al*. 2019), the typology of data obtained may have relevant implications on the implementation of the methodology required for better estimating these biogenic structures and consequently the ecosystem services they provide. Indeed, one of the most important values of the *P. oceanica* ecosystem is represented by the vast long-term carbon stock accumulated over millennia within the *matte* (Mateo *et al*. 2006). In reef areas where *P. oceanica* meadows regressed and the leaf canopy disappeared, the underlying *matte* is no longer protected against erosion by high-energy waves (Boudouresque *et al.* 2012). Surveys carried out with the same methodology as that adopted in the present study will therefore allow repeated *matte* thickness measurements in order to improve global estimates of Mediterranean seagrass Blue Carbon sinks and to highlight eventual erosion phenomena, as urgently recommended by the scientific community (Pergent *et al.* 2014).

The use of very high-resolution images made it also possible to identify atoll-like formations. These structures have been observed in small areas of the Mediterranean Sea, along the Tunisian, Turkish and Corsican coasts (Blanpied *et al.* 1979, Boudouresque *et al.* 1990, Pasqualini *et al.* 1995), and in Sicily, so far, exclusively at the Stagnone di Marsala, a lagoon-like coastal basin on the Eastern coast (Calvo & Fradà Orestano 1984). The results of the present study indicate that atoll formations may be more frequent than previously thought. Increasing evidences suggest that *Posidonia* atolls are the result of particular dynamics occurred in very shallow meadows, often subjected to stressful conditions where seagrass meadows can live at the extreme of their environmental tolerance (La Loggia *et al*. 2004, Tomasello *et al.* 2009). Pergent & Pergent-Martini (1995) and Boudouresque *et al.* (2012) hypothesized that atolls origin from nearly circular patches of *P. oceanica*, where plagiotropic (horizontal) shoots only grow outwards, whereas the shoots on the central portion of the patch die. Notoriously, *P. oceanica* is considered as an 'ecosystem engineer' species, given its ability to affect significantly physical, chemical and biological features of their environment up to determine inhospitable conditions for itself (Boudouresque *et al.* 1975). During *matte* elevation, shoots and leaves can rise up to the sea surface forming small lagoon-like environments, within which the hydrodynamic regime reduces abruptly inducing the increase in temperature and salinity variability in comparison with the adjacent open sea. The atolls detected in this study lie right at the inner edge of the reef and in very shallow waters where water circulation is even more affected by seagrasses, especially at low tide when the canopy can occupy the entire water column (Koch *et al.* 2006). Under this circumstance one important consequence is that hydrodynamic regime (and covariates), considered a prominent factor in shaping seagrass landscape (Bell *et al.* 2006), within seagrass patches located at the inner edge of reefs, falls further down greatly affecting shoot vitality. Continuous measurements of environmental variables (especially temperature and salinity) are clearly necessary to characterize the environment inside the atolls. Another important finding that arose from our analysis is that atolls appear very irregularly shaped. Although spatially explicit models have not yet been developed in a way that would explain such complex patterns (Duarte *et al.* 2006), a new hypothesis can explain the phenomenon underlying atoll formations. In a diachronic study, Bonacorsi *et al.* (2013) observed that atolls origin and develop from a single self-maintaining patch, more or less regularly shaped across time. However, this model does not support the complexity of atolls recorded in the present study, where, indeed, atolls appear to be rather elongated in shape or with multiple arcs of different amplitude delimiting their contours. According to Bonacorsi *et al.* (2013), such morphological complexity may be explained only assuming a very variable speed of horizontal growth of the rhizomes placed at the periphery of the patches. Alternatively, a new hypothesis on atolls formation can be formulated, by considering such structures as the results of the union of several patches that came into contact during clonal expansion, still maintaining the geometric memory of their union for a certain time interval. Previous studies carried out along Sicilian coasts seem to support the multi-patch origin here hypothesized, since the genetic structure of atolls was demonstrated to be composed by multiple clones (Tomasello *et al.* 2009).

The *P. oceanica* reef model here presented, obtained by integrating a large amount of extremely accurate photographic data and derived bathymetry, allowed to realize a 3D reconstruction with great realism. The results obtained made it possible to graphically reproduce a spatially heterogeneous mosaic with accuracy unthinkable a few years ago. The devices here employed are being continuously implemented in terms of miniaturization and lightening of vehicles, sensor resolution and powerful softwares. Surely the advent of these technologies represents the beginning of a new era in the study and monitoring of these important natural monuments, since many of the limitations of classical methods applied at seagrass landscape level such as incorrect positioning, low resolution and consequent misleading interpretation of data (Bell *et al.* 2006), may now be considered overcome.

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BRIDGING RISK ASSESSMENT OF HUMAN PRESSURE AND ECOSYSTEM STATUS

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ENVIRONMENTAL RISK ASSESSMENT
ANTHROPOGENIC PRESSURES
STATUS – PRESSURE LINK
EBQI
MEDITERRANEAN ECOSYSTEM

ABSTRACT. – Within the frame of a natural environment impacted by anthropogenic activities, assessing and quantifying their related pressures are essential to its management and are linked to its status. However, there are no, or very few, geographical areas where thorough knowledge of human uses and sources of impact is available as a basis to quantify these pressures. For this reason, we propose to grade impact sources based on risk assessment index using semi-quantitative rating grids. The impact source is defined as the environmental factor responsible for the impact (*e.g.*, sewage, fishing activity or coastal development). The environmental Risk Assessment of Marine Ecosystems (RAME) is based on several combined rating criteria in order to obtain a criticality score. These semi-quantitative criteria are: (S) the sensitivity of the environment, the ecosystem or the species; (I) the importance of the impact source; (D) the distance from the impact source, and (O) the frequency of occurrence of the pressure. Thereafter, the index is weighted by a criterion of control that is related to the environmental management (M). This method can be adapted to all types of pressure and is not specific to any situation.

INTRODUCTION

To confront strong anthropogenic pressures and conservation efforts, management procedures are necessary but must be tailored to the environmental risks. Effective management of the natural environment and human pressure requires assessment of the ecological status of the ecosystem. In the European Union (EU), within the framework of the Habitats Directive (HD, 92/43/EC) and the Marine Strategy Framework Directive (MSFD, 2008/56/EC), measures related to the marine environment are meant to enhance its conservation, in the context of sustainable use of ecosystem services. Ecosystem services form a basis for the recognition and the economic valuation of environmental processes that have beneficial consequences for human wellbeing (Millennium Ecosystem Assessment 2005). The ecosystem approach offers the best response possible for the assessment of the ecological status, taking into account the functioning of the ecosystem as a whole on based on several functional compartments (Laffoley *et al*. 2004, Boudouresque *et al*. 2020b). For the northwestern Mediterranean Sea, the Ecosystem Based Quality Index (EBQI) is representative of this approach (Personnic *et al*. 2014, Ruitton *et al*. 2014, Rastorgueff *et al*. 2015, Thibault *et al*. 2017). The management and the conservation of the marine environment require a good understanding of the scope and the intensity of human activities. The prerequisite for linking ecological status and anthropogenic pressures is proper assessment of the extent and intensity of human threats to the environment.

How to assess anthropogenic pressures in the marine realm?

The choice of the working scale is essential to properly consider the pressure and its point source or widespread nature as well as its local or global scope. A global approach involves the use of extensive spatial data and their overlap onto the status of marine ecosystems. This analytical process provides tools allowing for a global-scale approach to allocate conservation resources or implement broad ecosystem-based management (Halpern *et al.* 2008). But to act locally, more specific information on the pressures are needed. In the marine realm, precise information on pressures in terms of importance, location and field of influence is scarce. Among the existing sources, Medtrix (Andromède Océanologie 2016) and Medam (Meinesz *et al*. 2013) provide a basis for locating a range of pressures such as coastal development or sewage outfalls in coastal areas. Holon *et al.* (2018) have proposed a spatial statistical approach based on a predictive model of multiple coastal anthropogenic pressures which was compared to maps of living and dead *Posidonia oceanica* (Linnaeus) Delile beds but not with their ecological status. In contrast, Giakoumi *et al.* (2015), who took into account the state of the food web to assess the cumulative effect of human impacts on the *P. oceanica* meadows, reached very different conclusions. It emerges that the choice of the working scale and the indicators of the ecological status are essential factors for understanding the relationships between status and pressure. In the terrestrial realm, data on anthropic pressures are more readily available despite uncertainty regarding the carrying capacity of ecosystems and their ability to tolerate a given level of pressure (Fanelli *et al*. 2006, Bartalev *et al.* 2007, Di Bitetti *et al*. 2013, Maraux *et al*. 2013).

Existing indicators or assessment methods are generally specific to one case study. The LUSI is based on terrestrial uses that are not clearly linked to the environmental status (Gardi *et al*. 2010), the HAPI is based on a few human terrestrial and marine pressures related only to the subtidal rocky shore (Blanfuné *et al*. 2017), while the pressure index of Ar Grall *et al*. (2016) only concerns intertidal communities. AFB *et al.* (2019) focused on the impacts on the physical integrity of habitats resulting from professional fishing activities. The 'cocktail effect' is another difficult aspect to consider (human activities can overlap and their impacts on marine ecosystems can be more harmful when combined). Furthermore, global change issues also contribute to this 'cocktail effect', in combination with direct human pressure, making it even harder to assess (Boudouresque *et al.* 2017).

Risk assessment methods

Risk assessment involves various methods, which provide a basis for assessment of pressures and even anticipation of their effects in many technical fields. Assessment methods of this type are already used to consider the environmental risk in the industrial field, when dealing with genetically modified organisms and even with public health. These methods can be adapted to all types of pressure and are not specific to a particular situation. The principles and methods of risk assessment provide the means to conceptualize and assess the risk in the purpose of its management (Aven 2016). Its implementation within the framework of the assessment of the risk for a natural environment exposed to human activities provides a basis for environment assessment, prediction, and management.

The aim of the present work is to adapt the risk assessment approach to marine coastal ecosystems in a Mediterranean context and to define a suitable new indicator. Hitherto, in the marine realm, human pressures have generally been managed with hindsight once the deleterious effects are felt. Even if we try to embed the tripartite 'avoid-reduce-compensate' approach in an environmental management system, certain pressures still persist, gradually intensify and go beyond what the environment can tolerate (Treweek *et al.* 2005, Michelot & Aseeva 2017). Adopting a risk assessment approach makes it possible firstly to connect pressures with the status of the environment and above all provide the means to predict any potential future effects and to plan ahead in order to develop good management decisions.

MATERIALS AND METHODS

A source of impact is any form of human activity that can interact with the natural environment. These sources are human activities with a negative effect (*e.g.,* sewage, fishing activity and coastal development). One impact source can be responsible for one or several impacts. For example, coastal development may lead to the destruction of a natural area but also to the release of substances into the environment that will affect organisms. An impact source can act directly or indirectly on the environment and its effect can be rapid or delayed over time. Conversely, when pressures stop or management measures are implemented, the benefit may take time to become apparent. In the marine realm, it is very complicated to quantify the impacts while it is much more easily achievable to identify the sources. For all these reasons, we propose here to address the issue of pressure from the angle of risk assessment.

Any source of impact can affect the environment depending on its sensitivity, as well as the importance, the distance, and the occurrence of the phenomena. The environmental Risk Assessment of Marine Ecosystems (RAME), by considering several rating grids, combines semi-quantitative criteria that will enable us to obtain a criticality score. These criteria are: (S the sensitivity of the environment, ecosystem or species; (I) the importance of the impact source; (D) the distance from the impact source and (O) the occurrence of the pressure. Thereafter, the index is weighted by a criterion of control that is related to the environmental management system (*e.g.,* level of protection, sewage treatment, regulation). Each criterion is assessed by means of semi-quantitative rating grids with scores ranging from 1 to 4. A score of 1 corresponds to low sensitivity, importance, or occurrence and a significant distance, and good environmental management. A score of 4 highlights a high sensitivity, importance, or occurrence over a short distance, with an ineffective management.

The sources of impact that will be considered here are as exhaustive as possible and include discharges, global change, physical destruction and degradation, noise and vibration, fishing and all other activities not explicitly mentioned but generating pressure which can be compared to that from other activities.

Sensitivity (S): Sensitivity refers to the combination of the ability to tolerate the pressure (resistance) and to recover from a disturbance (resilience) (Holling 1973). The sensitivity (S) of the environment towards a source of impact depends on the species, the community or the ecosystem, but also on specific conditions of the environment, or geographical area, such as exposure of the environment, current, depth, slope, etc. Sensitivity offers groundwork in a local context. For example, a *Posidonia oceanica* seagrass meadow will be more sensitive to the anchorage pressure than a soft bottom while cetaceans will be more impacted by underwater noise than macrophytes. The rating grids (Table I to VIII) attempt to be as exhaustive as possible while retaining a general focus so that each scenario can fall into a category without being based on in-depth knowledge of the biology of the species or the technical details of the sources. The choice of the score for the sensitivity of the species and / or the

Table I. – Rating grid of sensitivity to a discharge altering the salinity or the temperature of the environment.

Score	Characteristics of species or ecosystems concerned by a discharge altering the salinity or the temperature of the environment
4	– Fixed species or ecosystem sensitive to variation in salinity and / or temperature and exposed directly to the plume of water that is desalinated or hypersaline or with different temperature – Fixed species with very slow dynamics: low growth rate for example
3	– Fixed species but tolerant to a range of variations in salinity and / or temperature. The risk being that occasional variations may exceed the tolerance range – Fixed species sensitive to variations in salinity and / or temperature but not directly under the influence of discharge. For example, a species attached to the bottom subjected to a layer of freshwater which is usually to be found near the sea surface and conversely a species of shallow habitats subjected to brine which is generally to be found on the bottom (*e.g.*, discharge from a desalination plant). The risk being that the species is occasionally in contact with the modified water plume
2	– Mobile species but with low mobility, for example not swimming – Stenohaline and / or stenothermal mobile species. These species will be able to flee the impacted area but will not be able to survive there because of the modification of the specific composition of the community
1	– Very mobile, migratory, and swimming species – Species very tolerant to variations in temperature and salinity, euryhaline and eurytherm

Table II. – Rating grid of sensitivity to discharge containing nutrients and/ or organic matter and / or contaminants.

Score	Characteristics of species or ecosystems concerned by a discharge containing nutrients and / or organic matter and / or contaminants
4	– Fixed species / ecosystem, characteristic of clean water, sensitive to eutrophication and contamination even if occasionally – Not competing species and with slow growth rate
3	– Fixed species / ecosystem of clean water, sensitive to eutrophication and contamination but resistant if occasionally exposed to the discharge
2	– Fixed species / ecosystem tolerant to contamination and to an increase in nutrient exposure – Mobile species but with low mobility, for example not swimming, tolerant to pollution – Mobile species sensitive to contamination
1	– Fixed or mobile species tolerant to eutrophication and contamination, or even favored by them – Ecosystem tolerant to contamination

Table III. – Rating grid of sensitivity to discharge containing suspended terrigenous and mineral matter.

Score	Characteristics of species or ecosystems concerned by a discharge containing suspended terrigenous and mineral matter
4	Substratum with slope < 45° and/or calm water conditions with little or no current – Photophilic photosynthetic species at a depth > 10 m – Fixed heterotrophic species with very low dynamics: for example, low growth rate (less than 1 cm/year)
3	Substratum with slope < 45° and/or calm water conditions with little or no current – Photophilic species at a depth < 10 m – Fixed heterotrophic species, with low tolerance to sedimentation – Heterotrophic species with low mobility and sensitive to sedimentation Substratum with slope > 45° and/or exposed, open environment, strong current – Photophilic photosynthetic species at a depth > 10 m – Fixed heterotrophic species with very low dynamics: for example, low growth rate (less than 1 cm/year)
2	Substratum with slope < 45° and/or calm water conditions with little or no current – Sciaphilous photosynthetic species with high growth rate – Heterotrophic fixed species with high growth rate, tolerant to sedimentation – Heterotrophic mobile species Substratum with slope > 45° and/or exposed, open environment, strong current – Photophilic photosynthetic species at a depth < 10 m – Fixed heterotrophic species, with low tolerance to sedimentation – Heterotrophic species with low mobility and sensitive to sedimentation
1	Substratum with slope > 45° and/or exposed, open environment, strong current – Sciaphilous photosynthetic species with high growth rate – Heterotrophic fixed species with high growth rate, tolerant to sedimentation – Heterotrophic mobile species

Table IV. – Rating grid of sensitivity to global change (especially increase in temperature, decrease in pH, rise in sea level, spread of invasive species).

Score	Characteristics of species or ecosystems concerned by global change
4	– Ecosystem / ecosystem engineer species at the limit of its thermo-tolerance range – Calcified fixed species playing the role of ecosystem engineer, sensitive to low pH variation – Engineer species with a very limited range of distribution and/or living at a depth level overwhelmingly impacted by the rise in sea level (*e.g.*, mediolittoral species – sensu Pérès & Picard 1964). The sensitivity increases if the species is slow growing – Specialized species, low tolerance for environmental changes including invasion
3	– Ecosystem / ecosystem engineer species not at the limit of its thermo-tolerance range – Fixed species (non-ecosystem engineer) at the limit of its thermo-tolerance range – Calcified fixed (non-ecosystem engineer) species sensitive to low pH variations – Non-ecosystem engineer species with a very limited distribution area and living at a depth level overwhelmingly impacted by the rise in sea level – Species / ecosystem in competition with invasive species for space but not for trophic, pathological, or physiological interactions
2	– Mobile species at the limit of its thermo-tolerance range – Calcified mobile species sensitive to low pH – Species whose range is limited to a sea level which is partially impacted by the rise in sea level
1	– Thermophilic species / ecosystem – Species not sensitive to decrease in pH – Species with wide range and not affected by the rise in sea level – Competitive species (high growth rate and generalist strategy in life history traits)

Table V. – Rating grid of sensitivity to physical destruction and degradation (burial by coastal development, dredging discharge, anchorage, use of fishing gear, etc.).

Score	Characteristics of species or ecosystems concerned by physical destruction and degradation
4	– Physical destruction of an ecosystem engineer species – Very slow ecosystem recovery (> 10 years) – Ecosystem very sensitive to the use (*e.g.*, trawling) or loss of fishing gear (abrasion, fixed species removed) (*e.g.*, *Posidonia oceanica* seagrass meadows and coralligenous habitat, respectively)
3	– Physical damage of an ecosystem engineer species – Fixed species, non-ecosystem engineer, with low growth rate – Slow ecosystem recovery (5 to 10 years) – Ecosystem moderately sensitive to the use (*e.g.*, nets) or loss of fishing gear (e.g. *Cymodocea nodosa* meadows and coastal detritic bottoms)
2	– Fixed species, non-ecosystem engineer, with high growth rate – Slow moving and fast-growing species – Moderately fast ecosystem recovery (1 to 5 years) – Ecosystem weakly sensitive to the use (*e.g.*, nets, trawling) or loss of fishing gear (*e.g.*, sandy bottom)
1	– Very mobile species (swimming) – Ecosystem relatively insensitive to physical destruction and degradation – Fast ecosystem recovery (< 1 year) – Ecosystem non-sensitive (physically) to the use of fishing gear (*e.g.*, muddy bottom, open water column)

Table VI. – Rating grid of sensitivity to acoustic pollution.

Score	Characteristics of species or ecosystems concerned by acoustic pollution
4	– Species using communication systems for the social organization of the population and echolocation (*e.g.*, cetaceans)
3	– Noise-sensitive species showing behavioral changes (*e.g.*, fish)
2	– Species impacted through physiological or other mechanisms at the individual level that could have long-term consequences
1	– Acoustic pollution-tolerant species

ecosystem towards a source of impact will be made based on the most damaging rating.

A discharge that alters the salinity or the temperature of the environment (Table I) implies that there is no contaminant in the effluent. Only salinity or temperature values can be modified. The salinity of the receiving environment can be altered by a freshwater discharge (*e.g.*, stormwater drainage) or a hypersaline outfall (*e.g.*, desalination plant). The temperature of the receiving environment can be altered by warmer water (*e.g.*, from a water-cooling system) or a cold effluent (*e.g.*, water effluent from a methane gas terminal).

A discharge containing an excess (compared to the ‘natural’ content of the habitat) of nutrients, organic matter or contaminants (Table II) corresponds for example to untreated sewage,

Table VII. – Rating grid of sensitivity of the resource in relation to fishing pressure.

Score	**Characteristics of the resource in relation to fishing pressure**	
	One species or a small number	Fish assemblage
4	– Long life expectancy species > 50 years (*e.g.*, *Epinephelus marginatus*) – Later age at first spawning > 5 years (*e.g.*, *Anguilla Anguilla*) – Slow growth rate species – Target species for fishing – Commercially exploited marine species – Piscivorous species	– Fish assemblage of ultra-oligotrophic waters (*e.g.*, south-eastern Mediterranean) – Deep ecosystem – High mean trophic level of the fish assemblage based on biomass
3	– High life expectancy species (10 to 50 years) – First spawning between 3 to 5 years old (*e.g.*, *Diplodus sargus, Thunnus thynnus*) – Macrocarnivorous species	– Fish assemblage of oligotrophic waters (*e.g.*, Gulf of Lions, north-western Mediterranean)
2	– Medium life expectancy species (2 to 10 years) – First spawning between 1 and 2 years old (*e.g.*, *Symphodus ocellatus*, *S. tinca, Sardina pilchardus*) – Non-target fish but often caught incidentally – Mesocarnivorous species	– Fish assemblage of mesotrophic waters (*e.g.*, Bay of Biscay, north-eastern Atlantic) – Estuarine areas – Medium mean trophic level of the fish assemblage based on biomass
1	– Short life expectancy species usually < 2 years (*e.g.*, *Atherina* spp.) – Early age at first spawning < 1 year (*e.g.*, *Octopus vulgaris*) – High growth rate species – High level of reproduction – Non-target species – Non-commercial species – Planktivorous and herbivorous species – Low trophic level species	– Fish assemblage of eutrophic zone – Upwelling systems fish assemblage (*e.g.*, Peruvian coast) – Low mean trophic level of the fish assemblage based on biomass

wastewater from treatment plant outfalls or industrial discharge. The sensitivity of organisms and ecosystem depends on their tolerance to contaminants, nutrient enrichment, and their mobility.

A discharge containing suspended terrigenous and mineral matter (Table III) implies that no significant quantities of contaminant or organic matter or nutrient are present in the effluent. This discharge may be remarkable for the quantity of mineral suspended matter, such as during coastal work generating a lot of fine particles in the environment or an estuary carrying a man-induced excess load in sediments (*e.g.,* linked to soil erosion following deforestation). The sensitivity of organisms and ecosystems to suspended terrigenous matter depends on their light requirements and their ability to resist burial and therefore in particular their growth rate and the substratum slope.

Global change (Table IV) corresponds to the current climate change, including effects on pH (acidification), oxygen concentration, temperature, as well as biological invasions, rise in sea level, increase in extreme event frequency, etc.

The sensitivity to physical destruction and degradation includes all physical damage, reversible or not, such as anchorage, trawling and the use and loss of fishing gears (Table V). Physical destruction such as burial occurs for example during coastal development, inclusion within a port basin, and the discharge of dredging material. Full recovery is a return to the former state of the habitat, prior to the impact, *i.e.,* to a structurally and functionally recognizable habitat with its associated biological community (La Rivière *et al*. 2018). The assessment of habitat sensitivity to physical pressures is essentially based on expert judgment; the study led by La Rivière *et al.* (2018) can constitute a reference document for coastal habitats. A description of the sensitivity of ecosystems to the loss of fishing gear is available following completion of the methodological guide on the impact of fishing gear (Belloni *et al*. 2019).

Anthropogenic underwater noise is now recognized as a worldwide issue (Williams *et al.* 2015). Most human activities generate noise. Many species of fish and cetaceans are sensitive to sounds because they use them to orient themselves, to communicate with each other, to avoid predators and to feed. Some noises can disorient these species, change their behavior, and even kill or deafen cetaceans. For fish assemblages, several studies have shown that intense noise can have negative effects on certain species such as habitat abandonment, reduced reproductive capacity and increased susceptibility to disease. For example, noise generated by underwater oil exploration that generates powerful sound sources, particularly when using air cannon, shows that fish exposed have sustained significant damage to their auditory sensory epithelium (McCauley *et al*. 2003). The sensitivity toward anthropogenic noise depends on the species and particularly the use of sounds and vibrations in its biology and physiology and its ability to perceive noise (Table VI).

The sensitivity of resources to fishing pressure concerns all marine phyla (*e.g.,* fishes, crustaceans, mollusks). The sensitivity of the resource depends on whether a single or a small number of species or the whole assemblage is considered. Table VII therefore presents the two cases. On the one hand, if we consider only a small number of species, it will be the biological characteristics of the species and its life history traits that will be important. On the other hand, if we consider the fish assemblage, it will be the productivity of the area (*e.g.,* inputs in nutrients, primary production) and the interspecific relationships

Table VIII. – Rating grid of sensitivity to other activities not cited above.

Score	Characteristics of species or ecosystems concerned by other activities
4	– Species/ecosystem whose presence is incompatible with disturbance caused by these other activities
3	– Species/ecosystem highly sensitive to these other activities, human presence, disturbance
2	– Species/ecosystem not very sensitive to these other activities, human presence, disturbance
1	– Species/communities not sensitive to these other activities, human presence, indifferent to disturbance

Table IX. – Rating grid of importance of a discharge.

Score	Characteristics of the importance of a discharge
4	– Discharge of toxic substances known to be dangerous, toxic, which can lead to mortality at the doses contained in the effluent – Discharge containing radioelements – Industrial discharge obtained with derogation and from an ICPE (Installation Classified for the Protection of the Environment) – Discharge from untreated sewage outfall
3	– Discharge of CMR (Carcinogenic, Mutagenic and Reprotoxic), proven endocrine disruptors – Trace elements and persistent organic pollutants that can be biomagnified – Sewage treated by a sewage treatment plant but with high flow rate (> 10 000 population equivalent) – Presence of macro-waste altering the natural habitat – Industrial release to ICPE standards – Organic matter and / or suspended matter in high quantity
2	– Discharge of substances without proven toxicity but potentially biomagnified – Sewage treated by a sewage treatment plant but with low flow (< 10 000 population equivalent) – Presence of macro-waste that does not alter the functioning of the natural habitat – Industrial discharge compliant with standards and not concerning an ICPE – Discharge containing no pollutant substance but with a different temperature or salinity from the environment – Organic matter and / or suspended matter in low quantity
1	– Water discharge without pollutant, organic matter, or nutrient, in very low quantity that cannot cause variation in salinity or temperature of the receiving environment

(trophic, mutualistic, etc.), which will influence this sensitivity to the exploitation of the resource. The ratio fish production/primary production takes into account the nutrient richness (Cresson *et al*. 2020), while the fish production is also linked to the length of the food web (Sommer *et al.* 2002). In the Mediterranean, we can distinguish two zones according to their richness in nutrients, the southeast with ultra-oligotrophic waters, and the northwest with oligotrophic waters (Moutin *et al.* 2012).

Other impacts can result from human activities such as scuba diving, snorkeling, or other recreational activities (Table VIII). Note that most of these activities can be included among the activities cited above (Tables I through VII). For example, sensitivity to boat anchorages is dealt with in the rating grid of sensitivity to physical destruction (Table V). Sensitivity to yachting activities will be concerned both in the noise grid (Table II and in the discharge grid (Table VI) for pollution generated by grey and black water effluents and hydrocarbons.

Importance (I): The importance (I) of an impact source reflects the harmfulness of the source for a species or an ecosystem (Tables IX through XV). It is linked to its nature (*e.g.*, toxicity), its flow and intensity (*e.g.*, quantity, level). For example, information on wastewater treatment plant discharges in France is available on the website of the French Ministry of the Environment (*Ministère de l'Écologie et de la Transition Solidaire)* (http://assainissement.developpement-durable.gouv.fr/index.php). It should be noted that even at a relatively low levels of importance when compared to other anthropogenic sources, it could still lead to long-term exposure to sessile marine organisms and cause significant damage. This is accounted for by the sensitivity (Tables I to III) and occurrence criteria (Table XVII).

Global change includes several phenomena such as increase in temperature, decrease in pH, rise in sea level, invasive species. For the Mediterranean Sea, T-Mednet, an observation network on climate change impact in marine coastal ecosystems, collects seawater temperature and mass mortality events data from scientific observers all around the Mediterranean (Garrabou *et al.* 2018, 2019). The rating grid of the importance of global change is based on the RCP (Representative Concentration Pathway) climate scenario (Guiot & Cramer 2016). The baselines for the Mediterranean, reference status and predictions are to be found in Shaltout & Omstedt (2014) for sea surface temperature, in Jackson & Jevrejeva (2016) for the sea level and in Zunino *et al.* (2017) for acidification. For the NIS (Non-Indigenous Species), UNEP/MAP-RAC/SPA (2008) gives examples of impact of NIS on ecosystems under consideration.

Coastal areas play an essential economic, social, and political role in most countries that are conducive to extensive artificialization of the shoreline at the expense of littoral underwater ecosystems. Coastal development damage may be direct (*e.g.*, burial, destruction) or indirect (*e.g.*, sedimentation, contaminant input, erosion, and increased turbidity). Moreover, other activities can cause physical destruction and degradation of ecosystems such as dumping of dredged material, sand replenishment

Table X. – Rating grid of the importance of global change (increase in temperature, decrease in pH, rise in sea level, Non-indigenous/invasive species - NIS).

Score	Characteristics of the importance of global change
4	– Non-indigenous species (NIS) profoundly altering the functioning of the ecosystems in the area – Increase in mean temperature > RCP8.5 climate scenario – Frequent (more than 1 every 5 years) and intense thermal anomalies and related mass mortality events in invertebrate communities – Decrease in pH > 0.2 compared to the baseline – Rise in sea level > RCP8.5 climate scenario
3	– NIS modifying several interactions between species within the ecosystem – Increase in mean temperature between RCP8.5 and RCP2.6 climate scenario – Thermal anomalies (frequency from 1 every 5 years to 1 every 10 years) and related mass mortality events in invertebrate communities – Decrease in pH between 0.1 and 0.2 compared to the baseline – Rise in sea level between RCP8.5 and RCP2.6 climate scenarios
2	– NIS modifying some interactions between species within the ecosystem – Increase in mean temperature close to RCP2.6 climate scenario – Rare thermal anomalies (less than 1 every 10 years) – Decrease in pH close to 0.1 compared to the baseline – Rise in sea level close to RCP2.6 climate scenario
1	– No significant changes in the ecosystems due to NIS – Increase in mean temperature < RCP2.6 climate scenario – No significant decrease in pH – Rise in sea level < RCP2.6 climate scenario

Table XI. – Rating grid of the importance of coastal development, burial (dredging discharge) and physical destruction and degradation.

Score	Characteristics of the importance of physical destruction and degradation
4	– Irreversible destruction (on the scale of a human life) by coastal development, burial, etc. and affected area ≥ 10 m^2 – Reversible degradation and affected area ≥ 100 m^2; reversible degradation is for example, temporary abrasion of the substrate or a rearrangement of a sandy bottom
3	– Irreversible destruction (on the scale of a human life) by coastal development, burial, etc. and affected area from 1 to 10 m^2 – Reversible degradation and affected area from 10 to 100 m^2
2	– Irreversible destruction (on the scale of a human life) by coastal development, burial, etc. and affected area < 1 m^2 – Reversible degradation and affected area from 1 to 10 m^2
1	– No direct destruction by coastal development – Reversible degradation and affected area < 1 m^2

of beaches, and low-crested structures. The importance of coastal development, burial and all physical destructions depends on the affected surface and whether it is reversible or not.

The importance of anchorage pressure can be based on the daily mean number of anchorages during the peak frequentation period and the size of the boat's anchorage in the studied area (Abadie *et al*. 2016, 2017). To collect such information, it is necessary to monitor the boats frequentation, which has been rarely attempted. In other studies, it is only the number of boats per day and surface unit that is considered to quantify the anchorage pressure (Francour *et al*. 1999, Boudouresque *et al*. 2012, Frachon *et al.* 2013, Rouanet *et al.* 2013, Claeys *et al*. 2017) or the use of AIS data (Automatic Identification System; Deter *et al*. 2017). Thresholds have been proposed by Boudouresque *et al.* (2012) such as a maximum density of 10 anchorages per day and per hectare during the peak period and a mean of 2 anchorages per day and per hectare (annual mean). This threshold does not consider the boat size and is only suitable for small and medium size boats (less than 24 m-80 feet in length). The rating grid for the importance of anchorage is therefore based on criteria for which we can easily provide answers based on occasional observations or field knowledge by managers and based on the studies cited above.

Ambient ocean noise is generated by a variety of sources of both natural (biological and ambient ocean noise) and anthropogenic origin. Ambient noise levels in the open ocean increased approximately by 3.3 dB per decade during the period 1950-2007 and can be attributed primarily to commercial shipping activity (Frisk 2012). It is estimated to be ~ 90 dB in 2007 in the open ocean (55 % from natural noise and 45 % from shipping noise; Frisk 2012). In coastal areas, noise can locally increase above this ambient noise, depending on the anthropogenic activities. Hermannsen *et al* (2019) underline that small recreational motorized vessels dominate the anthropogenic noise in the shallow water soundscape especially in coastal areas. In the framework of the MSFD, the noise is considered in terms of intensity but also according to its duration. A distinction must be made between impulsive emissions (energetic noise emissions of very short duration) and continuous emissions (permanent noise emissions), and this is examined within the occurrence grid (Table XVII). The importance of noise pollution is therefore assessed considering the noise source level (decibels; Table

Table XII. – Rating grid of the importance of anchorage. The numbers given in the table represent the mean number of boats anchored/day/km^2 during the peak frequentation period (July and August for the Mediterranean).

Score	Characteristics of the importance of anchorage	
	Monitoring of anchorage during the peak frequentation period	No monitoring of the anchorage, occasional observations or managers' field knowledge
4	– Boats ≥ 200 m long, ≥ 2/day/km^2 (mainly cruise vessels) – Boats 21-200 m long, ≥ 7/day/km^2 – Boats 10-20 m long, ≥ 16/day/km^2 – Boats <10 m long, ≥ 60/day/km^2 – Total number of boats ≥ 50/day/km^2 (mainly small boats but of unknown length)	– Boats > 200 m long are regularly in the area – Boats 21–200 m long are frequent in the area – Boats 10-20 m long are numerous during the peak season – Boats < 10 m long are very abundant during the peak season – The whole area is occupied by moored boats, anchorage carrying capacity reaches its limits during the peak season
3	– Boats ≥ 200 m long, 1/day/km^2 – Boats 21-200 long, 2 to 7/day/km^2 – Boats 10-20 m long, 8 to 16/day/km^2 – Boats < 10 m long, 30 to 60/day/km^2 – Total number of boats 20-50/day/km^2 (mainly small boats but of unknown length)	– Boats > 200 m long are occasional in the area – Boats 21-200 m long are occasional – Boats 10-20 m long are frequent – Boats < 10 m long are numerous – Anchorage carrying capacity occasionally reaches its limits
2	– Boats 21-200 m long, ≤ 2/day/km^2 – Boats 10-20 m long, 3 to 8/day/km^2 – Boats < 10 m long, 10 to 30/day/km^2 – Total number of boats 8 to 20/day/km^2 (mainly small boats but of unknown length)	– Boats 21-200 m long are very occasional in the area – Boats 10-20 m long are occasional – Boats < 10 m long are frequent – Anchorage carrying capacity never reaches its limits
1	– Boats 10-21 m long, ≤ 2/day/km^2 – Boats < 10 m long, maximum 10/day/km^2 – Total number of boats ≤ 8/day/km^2 (mainly small boats but of unknown length)	– Boats 10-20 m long are very occasional – Boats < 10 m long are occasional

Table XIII. – Rating grid of the importance of noise pollution.

Score	Characteristics of the importance of acoustic pollution
4	– Sound level above 180 dB (*e.g.*, supertanker, more than 200 dB for active sonar or seismic airgun array)
3	– Sound level from 150 to 180 dB (*e.g.*, frigate, dredger, echo sounder)
2	– Sound level from 110 to 150 dB (*e.g.*, sidescan, small motorized vessels at speed > 9 km/h)
1	– Sound level below 110 dB (*e.g.*, equivalent to sailing, submarine, small motorized vessels at speed < 9 km/h)

Table XIV. – Rating grid of the importance of fishing activities.

Score	Characteristics of the importance of fishing activities
4	– Industrial fishing activities using bottom-contact fishing gear (*e.g.*, trawling, dredging) – Extensive artisanal fishing activities, using active bottom-contact fishing gear (*e.g.*, coastal trawling, 'gangui' in the fisher's local dialect of Provence) – Extensive recreational fishing activities, spearfishing, and jig fishing – Fishing techniques with high level of by-catch – Fishing activity that causes disturbances greater than the population's renewal capacity (over-exploitation of a fish stock)
3	– Industrial fishing activities using fishing gear in the water column – Artisanal fishing using passive and selective fishing gear (*e.g.*, fixed net and bottom longline) – Intensive recreational fishing activities: angling on the bottom
2	– Occasional artisanal and recreational fishing activities; the occasional nature of the practices will be judged according to the frequentation of the area as reported by observations made in the field either by the managers or by people used to frequenting the area – Recreational fishing activities: trolling fishing
1	– Rare fishing activities

XIII; Boyd *et al* 2008). The frequencies of the noises are not considered even if this parameter is important for the effects and for the propagation of the signal, but this information is generally absent not available.

To assess the importance of fishing activities, we distinguish recreational, artisanal (*i.e.*, small scale) and industrial fishing and the type of fishing gear (Table XIV). A study by IFREMER (2008) lists the impacts of professional fishing gears on habitats and species. The degree of impact depends on the gear and the type of habitat, therefore fishing techniques can be classified according to the potential damage they can have on the habitat. Fishing gear in contact with the bottom can disturb it. Substrate shifts, destruction of carrying capacity and reduction of the complexity of habitats (uniformization of the bottoms) can be induced. Among biological impacts, fishing gear can destroy organisms fixed on the bottom (*e.g.*, the giant mollusk *Pinna*

Table XV. – Rating grid of the importance of other activities or pressures not cited above.

Score	Characteristics of the importance of other activities
4	– Activity that causes disturbances going beyond the population's renewal capacity (*e.g.*, permanent trampling of an area) – Activity that creates a continuous and permanent disturbance of the species – Disturbances going beyond the resilience capacity of the ecosystem
3	– Activity that creates frequent disturbances of the species – Effect on the population (recruitment, abundance, sex-ratio, demographic structure, etc.)
2	– Activity that creates a temporary disturbance – Physiological effects on certain individuals without endangering the population – Vital needs of species disturbed but reversible, less than resilience
1	– Activity that does not create any disturbance for the communities – Neutral activity for populations or the ecosystem – No impact on the vital needs of individuals (O_2, light, nutrient, etc.)

Table XVI. – Rating grid of distance from impact source.

Score	Distance between a point-source pressure and a point impact (see Fig. 1C)	Distance between a point-source pressure and a diffuse impact (see Fig. 1A)	Distance between a diffuse-source pressure and a diffuse impact (see Fig. 1B)
4	0	0 to 1 km	0 to 1 km
3	0 to 0.1 km	1 to 3 km	1 to 3 km
2	0.1 to 1 km	3 to 6 km	3 to 6 km
1	> 1 km	> 6 km	> 6 km

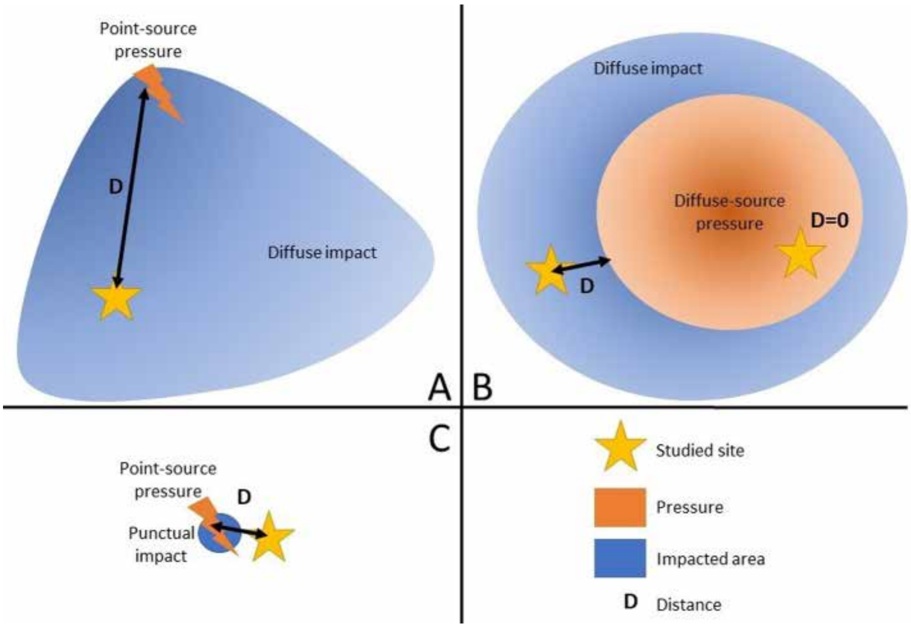

Fig. 1. - Distance measurement between pressure and the impacted studied site. **A**: Distance between a point-source pressure and a diffuse impact; **B**: Distance between a diffuse-source pressure and a diffuse impact; **C**: Distance between a point-source pressure and a point impact.

nobilis Linnaeus, 1758), move animals outside of their natural habitat, dig up individuals, contribute to the dissemination of invasive species and exhaust resources. Physical and chemical effects are also possible with the physical contact of the fishing gear with the substrate, which can induce a resuspension of the sediment, turbidity, and chemical effects (impact on biogeochemical

Other impacting activities, including scuba diving, snorkeling, or other recreational activities, are important in coastal areas of the Mediterranean Sea (Rouanet *et al.* 2017). New activities are also regularly created, can establish a trend, and be practiced for a few years along the coast. Table XV sum up the importance of all activities, which could not be taken into account in the other table.

Distance (D): The distance (D) concerns the distance between the source and the impacted environment (Fig. 1). A point-source pressure is in a limited area unlike a diffuse-source pressure, which cannot be located precisely, and which concerns a large area; however, both can have widespread effect. The distance is easy to measure if it is related to a point-source pressure with a geographically limited impact (*e.g.*, dredging, anchorage; Fig. 1C), but more difficult to estimate if the impact is diffuse. The origin of diffuse impact can be both diffuse-source pressure

Table XVII. – Rating grid of occurrence of an activity, discharge, or coastal development.

Score	Occurrence of activity / discharge / development works
4	Activity/discharge: – Daily activity – Continuous or daily discharge or activity (*e.g.*, sewage outfall, commercial shipping) – Noise emissions more than half the time (*e.g.*, > 12 h/day or 15 days/month) Coastal development: – Development works: ≥ 2 events every year
3	Activity/discharge: – Seasonal activity but frequent in the season concerned – Discharge less than 1 time per week – Noise emissions between half the time and 1/10th of the time (*e.g.*, between 2.4 h to 12 h/day or between 3 to 15 days/month) Coastal development: – Development works: 1 event every year
2	Activity/discharge: – Activity ≤ 1 time per month – Discharge less than 1 time per month – Noise emissions of short duration, between 1/10th to 1/30th of the time (*e.g.*, between 0.8 h to 2.4 h/day or between 1 to 3 days/month) Coastal development: – Development works: 1 event every 2 to 5 years
1	Activity/discharge: – Activity ≤ 1 time per year – Discharge occur less than once a year – Noise emissions of short duration, less than 1/30th of the time (*e.g.*, less than 0.8 h/day or 1 day/month) Coastal development: – Development works: < 1 event every 5 years

Table XVIII. – Rating grid of environmental management.

Score	Characteristics of environmental management
4	– No management measures exist – No specific regulations for uses and discharges – Unsuitable actions for the protection of the environment – No fishing quota, fishing regulation and fishing labor regulations
3	– A few management measures have been introduced but are insufficient, no policing or field inspections – Management based on mitigation – Only fishing effort regulation or fishing quotas
2	– Species-centered management actions with policing and field inspections – Management based on reducing the importance of impact sources or control – Local fishing regulation (*e.g.*, ban on spear fishing and trawling, artisanal and recreational fishing regulations, fishing charter more restrictive that national and local regulations)
1	– Management measures have been introduced and seem to be effective, with policing and field inspections – Ecosystem-based management with field monitoring to survey the effectiveness of the management – Prevention measures to limit further impact – No-Take-Zone (artisanal and recreational fishing are banned)

(Fig. 1B) or point-source pressure (Fig. 1A). In the case of diffuse-source pressure such as shipping or global change (Table XVI), the distance is measured between the studied site and the closest point of the diffuse-source (*e.g.*, the border of the shipping lane). If the studied area is within the pressure area, the distance is then equal to zero. A discharge is a point-source when the outfall can be located precisely but can be diffuse-source when the discharge affects a large area (runoff water and greywater/blackwater discharges from vessels in a mooring area).

Occurrence (O): To estimate the occurrence of a phenomenon, it is essential to adopt different scales depending on the source related to an activity, a discharge or a coastal development (Table XVII). Coastal development includes all coastal development works such as deployment of coastal constructions (*e.g.*, harbor, dikes, piers), strengthening and extension of external seawalls, rehabilitation of wharfs, public access, roads, and offshore wind turbine arrangements. The rise in sea level, flooding and coastal erosion represent serious threats that could, in the future, increase the need for coastal structure reinforcement. Obviously, depending on the nature of the work, the impact will differ in extent and this notion is taken into account by the importance criterion (Table XI).

Environmental management (M): Environmental management (M) corresponds to all the management measures already existing at the time of the analysis. Management measures can be of several kinds depending on the status of the area (*e.g.*,

Table XIX. – List of the XIII groups of sources of impact, the corresponding DCSMM descriptor, tables to use for the RAME calculation and the name of the RAME.

Sources of impact	Corresponding DCSMM descriptor	Tables					RAME name
		S	I	D	O	M	
Suspended matter (SM) discharge	Sea-floor integrity	III	IX	XVI	XVII	XVIII	SM
Anchorage	Sea-floor integrity	V	XII	XVI	XVII	XVIII	Anchorage
Physical destruction and degradation	Sea-floor integrity	V	XI	XVI	XVII	XVIII	Degradation
Use of fishing gear	Sea-floor integrity	V	XV	XVI	XVII	XVIII	Fishing gear
Discharge of water with different salinity or temperature than the environment	Hydrography	I	IX	XVI	XVII	XVIII	Hydrography
Contaminant discharge	Contaminants	II	IX	XVI	XVII	XVIII	Contaminant
Nutrients and organic matter discharge	Eutrophication	II	IX	XVI	XVII	XVIII	Eutrophication
Waste discharge	Waste	V	IX	XVI	XVII	XVIII	Waste
Acoustic pollution	Energy input	VI	XIII	XVI	XVII	XVIII	Acoustic
Non-indigenous species	Invasions	IV	X	XVI	XVII	XVIII	Invasion
Temperature increase, pH decrease, and sea level rise		IV	X	XVI	XVII	XVIII	Global change
Marine resources	Fishing	VII	XIV	XVI	XVII	XVIII	Fishing
Other activities		VIII	XV	XVI	XVII	XVIII	Other

marine protected area, EU Natura 2000 site, national park) and the ecological status of the ecosystem. The management measures may relate to (i) prevention; (ii) decreasing the importance of impact sources or control and (iii) mitigation measures.

Species-centered management actions can be considered today as inappropriate (Boudouresque *et al*. 2020a, b) as an ecosystem is a complex system of species interactions and the consideration of only one species or a group of species cannot solve ecosystem issues. Consequently, the management actions that can have applicability in the context of ecosystem-based management are of particular relevance. The risk assessment of a marine ecosystem is therefore weighted by a criterion of control that is related to the environmental management (Table XVIII).

Calculation of the Risk Assessment of Marine Ecosystem (RAME): The Risk Assessment of Marine Ecosystem (RAME) for an impact source and an area is calculated by the multiplication of the score for each criterion.

$$RAME = S \times I \times D \times O \times M / 1024$$

The result of the multiplication of the five criteria varies from 1 to 1024. The value obtained is then divided by 1024 to give a score on a scale of 0 to 1. A value close to 0 corresponds to a weak impact of the source and a value close to 1 corresponds to a huge impact.

The different types of pressure exerted in the marine environment could be grouped in 13 sources of impact (Table XIX). The scores of the cumulative value of RAME for each pressure at one site ($RAME_{total}$) range from 0 to 13.

RESULTS

In a given area, generally several anthropogenic pressures are exerted and the RAME must be estimated for each related impact source. For example, for an area subject to sewage outfall, fishing activities and anchorage, a RAME must be estimated for those three pressures. We obtain 3 RAME values: $RAME_{contaminant}$, $RAME_{fishing}$, $RAME_{anchorage}$. For the considered area, we can aggregate the 3 values or use them separately to analyze more precisely the relationships between ecological status and pressures. This approach is particularly relevant if an Ecosystem-Based Quality Index (EBQI) is used allowing assessment of the status of multiple functioning compartments (Personnic *et al*. 2014; Ruitton *et al*. 2014; Rastorgueff *et al*. 2015; Thibaut *et al*. 2017).

In the Bay of Marseille, which is under pressure from multiple sources, the ecological status of three *Posidonia oceanica* seagrass meadows has been assessed using the EBQI method in 2019. The first site located on the 'Plateau des Chèvres' is affected by fishing activities and is located next to the sewage outfall of the Marseille sewage treatment plant. The second site, 'Moyade', is in the core of the Calanques National Park, in a no-take zone since 2012. The third site in the Marseille Prado Bay is subject to various discharges from the city, in particular runoff urban wastewater and occasionally bypass sewage water after a severe storm, and to fairly intensive fishing and boating activities and anchorages.

The results of the EBQI assessments (Personnic *et al*. 2014) give five ecological status classes, from Bad to High: (i) Bad ($EBQI < 3.5$); (ii) Poor ($3.5 \geq EBQI < 4.5$); (iii) Moderate ($4.5 \geq EBQI < 6$); (iv) Good ($6.0 \geq EBQI < 7.5$) and (v) High ($EBQI \geq 7.5$) (Table XX). The ecological status for the 3 sites 'Plateau des Chèvres', 'Moyade' and 'Prado Bay' are respectively poor, good and moderate (Table XX).

The RAME is assessed for each pressure taking into account its importance, its distance, its occurrence, its environmental management and the sensitivity of the ecosystem (Table XX to Table XXII).

The total RAME (cumulative value of RAME for each pressure, $\text{RAME}_{\text{total}}$ in (Table XXII) for the 3 sites 'Plateau des Chèvres', 'Moyade' and Prado Bay' are respectively 3.14, 1.48 and 3.51. The lowest value of risk corresponds to the site of Moyade with the best EBQI status (Table XX and Table XXII). The total RAME values for the other two sites are similar but the contribution of each pressure is rather different (Fig. 2).

The RAME for each human pressure reflects the fishing pressure at both the 'Plateau des Chèvres' and the 'Prado Bay' (Fig. 2). The low fishing pressure observed since 2012 at 'Moyade' explains the good status of the fish assemblage (Fig. 3). This assemblage is not yet at its optimum but is gradually improving as shown by the fish censuses conducted over the last few years at this site (GIS Posidonie, comm pers).

Another major difference between sites is the extent of waste waters discharges at the Plateau des Chèvres site, linked to contaminant inputs

Table XX. – Ecological status assessment by the EBQI method of the *Posidonia oceanica* meadows at the 3 sites. Each functional compartment is assessed according to the ecological status from 0 to 4. HOM: High level of organic matter in the water filter feeders' indicators. LOM: Low level of organic matter in the water filter feeders' indicators. SRDI: Specific Relative Diversity Index is the mean number of species of teleosts observed per transect. Compartments 10 to 12 concern teleosts. EBQI: Ecosystem-Based Quality Index (0 through 10). CI: confidence index. For more details on the method, see Personnic *et al*. (2014).

		Ecological status of functional compartment		
N°	**Functional compartment**	**Plateau des Chèvres**	**Moyade**	**Prado Bay**
1	Rhizomes	4.0	4.0	4.0
2	*Posidonia* leaves	2.5	3.0	3.0
3-4	Leaf epibiota	1.0	3.0	3.0
5	*Pinna nobilis*	1.0	0.0	0.0
6	HOM/LOM	1.0	3.0	1.5
7	Litter	3.0	2.0	4.0
8	*Holothuria* spp.	2.0	3.0	4.0
9	Herbivorous	1.5	1.5	2.5
10	Predators	1.0	2.0	0.0
11	Piscivorous	0.0	2.0	0.0
12	Planktivorous	2.0	2.5	2.0
10-12	SRDI	1.0	3.0	2.0
13	Sea birds	1.5	2.0	2.0
	EBQI	3.9	6.0	5.0
	CI (%)	99	97	100
	Ecological status class	Poor	Good	Moderate
	$\text{RAME}_{\text{total}}$	3.14	1.48	3.51

Table XXI. – Related sources of pressure in each site and information about human pressures. SM: Suspended Matter. NC: Not concerned. See Fig. 2 for values.

RAME	**Plateau des Chèvres**	**Moyade**	**Prado Bay**
SM	Suspended matter from the outfall and the Huveaune River	Suspended matter by runoff during storms	Rhone River diluted water intrusion in Marseille's Bay, runoff from Hubeauve
Anchorage	Small boats, occasionally	Small boats, frequent from spring to autumn for diving activity	Small boats, occasionally
Degradation	NC	NC	Coastal development at 2 km from the site
Fishing gear	Net fishing, spear fishing and angling	NC: No-take zone since 2012	Net fishing, spear fishing and angling
Hydrography	Fresh water from the sewage outfall in surface	NC	Intrusion of the Rhône River fresh water in surface
Contaminant	Sewage outfall at 2900 m	Sewage outfall at 4 700 m	Runoff urban wastewater and occasionally bypass sewage water
Eutrophication	Sewage outfall at 2900 m	Sewage outfall at 4 700 m	Runoff urban wastewater and occasionally bypass sewage water
Waste	Some macro-waste from the outfall	No macro-waste observed underwater	Some macro-waste from the city
Acoustic	Small motorized vessels, no limited speed	Small motorized vessels at limited speed	Small motorized vessels at limited speed, big ships daily traffic (cruise and commercial)
Invasion	Scarce patches of *Caulerpa cylindracea*	Scarce patches of *Caulerpa cylindracea*	Scarce patches of *Caulerpa cylindracea*
Global change	Thermal anomalies	Thermal anomalies	Thermal anomalies
Fishing	Net fishing, spear fishing and angling	NC: No take zone since 2012	Net fishing, spear fishing and angling
Other	NC	Diving	NC

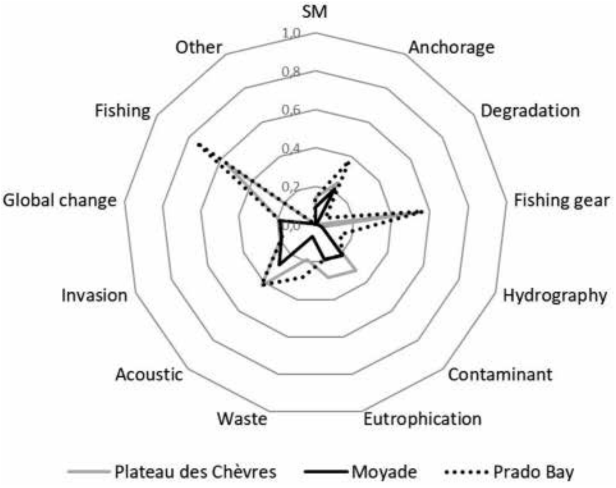

Fig. 2. – Example of RAME assessment (Risk Assessment of Marine Ecosystems) for each human pressure at three sites in the Bay of Marseille. SM: Suspended Matter.

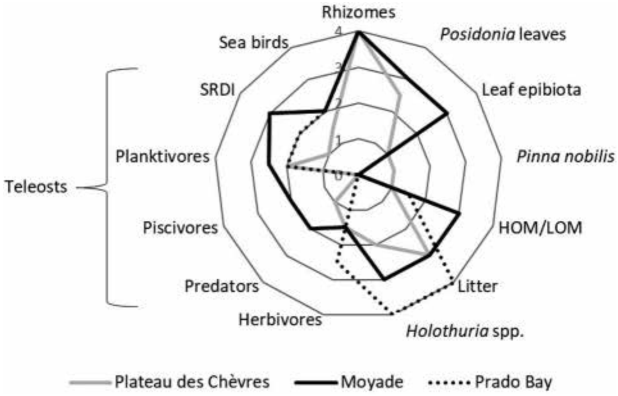

Fig. 3. – EBQI (Ecosystem Based Quality Index) assessment for each functional compartment, at three sites in the Bay of Marseille. HOM: High level of organic matter in the water filter feeders' indicators. LOM: Low level of organic matter in the water filter feeders' indicators. SRDI: Specific Relative Diversity Index is the mean number of species of teleosts observed per transect.

and eutrophication. These inputs are long-standing in the area and certainly explain the important high level of organic matter in the filter feeders' indicators of filter-feeder invertebrates ('bad' status of the LOM/HOM). The low status of the *Posidonia oceanica* leaf compartment at this site (density of shoots and cover) is however a consequence of the local degradation of the seawater quality.

Acoustic pollution and anchorage are present at all sites, although to a lesser extent at 'Moyade'. Global change as well as invasions is similar at all sites as they are within the same water body and habitat. The 'bad' status of *Pinna nobilis* is due to mass mortality events since 2018 due to the unicellular parasite *Haplosporidium pinnae* (Catanese *et al.* 2018).

DISCUSSION AND CONCLUSIONS

The presented methodology may apply to all type of pressures and environments, by simply adapting rating grids and topics. Several sources of human pressures can be assessed in an independent or combined manner and the results can be presented in an integrative way (RAME$_{\text{total}}$) or as single values for each pressure (RAME$_{\text{SM}}$, RAME$_{\text{anchorage}}$, RAME$_{\text{fishing}}$, etc.). In the framework of an ecosystem-based approach for the assessment of the ecological status of a given environment, this method allows a multifactorial analysis.

The field of research on ocean impacts has been growing rapidly over the last decades, but in the meantime the ocean environment has been becoming increasingly degraded. Our scientific knowledge based on the good ecological status of an ecosystem is often far from the pristine state. So, we must work on how to choose a relatively good ecological status so that our analyses on the status – pressure link could be accurate and not underestimated. Furthermore, the relationships between the intensity of the anthropogenic pressures and the ecosystem response is often not linear but features tipping points (*i.e.*, thresholds) that involve dramatic changes from a healthy to a degraded ecosystem or from one status to an intermediate one (Conversi *et al* 2010, Lejeusne *et al* 2010).

Table XXII. – RAME value for each source of pressure and RAME$_{\text{total}}$ and EBQI for each site.

RAME	**Plateau des Chèvres**	**Moyade**	**Prado Bay**
SM	0.141	0.094	0.141
Anchorage	0.250	0.211	0.375
Degradation	0.004	0.004	0.063
Fishing gear	0.563	0.008	0.563
Hydrography	0.035	0.035	0.141
Contaminant	0.316	0.211	0.211
Eutrophication	0.281	0.188	0.188
Waste	0.188	0.063	0.281
Acoustic	0.422	0.281	0.422
Invasion	0.188	0.188	0.188
Global change	0.188	0.188	0.188
Fishing	0.563	0.012	0.750
Other	0.002	0.002	0.003
RAME$_{\text{total}}$	**3.139**	**1.482**	**3.511**
EBQI	**3.9**	**6.0**	**5.0**

This regime shift generally implies a rupture in the resilience of an ecosystem. A non-linear relationship between human threats and the structural status of the *P. oceanica* meadows was detected by Holon *et al* (2018) which add complexity for establishing comprehensive models on relationships between human pressure and ecological status.

How can we take into account the time lag in the response of an ecosystem to a pressure?

The response time is defined as the time it takes for an indicator to record changes (degradation or recovery) in ecosystem health (Contamin & Ellison 2009). The environmental response to a pressure or ecological restoration is generally delayed in function of its intensity, the delay in biochemical and physiological processes and the resilience of the ecosystem (Hamilton 2012, Morales *et al*. 2012). Moreover, studies showing non-injurious effects at the population or the ecosystem level do not mean that there is no impact mediated through physiological or other mechanisms at the individual level that could have long-term consequences (Moore *et al*. 2004). The implications of such time lags in response to degradation or ecosystem restoration are difficult to estimate accurately. Risk assessment methods can overcome this difficulty in ecosystem-based management systems. A major challenge in impact and risk assessment is to link ecological consequences and the impact of pressures. Only the analysis of multiple datasets will be able to provide the means to bridge the ecological status of the ecosystem and the pressures estimated by the RAME. Following the pattern of risk assessment enabling preventive measures when human health is at stake, we could establish preventive rules for environmental management to prevent its degradation and to secure the sustainability of the environment.

How can the impacts be managed?

Finally, the purpose of these analyses is to identify the main sources of impact at a given location and to determine whether their level is bearable by the environment (carrying capacity), and then mitigate the effect with appropriate management measures (Guarnieri *et al* 2016). These answers can be threefold. Firstly, in some cases, countervailing or offset measures may be considered (Hrabanski 2015). Secondly, only the reduction of the source of impact can allow a return to good ecological status. And finally, management aims to continue activities but with preventive measures to mitigate the effects.

Ecosystem-based management of marine ecosystems considers impacts caused by complex interactions between environmental and human pressures (*i.e.*,, oceanographic, climatic, socio-economic) and marine ecosystems. Understanding ecosystem responses to multiple human threats is a major challenge for the implementation of sustainable natural resource management. Risk assessment is a preventive approach allowing the management of human pressures upstream of the damage they could cause. Even more effective ecosystem-based management methods should anticipate the impacts and only a risk assessment approach can make this possible to achieve.

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WHAT BIOTIC INDICES TELL US ABOUT ECOSYSTEM CHANGE: LESSONS FROM THE SEAGRASS *POSIDONIA OCEANICA*

INDICES APPLICATION ON HISTORICAL DATA

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POSIDONIA OCEANICA
SEAGRASS
LIGURIAN SEA
ECOLOGICAL INDICES
HISTORICAL DATA

ABSTRACT. – Anthropogenic pressure on marine ecosystems is affecting water quality and seafloor integrity. Mediterranean seagrass meadows of endemic *Posidonia oceanica* (Linnaeus) Delile are considered a priority habitat under the European Directive 92/43/CEE, given their ecological and economic importance and being an environmental quality indicator. Availability of historical data on three *P. oceanica* meadows along the Ligurian coast (NW Mediterranean Sea) allowed assessing change in the meadow status over time. A number of indicators and ecological indices at different levels of ecological complexity, including the multimetric PREI (Posidonia Rapid Easy Index) adopted by the environmental agencies, were employed and compared. This paper aims at a) defining the health status of the *P. oceanica* meadows and measuring their changes through time collating available historical information; b) evaluating the discriminating power of the different indices and assessing their consistency with each other. The different indices adopted revealed little consistency thus suggesting that no single index can define the health status of *P. oceanica* meadows; as a consequence the use of an indices set is highly recommended to monitor meadow evolution over time.

INTRODUCTION

Biotic indices have long been used to assess the environmental quality of fresh water and terrestrial ecosystems (Cairns & Pratt 1993, Andreasen *et al.* 2001). The application of naturalistic information to elaborate biotic indices in support of marine management is still under development, while it is current practice on land (Bianchi *et al.* 2012). In the last decades, the use of indices to assess marine environmental status in Europe has become frequent, spurred by European directives. Indeed, EU rules (see Directive Proposal 1999/C 343/01, Official Journal of the European Communities 30/11/1999) emphasized the importance of biological indicators to establish the ecological quality of European seas and estuaries (Borja *et al.* 2000). The EU Water Framework Directive 2000/60/EC (WFD) underlined the need for biotic indices, which have been introduced in considerable number. More recently, the EU Marine Strategy Framework Directive 2008/56/EC (MSFD), introduced the concept of "seafloor integrity" to improve the assessment of ecological quality (Bianchi *et al.* 2012). Therefore, the use of seagrass as biological indicators has become a common practice to assess the environmental quality of coastal seas (Pergent *et al.* 1995, Short & Wyllie-Echeverria 1996, Hemminga & Duarte 2000).

Seagrass meadows are declining worldwide due to natural and human-induced events (Short & Wyllie-Echeverria 1996). The endemic Mediterranean seagrass *Posidonia oceanica* (Linnaeus) Delile, 1813 is the most important and abundant seagrass and it is considered a priority habitat for both animals and plant communities, covering different substrata from the sea level down to 40 m depth (Boudouresque *et al.* 2006). There are many ecosystem services provided by *P. oceanica*: it represents an origin of food for many marine and shore organisms, but also plays fundamental roles such as nursery areas for fish and invertebrates and shoreline protection (Vassallo *et al.* 2013). Thus, *P. oceanica* is mentioned in the Habitat Directive 92/43/CEE and, since 1991, is included in the Red List of the International Union for Conservation of Nature (IUCN) as a threatened species of the Mediterranean Sea (www.iucnredlist.org).

A general regression of *P. oceanica* meadows has been recorded in the Ligurian Sea (Bianchi & Morri 2000). In fact, it has been estimated that in the past century the Ligurian *P. oceanica* meadows lost 50 % of their original extent (Peirano & Bianchi 1997, Peirano *et al.* 2005, Burgos-Juan *et al.* 2016). However, as a consequence of conservation policies enforced in the last decades, the regression of *P. oceanica* in the Ligurian Sea has possibly ceased or at least slowed down (Burgos *et al.* 2017, Bianchi *et al.* 2019) as in several places in the Mediterranean Sea (De los Santos *et al.* 2019).

This paper aims at a) defining the health status of three Ligurian *P. oceanica* meadows, using ecological indices and descriptors that work at different levels of complexity (*i.e.*, individual, population, community, and seascape), and measuring their change through time collating all available historical information; b) evaluating the discriminating power of the different indices and assessing their consistency with each other.

MATERIAL AND METHODS

Study area: This study was carried out in Liguria, an administrative region in NW Italy, whose coast border the central and largest part of the Ligurian Sea. Three *Posidonia oceanica* meadows were analyzed, located in Monterosso (MM), Prelo (PR), and Bergeggi (BE) (Fig. 1), where historical data collected along underwater transects were available since the 1980s (Table I).

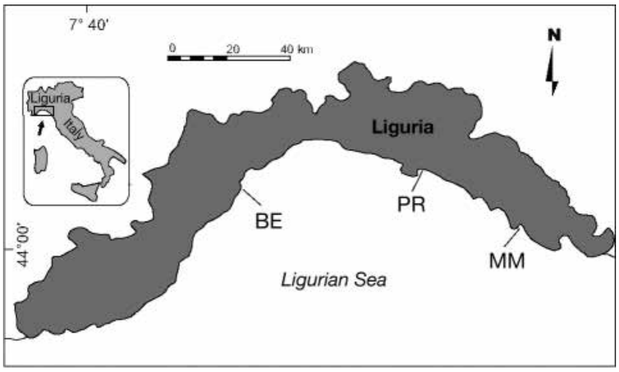


Fig. 1. – Map of the study area and location of the three meadows investigated: Bergeggi (BE), Prelo (PR), and Monterosso (MM).

Table I. – Data sources for the three *Posidonia oceanica* meadows investigated.

Meadow	Year	References
Prelo	2002	Lasagna *et al*. 2006a, b
Prelo	2003	Lasagna *et al*. 2006a ,b
Prelo	2004	Lasagna *et al*. 2006a, b
Prelo	2005	Lasagna *et al*. 2006a, b
Prelo	2006	Lasagna *et al*. 2011
Prelo	2013	Bianchi *et al*. 2019
Prelo	2017	Rigo *et al*. 2019
Bergeggi	1987	Vetere *et al*. 1989
Bergeggi	1992	Sandulli *et al*.1994
Bergeggi	2004	Montefalcone *et al*. 2007
Bergeggi	2009	Montefalcone *et al*. 2010
Bergeggi	2012	Oprandi *et al*. 2014b
Bergeggi	2016	Bianchi *et al*. 2019
Monterosso	1991	Peirano *et al*. 1999
Monterosso	1992	Peirano *et al*. 1999
Monterosso	1994	Peirano *et al*. 2001
Monterosso	1996	Peirano *et al*. 2001
Monterosso	1997	Peirano *et al*. 2011
Monterosso	2002	Montefalcone *et al*. 2007
Monterosso	2008	Bianchi *et al*. 2019
Monterosso	2017	Bianchi *et al*. 2019

Field and laboratory activities: Historical data were compared with data collected more recently (2016 and 2017). All the historical and recent data considered in the analyses were collected during summer season, along underwater depth transects (Bianchi *et al.* 2004) located in the same area of previous studies (Table I). The sampling activity carried out along each transect consisted in a visual estimation, every 10 m along the marked line, of the percentage cover of the seafloor by living *P. oceanica*, dead matte, sand, rock, and possible substitutes (*i.e.*, *Cymodocea nodosa* Ucria, 1870, *Caulerpa taxifolia* (M. Vahl) C. Agardh, 1817, *Caulerpa cylindracea* Sonder, 1845 and *Caulerpa prolifera* (Forsskål) J. V. Lamouroux, 1809). The meadow shoot density was measured at 15 m depth, as recommended by ISPRA (Italian Higher Institute for Environmental Protection and Research (www.isprambiente.gov.it/files/icram/scheda-metodologia-posidonia-new.pdf). Altogether, 18 shoots were sampled for laboratory analyses through plant phenology (Giraud 1977) and lepidochronology. Further analyses were also conducted on the associated epiphytic community: all the epiphytes were scratched from the leaves, then dried and weighed to assess their biomass.

Ecological indices and descriptors assessment: Data obtained from field and laboratory activities were used to define the health status of the three *P. oceanica* meadows, through a set of descriptors and ecological indices working at different ecological complexity levels:

1) Leaf surface (cm^2 $shoot^{-1}$) at the individual level, to describe the physiological status of the plant (Leoni *et al.* 2007);

2) Shoot density (shoots m^{-2}) and lower limit depth (m) at the population level, to understand dynamics in the meadow structure (Pergent *et al.* 1995);

3) Epiphyte biomass (mgDW cm^{-2}) at the community level, which provides rapid information on changes in the water quality (Giovannetti *et al.* 2010);

4) Conservation Index (CI), Phase Shift Index (PSI), and Substitution Index (SI) at the seascape level, to evaluate changes over time of the meadow composition (Moreno *et al.* 2001, Montefalcone 2009);

5) Posidonia Rapid Easy Index (PREI, Gobert *et al.* 2009), which integrates different metrics (shoot density, leaf surface, epiphyte biomass, leaf biomass, and depth and type of the lower limit) and thus encompasses multiple ecological levels.

The above listed indices and descriptors were calculated also from historical data when information was available and complete. Results obtained for each index were then classified in five classes using the reference classifications of ecological quality status (Gobert *et al.* 2009, Montefalcone 2009, UNEP/MAP-RAC/SPA 2011-2015): bad, poor, moderate, good, and high. Only for epiphyte biomass no classification is available.

RESULTS

Meadows ecological quality and trend over time

At the individual level, analysis of the leaf surface (Fig. 2) showed a worsening in the ecological status in all the three meadows. The status of the *Posidonia oceanica* meadow at Prelo changed from good to poor in the last fifteen years. Also in Monterosso a decline in the meadow ecological status from good to moderate occurred from 1991 to 2017. Regarding Bergeggi, data on leaf surface was available only for two periods: the meadow showed a moderate ecological status in 1992, a poor status in 2016.

At the population level (Fig. 3), all indices evidenced stability or even improvement, of the ecological status. Shoot density increased in Monterosso and Prelo meadows, and the ecological status passed from moderate to good in both meadows. Bergeggi maintained a moderate status, notwithstanding a reduction in shoot density values over time. With regard to the lower limit depth, the status remained poor over time in all the three meadows.

At the community level, epiphyte biomass showed a decrease of its values in all meadows, notwithstanding high temporal variability (Fig. 2).

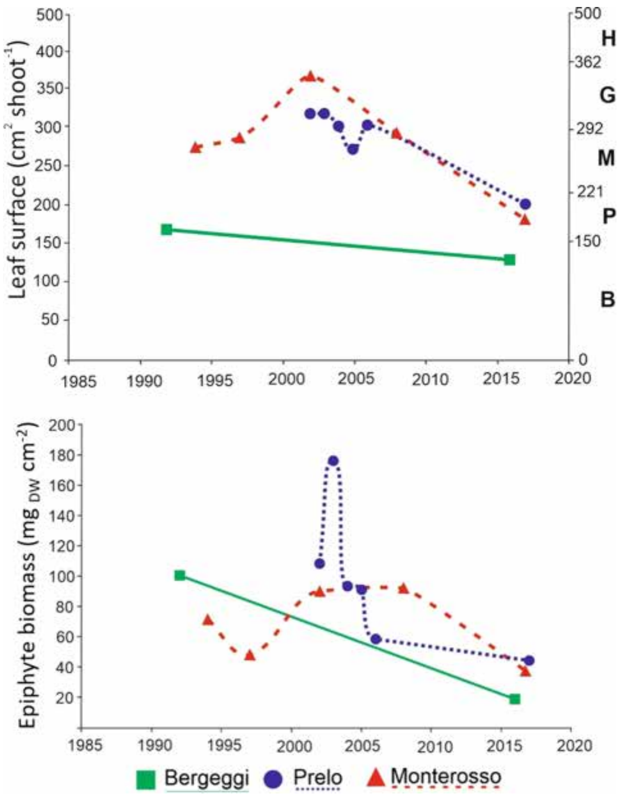


Fig. 2. – Trend over time of the indices at the individual (upper panel) and community (lower panel) level in the three meadows investigated. The y axis on the right side of the leaf surface graph reports reference values of the UNEP/MAP-RAC/SPA (2011-2015). Classification: B: bad, P: poor, M: moderate, G: good, and H: high.

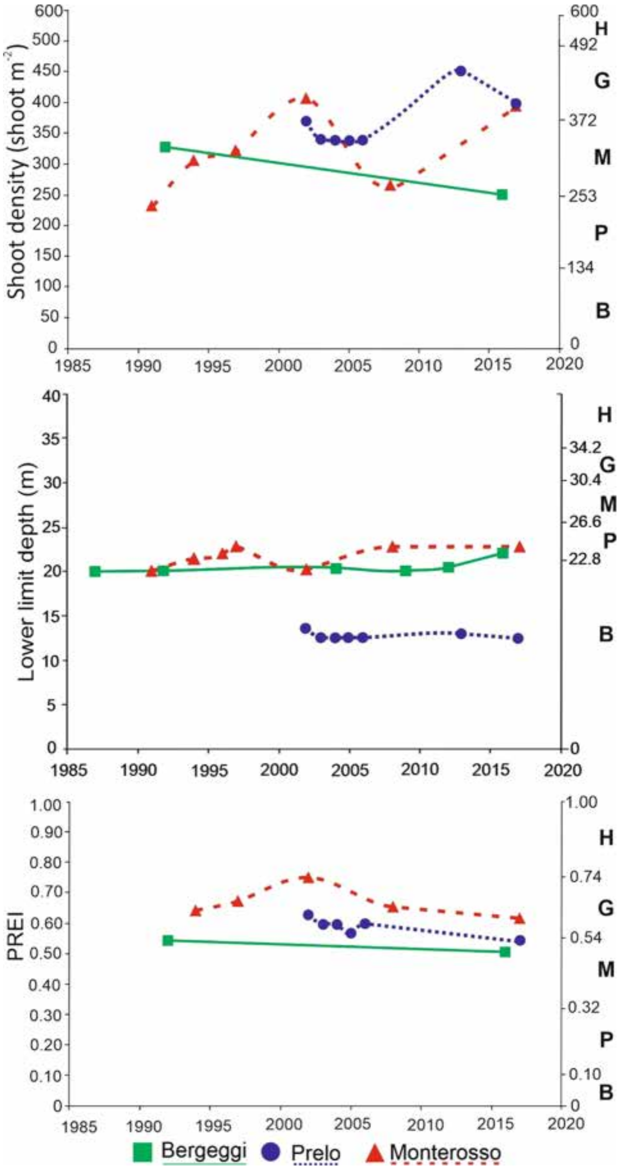


Fig. 3. – Trend over time of the indices at the population level (upper panel: shoot density; middle panel: lower limit depth) and of the multimetric index PREI (lower panel) in the three meadows investigated. The y axes on the right side of the shoot density and the lower limit depth graphs report reference values of the UNEP/MAP RAC/SPA (2011/2015) classification, whilst that of PREI from Gobert *et al.* (2009). Classification: B: bad, P: poor, M: moderate, G: good, and H: high.

At the seascape level, SI and PSI displayed highest temporal variability (Fig. 4). According to CI, Prelo meadow maintained a moderate ecological status, Monterosso showed a slight improvement passing from good to high status, whilst Bergeggi revealed a slight worsening of its status passing from good to moderate. SI evidenced a steady high ecological status in Prelo and Monterosso, but showed a high variability through time in Bergeggi, where it passed from high (1987), good (1992), and moderate (2004), to return again to a high ecological status in 2016. PSI showed different situations in the three mead-

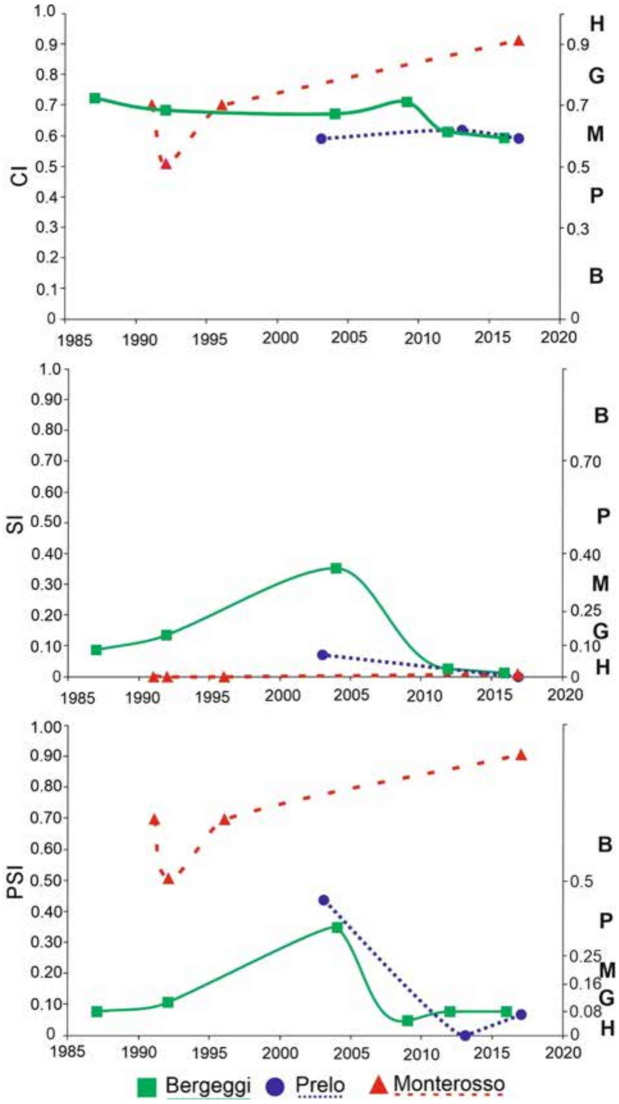
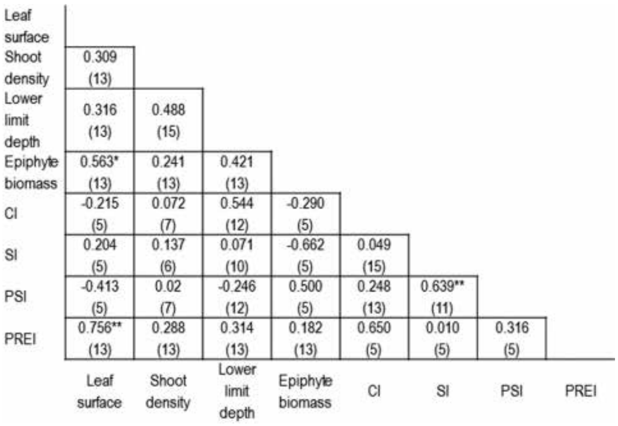


Fig. 4. – Trend over time of the indices at the seascape level (upper panel: CI; middle panel: SI; lower panel: PSI) in the three meadows investigated. The y axes on the right side of the CI, SI, and PSI report values of the classification by Montefalcone (2009): B: bad, P: poor, M: moderate, G: good, and H: high.



	Leaf surface	Shoot density	Lower limit depth	Epiphyte biomass	CI	SI	PSI	PREI
Leaf surface								
Shoot density	0.309 (13)							
Lower limit depth	0.316 (13)	0.488 (15)						
Epiphyte biomass	0.563* (13)	0.241 (13)	0.421 (13)					
CI	-0.215 (5)	0.072 (7)	0.544 (12)	-0.290 (5)				
SI	0.204 (5)	0.137 (6)	0.071 (10)	-0.662 (5)	0.049 (15)			
PSI	-0.413 (5)	0.02 (7)	-0.246 (12)	0.500 (5)	0.248 (13)	0.639** (11)		
PREI	0.756** (13)	0.288 (13)	0.314 (13)	0.182 (13)	0.650 (5)	0.010 (5)	0.316 (5)	

Fig. 5. – Correlation matrix among the indices used in this study. *: $p < 0.05$; **: $p < 0.01$. Numbers in parentheses are numbers of cases.

ows. A clear increase of its values was observed in Prelo, which changed from poor to high status. The status of the Bergeggi meadow dropped from high in 1987 to moderate in 2004 and 2017 according to PSI. The Monterosso meadow showed little variation in PSI values over time, always remaining in a bad status.

The PREI (Fig. 3) showed little or no important change in any of the three meadows. Monterosso ecological status was always classified as good, Bergeggi as moderate, whilst Prelo decreased from good to moderate with time.

Consistency among indices

A significant correlation was found between PREI and leaf surface ($p < 0.01$, n = 13), between PSI and SI ($p < 0.05$, n = 11), and between leaf surface and epiphyte biomass ($p < 0.05$, n = 13) (Fig. 5).

DISCUSSION AND CONCLUSIONS

Most of the indices adopted were consistent in displaying a worsening in the ecological status of the Bergeggi meadow, even though the sampling area is today located within the Marine Protected Area "Isola di Bergeggi". Only SI showed a recovery trend from 2009, thanks to the reduction of the two substitutes *Caulerpa cylindracea* and *Cymodocea nodosa* (Montefalcone *et al.* 2007, Oprandi *et al.* 2014a, b). The Monterosso meadow showed different situations according to the index taken into account. All indices working at the individual and the community levels displayed a worsening trend in the last thirty years. The two indices at the population level, *i.e.*, shoot density and lower limit depth, were discordant: the former showed a recovery, the latter a steady trend. At seascape level only CI vas consistent in showing an increase in the ecological quality of this meadow, while PSI and SI did not show any variations in the ecological status trough the time. The Prelo meadow is affected by a high level of anthropogenic pressures (Lasagna *et al*. 2011). Only indices working at the individual levels showed consistently a worsening in its ecological status during the last fifteen years. On the contrary, the two indices working at the population level and one at the seascape level (CI) showed a steady condition, whilst SI and PSI increased thanks to the reduction of substitutes.

Our results showed that it is not possible to define univocally a trend in the health status of the three *P. oceanica* meadows investigated. The consistency among all the indices was often low (Fig. 6), either considering the same sampling period or among different sampling periods, making even more difficult to identify clear trends over the last thirty years. Indices working at the individual level and the community level often displayed a similar behavior. These indices can be viewed as early warning indicators (Giovannetti *et al*. 2010), responding quickly

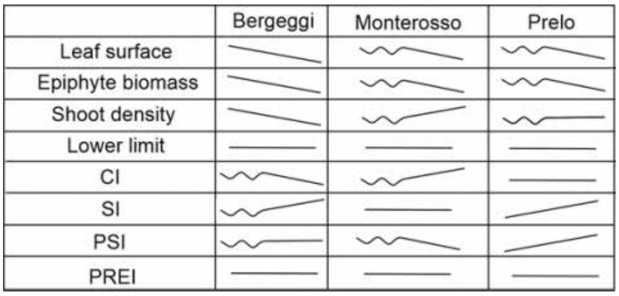


Fig. 6. – Graphical representation of trend over time of each index applied in the three meadows investigated.

to disturbances and to environmental change, and thus being good indicators in short time-scales. Longer times are indeed necessary for descriptors and indices working at the population and the seascape level to show change in the meadow status. Shoot density and lower limit, despite referring to the same ecological level (*i.e.*, population), did not show consistency, probably because the shoot density reacts faster than the lower limit. It should also need to consider that the bad lower limit status could be charged to the climate change occurred in the 21th century that caused a shift in sea level (Bonacorsi *et al.* 2013). Despite the lower limit stability, it should be considered that its limit values, identified by the Mediterranean classification (UNEP/MAP-RAC/SPA 2011-2015) have been recently criticized. In particular they have been considered not completely suitable for the Ligurian coast (Bianchi & Peirano 1995, Oprandi *et al.* 2019) since here *P. oceanica* meadows hardly exceed 30 m depth. This results in a uniform bad status of the analyzed meadows. Local variability in this descriptor could be hidden if the classification at the Mediterranean spatial scale is used. This notwithstanding, the slight increase in the lower limit depth observed in Bergeggi and Monterosso may be a positive signal of meadows recovery.

Shoot density was often consistent with CI. Some authors recognized CI as an index working at population level (Romero *et al.* 2007a), being based on the evaluation of living *P. oceanica* cover along underwater depth transects. It may happen that when the meadow shoots density is high, divers record higher cover values by living plants. Concerning the other seascape indices, *i.e.*, SI and PSI, they react to change only when substitutes are established and a shift in the *P. oceanica* meadow has occurred (Montefalcone 2009).

The multimetric PREI (Gobert *et al.* 2009) always showed a steady trend in all the three meadows, although the single metrics showed changes over time. This result can be due to different situations: i) all the composing metrics display the similar steady behavior; ii) the metrics are discordant, so that a change in each single metric can be mutually counterbalanced by the other metrics in the PREI algorithm, and this is what happened in Monterosso and Prelo meadows; iii) when the majority of the metrics are concordant, as it happened in the Bergeggi meadow, a steady trend of PREI may result when the change is limited. It can thus be concluded that PREI averages and dampens the variability of the single metrics. Integrating a number of metrics into a synthetic index is very useful and effective for monitoring purposes, but it may homogenize the meadow condition. This result pointed out that PREI could not be appropriate, if used alone, to evidence punctual changes in the meadow health. PREI was also significantly correlated only with leaf surface, being the leaf surface one of the metrics used to build this index. Also PSI and SI were highly correlated, considering that PSI contains SI in its formula. Finally, correlation between leaf surface and epiphyte biomass is reasonable as the more the leaf grows the more the epiphytes have space to colonize.

In conclusion, it must be pointed out the importance of long time-series, which are usually very scarce, fragmentary and inhomogeneous, since historical data were collected under different kind of studies and sometimes also with different methodologies. Comparison of data trough time showed inconsistency among most indices, making it difficult to identify a consistent dynamic of Ligurian meadows over time. As a plethora of existing indices, adopting a single index is not recommended to assess the ecological status of *P. oceanica* meadows. All the indices investigated in this paper work at different ecological levels, thus providing different information. This is why many multimetric indices, such as the PosWare (Buia *et al.* 2005, Silvestre *et al.* 2006), the PoMi (Romero *et al.* 2007a, b), the Valencian CS (Fernandez Torquemada *et al.* 2008), the BiPO (Lopez y Royo *et al.* 2010), and PREI have been recently developed to address requirements of the EU directives. However, our study evidenced that using PREI alone would not have evidenced changes at both spatial and temporal scales. For this reason, flanking a multimetric index with other indices, such as for instance the seascape indices here adopted, should be recommended to collect complementary information and to better understand specific drivers of change in seagrass ecosystem.

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LAYING A GAS PIPELINE THROUGH A *POSIDONIA OCEANICA* MEADOW: AN EXAMPLE OF ITS EFFECTS ON PLANT RECOVERY AND EPIFAUNAL DIVERSITY

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LONG TERM RESEARCH
EPIFAUNA
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DREDGING

ABSTRACT. – We report on a case of natural *Posidonia oceanica* recovery after partial meadow destruction due to the laying of a submarine gas pipeline covered with rubble. Ten years later, the new patches were mapped combining the use of an underwater photogrammetry technique and GIS; at the same time, the ecosystem function of the new settled plants was assessed by analyzing major elements of the *Posidonia* motile invertebrate fauna (amphipods, isopods and mollusks). Recruitment of detached plants from nearby meadows was only successful on rubble compared with on adjacent sandy zones. In a subarea of 736.83 m^2, 184 new *Posidonia* spots were established. A high complexity of patch structure (*i.e.*, shoot/unit area) and community richness (*i.e.*, number of species, abundance and diversity) was detected in comparison with other historic shallow stands with the same geographical exposure. The richness in the epifaunal population and the assemblage composition of the three main epifaunal taxocenes also point to a good recolonization capacity. These results call for additional investigations to assess the functioning of the *P. oceanica* ecosystem through the associated epifauna; however, the success of reforestation on rubble along a channel with intensive shipping activity can suggest a solution to manage human requirements and landscape integrity at low cost and with natural donors.

INTRODUCTION

Posidonia oceanica (Linnaeus) Delile meadows represent one of the most productive coastal systems in the Mediterranean basin, but for almost 30 years the seagrass has been included in the Habitat Directive (1992/43/EEC) list and its habitat has been under legal protection, as it is continuously facing significant threats, driven by increasing human activity as well as global warming (Pergent *et al.* 2012). The fragmentation of the *P. oceanica* meadows is considered one of the main issues relative to their decline, also affecting the connectivity and diversity of associated communities and therefore the main trophic fluxes and ecosystem functioning (Mazzella *et al.* 1992, Boudouresque *et al*. 2006, Personnic *et al*. 2014). For these reasons, in order to mitigate *P. oceanica* loss caused by coastal works and infrastructure settlements and to renovate its ecosystem goods and services, several restoration operations have been undertaken (Cunha *et al.* 2012). As this seagrass is a slow-growing species and natural recovery by damaged plants may require decades, several transplanting techniques have been employed involving the introduction of shoots or meadow blocks (Bacci *et al.* 2014). Varying rates of success have been achieved but always at high financial cost when applied at large scales, because of the high work time requirements (both in diving activities and in the lab). Moreover, the large number of shoots that have to be removed from a donor meadow, especially in a marine protected area, is a major concern. It has already been reported (Balestri & Lardicci 2008, Balestri *et al.* 2011) that the use of plant fragments detached after storms seems to have major advantages over traditional restoration techniques.

Here we report preliminary results on the long-term success of ramets naturally arriving on a trench dredged through a *P. oceanica* meadow to lay a gas pipeline between the Phlegrean island of Ischia (Gulf of Naples, Italy) and the mainland. Moreover, the ability of new patches to support an epifaunal community comparable with that typically associated with this seagrass system has been assessed for the first time. Results could be useful as a basis for integrating other ecosystem-based approaches for the purpose of assessing the ecological functioning of this key ecosystem.

MATERIALS AND METHODS

The study sites: The submarine gas pipeline was deployed in 2009 between the island of Ischia (Gulf of Naples, Italy) and the mainland (Fig. 1). It runs on the sea bottom up to the entrance of the harbor of Ischia, where a *Posidonia oceanica* meadow had developed (GAS) (Fig. 1). Along its shallowest stands (from 7.5 to 5 m depth), settled on a matte 1.5 m high, the *P. oceanica* system was removed, and in a channel about 300 m long and 6 m wide the pipeline was laid and covered by rubble.

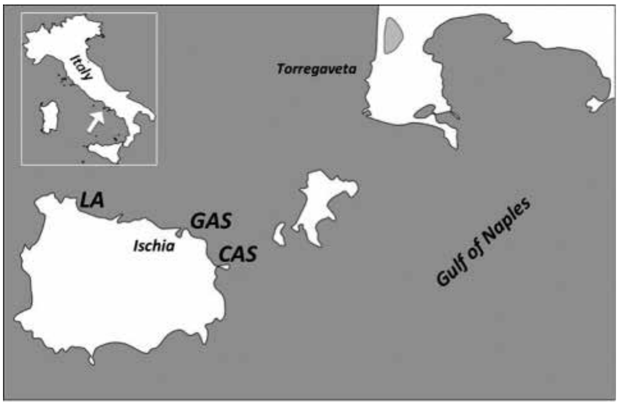

Fig. 1. – Map of the study area in the Gulf of Naples (Italy) with the locations of the monitored *Posidonia* recovery in front of the harbor of Ischia (GAS) and of the other two compared meadows at Lacco Ameno (LA), and Castello Aragonese (CAS).

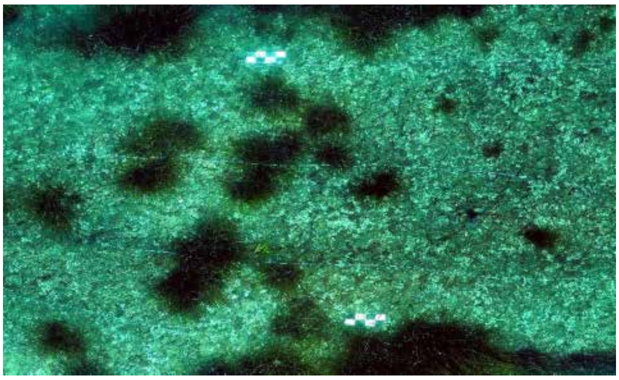

Fig. 2. – Scuba diving operations to map the new *Posidonia* patches and an example of the resulting ortho-photomosaic.

Mapping of Posidonia oceanica *recovery*: After ten years (2009-2019), the results of the natural recovery of the *Posidonia oceanica* by resettlement plants coming from nearby stands were quantified and georeferenced in its deepest sector. The new patches were mapped by the Digital Photogrammetry technique and their cover was determined by GIS according to Cotugno *et al.* (2019). Generally, 5 ropes per time were displayed on the bottom, 2-3 meters apart from each other. Four white/black targets were placed on the corners of the working sub-area and their positions were georeferenced using a GPS (model Garmin Etrex 30), kept on the sea surface, directly above each underwater target. A small camera (Garmin Virb Ultra) able to record video in 4 k at 25 fps was used by a diver moving along the ropes at a constant distance from the bottom in order to obtain at least 70 % of overlapped pictures (Fig. 2). In the lab, 300 to 1,800 frames were extracted from each film sequence and then processed with the Agisoft Photoscan software that also provided us with the automatic calibration to correct focal length and lens aberration. By using structure from motion algorithms (SfM), 3D models of the bottoms were produced and the Orthophoto mosaics were georeferenced using the open source software QGIS. By means of the geometrical QGIS tools, we obtained the surface area and the total cover of the newly settled *Posidonia oceanica* patches.

Patch structure and epifaunal community: Patch structure (number of shoots per unit area) and epifaunal community were sampled in June 2019 within the same two randomly selected plots (0.16 m^2 each). Epifauna was collected first using a diver-operated suction sampler (for a description of the method used, see Garrard *et al.* 2014). The material obtained was stored in ethanol and then sorted into coarse taxonomic groups. Overall, samples were numerically dominated by three groups (mollusks, amphipods and isopods), which together made up 76 and 75 % of the total epifaunal abundance from the two replicates, respectively. Following this, the three groups were identified at the LPT (lowest possible taxon) on an expert basis.

The composition and structure of the three taxocenes was then compared with those from two historically established beds located on the northern coast of Ischia, *i.e.*, off Lacco Ameno (denoted as LA) and Castello Aragonese (CAS), respectively (Fig. 1), which were sampled using the same method and in the same season as the GAS one, although in a different year and at a shallower depth (3 m depth) (Garrard 2013). Notwithstanding the difference in depth, the communities from the three sampled stations can be considered as belonging to a same coenotic unit (shallow stand; Mazzella *et al*. 1992).

Summary variables of the three selected taxocenes (N, number of individuals; S, number of species; H'log$_e$, Shannon-Wiener diversity index) are presented using bar graphs. A statistically reliable data analysis was not possible owing to the small sample size (two replicates). However, standard deviations are shown in the graph as an indication of sample variability.

Multivariate analyses of the structure of assemblages were conducted following PRIMER v6 (Primer-E Ltd., Plymouth, UK) procedures on square root-transformed abundance data. SIMPER (Similarity Percentage test) was used to determine the species that contributed the most to similarity within each assemblage as an indication of their typification ability. nMDS (non-metric multidimensional scaling) plots, overlaid with circles resulting from a previous CLUSTER analysis tested for significance with SIMPROF (Similarity Profile routine), were used for a graphical representation of similarities between samples.

RESULTS

Mapping and patch complexity

The deepest section of the channel (7.5-6.3 m depth) was measured and mapped using the Digital Photogrammetry technique. It measured 128 m in length with a surface area of 736.83 m^2: 184 new patches, of different size, had been settled, covering a surface area of 67.76 m^2 (9.2 %) (Fig. 3). The structural complexity of the patches was found to be high, with a shoot density higher than those recorded in LA and CAS meadows (Table I). Since 2009, no new ramet had settled on bare sand but they only colonized the rubble (Fig. 3).

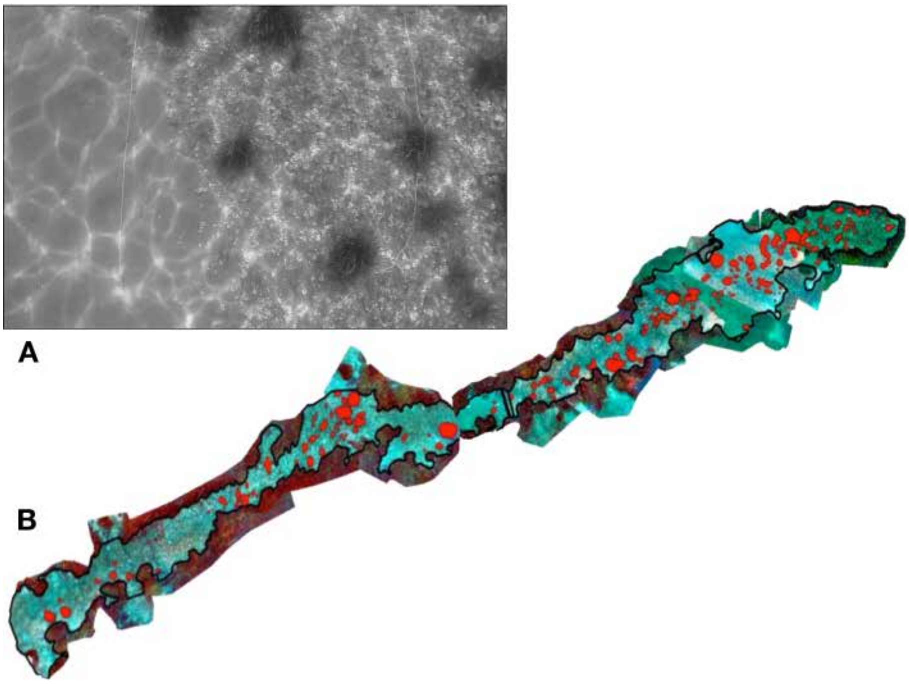

Fig. 3. – Colonization of drifted ramets on rubbles only (**A**) and map of the new *Posidonia* patches (red spots) along the monitored track (**B**).

Table I. –Values of the two replicates of *Posidonia oceanica* density at the three different sites.

	GAS		LA		CAS	
	a	b	a	b	a	b
No. shoots in 0.16 m^2	143	134	49	55	62	56

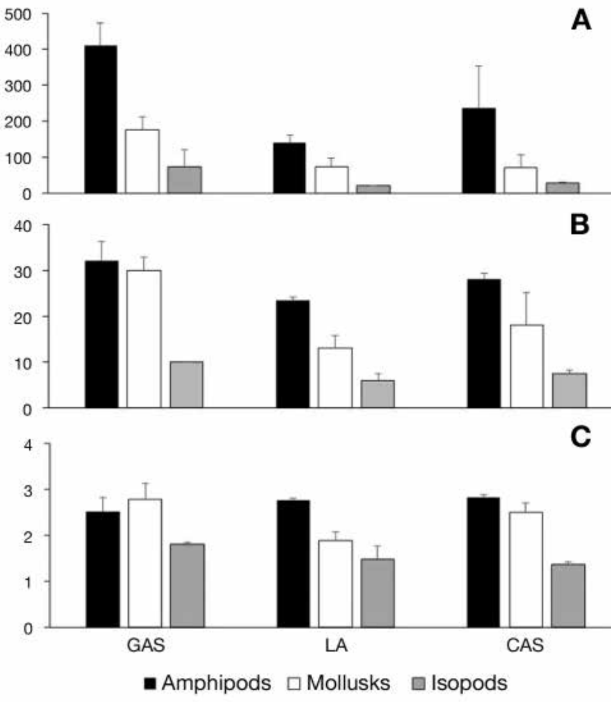

Fig. 4. – Abundance (**A**), number of species (**B**), and Shannon Diversity (**C**) of the three taxocenes at the three sites.

Epifaunal community

On the whole, GAS featured the highest abundance of all three taxocenes with respect to the other sites (Fig. 4A). The same was the case for the number of species (Fig. 4B) while the Shannon diversity did not show a clear difference (Fig. 4C).

In the ordination nMDS plots of the amphipod taxocene, samples from GAS were clearly separated (47 % similarity) from the other two sites, which in turn showed a significant separation at the 53 % level of similarity (Fig. 5a). Species which most characterize the assemblages are shown in Table II. Major contributors to the typification of GAS were a suite of species dominated numerically by *Apolochus neapolitanus* (Della Valle, 1893) and *Orchomene humilis* (Costa, 1853). *A. neapolitanus* was also the major contributor to similarity at CAS, followed by *Elasmopus pocillimanus* (Spence Bate, 1862). At LA, *Lembos websteri* (Spence Bate, 1857) ranked first in both abundance and contribution to similarity within the taxocene.

Also in the case of mollusks, GAS samples grouped separately in a significant manner (14 % similarity) from the other two sites (Fig. 5B). GAS was characterized by the high contribution of *Vitreolina philippi* (de Rayneval & Ponzi, 1854) whereas *Rissoa italiensis* Verduin, 1985 dominated at LA and a group of species including *Steromphala umbilicaris* (Linnaeus, 1758), *Rissoa auriscalpium* (Linnaeus, 1758) and *Alvania lineata* (Risso, 1826) were the main contributors to similarity at CAS.

No significant grouping was found for isopods (Fig. 5C). Juveniles of the species *Cymodoce hanseni* Dumay, 1972 accounted for the highest contribution

Table II. – Species which most contribute to the typification of the sites (based on SIMPER, cut-off 50 %). a: average abundance (mean of two replicates), b: percent contribution to total similarity (species nomenclature based on International Commission on Zoological Nomenclature).

	GAS		LA		CAS	
Species	a	b	a	b	a	b
Amphipods						
Apolochus neapolitanus	135.5	16.66	8.5	7.35	27	10.33
Orchomene humilis	56	10.47	0.5		11.5	6.08
Liljeborgia dellavallei	5		9.5	7.35	27.5	7.30
Ericthonius spp.	11.5	5.29	5		15.5	7.84
Aora spp.	13.5		11.5	8.22	6.5	
Megamphopus cornutus	11.5		11.5	7.80	8.5	
Apherusa cfr. *chiereghinii*	18	6.38	7.5		4.5	
Monocorophium sextonae	0		0		29.5	7.02
Lembos websteri	0		26	13.00	2.5	
Gammaropsis palmata	17.5	6.38	2		8	
Quadrimaera cfr. *inaequipes*	2		6		19	7.84
Iphimedia minuta	7.5		7	6.88	6.5	
Elasmopus pocillimanus	1		1.5		18.5	8.59
Peltocoxa marioni	12.5	5.53	2.5		3.5	
Mollusks						
Vitreolina philippi	43.5	13.94	0		0	
Rissoa italiensis	0		27	31.10	3	
Alvania lineata	0		11		12	11.75
Rissoa auriscalpium	10	6.36	0.5		8	12.87
Steromphala umbilicaris	0		4.5		9	13.90
Jujubinus striatus	0		7.5	14.66	5	
Tricolia pullus	8	7.53	1		2	
Alvania cimex	1		0		9	10.51
Parvicardium exiguum	0		9	15.84	0	
Nassarius incrassatus	8.5	6.36	0		0	
Chauvetia brunnea	8	6.36	0		0	
Calliostoma laugieri	4		0		3.5	9.10
Rissoa guerinii	6.5	6.36	0.5		0	
Limaria tuberculata	4.5	5.69	0.5		0.5	
Isopods						
Cymodoce hanseni	27.5	26.37	9.5	53.95	3	16.97
Joeropsis brevicornis	18.5	21.53	3		16.5	36.66
Gnathia juv indet.	5.5	15.23	0.5		1	

to similarity within the taxocene at GAS and LA, while the asellote *Joeropsis brevicornis* Koehler, 1885 ranked first in abundance and contribution to similarity at CAS (Table II).

DISCUSSION

The natural recovery of *Posidonia oceanica* after the deployment of the gas pipeline ten years ago represents a relevant result, due to the paucity of long term data on this topic, apart those recorded at Capo Feto (Sicily) (Badalamenti *et al.* 2006), and to the peculiar environmental conditions of the site, located at the entrance of the harbor of Ischia and along a channel with a very intense vessel traffic. Results confirm the success of the natural recruitment of this seagrass by propagules drifted from nearby stands (Balestri & Lardicci 2008) and often observed in these years along the trench. The role played by the stones to capture and trap the buoyant ramets testifies to the importance of the type of substratum in determining the success of new plant settlement (Di Carlo *et al.* 2005, Badalamenti *et al*. 2011). In particular, the higher rate of plant recovery observed at Capo Feto than at Ischia may be due

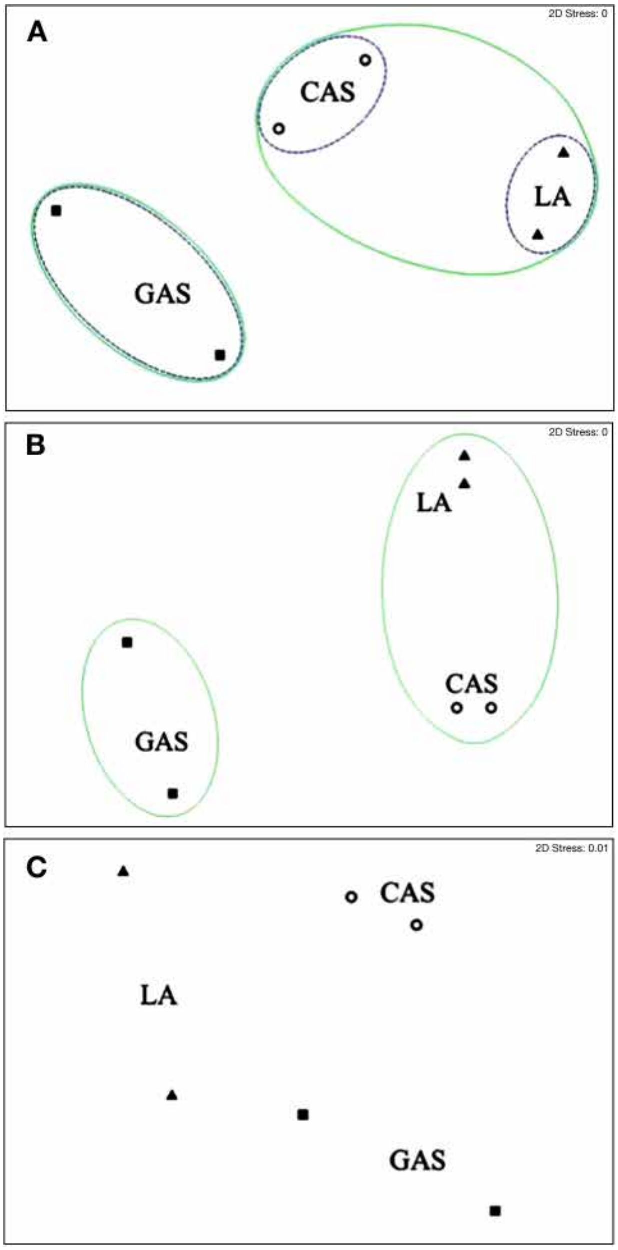

Fig. 5 –- Multidimensional scaling (nMDS) plots on Bray-Curtis similarity matrix of assemblage data (**A**: Amphipods; **B**: Mollusks; **C**: Isopods). Symbols represent plots sampled at the three sites (squares: GAS; triangles: LA; circles: CAS). Ovals include points, which show a significant similarity in cluster analysis.

to the deeper location of that meadow and to the different geological origins of the rubble. In Sicily, the trench was covered by calcareous stones while at Ischia volcanic rubble was used, the latter probably more economic as characteristic of the Phlegrean area. Volcanic rocks, more consistent, probably needed more time to be suitable for colonization by pioneer species before being able to facilitate the settlement of the seagrass.

The method used to map the *P. oceanica* recolonization has been previously applied in different benthic systems. In the Mediterranean Sea, photogrammetry has been used for studies on coralligenous ecosystem and few experiments have been done to map seaweeds and to follow structural and morphological characteristics of the *P. oceanica* meadow (Rende *et al.* 2015, Abadie *et al.* 2018). The results we obtained, although on a limited area, indicate that underwater photogrammetry can be a useful technique applied to conservation purposes and therefore can be coupled with classical monitoring procedures.

While aware of the exploratory nature of our study of epifauna, a number of observations can be made. First, attributes such as the number of individuals and of species from the new formation show higher values compared with those from the "old" beds while levels of diversity are within the same range. Higher shoot density at GAS, as a descriptor of habitat complexity (Attrill *et al*. 2000), may be one determinant of this relative richness, possibly coupled with the effect of patch size and the inputs from the nearby meadow edges (Tanner 2006, Bostrom *et al*. 2006). A major effort was allocated to the taxonomic resolution of the assemblage analysis. This is warranted by the importance of detecting those species, which may be associated with a particular condition of the beds and may point to the restoration of a healthy/typical population in new *P. oceanica* formations. The structure of amphipod and mollusk assemblages at GAS differs from those from the other two sites, which in turn show some degree of similarity with each other. Overall, dominant species at all three sites are known to be commonly associated with *P. oceanica* meadows although with specific peculiarities. Among amphipods, *A. neapolitanus* is a small-sized species which may be favored by the high complexity of the new patch while the greater weight of *L. websteri* at LA and of species such as *E. pocillimanus* and *Monocorophium sextonae* (Crawford, 1937) at CAS may be explained by a substantial presence of detritus and the relative proximity of rocky reefs, respectively. This is consistent with the notion that local factors may influence epifaunal composition (*e.g.*, Borg *et al.* 2010). On the whole, carnivores and scavengers (*A. neapolitanus*, *O. humilis*) seem to prevail in the new formation while tube-building detritivores and deposit-suspension feeders (*L. websteri*, *M. sextonae*, *Ericthonius* spp.) dominate the other sites possibly benefitting from the presence of an established *matte*, which is lacking at GAS. Species dominating the mollusk assemblages at the three sites also mostly belong to the typical stock of *P. oceanica* (*e.g.*, Russo *et al.* 1984, Albano & Sabelli 2012). However, the new patch is characterized by the small eulimid *V. philippi*, which is a parasite on sea urchins and ophiuroids (Oliverio *et al*. 1994). Its presence should be indicative of a remarkable echinoderm frequentation. Isopods do not show a clear distinction between assemblages, seemingly due to the consistent occurrence of *C. hanseni*, which constitutes a substantial component of the isopod taxocene in northern Ischia meadows (Gambi *et al.* 1992, Garrard 2013), and of *J. brevicornis* which features a broader habitat range and whose presence may be favored by the relative nearness of rocky reefs similar to the above mentioned amphipod species.

In conclusion, while differences occur seemingly related to the fragmentation status, the age and the location of the studied beds, we may say that the new formation is able to host a rich and diversified epifauna with compositional traits of a typical *P. oceanica* community. The next step, which is in progress, is the analysis of the nearby meadow, which may act as a source of epifaunal forms.

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OIL SPILL RESPONSE IN MANGROVES: WHY A SPECIFIC ECOSYSTEM-BASED MANAGEMENT IS REQUIRED? THE CASE OF FRENCH GUIANA – A MINI-REVIEW

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MANGROVES
ECOSYSTEM-BASED MANAGEMENT
OIL SPILL

ABSTRACT. – Mangrove forests are formed by mangrove trees and shrubs that grow in the intertidal zone at the sea-continent interface. They constitute major ecosystems of tropical to subtropical muddy coasts that perform several ecological functions, including: mitigation of coastal erosion and flooding hazards associated with storm waves, extreme tides and tsunami, providing nurseries for some estuarine and coastal species (*e.g.*, shrimps, fishes), production and recycling of organic matter, carbon storage, functioning as long-term sinks for several contaminants. World mangroves face a number of threats with increasing habitat destruction caused by direct and indirect anthropogenic pressures coupled with global climate change. They are known to be extremely vulnerable to oil spills. Even if the fate and impact of oil spills in such ecosystems have been partially monitored and experimentally studied (*e.g.*, 30-year TROPICS field experiment in Panama islands, replicated field trials conducted in central Queensland, Australia) significant gaps in knowledge remain. The oil dynamic in such ecosystem is complex and depends on the abiotic-biotic processes interactions. Understanding the fate and impact of the oil spill thus requires an integrated approach of the functioning of the whole mangroves system facing the pollution. The case of the French Guiana mangroves, subjected to intense hydro-morpho-sedimentary dynamics under the direct influence of the massive discharge of suspended sediments from the Amazon River, will serve as conceptual model to highlight the importance of the need for a specific Ecosystem-based Management response in case of oil spill.

MANGROVES ECOSYSTEMS

Mangroves constitute unique ecosystems that are some of the most productive ecosystems on the planet with biomass levels similar to those observed in tropical rainforests (Donato *et al.* 2011). They develop in the tropical and subtropical intertidal coastal regions of the world (*e.g.*, mudflats, estuaries, deltas, lagoons, etc.) occupying a global extent of 137,600 km^2 (Bunting *et al.* 2018) and almost 75 % of the worldwide coastline (Duke *et al.* 1998). Mangroves are dicotyledonous woody trees and scrubs (Viridaeplantae, Spermatophyta) that are highly adapted species to the variable flooding, high temperatures, high sedimentation, anoxic and salinity stress conditions prevailing in low latitudes muddy intertidal zones (Alongi 2002). They colonize these harsh environments thanks to several structural and ecophysiological adaptations including: aerial roots to breath in anoxic sediments, impermeable layer within the exodermis to mitigate radial oxygen loss during diffusion into the underground roots, structure of buttresses, xerophytic water conserving leaves, low water potentials and high intracellular salt concentrations, salt exclusion and salt secretion, vivipary and tidal dispersal of water-buoyant propagules (Duke *et al.* 1998, Shi *et al.* 2005, Alongi 2016, Srikanth *et al.* 2016).

There are roughly 70 mangrove species and 55 "true mangrove" main species (*sensu* Tomlinson 2016): *Nypa fruticans* Wurmb and *Laguncularia racemosa* C. F. Gaertn and all the species belonging to the genera *Avicennia*, *Lumnitzera*, *Bruguiera*, *Ceriops*, *Kandelia*, *Rhizophora* and *Sonneratia* (Tomlinson 2016, Quadros & Zimmer 2017). At the global scale, two biogeographic regions can be distinguished, the Atlantic-East Pacific (AEP; including eastern South America) and the Indo-West Pacific (IWP; including: eastern Africa and Madagascar, Indo-Malaysia and Asia, and Australasia) (Duke *et al.* 1998, 2002, Duke 2006, Van der Stocken *et al.* 2019). The mangroves of IWP region are pretty much more diversified than those of AEP region with a total of 54 and 17 species (counting all mangrove species), respectively (Tomlinson 2016). However, despite mangrove forests are formed by a relatively small number of rooted vascular species, they create a unique habitat for numerous terrestrial, estuarine and marine species (Robertson & Duke 1987, Primavera 1998, Kathiresan & Bingham 2001, Kerry *et al.* 2017). They also deliver several supporting, regulating, provisioning and cultural services (Table I). Mangroves ecosystem services worth US$ 33,000-57,000 per hectare per year (UNEP 2014) that is, considering they cover around 14 million hectares (Giri *et al.* 2011), up to US$ 800 billion per year.

FRENCH GUIANA MANGROVES

In French Guiana (FG), mangrove forests occupy about 80 % of its 350 km long coast with a total extent of about 70,000 ha (Proisy *et al.* 2003, Fromard *et al.* 2004). They are considered as one of the best-preserved mangroves in the world (Fromard & Proisy 2010, Olagoke 2016). As it is characteristic of the AEP region, FG mangroves exhibit low mangrove diversity with the dominance of three main species, *Avicennia germinans* (Linnaeus) Stearn (about 80 % of mangroves stands) and two species of *Rhizophora* (*R. racemosa* and, predominantly, *R. mangle*). *Avicennia germinans* can form monospecific and even-aged coastal forests sometimes in association with *L. racemosa*. Along riverbanks, at the limit of tidal influence (*i.e.*, polyhaline area), *Rhizophora* species grow in mixed mangrove swamp forest communities (Fromard *et al.* 1998, Fromard & Proisy 2010).

FG mangroves belong to the most dynamic coastline of the World, along the North of South America between the Amazon River mouth in Brazil to the Orinoco one in Venezuela. As a matter of fact, the FG coast is under the direct influence of the massive suspended-sediment discharge from the Amazon River (754,106 tons yr^{-1}) with a very active morpho-sedimentary dynamics characterized by the migration of mud banks along the coast from east to west toward the Orinoco River (Anthony *et al.* 2010). The North Brazil current feeds, with the North Equatorial Counter Current, the Guiana current flowing northwestward along the northeastern coast of South America (Condie 1991). Both these strong coastal currents are annually responsible of the transportation of around ~300 million m^3 of sediments from the Amazon, generating a heterogeneous remodeling of the coastline (Anthony *et al.* 2014). Marked deposition phases with the formation of mud banks alternate with erosion phases, deeply affecting the coastline. Together with the input of Amazonian and Guianese freshwater plumes, waves and tidal currents, the FG coast is therefore characterized by a highly variable salinity (Lambs *et al.* 2008), and a high and changing turbidity, making mangroves the only adapted natural community to this unstable environment (Fromard *et al.* 2004). The development and maturation of the coastal mangrove forests are closely related to the mud banks dynamics and are divided in five successive steps: bare mud, pioneer mangrove (propagules settlement), young, mature, and senescent forest (Fromard & Proisy 2010, Toorman *et al.* 2018) (Fig. 1). As soon as the mud is sufficiently consolidated, it is rapidly colonized by the *A. germinans* and *L. racemosa* propagules (Gratiot *et al.* 2007). As a selective strategy to the recurrent sediment instability, FG mangroves are characterized by neoteny phenomenon (early flowering and fruiting) and a timing of dispersal processes coinciding with favorable sedimentological conditions (Fromard *et al.* 2004, Fromard & Proisy 2010). *Avicennia germinans* seedlings will form in less than two years young mangrove forests. Thanks to this high growth rate (up to 2 m. yr^{-1}), FG mangroves constitute one of the most abundant aboveground biomass worldwide (Fromard *et al.* 2004). Sediments are OM-enriched as the vegetation grows, with degradation by suboxic processes in the young facies then by anaerobic bacterial metabolisms, sulfato-reducers in more mature mangroves, resulting in an accumulation of dissolved organic carbon (DOC) in the deep sedimentary layers (Marchand 2017, Marchand *et al.* 2003, 2004, 2005, 2006). Despite the strong instability of the Guianese coast, these mangroves shelter benthic infauna biodiversity with high bioturbation activities (Aschenbroich *et al.* 2016, 2017), which also depend on the local geomorphology patterns, and reciprocally (Brunier *et al.* 2020).

Table I. – Main mangrove ecosystem services (adapted from Mitra 2020).

Provisioning services	Regulating services	Cultural services	Supporting services
– Wood provisioning – Food provisioning – Water provisioning – Raw material – Medicinal Resources / Biochemicals – Ornamental resources – Genetic resources	– Climate regulation – Natural Hazards regulation – Purification and detoxification of water – Air and soil – Water / Water flow – Erosion and soil fertility – Pollination – Pest and disease regulation	– Opportunities for recreational and tourism – Aesthetic value – Inspiration for arts – Information foreducation and research – Spiritual and religious experience – Cultural identity and heritage – Mental well-being and health	– Ecosystem process maintenance – Life cycle maintenance – Biodiversity maintenance and protection (breeding ground and nursery habitat) – Support of coastal and marine fisheries

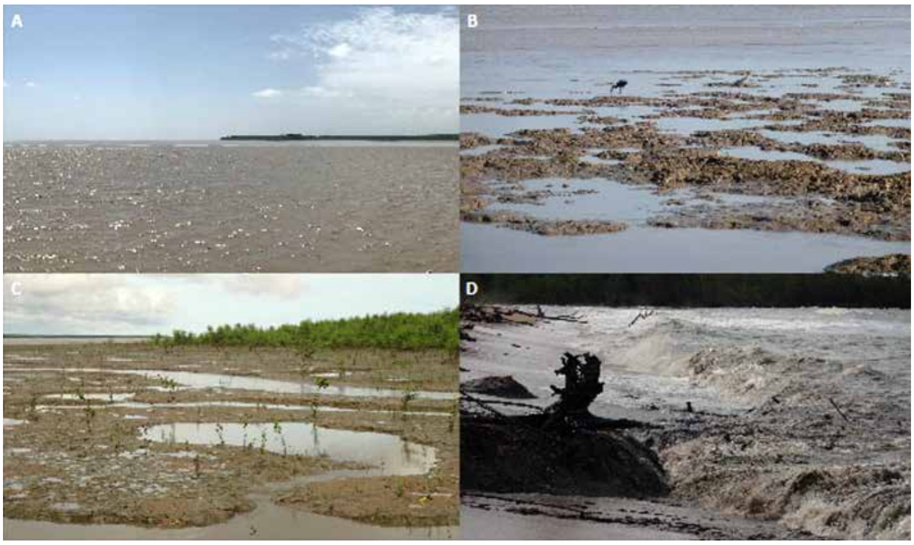

Fig. 1. – French Guiana mangroves dynamics driven by the alternate phases of accretion and erosion linked to the Amazonian dispersal system. **A**: From bottom to top: overall view of the transition between turbid waters, mud bank and mature *Avicennia germinans* mangrove forest (Kourou coast). **B**: Consolidated mud bank (Awala beach). **C**: Pioneer and young mangrove forest of *A. germinans* (Sinnamary estuary). **D**: Erosion and destruction phase of mangrove forest (Mana paddy field area). Photo credits: P. Cuny.

MANGROVE UNDER THREATS

Mangrove forests are one of the world's most threatened tropical ecosystems (Duke *et al.* 2007). In the half-past century, mangrove forests surface has declined by 30-50 % (FAO 2007, Polidoro *et al.* 2010). Thanks to environmental awareness and management strategies this rate has however markedly decrease. The calculated average deforestation rate from 2000 through to 2012 was between 0.16 % and 0.39 % per year, still reaching nonetheless values up to 8.08 % in some coastal areas of Southeast Asia (Hamilton & Casey 2016). This annual average rate is often higher than for tropical continental forests, which is about 0.5 % since the 1990's (Achard *et al.* 2014). Mangroves destruction is mainly due to littoral development and to global demand for commodities (*e.g.*, expansion of aquaculture and rice culture, conversion of mangroves to oil palm plantations) (Richards & Friess 2016). Direct destruction is due to clearing or overharvesting but indirect anthropogenic pressures like modified river discharge and/or pollution can also lead to mangroves habitat loss (Carugati *et al.* 2018).

OIL SPILLS: AN IMPORTANT THREAT FOR MANGROVE ECOSYSTEMS

Mangroves are highly vulnerable to oil spills. They deeply affect ecosystem services of mangroves, like fisheries production and shoreline protection. Between 1958 and 2015 a review of current literature and public databases have shown than more than 1.94 million ha of mangroves habitat have been oiled and more than 126,000 ha destroyed (Duke 2016). Oil deposits on sensitive plant surfaces, affecting sediments and benthic communities causing death and sublethal impacts (Duke *et al.* 1999, Kathiresan & Bingham 2001). Oil spill response and clean-up are particularly difficult in mangrove and may significantly damage roots and seedlings, but also bury oil deeper into sediments where oil biodegradation processes are slower (Machado *et al.* 2019). In some cases, following initial cleanup of the major part of the pollution, the best response to treat the residual pollution is the "walk-away" strategy that is to say doing nothing and allowing natural attenuation processes like biodegradation to act (Duke 2016). Depending on the amount and type of oil spilled but also the surface of mangrove impacted as well as the existence of other stressors (*e.g.*, herbivory, storms, diseases, pollutants) recovery, when it happens, can last more than 30 years (Duke 2016). The last severe oil spill impacting mangrove forests occurred in Brazil's north-eastern tropical coast in late August 2019 affecting more than 3,000 km of coastal ecosystems (Soares *et al.* 2020). At that time there were concerns that the Brazilian oil spill could reach FG coasts. Indeed, Brazilian oil offshore exploitations represent the main risk of oil spill for FG mangroves.

BRAZILIAN OIL: A MAJOR RISK FOR THE FG MANGROVES

Due to the intense North Brazil current that flows northwestwardly, oil spills originating in the Brazilian Equatorial Margin would reach, depending of the time of the year, the French Guianese coast. Indeed, modeled dynamics of an oil spill occurring in the coastal region of Amapá (Foz do Amazonas basin) in Brazil coastal waters according to time of year showed that an oil spill happening at the end or at the beginning of the year would deeply impact FG coast (Chevalier *et al.* 2020). Though, effects of spilled oil on FG coastal ecosystems remain largely unknown but could have dramatic effects on this ecosystem, its functioning and its ecosystemic services.

So far, there is a weak oil spill response readiness in FG. A clean-up guide does exist (Colombier 2015) but it was elaborated from data from the literature based on studies carried out in other areas of the world. Most of the literature on the effects of oil on mangroves report studies carried out on single species seedlings or propagules. The results obtained show in particular the levels of toxicity of oil hydrocarbons with contrasted results depending on the mangrove species (*e.g.*, Proffitt *et al.* 1995, Zhang *et al.* 2007, Ke *et al.* 2011, Naidoo 2016, Guedes *et al.* 2018). There have been two major field experiments at larger scales that took into account not only the effects of oil on trees but also on part of benthic organisms: (i) replicated field trials conducted in Port Curtis, central Queensland, Australia, which lasted 22 months – experimental plots of about 35 m^2 dominated by *R. stylosa* were contaminated with 200 L of oil and dispersed oil (Duke *et al.* 1999) and, (ii) the TROPICS field experiment in Panama islands which lasted 32 years – two sites of 30 m^2 of *A. germinans*, *L. racemosa*, *R. mangle* forest were contaminated with crude oil and crude oil pre-mixed with dispersant (approximately 1000 L per site over 24 and 48 hours; Renegar *et al.* 2017). The latter experiment is the one that seemingly could best apply to FG mangroves, as mangrove species are the same, however, the ecosystemic conditions prevailing in the study area in Panama are very different as mangroves are connected to seagrass and coral reef ecosystems.

In FG, only a preliminary *in situ* study was conducted so far (Jézéquel *et al.* 2016). The objectives of the study were to assess the effects of oil on benthic communities and to evaluate the oil weathering processes in the sediments of a young *A. germinans* mangrove, one month after the contamination (≈ 20,000 ppm of Brazilian light crude oil topped at 250 °C). The results highlighted the high bioremediation potential of the autochthonous microbial community, which exhibited a high biodegradation activity on aliphatic and aromatic hydrocarbons (biodegradation rates higher than 90 %). However, it is worth noting that about 80 % of total petroleum hydrocarbons still remained in the sediment. Furthermore, added oil also induced a 90 % decrease in the mean densities of meso- (> 250 mm) and macro-benthic organisms (> 1 mm) within the contaminated surficial layer of the sediment (0-4 cm). The oil had also marked effects on the composition of the benthic communities (micro-, meso- and macro-benthos) showing, on the other hand, the important benthic compartment vulnerability to oil spills.

Whatever, all the existing data give only some leads about the fate and effects that would have oil spill in FG mangroves. For instance, it is absolutely not known how would interact the oil spill with the turbiditic waters and mud banks of the coastal area, while such mobile banks are huge reactive natural incubators preluding high mineralization processes (Aller & Blair 2006). Such uncertainty thus excludes any possibility to predict the fate of oil slicks in this environment. Similarly, it is not known how oil contamination of the mobile muds would affect the early stages of mangrove development (propagules settlement and pioneer mangrove), and its associated benthic system, which is a crucial step toward the growth of the older mangrove stages. These early stages play a major role in the overall dynamics of mangroves in Amazon-influenced coast of South America; they are also the most oil-sensitive (Duke 2016). In fact, many, if not most, of the various factors and interconnected biotic and abiotic processes that would govern the fate and effects of hydrocarbons in such ecosystems remain to be studied in a holistic and integrated way (Cuny *et al.* 2011). For instance, sediment hydrocarbon-degrading bacteria activity may be controlled by several factors such as plant roots-bacteria interactions (Gomes *et al.* 2010, Gkorezis *et al.* 2016, Sampaio *et al.* 2019), bioturbation (Cuny *et al.* 2011), or meiofauna grazing (Näslund *et al.* 2010, Louati *et al.* 2013, Pusceddu *et al.* 2014) in turn controlled by macrofauna (Braeckman *et al.* 2011, Urban-Malinga *et al.* 2014). It is very likely that the activity of these benthic bacteria is also controlled by viruses (Head *et al.* 2006, Cuny *et al.* 2011). Viruses appear to be particularly diversified and uncharacterized in mangrove ecosystems (Jin *et al.* 2019). They have been shown to be active even in deep sub-seafloor marine sediments, controlling microbial community (Cai *et al.* 2019). It is also probable that hydrocarbon-degrading bacteria would interact with fungi. Indeed, fungi-bacteria consortium was shown to be efficient for the bioremediation of mangrove oil-contaminated sediments (Li & Li 2011).

From a broader standpoint, it is only by taking into account the realities of ecosystem functioning (*e.g.*, the ecological networks) that any oil spill response could be efficient in particular in such dynamic coastal system like FG. That means that not only we have to study the several abiotic and biotic processes controlling the dynamics of oil in marine ecosystems, but more importantly, we have to understand the overall functioning of these ecosystems resulting from the multi-scale interactions of the different processes operating within, not only the ecosystem, but indeed the socio-ecosystem. As a matter of fact, besides aspects related to the own natural dynamics of spilled oil in marine systems and the knowledge we have about it, the vulnerability of mangrove ecosystems to oil spills finally relies on societal and institutional readiness and management strategies.

CONCLUSION: WHY A SPECIFIC ECOSYSTEM-BASED MANAGEMENT (EBM) IS REQUIRED?

An EBM, recognizing the full array of interactions within an ecosystem, including humans, is required because mangroves are not "just a set of trees" but complex systems influenced by numerous components defin-

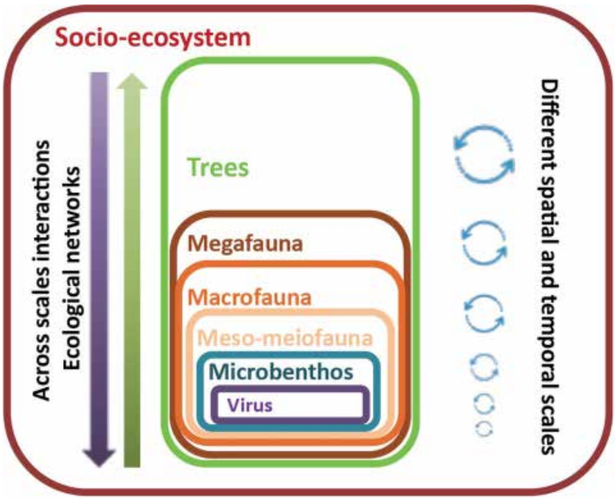

Fig. 2. – Schematic representation of the different levels of organization of mangrove ecosystem interacting in different ways at several spatial and temporal scales. Microbenthos include bacteria, archaea, fungi and other micro-eukaryotes. The lower the level, the faster are the associated processes and vice versa.

ing their functions and services (Lee *et al.* 2014). For instance, without the benthic compartment mangrove trees could hardly develop. The mineralization of organic matter and, particularly, of dead mangrove vegetation is indeed closely related to the activity of a diverse benthic microbial community that deliver sources of nitrogen, phosphorus, and other nutrients to mangrove seedlings. In turn, plant-root exudates and other plant material like leaves enable the growth of several microorganisms and larger organisms like crabs, respectively (Holguin *et al.* 2001). The fungi play a pivotal role in coordinating the entire microbial community by controlling the structure of functional networks and the microbial-based nutrient cycling on overall sediment (Booth *et al.* 2019). In addition, the diverse mangrove viruses probably directly manipulate carbon cycling through the release of important amounts of organic carbon and nutrients from hosts but also, by assisting microorganisms in driving biogeochemical cycles by transferring to them auxiliary metabolic genes like those involved in biolysis of complex polysaccharides (Jin *et al.* 2019).

Mangrove forests degradation can result in a benthic biodiversity loss of about 20% and a loss of 80 % of microbial-mediated decomposition rates, of the benthic biomass and of the trophic resources (Carugati *et al.* 2018). As pointed out by Borges *et al.* (2017), in order to reconcile mangrove conservation with resource use, mangrove should be treated as an integrated system and not divided in subsystems. This statement also applies to pollution management and oil spill readiness. Ecosystems are complex systems; complexity is not just a buzz word but a "new way" to think and managed natural systems considered as a hierarchy of interrelated organizational levels exhibiting emergent, self-organized, and adaptive behaviors (Levin 2005, Eppel & Rhodes 2018). The need of holistic approaches in ecology has been acknowledge since several decades (Lefkaditou 2012). The need to think and manage the ecosystems, admitting that they are complex systems, was further pointed out since the 1990's (Reason & Goodwin 1999). It is obvious that holistic approaches are very challenging as they require specific methodological developments, multi-scale studies and, ideally, the simultaneous analysis of the different levels of organization of an ecosystem (Witman *et al.* 2015) (Fig. 2). Nonetheless, it is the only way to improve environmental management strategies and particularly oil spill readiness. Otherwise, partial knowledge, based on results of the literature obtained in different mangrove ecosystems from other biogeographical areas or, studies on specific communities or specific organizational levels (*e.g.*, propagules, seedlings, microbenthos) at limited spatial and temporal scales can only bring a limited capacity to model and predict the fate and effects that would have an oil spill in FG mangroves.

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AN ECOSYSTEMIC APPROACH FOR AN ECOLOGICAL CRISIS IN BERRE LAGOON

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LAGOON ECOLOGY
ECOLOGICAL CRISIS
ZOSTERA MEADOW
ECOSYSTEM RESILIENCE

ABSTRACT. – Berre lagoon is a Mediterranean lagoon deeply impacted by industry and urban activities. Since 1966, a hydroelectric powerplant has discharged large quantities of freshwater and nutrients into the lagoon, inducing major ecosystemic changes. The lagoon ecosystem has declined to a eutrophic state with the loss of *Zostera* meadows and marine macrofauna. In 1994, an extensive monitoring network for Berre lagoon was set up. Different compartments of the ecosystem were measured: water quality, sediment quality, macrophytes (including Magnoliophyta), benthic macrofauna, fisheries, and ichthyofauna. Results show a pattern of change in the ecosystem linked to the different phases of eutrophication reduction. However, in 2018, a major ecological crisis occurred, inducing anoxia over more than 90 % of the lagoon surface area. Analysis of data from the monitoring network during and after this crisis, taking into account environmental and climatic factors, provides a basis for understanding the degradation of the different compartments of the ecosystem. The origin of this crisis is a 'cocktail effect' of high spring nutrient inputs, high water temperature, strong water stratification, lack of wind, lack of Zostera meadows and high benthic biomass. This crisis highlights the extreme fragility of the Berre lagoon ecosystem and shows the importance of an ecosystemic approach for the monitoring network.

INTRODUCTION

Coastal lagoons and estuaries have, since the early or middle 20th century, become among the coastal ecosystems the most impacted by disturbances worldwide (Valiela *et al*. 1997, Cardoso *et al*. 2004). Berre lagoon is one of the largest Mediterranean deep lagoons (155 km², maximum depth 9.5 m; mean depth: 6 m). It communicates with the Mediterranean Sea through the Caronte Channel and receives freshwater from several natural rivers (Deslous-Paoli 1996). At the beginning of the 20th century, Berre lagoon was a biodiversity hot spot, with an abundance of marine species and large *Zostera* meadows from the surface to 6 m depth (Rioual 1972, Gourret 1907, Roux *et al*. 1985, 1993). During the industrial revolution, Berre lagoon was impacted by severe chemical pollution resulting from industry, agriculture and urbanization (Arfi 1989, GIPREB 2012). Despite this industrial impact, the ecology of the lagoon remained in good condition with high biodiversity and extensive *Zostera* meadows (Bernard 2007).

Since 1966, the installation of a hydroelectric power plant induced high freshwater and nutrients inputs into the lagoon (Roux *et al.* 1985). Initially, this disturbance caused major changes in the Berre ecosystem: the heavy inputs of freshwater (up to seven times the volume of the lagoon per year) induced a decline of the surface water salinity from 24-36 to 1-22) and a water column stratification with low salinity water down to 5 m and more salty water at depth (under calm conditions) (Kim 1985). The associated nutrient inputs caused the decline of the ecosystem to a eutrophic state with high chlorophyll *a* concentration, anoxic episodes at depth, benthic macrofauna biodiversity loss (Stora and Arnoux 1983, Stora 1995, Zaghmouri *et al*. 2013) and a dramatic loss of the *Zostera* meadows from 6,000 ha in 1960 to 1.5 ha in 1998 (Bernard *et al.* 2007) France. A first limitation of freshwater inputs was initiated in 1994 (2.5 Gm^3 per year), and a second in 2005 after European litigation (Truilhé-Marengo 2013). Since this litigation, the freshwater inputs have been limited to 1.2 Gm^3 per year, and in addition the salinity must be controlled to avoid high variations (75 % of the time above 20, 95 % of the time above 15). In parallel, since the 1990s, a major effort has been deployed throughout the watershed to reduce nutrient inputs resulting from urban and industrial activities (Gouze *et al.* 2008a, b).

Overall, these input reductions have led to a major change in the ecosystem as a whole. The Berre lagoon-monitoring network, which has been in existence since 1994, has shown improvement in most of the ecosytem components. After a phase of instability, the lagoon eutrophication level has declined: the chlorophyll *a* has decreased, the macrophyte community has become more diversified, the *Zostera marina* Linnaeus meadows have become more extensive (17.93 ha in 2017), the shore benthic macrofauna biodiversity has increased (GIPREB 2019). In particular, a large manila clam *Ruditapes philip-*

pinarum (A. Adams & Reeve, 1850) population is present along the shore (up to 4-5 m), with very high density at some points (Mahé *et al.* 2020).

But the restoration trajectory of a lagoon is complex (Bettinetti *et al.* 1996, Derolez *et al*. 2019, Leruste *et al.* 2019a). Anoxic crisis episodes could occur and impact the ecosystem in various ways and could be recurrent as in some other French Mediterranean lagoons such as Thau lagoon (Souchu *et al.* 1998, Harzallah & Chapelle 2002) located in southern France, suffers episodically in summer from anoxic crises known as 'malaïgues'. Such crises mostly occur under warm conditions and low winds. In this paper we investigated effects of local weather conditions (air temperature, wind speed and precipitation over southern France. The origin of these crises could be diverse such as climatic conditions, pollution, high nutrient inputs, and may be difficult to explain (Harzallah & Chapelle 2002).

During the summer-autumn 2018, a major ecological crisis occurred in Berre lagoon and affected the whole ecosystem. This crisis impacted all the ecosystem compartments at different levels. On the basis of the monitoring network results and the environmental data, might it be possible to understand the mechanisms of this crisis?

MATERIALS AND METHODS

Water quality: physical and chemical parameters: Ten stations were sampled monthly in Berre lagoon since 1994. At each point, a TSO (temperature, salinity, % of O_2 saturation) profile is established using a multi-parameter probe (hydrolab DS5). Depth and surface water were sampled using a Niskin bottle and then analyzed to measure nutrient concentrations (NO_3, NO_2, NH_4, total nitrogen, PO_4, total phosphorus) and chlorophyll *a* (Gouze *et al.* 2008b).

Biological parameters: Phytoplankton analysis is based on monthly samples at two stations. Macro- and nano-phytoplankton are identified and counted under the microscope.

The benthic macrophytes are monitored across 31 stations close to the shore every year. At each station, a survey is carried out by SCUBA diving transects perpendicular to the shore over a distance of 100 m. The abundance of each macrophyte group is noted using a semi-quantitative method from 0 (species absent to 500 for 100 % cover at the station; Astruch & Schohn 2019). For each macrophyte or group of macrophytes, an abundance index is calculated based upon the presence frequencies and the abundance along the transect (Astruch & Schohn 2019). The macrophyte survey was performed in June 2018, before the crisis, and was compared to the survey performed in June 2019 after the crisis.

A more specific survey was carried out on the *Zostera noltei* Hornemann meadows. In order to calculate their covered surface, aerial or satellite photography was used in 1998, 2009, 2014, 2017 and 2019. The photographs used were taken in June at the period of maximum growth of the *Zostera* meadows with a 30-cm resolution. A photo-interpretation was performed and the meadow was mapped using a GIS (QGIS v3.4). Verification by diving was undertaken to validate the interpretation and to estimate visually the meadow vitality (intermattes, coverage, epiphytes).

Shore benthic macrofauna has been monitored across 10 stations twice a year (summer and winter) since 2005. The stations are at around 4-5 meter depth. At each station, three replicates of sediments were sampled with an orange-peel bucket and were sieved at 1 mm. The surface area sampled at each replicate is 208 cm². At the laboratory, the living organisms were identified and counted. The species richness and abundance per species (number of individuals per m²) were calculated.

To estimate the surface impacted by the crisis, different transects were undertaken around the lagoon to determine the minimum depth where living macro-organism were observed. *In situ* observations were performed in September, after the first period of anoxia.

Meteorological data: Meteorological data (pluviometry, wind, air temperature) are based on data collected by Infoclimat (infoclimat.fr) every 3 hours at the Marseille-Marignane weather-station located to the south of Berre lagoon.

RESULTS

Water quality: physical and chemical parameters

The water temperature was particularly high during summer 2018. Temperatures over 30 °C were recorded in August. The average temperature in August was 28.1 °C for all the stations and all depths. A high water column stratification was observed with very wide differences in salinity between the surface and the bottom layer. The difference of salinity between these layers was more than 10 PSU during the whole summer (June to August). In June, the surface salinity was around 20 and increased

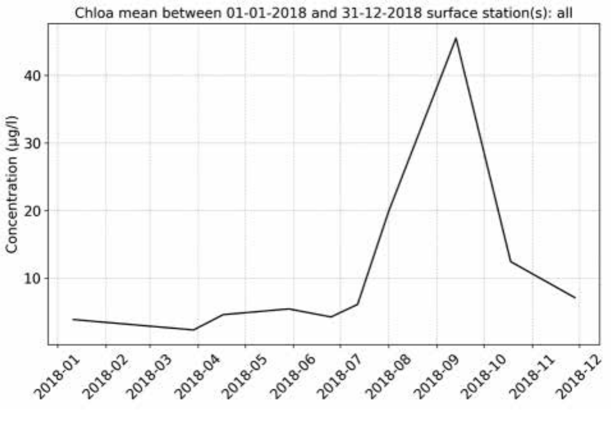

Fig. 1. – Mean concentration of Chlorophyll *a* (µg/l) at the surface in 10 stations in Berre lagoon in 2018.

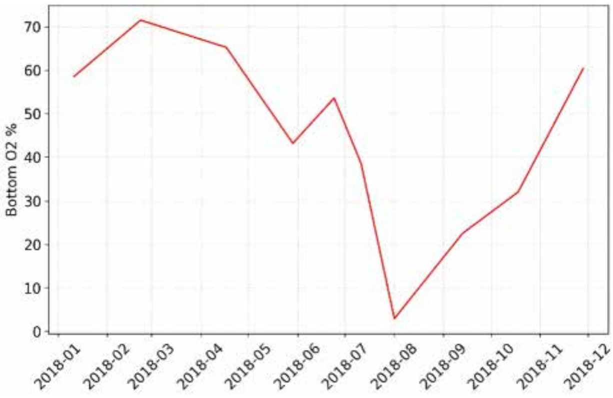

Fig. 2. – Mean percentage of bottom (last meter) dissolved oxygen (% of O_2 saturation) in 10 stations in Berre lagoon in 2018.

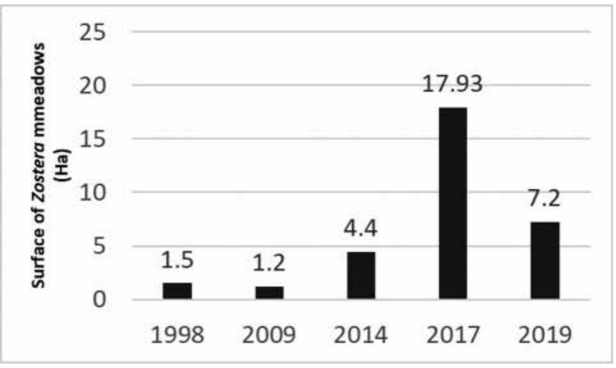

Fig. 3. – *Zostera* meadows surface (ha) in Berre lagoon between 1998 and 2019.

slowly to 25 in August. The bottom salinity (below 8 m) was quite constant, around 35, close to the marine salinity.

The results of the monitoring during the year show a high state of water eutrophication during summer 2018 with a very high concentration of chlorophyll *a* (Fig. 1). The spatial average surface concentration in September reached 45.54 µg·h^{-1} (standard deviation = 24.81 µg·h^{-1}). Such a high value has not been recorded by the GIPREB monitoring network since 1998. Similar observations were made for suspended matter, total nitrogen and PO_4 concentrations with high concentrations during the late summer (September-October). Continuous recording of dissolved oxygen (S. Rigaud, unpubl data) show up to 25 consecutive days of anoxia (0% of O_2 saturation) at 9 m depth, 10 days at 5 m depth and 5 days at 3.5 m. From September to December, hypoxic and anoxic conditions were recorded in the bottom layer (last meter; Fig. 2).

Biological parameters

Phytoplankton community analysis showed in August an efflorescence of nanoflagelates (< 10 µm). In September, a bloom of dinoflagellate *Gymnodinium impudicum* (S. Fraga & I. Bravo) Gert Hansen & Moestrup (more than 2.5 million cells per liter) was observed. According to G. Gregori (MIO, pers comm), during the same period, high concentrations of picoplankton were observed in the samples analyzed by cytometry.

The macrophyte populations after the crisis (in June 2019) compared to before (in June 2018) showed lower abundance of *Enteromorpha* species and regression of some marine species such as *Codium fragile* (Suringar) Hariot or *Bryopsis hypnoides* J. V. Lamouroux. *Ulva* sp. stayed abundant and showed no variation. *Cladophora* sp. showed a decrease after this episode. But the most important variation occurred in *Zostera noltei* with an abundance index reduced by 5. In terms of surface area, the estimated surface area of the *Zostera* meadows declined from 17.93 ha in June 2017 to 7.2 ha in June 2019 (Fig. 3).

The shore benthic macrofauna showed a strong decline in September 2018. The mean species richness dropped from 11.4 to 0.9 (Fig. 4). At the 10 stations monitored, only 2 still had living organisms. These 2 stations were located near the seawater entrance (Caronte Channel). The abundance of benthic organisms also decreased from 4,100 ind·m^{-2} to 162 ind·m^{-2}.

The *in situ* observations carried out by diving at different depths and locations around the shore enabled us to estimate that anoxic condition impacted more than 90 % of the lagoon's surface area (Fig. 5). Depending on the zone, anoxia impacted shallow areas down to 1 m depth. For a few zones such as south of Vaïne lagoon or Saint-Chamas Bay (at the north), anoxia impacted the whole

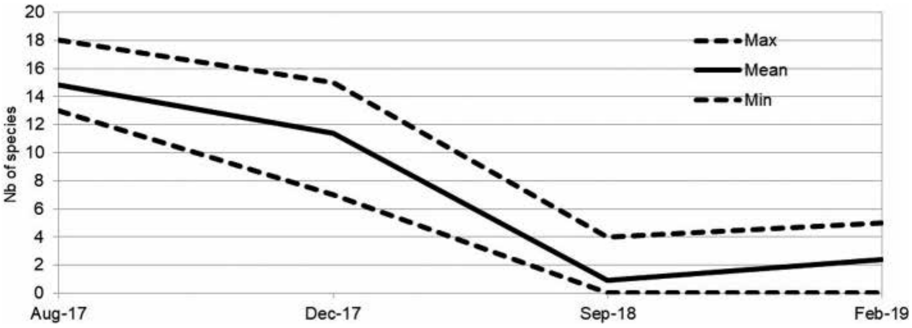

Fig. 4. – Maximum, mean and minimum shore (4-5 m depth) benthic macrofauna species richness (number of species) in 10 stations in Berre Lagoon.

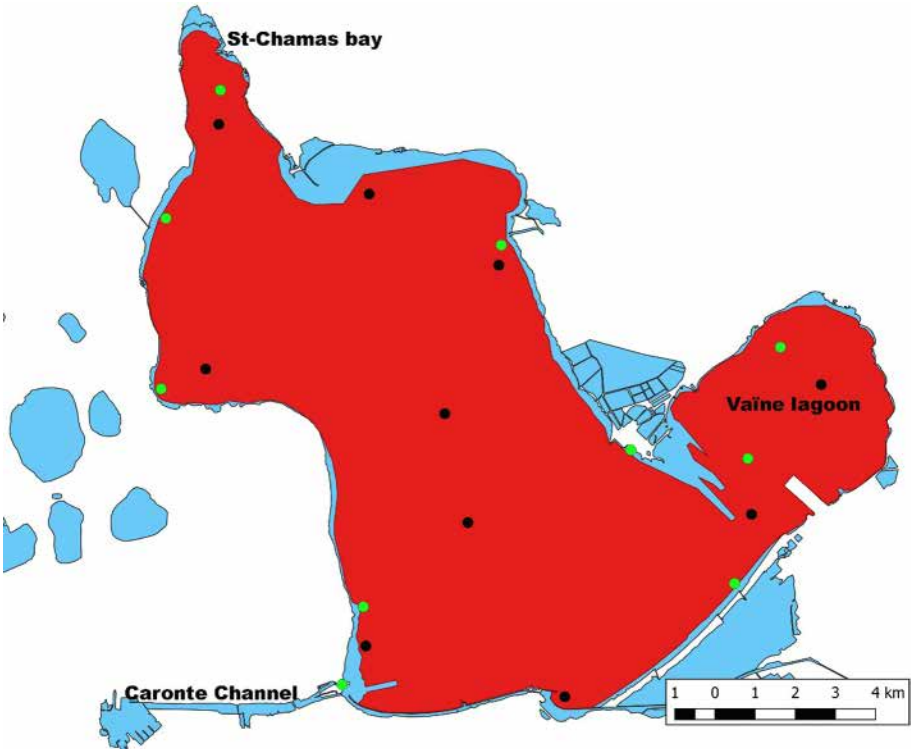

Fig. 5. – Estimated surface impacted by anoxia during the 2018 crisis in Berre lagoon (in red). The surface impacted is estimated at 14,400 ha representing 93 % of the total surface area.

of the water column (phenomenon called *malaïgue* in Provence).

Meteorological and environmental data

During summer 2018, the air temperature was quite high with an average of 25.87 °C (based on mean daily temperature, June to September). The temperature anomaly was +2.2 °C (compared to the 1981-2010 period). According to Meteofrance, summer 2018 was the second warmest summer in France since the beginning of the 20th century. The sunshine duration was 1,061.3 hours during the three summer months (July, August and September). During summer 2018, the wind velocity was quite low and an episode of 19 days without wind (above 23.4 $km \cdot h^{-1}$) was recorded. Twenty-two high wind episodes (*i.e.*, > 23.4 $km \cdot h^{-1}$) occurred during the summer period from June to September. The pluviometry during the first six months of 2018 was higher than normal (331 mm, + 35 % of the 1981-2010 climatology). Some rain episode also occurred during the summer (July: +121 % and August +103 %). High pluviometry was also recorded in autumn (October-November and December) (446.6 mm, +265 %).

The freshwater inputs during the six first months of 2018 represented 864 billion m^3, with 85.5 % coming from the hydroelectrical powerplant (the rest was accounted for by the three main rivers, direct watershed and pluviometry). These freshwater inputs also represent input into the lagoon of 36 tons of total phosphorus and 952 tons of total nitrogen (Gouze *et al.* 2014).

DISCUSSION

The Berre lagoon-monitoring network made it possible to record the impact of the 2018 crisis on several ecological compartments. The results show a high level of eutrophication of the water with high concentrations of nutrients and chlorophyll *a*. Anoxic conditions were observed in most of the lagoon (more than 90 %) and at very low depths (1.5 m). In some cases, anoxia impacted the whole of the water column and white water was observed, caused by the presence of green sulfur bacteria (*Chlorobiaceae*, Souchu *et al.* 1998).

As a consequence of these anoxic conditions, massive benthic mortalities were observed through macrofauna monitoring. In September 2018, only 2 of the 10 shore stations (4-5 m depth) presented living organisms. A stock survey of the manila clam (*Ruditapes philippinarum*) estimated a loss of more than 75 % of the population abundance during this crisis (Mahé *et al.* 2020). During the spring of 2019, a survey of the macrophytes showed a community change: less of *Enteromorpha* and a strong decline of abundance of Magnoliophyta such as *Zostera noltei*. The effects of this crisis were more severe with regard to the surface area of the *Zostera noltei* meadow. The 2019 survey showed a surface area loss estimated at 60 % of the 2017 surface. This Zostera meadow degradation could have been caused by the low transparency of the lagoon water (high concentrations of chlorophyll *a* and suspended matter) that reduced photosynthesis (Santos *et al*. 2010). In addition, the *Zostera noltei* meadows and in particular the rhizomes could have been stressed by the anoxic conditions and burned by the released sulfurdihydrogen (H_2S) (Pulido & Borum 2010). The meadows located where white waters occurred completely disappeared, confirming this link between H_2S and the decline of the meadows.

On the basis of the environmental parameters, how could the occurrence of such a crisis be explained?

At the beginning of the summer, the surface salinity was relatively low (22) due to the high freshwater inputs during the winter and spring. The water column was stratified, and the delta of salinity between the surface and the bottom was high (> 10 points of salinity for the deeper stations). The freshwater input was due to the direct and indirect (through the natural rivers) watershed and from the hydroelectric powerplant. As a consequence, the lagoon surface salinity stayed relatively low (20-22) in June 2018 compared to previous years. These freshwater inputs also represent an input of nutrients (nitrogen and phosphorus) into the lagoon ecosystem. These nutrient inputs increased the growth of phytoplankton during the summer. However, such high inputs have already been recorded without causing an ecological crisis later. Similar observations could be made for the air and water temperature. If the temperature recorded during that summer was particularly high, similar temperatures have already been recorded without causing any crisis. For example, in 2019, high air temperatures were also recorded.

The climatic parameter that could have been quite exceptional during summer 2018 is the low wind activity resulting in reduced water column mixing. The freshwater input installed a pattern of stratification. The lack of wind, coupled with high temperatures, high phytoplanktonic production and benthic consumption generated oxygen depletion at depth. In the bottom water layer, dissolved oxygen is consumed by benthic organism respiration and organic matter degradation. Part of the organic matter results from the sedimentation of dead phytoplankton cells. Such stratification linked with in depth anoxia is unfortunately recurrent phenomenon in Berre lagoon (GIPREB 2017). But in 2018, during this crisis, the absence of strong wind prevented the mixing of water, and the anoxic layer increased and impacted an increasingly extensive surface area. However, winds of 6.5 $m.s^{-1}$ (*i.e.*, 23.4 $km{\cdot}h^{-1}$) are enough to mix the water column (Nerini *et al.* 2001) and such wind speeds were recorded during the summer (22 times). Moreover, the high primary production at the surface represented an important source of organic matter at the bottom and increased the biological oxygen demand. With high water temperatures, the dissolved oxygen concentration was lower. This oxygen consumption was higher at the shore, where there was a very high biomass of manila clams. As the anoxic layer increased, benthic mortality occurred and these dead organisms became a new source of organic matter, which also needed oxygen for its degradation. During the anoxic conditions, the sediment constituted a source of phosphorus (PO_4) for the water column and thus a new source of nutrient to sustain the eutrophic conditions (Rigaud *et al*. 2013, 2017). The system was thus locked in a self-reinforcing feedback loop.

Another factor, which may explain the severity of this crisis, compared to other lagoons with higher ecological status, is the absence, or at least the low abundance, of seagrass meadows, natural oxygen producers for the ecosystem. In fact, even if the *Zostera noltei* meadow cover was the most extensive observed over the last decades, it only represented 0.9 % of the 0-3 m depth surface area. This low abundance is insufficient to produce enough oxygen and to mitigate the anoxic crisis.

Thus, taken separately no single explanation parameter could explain the crisis. The 2018 crisis cause was a cocktail-effect of different environmental parameters that acted in synergy, with these dramatic results: high spring inputs, strong water stratification, high water temperature, lack of strong wind, absence of *Zostera* meadows, and relatively high shore faunal benthic biomass.

After the beginning of the crisis in late July, new freshwater inputs from the hydroelectric powerplant (in mid-August and September) represented a new source of nutrients and have led to the extension of the duration of the crisis. The bloom of dinoflagellates observed in autumn could be linked to these enrichments (Leruste *et al.* 2019b). Due to these freshwater inputs, even after a wind episode that mixed the water column, stratification was quickly restored. The high organic matter stock, due to the dead organisms, represents a sink of dissolved oxygen. For this reason, high concentrations of chlorophyll *a* in October and hypoxic or anoxic conditions up to November were still observed.

The occurrence of this ecological crisis in Berre lagoon illustrates the fact that the restoration policy (freshwater inputs reduction) is perhaps insufficient to avoid such a major crisis. This crisis illustrates that the Berre lagoon ecosystem remains unstable. Adverse climatic conditions, such as during summer 2018, can make the ecosystem vulnerable to a major ecological crisis that will affect all its compartments. Such a severe crisis is a step backwards after the improvements observed over recent years and the restoration of some major compartments such as the *Zostera* meadows is uncertain. To limit the risk of a new crisis, or to limit its impact on the ecosystem, a solution could be to limit the stratification and the nutrient inputs.

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SIX YEARS OF MONITORING OF FISH ASSEMBLAGES IN SHALLOW BOTTOMS AROUND THE EMBIEZ ISLANDS AND CAP SICIÉ (FRENCH MEDITERRANEAN SEA)

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FISH ASSEMBLAGES
MANAGEMENT
FISHERIES
NATURA 2000
MPAS

ABSTRACT. – Located between the Port-Cros and the Calanques National Parks, the Embiez Islands and the Cap Sicié massif have important natural heritage conservation challenges highlighted by the establishment of three Natura 2000 sites. Coastal ecosystems and associated fish assemblages are subject to various anthropogenic pressures. The fish assemblages of shallow bottoms have been surveyed over the 2012-2017 period by visual censuses in 7 sites, twice a year, in order to assess their structure and temporal changes. The results obtained during these 6 years of monitoring showed that the fish assemblages in this area were diversified and abundant but dominated by a few species of herbivores, mesocarnivores and planktivores. Fish species belonging to higher trophic levels were rare. The fish assemblages of this area differed from those observed in marine protected areas, where appropriate management measures of fishing activities are applied. Fish monitoring following an assemblage approach allowed discussing the efficiency of marine protected area management measures to sustain good functional ecosystem services.

INTRODUCTION

Fish assemblages play a crucial role in the functioning of Mediterranean coastal ecosystems (Sala *et al*. 2012). The Mediterranean Sea has long been overexploited (Vasilakopoulos *et al.* 2014). The most diverse and productive fish assemblages are found in shallow rocky bottoms between the surface and 20 m depth. These environments are the most impacted by human activities. Despite the increasing number of Marine Protected Areas (MPAs), a very small percentage of the coastline is actually efficiently protected (Meinesz & Blanfuné 2015). Consequently, most of these habitats are not subject to specific regulations to protect fish assemblages. Currently, most of the areas considered as MPAs in Europe are under Natura 2000 site classification. The aim of the network is to ensure the long-term survival of Europe's most valuable and threatened species and habitats, listed under both the Birds Directive and the Habitats Directive (92/43/CEE). Along the French Mediterranean coasts, these sites do not usually include strictly protected zones neither regulatory measures of potentially impacting fishing activities (Guidetti *et al.* 2019, Meinesz & Blanfuné 2015).

At West of the City of Toulon (Provence, France), marine areas around the Embiez Islands and the Cap Sicié are managed by three Natura 2000 sites with no regulation measures for fishing activities. Toulon area has a large population, close to 440,000 inhabitants, to which is added the flow of tourists in the summer period. Less than 50 km from either side of Toulon, there are two National Parks including no-take marine reserves. Located to the west, the Calanques National Park was created in 2012 and the benefits of the management measures for the marine environment are becoming conspicuous. To the east of Toulon, Port-Cros National Park, created in 1963, is one of the oldest and well enforced MPA in the Mediterranean Sea, with a multi-use management that generates a strong reserve effect (Astruch *et al*. 2018). Coastal ecosystems and associated fish assemblages are subject to various anthropogenic pressures. The Toulon area is subject to high fishing pressure by both professional and recreational fishermen (including spearfishing and handline fishing). The aim of this study was to perform the first characterization of the fish assemblages present on shallow rocky bottoms of the Natura 2000 sites around the Embiez Islands and Cap Sicié, and to compare them to those encountered in well-established Mediterranean MPAs.

MATERIAL AND METHODS

Study site: The study area is located to the west of Toulon (France, north-western Mediterranean) and extends on three Natura 2000 sites (Fig. 1). Habitat structure is highly diversified with large proportions of seagrass beds (*Posidonia oceanica* (Linnaeus) Delile) and rocky bottoms, and to a lesser extent coralligenous habitats and sandy bottoms.

Visual censuses and sampling design: Data sets were collected by underwater visual censuses (UVC) on transects according to the method developed by Harmelin-Vivien & Harmelin

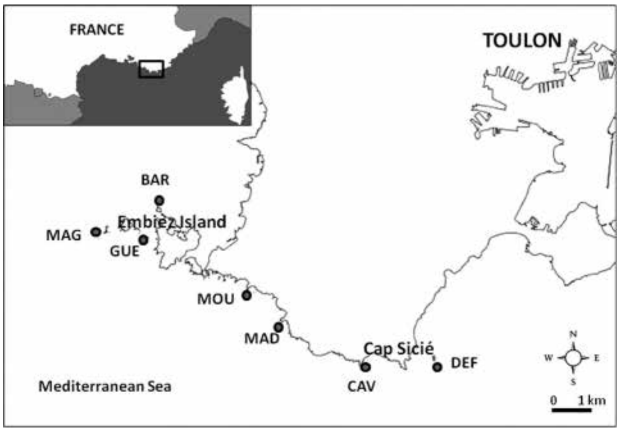

Fig. 1. – Map of study sites west of the City of Toulon. BAR: Basse Renette, MAG: Magons, GUE: Guenaud, MOU: Mouret, MAD: Mal Dormi, CAV: Cap Vieux, DEF: Deux Frères.

(1975) and Harmelin-Vivien *et al*. (1985), and usually used in most MPAs in the north-western Mediterranean (Harmelin-Vivien & Harmelin 2013). Seven sampling sites were considered in the present study. Fish counts were carried out during spring and autumn on a six-year period (2012-2017). At each site, fish assemblages were assessed on 8 transects of 25 m long and 5 m wide constituting replicates. All fish species, their abundance and size (cm) were recorded. All sites were chosen for their similarity in terms of type of substrates, mainly represented by rocky bottoms and seagrass beds to a lesser extent. Transects were placed between 9 to 16 m depth. Habitat characteristics were recorded on each transect, measured as visual estimates of the cover percentages of rock, boulders, rocky slab, sand, *Posidonia oceanica* bed and coralligenous concretions.

Data analysis: A global data analysis was performed combining all data sets (all years and seasons combined). Fish biomass was obtained from the estimation of wet mass of each individual on the basis of size/mass relationships available in FishBase (Froese & Pauly 2019). Density and abundance were processed for the whole assemblage (all species). Reduced density and abundance were calculated excluding planktivorous species (Centracanthidae, Pomacentridae and the sparid *Boops boops* (Linnaeus, 1758)). Density and biomass were also calculated for target species based on a list of 26 species as done by Astruch *et al*. (2018). Target species mainly belonged to the following fish families: Congridae, Gadidae, Labridae, Moronidae, Mullidae, Sciaenidae, Scorpaenidae, Serranidae, Sparidae and Sphyraenidae. Fish species were classified into 6 trophic groups according to their diet, following Astruch *et al*. (2018). Data being not normalized, differences in mean values were tested with non-parametric Kruskal-Wallis tests.

RESULTS

During these 6 years of monitoring, 48 taxa of coastal fishes have been observed in the study area (Table I). Total number of species encountered per site varied between 29 at Mal Dormi and 40 at Deux Frères (Table II). Mean point diversity in the area was 9.8 species/transect and varied significantly among sites (Kruskal-Wallis test, $p < 0.0001$) with the highest values at Deux Frères and Magnons, and the lowest at Basse Renette. Mean density (for all species combined) for the study area was 211.5 ind/100 m² and varied significantly among sites (Kruskal-Wallis test, $p < 0.0001$) with the maximum value at Deux Frères and the lowest value at Guénaud,

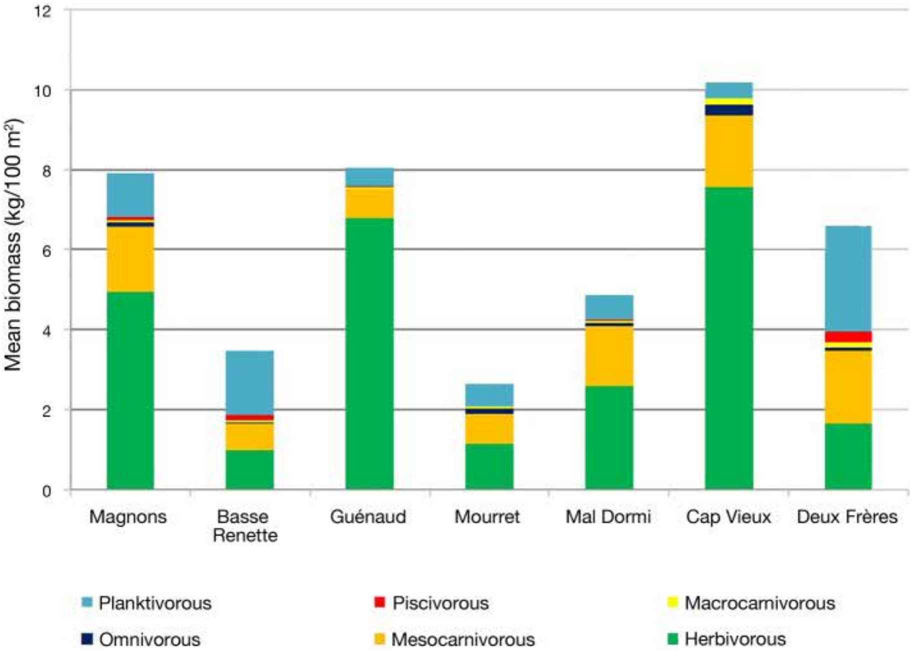

Fig. 2. – Mean biomass (kg/100 m²) of fish assemblages in each site and its distribution between the 6 trophic groups considered.

Table I. – List of the fish species recorded in the study sites located west of the City of Toulon with indication of their exploitation status by fisheries (target or no-target species) and trophic category (Species nomenclature based on International Commission on Zoological Nomenclature).

Families	Species	Target species	Trophic category
Apogonidae	*Apogon imberbis*		Planktivorous
Carangidae	*Seriola dumerili*	X	Piscivorous
	Trachurus trachurus		Macrocarnivorous
Centracanthidae	*Spicara sp.*		Planktivorous
Gadidae	*Phycis phycis*	X	Mesocarnivorous
Labridae	*Coris julis*	X	Mesocarnivorous
	Labrus merula	X	Mesocarnivorous
	Labrus mixtus	X	Mesocarnivorous
	Labrus viridis	X	Mesocarnivorous
	Symphodus cinereus		Mesocarnivorous
	Symphodus doderleini		Mesocarnivorous
	Symphodus mediterraneus	X	Mesocarnivorous
	Symphodus melanocercus		Mesocarnivorous
	Symphodus melops		Mesocarnivorous
	Symphodus ocellatus		Mesocarnivorous
	Symphodus roissali		Mesocarnivorous
	Symphodus rostratus		Mesocarnivorous
	Symphodus tinca	X	Mesocarnivorous
	Thalassoma pavo		Mesocarnivorous
Mugilidae	*Mugilidae sp.*		Omnivorous
Mullidae	*Mullus surmuletus*	X	Mesocarnivorous
Muraenidae	*Muraena helena*	X	Macrocarnivorous
Pomacentridae	*Chromis chromis*		Planktivorous
Sciaenidae	*Sciaena umbra*	X	Mesocarnivorous
Scorpaenidae	*Scorpaena maderensis*		Macrocarnivorous
	Scorpaena notata		Macrocarnivorous
	Scorpaena porcus		Macrocarnivorous
	Scorpaena scrofa	X	Piscivorous
Serranidae	*Anthias anthias*		Planktivorous
	Epinephelus costae	X	Piscivorous
	Epinephelus marginatus	X	Piscivorous
	Serranus cabrilla	X	Macrocarnivorous
	Serranus hepatus		Macrocarnivorous
	Serranus scriba	X	Macrocarnivorous
Sparidae	*Boops boops*		Planktivorous
	Dentex dentex	X	Piscivorous
	Diplodus annularis		Mesocarnivorous
	Diplodus cervinus	X	Mesocarnivorous
	Diplodus puntazzo	X	Omnivorous
	Diplodus sargus	X	Mesocarnivorous
	Diplodus vulgaris	X	Mesocarnivorous
	Oblada melanura		Omnivorous
	Pagellus acarne	X	Macrocarnivorous
	Pagrus pagrus	X	Macrocarnivorous
	Sarpa salpa		Herbivorous
	Sparus aurata	X	Mesocarnivorous
	Spondyliosoma cantharus	X	Mesocarnivorous
Sphyraenidae	*Sphyraena viridensis*	X	Piscivorous

with respectively 291.7 ind/100 m^2 and 153.3 ind/100 m^2. Among the 48 species observed, 5 of them represented more than 85 % of the total density. The most abundant species were *Chromis chromis* (Linnaeus, 1758), *Coris julis* (Linnaeus, 1758), *Sarpa salpa* (Linnaeus, 1758), *Spicara* sp. and *Boops boops*. When we excluded planktivorous species, the mean reduced density was 60.8 ind/100 m^2 in the study area. For target species (26 species out of the 48 observed in the study area) the mean density was 40.9 ind/100 m^2 and the mean biomass was 1.41 kg/100 m^2.

The mean biomass for all sites was 6.24 kg/100 m^2 when all species were considered and 5.19 kg/100 m^2 for the reduced biomass (without planktivorous species). Fish assemblages were dominated by herbivorous species which contributed mainly to the significant fluctuation of biomass between sites (Kruskal-Wallis test, $p < 0.0001$) (Fig. 2). *Sarpa salpa* was the only herbivorous species recorded in the study area and accounted for 59 % of the mean total biomass. Piscivorous fishes were represented by 6 species and accounted for 1.25 % of the biomass. Among the carnivorous species observed in the area, two serranids, the dusky grouper *Epinephelus marginatus* (Lowe, 1834) and *E. costae* (Steindachner, 1878), and the Sciaenidae *Sciaena umbra* (Linnaeus, 1758) are concerned by a moratorium banning angling and spearfishing. During this six-year study, 20 dusky groupers were observed. Among them, several individuals were probably observed several times during the different counting campaigns, which lowered the actual number of groupers present in this area. Only one individual of *E. costae* was observed. A total of 17 individuals of *Sciaena umbra* was observed in only 2 out of the 7 sites investigated: 16 individuals were observed at Cap Vieux during the different UVC campaigns and 1 individual was observed at Mouret.

Table II. – Characteristics of fish assemblages in the study sites located west of Toulon and in well-established Mediterranean MPAs.

	Alpha diversity per site	Point diversity per transect	Mean density (all species) (ind/100 m²)	Mean reduced density (ind/100 m²)	Mean density (Target species) (ind/100 m²)	Mean biomass (All species) (kg/100 m²)	Mean reduced biomass (kg/100 m²)	Mean biomass (Target species) (kg/100 m²)
Magnons	34	11.3	224.0	61.9	39.8	7.91	6.83	1.82
Basse Renette	34	8.6	222.3	29.9	20.9	3.48	1.84	0.83
Guénaud	31	9.2	153.3	45.4	28.8	8.04	7.59	0.76
Mourret	31	9.2	176.3	69.1	42.4	2.64	2.07	0.77
Mal Dormi	29	10.5	211.9	72.2	51.7	4.86	4.25	1.52
Cap Vieux	33	8.3	200.9	87.3	57.8	10.19	9.78	1.89
Deux Frères	40	12.1	291.7	60.1	45.2	6.60	3.97	2.22
Total area	**48**	**9.8**	**211.5**	**60.8**	**40.9**	**6.24**	**5.19**	**1.41**
BIOMEX (1) Inside MPA		10.1-14.1		48.9-90.9			3.6-22.6	
BIOMEX Outside MPA		7.7-13.9		25.2-78.6			0.56-3.2	
Port-Cros (2) (Rocky reefs 5-15 m depth)	23-34	10.7-16.7	158.4		37.5	6.58		4.74

(1) Harmelin-Vivien *et al.* 2008
(2) Astruch *et al.* 2018

DISCUSSION

Substrate type and depth are generally the main factors affecting fish assemblage composition (García-Charton & Pérez-Ruzafa 2001). Although all sites were chosen for their similarity in terms of substrate type and depth, some habitat differences might generate inter-sites variations in the fish assemblages' parameters recorded in the study area. Globally, West Toulon fish assemblages were well diversified with a high alpha diversity and a point diversity slightly lower than those encountered in well-established MPAs in the north-western Mediterranean Sea, such as around the Port-Cros archipelago (Astruch *et al.* 2018) and in the six MPAs studied in the BIOMEX project (Harmelin-Vivien *et al.* 2008). Values of mean density and mean biomass for all species combined and excluding planktivorous species in West Toulon sites were similar to those encountered in other well-established MPAs.

Pyramids of biomass of fish assemblages in West Toulon sites were dominated by herbivorous fishes (*Sarpa salpa*), while highest trophic level fishes including piscivorous species were scarce. On the contrary, the reverse situation with biomass pyramids of fishes dominated by high trophic level (macrocarnivores and piscivores) were encountered around Port-Cros Island MPA in similar conditions of habitat and depth (Astruch *et al.* 2018) and in other well-established reserves, approaching a pristine situation (Sala *et al.* 2012).

Target species were poorly represented in term of biomass in West Toulon area and among them the carnivorous species were particularly under-represented. Fish assemblages were largely dominated by species without economic interest in terms of density and biomass. *S. salpa* seemed to benefit from the low predation pressure to proliferate. This situation can have consequences on the whole ecosystem induced by a possible overgrazing of macroalgae (Pinnegar *et al.* 2000, Guidetti 2007, Vergés *et al.* 2009). In contrast, around Port-Cros Island, target species represented a high proportion of the total fish biomass, resulting from an effective management (Astruch *et al.* 2018). The strong differences observed between these two geographically close areas raises the question: does the West Toulon area have suitable habitats for fish assemblages like those of Port-Cros? No scientific data allows estimating the previous state of fish assemblages in this area. Fortunately, the film archive "*Par 18 m de fond*" was shot around the Embiez Islands in 1942. This film highlights that less than 80 years ago, fish assemblages in this area have seemed to be very productive and dominated by piscivorous species. The images shot in this film contrasted drastically with the current situation. In a few decades, fishing activities have sharply increased in this area, mainly for recreational fishing (angling and spear-fishing) (Font & Lloret 2014). Fish assemblages were also affected by the loss of habitats (particularly nurseries) and diverse anthropogenic pressures (pollution, over-frequentation, tourism activities, noise, etc.). This study highlighted that fish assemblages in the West of Toulon appeared to be disturbed and likely affected by fishing activities although the area is classified Natura 2000 site. In addition, even species concerned by special regulations such as groupers or *Sciaena umbra* were rare and exhibited escape behavior, which suggested illegal spearfishing activities and poaching. Natura 2000 sites are designated to protect a certain number of habitats but not specifically to protect coastal fish assemblages. Without any particular regulation of fishing activities, Natura 2000 sites do not effectively protect fish assemblages and target species, and consequently ecosystem-wide conservation (Meinesz & Blanfuné 2015, Guidetti *et al*. 2019). This area needs thus more effective management measures, such as the implementation of no-take areas, to protect the fish assemblages, which play a key role in the functioning of ecosystems. This would complement and reinforce the benefits of the actions supported by N2000.

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SPECIES-BASED OR ECOSYSTEM-BASED APPROACHES TO CONSERVATION PRACTICES: LESSONS FROM THE PORT-CROS NATIONAL PARK (SOUTH-EAST FRANCE, MEDITERRANEAN SEA)

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BIOLOGICAL INVASIONS
CONSERVATION
CONTROL OF INVASIVE SPECIES
ECOSYSTEM-BASED APPROACH
MANAGEMENT
MEDITERRANEAN
NATIONAL PARK

ABSTRACT. – For almost 60 years, the management policy of the Port-Cros National Park (PCNP) has steadily progressed, in phase with human impacts, changes in doctrines and goals in the field of nature conservation, what was socially acceptable at a given time, a very active Scientific Council and a gradual shift from species-centered to ecosystem-based management. Here, we illustrate problems faced by a national park, together with responses or practices that are suited, or not, to an ecosystem-based approach. PCNP's doctrine has been to explain that the abundance of species fluctuates naturally, and that we must not be too hasty to intervene, that nature sometimes does things better than humans, that a national park is neither a zoo nor a botanical garden, and the purpose is not to artificially increase the species diversity. The management approach of the PCNP can constitute a source of lessons for nature protection and management.

INTRODUCTION

In the 18th century, and for centuries earlier, the prevailing approach to nature was human-centered (Boudouresque *et al*. 2020). The French naturalist Buffon felt only contempt for the environments that we now refer to as 'natural' (Buffon 1764, 1767). It is clear that, for Buffon, the ideal nature was represented by the royal gardens of Versailles, its fountains and canals, and its adjacent forest dedicated to royal hunts. This vision of nature prevailed until the 19th century. During the first two-thirds of the 19th century, the philosophy of Saint-Simon, by promising the happiness of humanity through the scientific domestication of nature, helped to reinforce this vision. In Europe and North America, the 'Acclimatization Societies', which were set up in the 19th century and persisted until the mid-20th century, aimed to make Western nations profit from the natural wonders of the world, by introducing them to Europe and North America (*e.g.*, Matagne 1999, Planhol 2004, Faget 2007, Luglia 2014). The human-centered vision of nature persisted until the 20th century, in the guise of a dichotomy between useful species (for man) and pests (competitors of humanity); official lists of pests, the destruction of which was recommended, were published by European government bodies (De la Blanchère 1878, Faget 2016). It is worth noting that the interest for humans is today taken into account through the concepts of ecological goods and ecosystem services: the natural processes and component that benefit human needs (Costanza *et al.* 1997, 2014, Nordlund *et al.* 2016). The Japanese Satoumi approach based upon traditional use of the coastal waters by local people in the Edo period (1600-1868 CE) and still in use, may, in a way, be considered as a modern form of the human-centered approach (Berque & Matuda 2013, Henocque 2013, Yanagi 2013). The paroxysm of the human-centered approach was the eradication of predators (wolves, coyotes, grizzly bears), in the 19th century, in the famous Yellowstone National Park (USA). This destruction was not only the result of a naive approach by creationist do-gooders (carnivores are evil and were sent by God to punish men), but also of a naive approach to ecology. Predators are essential for the good health of their prey (see the Healthy Herd Hypothesis) and, in their absence, the Yellowstone ecosystem has collapsed. It was not until the middle of the 20th century that they were successfully reintroduced or naturally came back, partly restoring the natural character of the Yellowstone National Park (Brussard 1992).

From the 19th century on, and especially during the last decades, environmental protection and conservation have been mainly centered on the species, or on sets of species, corresponding to the specialization of taxonomists: bats, birds, sea mammals, flowering plants, etc. Of course, these taxonomists were also, in general, environmentalists, in the sense that they were also interested in habitat (a notion which should not be confused with that of the ecosystem). Together with artists, they played a major role in the emergence of the concept of nature protection (Matagne 1999, Jaffeux 2010). Species have been divided into two broad categories. On the one hand, outstanding species, including species with 'heritage' value, which deserve attention; on the other hand, the 'ordinary' species. The notion of outstanding species, together with its heritage value, is a fuzzy concept (Gauthier *et al.* 2010, Astruch *et al.* 2012, Boudouresque *et al.* 2020). It covers: (i) rare species (either threatened or not). (ii) Threatened species, according to the IUCN Red List criteria, or IUCN-like criteria at a regional scale. The IUCN Red List is often irrelevant: the IUCN is dominated by terrestrial lobbies (*e.g.*, birds, bats, flowering plants) or terrestrial-like lobbies (*e.g.*, sea mammals, marine turtles); as pointed out by Thibaut *et al.* (2016a) and by Verlaque *et al.* (2019), many marine species, such as the brown algae (kingdom Stramenopiles) *Cystoseira* and *Sargassum*, although on the brink of extinction and despite being the focus of extensive scientific literature, and fully meeting the IUCN criteria, are still considered by the IUCN as DD (Data Deficient) or NE (Non-Evaluated). (iii) Species protected by national or international legislation; unfortunately, protection is sometimes irrelevant and dependent upon taxonomic lobbies (Thibaut *et al.* 2016a, Mammides 2019, Verlaque *et al.* 2019); the seagrass *Cymodocea nodosa* (Ucria) Ascherson is protected in France, while it is an opportunistic species, that benefits from habitat degradation and climate warming, and is currently in expansion. The seabird *Larus michahellis* (Naumann, 1840) is also protected, while its proliferation is due to human impact and threatens a number of other seabirds[1]. (iv) Charismatic species (including iconic species) are lovable and/or aesthetically pleasing species that enjoy a coefficient of sympathy from the general public (whether threatened or not, whether or not they play an important functional role in the ecosystem) (Dubois *et al.* 2017, Thibot et al. 2018). Dolphins are the perfect example; the common bottlenose dolphin *Tursiops truncatus* and the striped dolphin *Stenella coeruleoalba* (Meyen, 1833) are far from being threatened; there is no evidence that their populations are declining at a global scale (LC – Least Concern) and in Europe (DD – Data Deficient) (but see Baş *et al*. 2017 for *Tursiops truncatus* (Montagu, 1821)); they may even benefit locally from human impact (*e.g.*, the decline of their competitors, such as sharks) (Cagnolaro & Notabartolo di Sciara 1992, Gannier 1995, Aguilar 2000, IUCN SSC Cetacean Specialist Group 2007, Baş *et al.* 2017, Braulik 2019). Whatever their actual population status, national legislations fully protect all Mediterranean dolphins.

Managers, stakeholders and environmentalists worldwide often prioritize the species-centered approach (or 'species-by-species' approach). The protection of an iconic and endearing species is obviously easier than that of tiny zooplankton species, or of parasites, although the latter may play a far more important role than the former in the functioning of 'healthy' ecosystems (Combes 2001). Within the framework of the Habitat Directive (1992) and of the Natura 2000 network of nature protection areas of the European Union (EU), the species-centered approach has been widely favored compared to the ecosystem-based approach, despite the name given to the Directive. It is worth highlighting that 'habitats', as understood by some managers, have nothing in common with ecosystems: they just correspond to the phytosociological units (Magnoliophyta) or species assemblages housing the species of interest. However, 'species-by-species' management is unrealistic, particularly when the emblematic species are either predators or prey. Obviously, the protection measures cannot lead to the increase in numbers of both predator and prey populations. The issue is that the management of natural habitats has often been driven by environmentalist 'lobbies', solely on the basis of taxonomical considerations. As every taxonomist specialist group focuses on its specific type of organism (*e.g.*, marine mammals, turtles and tortoises, birds, iconic fish such as the dusky grouper *Epinephelus marginatus* (Lowe, 1834), flowering plants), the management of natural habitats sometimes results in a layering of taxon-focused protection measures. All in all, the management of natural habitats is often reminiscent of the tale of the blind men examining the elephant (Boudouresque *et al.* 2020).

With the ecosystem-based approach (EBA), we moved from the notion of species, which of course play a role in an ecosystem, to that of an ecosystem in which species participate and interact. The distinction may seem tenuous. However, it is a true revolution, as important as the shift from the human-centered approach, which characterized the 18th century and is illustrated by Buffon (1764, 1767), to the species-centered approach. The EBA has several strong points (Boudouresque *et al.* 2020). (i) It allows the inclusion of humans in the functioning of the ecosystem, in a natural way, thus evolving from the notion of ecosystem to that of social-ecological system. Humans are no longer set aside but are within the system. (ii) While the species-centered approach often just con-

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[1] *Larus michahellis* is subject to a somewhat hypocritical system of 'adaptive management': although remaining protected, its populations are 'regulated' by officially mandated officials.

siders a collection of remarkable taxa, the EBA requires the construction of a framework corresponding to a conceptual model of the ecosystem. In this model, there are grounds for including even the unremarkable species (*i.e.*, species that are not rare, or threatened, or iconic). (iii) The conceptual model makes it possible to link the species together, following a network of interactions (predation, parasitism, mutualism), and to better interpret the possible fluctuations in their numbers. (iv) The conceptual model can be a stepping-stone towards analytical or numerical modeling, where flows (*e.g.*, C, N, P) between compartments are quantified. (v) The EBA also highlights the importance of tackling the coupling between adjacent ecosystems (including benthic and pelagic, terrestrial and marine ecosystems). (vi) The EBA enables the development of environmental quality indices that are much more significant and reliable than indices based on one or a few species: see *e.g.*, the Ecosystem-Based Quality Index (EBQI) for the *Posidonia oceanica* seagrass ecosystem (Personnic *et al.* 2014, Boudouresque *et al.* 2015). (vii) Invasive species constitute one of the most worrying aspects of global change, and the Mediterranean Sea is the area worldwide most hit by non-indigenous species (Galil 2008, Katsanevakis *et al.* 2013, Maxwell *et al.* 2016, Boudouresque *et al.* 2017a). Invasion issues are usually studied and managed in a single-species context: the interaction between an invasive species and a native one, the impact of an invasive species on point or alpha species diversity, etc. In fact, invasive species rarely act in isolation, but in packs; invasive species rarely have an impact on a species alone, but on entire communities; therefore, understanding their role and impact can only be achieved in the context of the whole ecosystem (Boudouresque *et al.* 2005a, 2011). (viii) Human activities (*e.g.*, fisheries, contamination) do have an impact on particular species; however, it is only within the framework of the whole ecosystem, and within its functional compartments, that these effects can be understood, managed and if possible mitigated (*e.g.*, Halpern *et al.* 2010, Cresson *et al.* 2014, Giakoumi *et al.* 2015, Ourgaud *et al.* 2015, Kincaid *et al.* 2017). Ecosystem-Based Fishery Management (EBFM) is obviously part of the EBA (*e.g.*, Rice 2005, Tudela & Short 2005).

The Port-Cros National Park (PCNP, Provence, France, Mediterranean) was established in December 1963 (Augier & Boudouresque 1973, Boudouresque 1976, Bougeant 1990). It encompassed the Island of Port-Cros and the neighboring island and islets (Bagaud Island, La Gabinière Islet and Le Rascas Islet), *i.e.*, the Port-Cros Archipelago, situated about 8 km off the continental coast of eastern Provence (Fig. 1). Together with the land areas of the island and islets, the PCNP included a 600-m wide belt of sea, corresponding to 1300 ha in surface area, surrounding the archipelago (Boudouresque *et al.* 2013). The

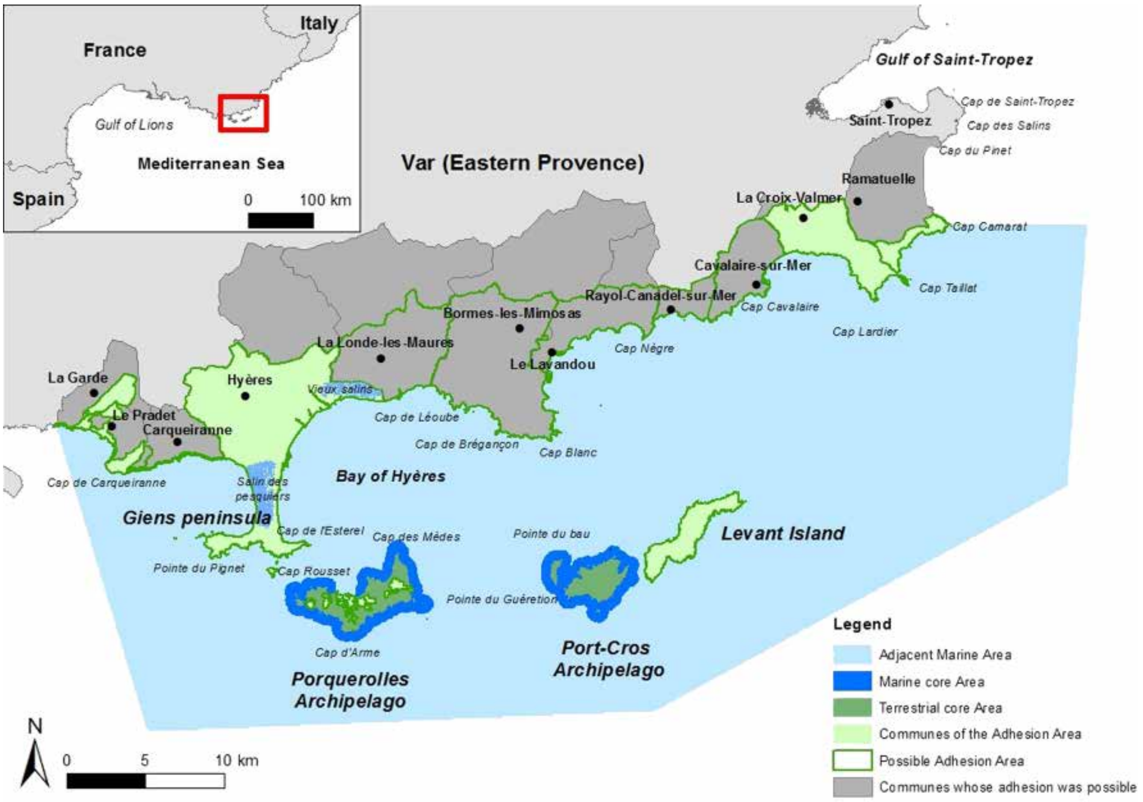

Fig. 1. – Map of the new Port-Cros National Park (N-PCNP), established in 2016. The initial PCNP, established in 1963, was restricted to the Archipelago of Port-Cros.

PCNP is one of the oldest terrestrial and marine national parks in the Mediterranean area. The oldest one is the Mljet National Park (Croatia), founded in November 1960, which stretches over 5,400 ha, including a marine area extending 500 m from the coastline (Kružić 2002).

Progressively, the PCNP has been entrusted with the management of an increasing number of territories outside the initial territory of 1963. Since 1985, the PCNP has managed the land (~950 ha), bought by the French state in 1974, situated on the neighboring island of Porquerolles. Since 1984, the park has managed the lands of the *Conservatoire de l'Espace Littoral et des Rivages Lacustres* (CERL) (Conservatoire of coastal areas and lake shores) situated at Cap Lardier, and since 1997 those situated on the island of Porquerolles (Grand Langoustier) and the Giens Peninsula (Escampobariou). Since 1999, the PCNP has run Natura 2000 for the islands of Port-Cros and Porquerolles and is the operator of Natura 2000 for the island of Le Levant and the salt marshes at Hyères. Since 2004, the PCNP provides technical and scientific support to the *Métropole Toulon Provence Méditerranée* (MTPM) for the management of the CERL territories of Les Pesquiers and Les Vieux-Salins (Barcelo & Boudouresque 2012, Astruch *et al.* 2018, 2019, Barcelo *et al.* 2018).

Following the redefinition of the national parks by French legislation in 2006, the PCNP engaged, between 2012 and 2016, in a major redefinition and extension of its territory; the new Port-Cros National Park (N-PCNP), established in 2016, includes the Port-Cros and Porquerolles Archipelagos as core areas (both terrestrial and marine), a vast Adjacent Marine Area (AMA – *Aire Maritime Adjacente*) including the Gulf of Hyères and extending seawards to the edge of the continental shelf, and a discontinuous continental area, the Adhesion Area (AA – *Aire d'Adhésion*) including five *communes* (municipalities; the *commune* is the smallest territorial division in France) (Fig. 1; Barcelo & Boudouresque 2012, Astruch *et al.* 2018, Barcelo *et al.* 2018). The *communes* of the AA have voluntarily joined the National Park, through the signing of a charter, which defines the objectives of the National Park regarding the conservation of the natural and cultural heritage and sustainable development (Thompson *et al.* 2011, Sellier 2015, Astruch *et al.* 2018, Hogg *et al.* 2018). In contrast, some *communes* of the Potential Adhesion Area (*Aire Optimale d'Adhésion*) did not wish to be part of the N-PCNP (Fig. 1).

The core area of the N-PCNP is therefore constituted by the archipelagos of Port-Cros and Porquerolles, off the coast of Provence. For almost 60 years, the management policy of the PCNP (initially restricted to the Port-Cros Archipelago) has steadily progressed, in phase with (i) changes in doctrines and goals in the field of nature conservation; (ii) what was socially acceptable at a given time; (iii) a very active scientific research background and a level of knowledge unique for Mediterranean protected areas (Farsac *et al.* 2013); (iv) improved governance, involving inhabitants, users and stakeholders (Hogg *et al.* 2016, Barcelo *et al.* 2018); and (v) a gradual shift from species-centered to ecosystem-based management. The PCNP Scientific Council, active, realistic and therefore respected and listened to by managers and stakeholders, has played an important role in this process (Barcelo *et al.* 2013a, Boudouresque *et al.* 2013).

Here, on the basis of several examples, we critically analyze earlier and recent issues faced by the PCNP, the N-PCNP and its Scientific Council, together with responses or practices that are suited, or not, to an ecosystem-based approach (EBA) (Tables I, II). It is worth noting that it was not our intention to provide a comprehensive list of the management issues faced by the PCNP and the N-PCNP over almost 60 years. Maintenance and restoration of the architectural heritage (forts dating from the 16th and 17th centuries), archaeological excavations and exploration of wrecks, such as that of *La Baleine* in the Bay of Port-Cros (*e.g.*, Guérout 1981, Caron 1983, Brun 1997, Long 2004, Ruitton *et al.* 2004), have not been considered. In addition, park management cannot be considered as limited to the chosen examples and the provided possible responses; scientific investigation, long-term monitoring of species, ecosystems, human uses, landscapes and seascapes are also part of the management process. For example, the mapping of the localization of fishing gear by PCNP and N-PCNP officials is pivotal for the updating of the fishing charter and the management of the artisanal fishery (see below). Finally, doing nothing is still a management response, often the best one, although sometimes difficult to explain to the public at large and policy makers who ask for visible actions (Pont 2003, Schnitzler *et al*. 2008). In any case, consideration should be given to past mistakes in any humility, firstly because it is easy to judge the past in light of today's knowledge and paradigms, and secondly because the truths of today will probably not be those of tomorrow.

SPECIES-BY-SPECIES MANAGEMENT ACTIONS THAT COULD BE CONSIDERED TODAY AS INAPPROPRIATE

The digging of a pond in the western part of Porquerolles Island is an example of the management errors driven by taxonomic lobbies, here the bat lobby (Fig. 2, Table I, T14). The probable goal was to increase the insect resource for bats (Éric Serantoni, pers. comm.), to the detriment of protected plants, which does not matter to bat lovers. It is important to note that at the time, in 2002-2003, the Scientific Council had not been consulted by the scientific service of the PCNP regarding this operation.

In the heart of the village of Porquerolles, in a cellar (known as the *'cave à vin'* – wine cellar), a colony of 65 individuals of the Geoffroy's bat *Myotis emarginatus*

Table I. – Management issues and responses from the PCNP and N-PCNP in the terrestrial realm (numbered T1 through T26). ‘No’ means that the management suggestion was not implemented, either because it was considered unnecessary or because it was deemed unrealistic, by the Scientific Council and/or by the management team of the PCNP. It is worth noting that Porquerolles has been managed by the PCNP since 1974, but it has only belonged to the national park (N-PCNP) since 2012. PC: Port-Cros Archipelago. PQ: Porquerolles Archipelago.

Nr, Dates	**Management issue**	**Responses, results and comments**	**References**
T1 1970 to 1975	Degradation of the habitat of the Tyrrhenian painted frog *Discoglossus sardus* (PC)	Construction of low walls across valleys to retain water. Failure: ponds quickly filled	Hervé Bergère (pers. comm.)
T2 1970s	Landscape enhancement? (PQ)	Plantation of ice plants *Carpobrotus* sp. on the back beach of Plage d’Argent	Nicolas Gérardin (pers. comm.)
T3 1974	An inappropriate initiative by a State official (not a PCNP staff member) at a loss – what to do with a gift from the President of an Asian country to the President of the French Republic – and said to have been responsible for the release of 3 individuals of the Sika deer *Cervus nippon* (one male and two females) (PQ)	Up to about 20 individuals. Gradual eradication by the ONCFS (Office National de la Chasse et de la Faune Sauvage – *National Office of Hunting and Wildlife*). Last sighting in 2016	Cheylan (1984), Cheylan & Geoffroy (2020), Alain Barcelo (pers. comm.)
T4 1975	Extinction of Hermann’s tortoise *Testudo hermanni* (PC)	Reintroduction of 46 individuals. See text for details	Besson (1975)
T5 Early 1980s	Providing shade for beach-goers on the Plage du Sud? Restoration of the supposed original features of the island landscape? (PC)	Plantation of Italian stone pines *Pinus pinea*	Hervé Bergère (pers. comm.)
T6 1982 to present	Degradation of the vegetation of the back beach (Plage du Sud) by tourists (bathers) (PC)	Construction of a low wall (1982), then installation of split stake fences (*ganivelles*) between 2002 and the present	Hervé Bergère (pers. comm.)
T7 Mid-1980s	Landscape enhancement (PC)	Plantation of ice plants *Carpobrotus* sp. in the village of Port-Cros (Port-Cros Island)	Nicolas Gérardin (pers. comm.)
T8 1995?	Worrying decline of 4 rare and protected species of Magnoliophyta (PC and PQ)	The authors suggested clearing, thinning of vegetation and nitrogen supply via anthropization. Some brushcutting operations carried out (PQ)	Médail *et al*. (1995)
T9 1995 to 2003	Too many footpaths: trampling, increased risk of fire, risk for hikers in the event of fire (PC)	Half a dozen paths closed	Hervé Bergère (pers. comm.)
T10 1995 to present	Worrying proliferation of invasive plants (e.g. *Carpobrotus* spp., *Cortaderia selloana, Lonicera japonica*)	Experimental manual control of *Carpobrotus* spp., *Senecio angulatus, Salpichroa origanifolia*, etc.	Aboucaya (2013)
T11 1997-1998 to 2007	Rescue of 40 individuals of Hermann’s tortoise *Testudo hermanni* kept by a resident of Saint-Tropez (eastern Provence) who was leaving the region (PQ)	Enclosure construction and feeding (lettuce, fruit)	Nicolas Gérardin (pers. comm.)
T12 2001	Predation of eggs and larvae of *Discoglossus sardus* by the introduced mosquitofish *Gambusia holbrooki* (PC)	Eradication of *Gambusia* by poisoning (rotenone)	Lim & Dauba (2001), Duguet *et al.* (2019)
T13 2002	Demolition of the cellar which housed a colony of the bat *Myotis emarginatus*, in the heart of the village (PQ)	Construction of a tower specially designed to house the bats. See text for criticism	Médard *et al* (1999), Quekenborn *et al.* (2004)
T14 2002 or 2003 to present	Unknown	Digging a pond to increase the insect resource for bats (PQ)	Éric Serantoni (pers.comm.)
T15 2003: No	Conservation of bat species (PC)	The authors suggested fitting out the buildings to enhance the abundance of bats	Stoecklé (2003)

Table I. Continued.

Nr, **Dates**	**Management issue**	**Responses, results and comments**	**References**
T16 2003 to present	Possible decline of shearwater (*Puffinus yelkouan* and *Calonectris diomedea*) (PC, PQ)	Setting up of artificial nest-cavities. Poor success: available nests are not a limiting factor. See text for criticism	LPO *et al.* (2007), Bourgeois *et al.* (2015)
T17 2002 to 2011	Predation of feral cat *Felis catus* on adult Mediterranean shearwater *Puffinus yelkouan* (PC, PQ)	Eradication of feral cats by cage traps: 112 cats caught and transferred to the mainland. Increase (by 1.5) of the reproductive bird population (PC)	Tranchant & Vidal (2003), Bourgeois & Vidal (2008), Bergère (2009), Médail *et al.* (2013) Clelia Moussay (pers. comm.)
T18 2004	Extinction of the ocellated lizard *Lacerta lepida* (PQ)	The extinction is natural (disappearance of suitable habitats). No management action suggested	Cheylan & Cluchier (2004)
T19 2006 to ?	Determine numbers, social structure and spatial distribution of visitors (bikers, boaters, divers, etc.)	Setting up of the 'Observatoire Bountîles'	Le Berre *et al* (2013), Brécard & De Luigi (2016)
T20 2007: ?	Predation of birds by the European hedgehog *Erinaceus europaeus,* non-indigenous in Porquerolles (PQ)	Suggested control of hedgehogs by trapping. See text for comments	Legrand *et al.* (2007)
T21 2011?	Beach erosion and beach-dune ecosystem impoverishment due to removal of *Posidonia* dead leaves (*banquettes*) (PC, PQ)	Manual removal of human-generated waste matter (*e.g.*, plastic and metallic debris, processed wood). No removal of *banquettes* and natural driftwood	Serantoni (2015), Boudouresque *et al.* (2016, 2017b)
T22 2011 to present	Over-frequentation and carrying capacity (PC, PQ)	Impact on the well-being of visitors, on safety (risk of fire, injured visitors) and on national park officials workload	Bergère & Le Berre (2011), Jolivet (2018), Deldrève & Michel (2019)
T23 2011 to present	Invasion by two exotic species, the ice plant *Carpobrotus* spp. and the black rat *Rattus rattus* (PC)	Co-eradication of *Carpobrotus* spp. and *Rattus rattus* (traps and bromadiolone)	Bourgeois *et al.* (2005), Passetti *et al.* (2012), Médail *et al*. (2013), Braschi *et al*. (2017), Buisson *et al.* (2018)
T24 2012 to present	Invasion by the Argentine ant *Linepithema humile* (PC and PQ)	Preventing its spread to remote areas of the islands, and to still non-colonized islets and Bagaud Island	Berville *et al.* (2012)
T25 2018	Light pollution (PQ)	Four bat species are good indicators of light pollution, which should be reduced	Geoffroy *et al.* (2018)
T26 2015 to present	Natural arrival of wild boar *Sus scrofa* on Port-Cros, Bagaud and Porquerolles Islands (PC and PQ)	Control *via* trapping and hunting. See text for comments	Cheylan & Geoffroy (2020), Hervé Bergère (pers. comm.)

Table II. – Management issues and responses from the PCNP and N-PCNP in the marine realm (numbered M1 through M20). 'No' means that the management suggestion was not implemented, either because it was considered unnecessary or because it was deemed unrealistic or unethical, by the Scientific Council and/or by the management team of the PCNP. It is worth noting that Porquerolles has been managed by the PCNP since 1974, but It has actually only belonged to the national park (N-PCNP) since 2016.

Dates	Management issue	Responses, results and comments	References
M1 1963 to present	Over-exploitation of fish	Ban on spear fishing and trawling	Boudouresque (1976)
M2 1970: No	Erosion of the *Posidonia oceanica* barrier reef in Port-Cros Bay by boats trying to cross it to reach the beach (PC)	The installation of a line of buoys and a rope across the bay was recommended (1970)	Augier & Boudouresque (1970a)
M3 1970s to present	Prevent anchoring by leisure boats and *P. oceanica* meadow degradation, setting up of dead weights, chains and mooring buoys (PC)	The impact of dead weights and chains was worse than that of anchors (Fig. 3). Should be removed. Dead weights were left in place while chains were removed	Robert (1983), Hervé Bergère (pers. comm.), Jean-François Magréaud (pers. comm.)
M4 Between 1982 and 1987 to present	Erosion of the *Posidonia oceanica* barrier reef of Port-Cros Bay by boats trying to cross it to reach the beach (PC)	The installation of a line of buoys and a rope across the bay of Port-Cros	Stéphane Penverne (pers. comm.)
M5 1985 to present	Setting up of artificial reefs, to improve fish stocks and artisanal fishery (La Palud, PC)	See text for criticism	Charbonnel *et al.* (2001)
M6 1987	Local extinction of the giant limpet *Patella ferruginea* (French continental coasts) (La Palud, PC)	Reintroduction of individuals from Corsica (Scàndula) to the Rascas Islet, Port-Cros Archipelago. Failed, see text for comments	Laborel-Deguen (1988), Laborel & Laborel-Deguen (1991)
M7 1989 to present	Only research goal. Cuttings of *P. oceanica* transplanted from 12 populations (Algeria, France, Greece, Italy, Spain and Turkey) (PC)	See text for criticism	Meinesz *et al.* (1993)
M8 1993 to present	Degradation of the *Posidonia oceanica* meadow by anchoring (PC)	Prohibition of anchoring in two areas of the Port-Cros Archipelago (Fig. 7)	Ganteaume *et al.* (2005)
M9 1994 to present	The attractiveness of the seascapes and the density of emblematic fish generate a risk of over-frequentation and habitat degradation (PC)	Each scuba diver must sign the diving charter each year and abide by its rules (*e.g.*, no feeding, no anchoring, mooring on specific buoys, no beginners)	Robert (2013a)
M10 1999 to present	Reconcile artisanal fishing with the protection of natural habitats, in the context of MUM management (PC)	The fishing charter is more restrictive that national and local regulations (see text)	Boudouresque *et al.* (2004), Cadiou *et al.* (2009), Robert (2013a)
M11 2002	Extinction of *Zostera noltei* and *Cymodocea nodosa* in the lagoon of the *Posidonia oceanica* barrier – reef (PC)	Transplantation of 71 cuttings of *C. nodosa* collected in the Bay of Port-Man, 10 m depth. Successful	Meinesz *et al.* (2005), Goujard *et al.* (2010)
M12 2003: No	Overfishing of *Homarus gammarus, Palinurus elephas* and *Maja squinado*?	No proposal	Noël 2003
M13 2004	Degradation of the *Posidonia oceanica* meadow by anchoring and dead weights (see M2) (PC)	Setting up of ecological mooring (Harmony®) in the Bay of Port-Cros	Francour *et al.* (1997), Hervé Bergère (pers. comm.)
M14 2009 to present	Presence of the invasive macroalga *Caulerpa taxifolia* (PC and PQ)	Manual, sometimes chemical, eradication. Successful (PC)	Cottalorda *et al.* (2011, 2012), Esposito *et al.* (2012), Barcelo *et al.* (2013b)

Table II. Continued.

Dates	Management issue	Responses, results and comments	References
M15 2010	Regression of shallow *Cystoseira* spp. in Provence	Installation of zygote traps (10 cm x 10 cm x 5 cm cement blocks) in littoral rockpools. Failure. See text for comments	Robvieux (2013)
M16 2012 to present	Promoting 'reserve effect' (PQ)	Regulating amateur fishing and leisure boating, according to a zoning system for the marine core area	Barcelo *et al.* (2018)
M17 2015 to present	Promoting 'reserve effect' and artisanal fishery (PQ)	Setting up of a No-Take-Zone (*zone resource*) in the south-eastern part of PQ	Rincé *et al.* (2015), Astruch *et al.* (2016, 2017), Barcelo *et al.* (2018)
M18 2016: No	Overgrazing of *Cystoseira brachycarpa* forests (PC)	Limitation of herbivorous populations (sea urchins *Paracentrotus lividus* and teleost fish *Sarpa salpa*)	Thibaut *et al.* (2016b)
M19 2018-2019	Mass mortality of the fan mussel *Pinna nobilis*	No transplantation of individuals to deeper areas, but physical protection of surviving individuals from anchoring and nets	Catanese *et al.* (2018), Cabanella-Reboredo *et al.* (2019)
M20 2018-2019	Oil spill of the CSL Virginia	Warning of the Scientific Council and the director of the N-PNPC against over-cleaning. See text for comments	Boudouresque *et al.* (2019b, c)

Fig. 2. – An artificial pond, on Porquerolles Island, dug in the early 2000s, possibly to provide an insect resource for bats. Photo © Charles-François Boudouresque.

(Geoffroy, 1806) had been established (Table I, T13). This bat species, which feeds primarily on spiders and flies, is an anthropophilic and cave-dwelling species; it is listed in the Annex II of the Habitats Directive of the European Union (Médard *et al.* 1999, Médard & Muratet 2000, Flaquer *et al.* 2008). The cellar was destroyed in 2002; a tower (Tour du Palmier), with a shelter, was specially designed and built to house the colony of bats (Quekenborn *et al.* 2004). Unfortunately, the bats preferred to settle in the stairwell of an apartment building, which caused nuisance for the inhabitants. Some individuals have taken up residence in a ruined fort, the Galéasson (Clélia Moussay, unpubl. data). It took three years for the colony to 'adopt' the tower. In Port-Cros Island, Stoecklé (2003) suggested fitting out buildings either recent, ancient or abandoned ruins, with shelters and water supply systems purpose-designed to enhance the colonies of bats. However, the Scientific Council firmly rejected these proposals, and the ruined Sardinière farm, when restored (in 2011-2012), was not equipped with features intended to increase bat populations (Table I, T15).

The establishment of artificial nest-cavities (artificial burrows) for two species of shearwater, *Puffinus yelkouan* (Acerbi, 1827) and *Calonectris diomedea* (Scopoli, 1769), also typically resulted from the species-by-species approach to management (Table I, T16). These two species may be locally in decline (Courbin *et al.* 2018), but are not classified as threatened (LC in the IUCN Red list of threatened species). Although natural nest-cavities are not a limiting factor (only 33 % are occupied), 95 artificial nest-cavities were installed in Porquerolles, Port-Cros and Bagaud Islands, between 2003 and 2013, in an attempt to increase the nesting population (Bourgeois & Vidal 2007, Bourgeois *et al.* 2015). Very few of the artificial nests have been occupied (none at Bagaud Island) and the number of fledgling chicks from these nests has been insignificant (LPO PACA *et al.* 2007); Bourgeois *et al.*

(2015) expressed more optimistic conclusions, although upon examination of their data, these can be challenged. Either way, whether it was a failure or a rather questionable success, that does not change the fundamental question: the role of a National Park is neither that of a zoo, nor a facility for breeding a species, even if it was threatened elsewhere (which, moreover, is not the case here).

Hermann's tortoise *Testudo hermanni* (Gmelin, 1789) had thrived at Port-Cros Island until the middle of the 19th century. Its extinction would have been caused by man (Jahandiez 1914). A first reintroduction project was rejected by the Scientific Council of the PCNP, in 1970. A second project was subsequently approved and 46 individuals were released in 1975 at two sites, Vallon de la Solitude and Plateau de la Marma (Table I, T4) (Besson 1975). The reintroduction was done using individuals donated by residents of the Massif des Maures (eastern Provence) (Cheylan 1983). At that time, the complex genetic structure of Hermann's tortoise populations was unknown; in fact, the populations of Provence and Greece belong to distinct haplotypes (subspecies? distinct species? see *e.g.*, Fritz *et al.* 2006 and Nivelle 2017, for discussion); however, the Hermann's tortoises, which were marketed in France and kept by local inhabitants, generally belonged to the oriental (Greek) haplotypes. It is therefore probably oriental haplotypes that have been reintroduced in Port-Cros, rather than the native western haplotype. The re-introduction seems to have been unsuccessful in the long term (Cheylan 1983). However, this example highlights the need to be extremely cautious when reintroducing species. Similarly, in Porquerolles, 40 Hermann's tortoises, abandoned by a resident of Saint-Tropez (Provence), were housed in enclosures and fed for several years (Table I, T11). Obviously, the role of a protected area is not to take over the role of the societies for the prevention of cruelty to animals (such as the *Société Protectrice des Animaux* in France and the Royal Society for the Prevention of Cruelty to Animals in England and Wales).

To prevent anchoring by leisure boats and degradation of the *Posidonia oceanica* meadow (Augier & Boudouresque 1970a, b, Boudouresque *et al.* 1995, Cossu *et al.* 2006, Boudouresque *et al.* 2012, Rouanet *et al.* 2012), 27 concrete deadweight, connected by chains, and mooring buoys, were set up in the Port-Cros Bay in the 1970s (Table II, M3) (Jean-François Magréaud, pers. comm.). In fact, the impact of dead weights and chains was worse than that of anchors (Fig. 3) (Robert 1983, Boudouresque *et al.* 2012, Robert 2013b). The Scientific Council of the PCNP therefore recommended the removal of dead weights and chains, and their replacement by 'ecological moorings' (Harmony® or Harmony-like). However, the removal of the dead weights would have aggravated the damage: it was decided to leave them in place and to remove only the chains, in 1982-1983 (Jean-François Magréaud, pers. comm.). Subsequently, ecological moorings (Harmony®) were set up in 2004-2005 (Jean-François Magréaud and Hervé Bergère, pers. comm.). These moorings, which appear to have been a success, are still in place.

Fig. 3. – Furrow dug in the *matte* of a *Posidonia oceanica* meadow by a mooring chain of a mooring. Anonymous photo.

Fig. 4. – Four cement blocks (10 cm × 10 cm × 5 cm) in a rock pool on Port-Cros Island, with a *Cystoseira crinita* forest. They were intended to trap *Cystoseira* zygotes. Photo © Thierry Thibaut.

The installation of cement blocks in littoral rock pools (Pointe de Malalongue, Pointe du Tuf, Port-Man Bay) was intended to trap zygotes of several *Cystoseira* species (Fucales, Phaeophyceae), such as *C. crinita*, *C. barbata* and *C. foeniculacea* (Robvieux 2013) (Fig. 4; Table I, M15). These species are declining in Provence, French Riviera, French Catalonia and other Mediterranean areas (Thibaut *et al.* 2005, 2015, Blanfuné *et al.* 2016). However, this is not the case at Port-Cros Archipelago (Thibaut *et al.* 2016b). The operation was a failure: no seedling of *Cystoseira* was observed on the blocks after the reproduction period, and the winter storms have washed away or thrown onto the coastal rocks 80 % of the blocks, which were not attached to the substrate (Robvieux 2013). Not only was the operation a failure, but also it is likely that these blocks, during storms and before being washed away, damaged the healthy *Cystoseira* stand. This operation is therefore emblematic of inappropriate actions for the protection of the environment: (i) the role of the PCNP is not to cultivate species, even if they are threatened elsewhere; (ii) very inexpertly designed operations can be counterproductive, and contribute to threats to healthy populations.

The case of the European hedgehog *Erinaceus europaeus* Linnaeus, 1758 in Porquerolles Island deserves special discussion (Table I, T20). The species is native and is widespread in Western Europe. However, it seems to have been originally absent, or became naturally extinct, from a number of small offshore islands, including Porquerolles (Legrand *et al.* 2007). In the latter, it was first observed in 1984 (Cheylan 1984), perhaps deliberately introduced, near the village, and subsequently spread to the whole island. Its diet is diverse, mainly constituted of plants and arthropods, but it can also prey on birds nesting on the ground. Three bird species, reared and released to the wild for hunting purposes, are particularly concerned (Tranchant *et al.* 2002, Legrand *et al.* 2007): (i) the pheasant *Phasianus colchicus* Linnaeus, 1758, native to Asia, was introduced to Europe in the ancient Greece era; (ii) Reeve's pheasant *Syrmaticus reevesii* (J. E. Gray, 1829), native to China, was introduced in Europe in the late 19th century; (iii) the red-legged partridge *Alectoris rufa* Linnaeus, 1758, is native to Europe (Peterson *et al.* 1993, Pascal *et al.* 2006). For three other bird species, the threats are more putative (Legrand *et al.* 2007): the Mediterranean shearwater *Puffinus yelkouan*, Cory's shearwater *Calonectris diomedea*, and the nightjar *Caprimulgus europaeus*. Finally, predation by hedgehog may putatively concern the corn bunting *Emberiza calandra* Linnaeus, 1758; however, this species is far from being threatened in Europe (Legrand *et al.* 2007). It is worth noting that the European hedgehog *Erinaceus europaeus* is a protected species in Europe. All in all, the destruction of a protected species (the hedgehog), native to Europe and the absence of which from Porquerolles could be relatively recent, in order to protect the hunting of non-native species, as pointed out by Legrand *et al.* (2007), can hardly be considered as a priority management response from N-PCNP.

The wild boar *Sus scrofa* Linnaeus, 1758 is native to Eurasia, including Europe. Its abundance in southern France is currently on the increase (Cheylan & Geoffroy 2020). Several causes can account for its proliferation; this is mainly due to the increase in the surface area of forest and maquis due to the severe reduction of agro-silvopastoral practices in Mediterranean France, but also the elimination of predation by the wolf *Canis lupus* Linnaeus, 1758, feeding in winter by hunters (in French: *agrainage*), climate warming that reduces the natural winter mortality and increases the breeding potential, etc. The wild boar is a good swimmer, able to naturally reach not too distant offshore islands. From the late 19th century, it had been absent from Port-Cros Island, but it perhaps (probably?) thrived in the island during the long time periods when Port-Cros was not permanently occupied by man. It was first sighted on Port-Cros Island in 2008; since 2015, a permanent population of wild boar has naturally colonized Port-Cros Island and Porquerolles Island (Table I, T26). In Port-Cros, it has been blamed for digging up the soil, in search of food, and as a result degrading some populations of rare and protected plants, notably geophytes, and of the rare and locally endangered Sardinian frog (*Discoglossus sardus*) located in some temporary ponds. The so-called 'negative' effects of wild boar must be compared with the 'positive' effects of wild boar, which are part of an ecosystem approach: (i) aeration of the soil; (ii) the return of coprophagous insects, eliminated by chemical vermifuges used for domestic livestock; boar droppings have become a real refuge for many of these endangered insects everywhere (Philippe Ponel, unpubl. data). These coprophagous insects are also widely exploited by bats; (iii) the return of necrophagous insects, for example *Necrodes litoralis* Linnaeus, 1758, specialized in large corpses. However, the permanent inhabitants of the island were disturbed by its unfamiliar presence and possible danger for people, including tourists. The Scientific Council of the N-PCNP was of the opinion that the role of a National Park is not to combat a natural process, and its impact on associated species (even if they are rare and protected plants), which would be in accordance with an ecosystem-based approach; however, taking into consideration the fact that the wild boar is not a threatened species, and that the well-being of inhabitants is an important feature of the N-PCNP governance, it was not opposed to the control of the wild boar, *via* trapping and hunting. This attitude is consistent with the principles for ethical wildlife control (Sellier 2015, Dubois *et al.* 2017). About 200 individuals have been killed in 2018 and 2019 (Hervé Bergère, pers. comm.).

The attempt to reintroduce the giant limpet *Patella ferruginea* Gmelin, 1791 to Port-Cros, from individuals from northern Corsica, can be *a priori* considered a good idea (Laborel-Deguen 1988, Laborel & Laborel-Deguen

Fig. 5. – The setting up of artificial reefs. La Palud, Port-Cros Island. Photo Philippe Robert © Port-Cros National Park (PCNP). Courtesy of the PCNP.

1991, Laborel-Deguen & Laborel 1991a) (Table II, M6). The species, which is long-lived and can reach a diameter of 11 cm, is a western Mediterranean endemic which has been present in the whole western basin of the Mediterranean. It dwells in the midlittoral zone and has been harvested by humans since the Neolithic period (Espinosa & Ozawa 2006, Colonese *et al.* 2011). It has become extinct in most of its original range, *e.g.*, in Provence and French Riviera (Laborel-Deguen & Laborel 1991b). The attempt does not seem to have been successful in the long term; the survival rate was 57 % after one day, 27 % after one year, 12 % after two years (Laborel-Deguen 1988, Laborel-Deguen & Laborel 1991a). The few very isolated individuals that have been observed in the Port-Cros archipelago, far from the Rascas islet, probably do not come from the small-reintroduced population, but from larvae from Corsica drifting with the currents (Meinesz *et al.* 2001).

In 1989, 278 cuttings and seedlings of the seagrass *Posidonia oceanica* originating from 12 populations in different parts of the Mediterranean (Algeria, France, Greece, Italy, Spain and Turkey) were transplanted to La Palud Bay (Port-Cros Island), at 11 m depth (Table II, M7). The goal was solely scientific: to check whether or not the morphological differences between these strains would be conserved when cultivated together in the same area and the same habitat (Meinesz *et al.* 1993). The choice of the Port-Cros National Park was purely technical: the ban on fishing in La Palud Bay (Astruch *et al.*, 2018) guaranteed that this underwater ‘botanical garden’ would not be disturbed by fishing gear. Today, at a time when biological invasions, including gene pollution, are considered as one of the most worrying environmental issues (Carlton & Hodder 1995, Schmitz & Simberloff 1997, Boudouresque *et al.* 2017a), it is difficult to understand why such a scientific experiment was authorized within a national park.

Is the goal of a national park the artificial increase in fish stocks, via artificial reefs? (Fig. 5, Table II, M5). The shutdown of sewage, and therefore the increase in domestic pollution, would have the same effect (see *e.g.*, Ourgaud *et al.* 2015). In fact, the installation of these artificial reefs corresponded to a demand from artisanal fishers, and it facilitated the subsequent establishment of the fishing charter (Robert 2013a, b). In addition, the goal was solely experimental (Charbonnel *et al.* 2001). The outcome could therefore be positive.

MANAGEMENT ACTIONS THAT CAN HAVE RELEVANCE IN THE CONTEXT OF ECOSYSTEM-BASED MANAGEMENT

The Tyrrhenian painted frog *Discoglossus sardus* Tschudi *in* Otth, 1837 is a species occurring in Corsica, Sardinia and on several small islands: Port-Cros, Le Levant (Provence), Giglio, Montecristo and Monte Argentano (Tuscany) (Knoepffler 1962). The species is not threatened in Corsica and Sardinia: it is listed by the IUCN as being of ‘Least Concern’ (LC) (Andreone *et al.* 2009). However, isolated populations on small islands can be more vulnerable and they could represent distinct conservation units. On Port-Cros Island, a small dam (30 m long, 6 m high) was built in the Vallon de la Solitude to create a water reservoir intended to irrigate crops, now abandoned, located downstream. This reservoir is used by *D. sardus* for its reproduction. The mosquitofish *Gambusia holbrooki* Girard, 1859, deliberately introduced in 1992, threatened the eggs and larvae of *D. sardus*; it was therefore eradicated in 2001 by poisoning (rotenone) (Lim & Dauba 2001, Duguet *et al.* 2019) (Table I, T12). The eradication was successful: it has never reappeared there (Hervé Bergère, pers. comm.).

The eradication of feral cats *Felis catus* Linnaeus, 1758 in Port-Cros (Table I, T17), predators of introduced rats together with the native Mediterranean shearwater *Puffinus yelkouan*, has been a success, insofar as it made it possible to improve the numbers and the reproductive success of the Mediterranean shearwater (Tranchant *et al.* 2002, Legrand *et al.* 2008). It also gave rise to a process of reflection on the functioning of the ecosystem and the interactions between cats, black rats *Rattus rattus* Linnaeus, 1758 (main prey of cats, with shearwater) and shearwater, a possible prey of rats (Bourgeois *et al.* 2005, Bergère 2009, Zarzoso-Lacoste *et al.* 2009, Médail *et al.* 2013). The management of the cat-rat-shearwater ‘triangle’ can be described as typical of an ecosystem-based management approach, and is consistent with ethical principles (see Dubois *et al.*, 2017).

The co-eradication of the ice plant *Carpobrotus* spp. and of the black rat *Rattus rattus* on Bagaud Island is also part of an ecosystem-based approach, since there are mutualistic interactions between ice plants and rats

(Bourgeois *et al.* 2005); in addition, all the compartments of the ecosystem were studied, before and after eradication (Table I, T23). Such a multidisciplinary approach has never been implemented in the context of the eradication of invasive species (Tranchant *et al.* 2002, Médail *et al.* 2013, Berville *et al.* 2015, Braschi *et al*. 2017, Buisson *et al.* 2018). Of course, the risk that co-evolution between black rats and native species, such as the European leaf-toed gecko *Euleptes europaea* (Gené, 1839), would have resulted in an adaptive equilibrium, putatively disrupted by the eradication, especially in the case of reinvasion by a different population of black rats, should be carefully considered (Delaugerre *et al.* 2019).

The control of invasive species also occurs in the AA of the N-PCNP: *e.g.*, (i) the destruction of the pond slider *Trachemys scripta* (Thunberg *in* Schoepff, 1792), a semiaquatic turtle native to southeastern United States and northern Mexico, which threatens the native European pond turtle *Emys orbicularis* Linnaeus, 1758 in the Vieux Salins saltmarsh of Hyères (Lascève 2014, Perrot *et al.* 2016); (ii) the uprooting of *Caulerpa taxifolia* at Le Pradet (eastern Provence) (Pironneau *et al.* 2014, Pironneau & Ringwald 2015, Barcelo *et al.* 2016).

There has been a myth, both among the public and among the first scientists working on this insular area, that the forest on the island of Port-Cros was an ‘original and pristine forest’, in a way a primary forest. In fact, the island has undergone strong human impact throughout its history, and especially over the last centuries: cultivation, massive deforestation to fuel a caustic soda plant or to export charcoal, overgrazing, etc. The holm oak *Quercus ilex* Linnaeus forest is therefore relatively recent and terrestrial ecosystems, their flora and fauna, are in a dynamic of rapid evolution (Médail *et al.* 2013). In contrast to the natural evolution of ecosystems over time, to ‘freeze’ landscapes in a state that is only a snapshot, fixed by the fragile memory of the observer, is a temptation that characterizes many environmentalists and even scientists. The PCNP, except perhaps at its very beginnings (Table I, T5, T8), did not fall into this trap, based on an archaic, naive and even completely wrong notion of biodiversity (see Boudouresque 2014 for the biodiversity concept). ‘Opening’ the habitats, in order to artificially increase the species richness, has not been the doctrine of the PCNP, N-PCNP and its scientific Council (see below for discussion). The best solution is probably to include natural disturbances as promoter of the patch dynamic of these terrestrial ecosystems (Médail *et al.* 2013).

The ecosystem constituted by the dune, the beach and the overlying layer of *Posidonia oceanica* dead leaves (*banquette*) (hereafter dune-beach-*banquette* ecosystem) harbors a unique fauna and flora and has a very high heritage value, in addition to providing valuable ecosystem services (Médail *et al.* 2013, Serantoni 2015, Boudouresque *et al.* 2017b). For the supposed pleasure of tourists, *banquettes* are removed from most Mediterranean beaches, treated as a waste and thrown on the garbage dump. This removal has dire ecological and economic consequences: the destruction of the ecosystem, erosion of the beaches, catastrophic attempts to restore them *via* riprap or sand replenishment; as the fate of beached dead leaves is to return sooner or later to the sea, and to feed (organic carbon, nutrients) coastal ecosystems, their destruction deprives coastal areas of fish, available to artisanal fishers (~35 kg wet mass of fish per metric ton of *banquette*) (Boudouresque *et al.* 2016). In addition, it seems that this ‘tourist demand’ is rather something dreamed up by tour operators and ill-informed mayors: even when uninformed, and massively when properly informed, tourists do not ask for the removal of the *banquette* (Boudouresque *et al.* 2017b). The PCNP strategy has been, in the core areas of Port-Cros and Porquerolles Archipelagos, to leave the *banquette* and the natural driftwood, to manually clean the beaches of human-generated waste matter (*e.g.*, plastic and metallic debris, processed wood) and to inform the general public about the ecological and economic issues (Fig. 6, Table I, T21) (Serantoni 2015). This concept of ‘ecological beach’, born in Port-Cros, is currently gaining ground in the Mediterranean (Borrello *et al.* 2019, Astier *et al.* 2020; Rotini *et al.* 2020).

Fig. 6. – Natural driftwood, on the beach of L’Oustaou de Diéu (Porquerolles Island), after an episode of extreme flooding (Spring 2012), left in place. Photo © Charles-François Boudouresque.

The management of the artisanal fishery around the Port-Cros Archipelago can be referred to as an ecosystem-based approach (Table II, M10). For most environmentalists, many stakeholders and some managers, the paradigm of a Marine Protected Area (MPA) is the banning of all human activities, mainly artisanal fishery (No-Take Zones, NTZs) (Boudouresque *et al.* 2005b). In fact, how ‘natural’ NTZs really are can be challenged. In the Mediterranean Sea, some major top predators, such as the monk seal *Monachus monachus* (Hermann, 1779), and sharks are locally or functionally extinct, respectively. Under these conditions, considering that the catch by an extinct population of monk seal was of the same order as

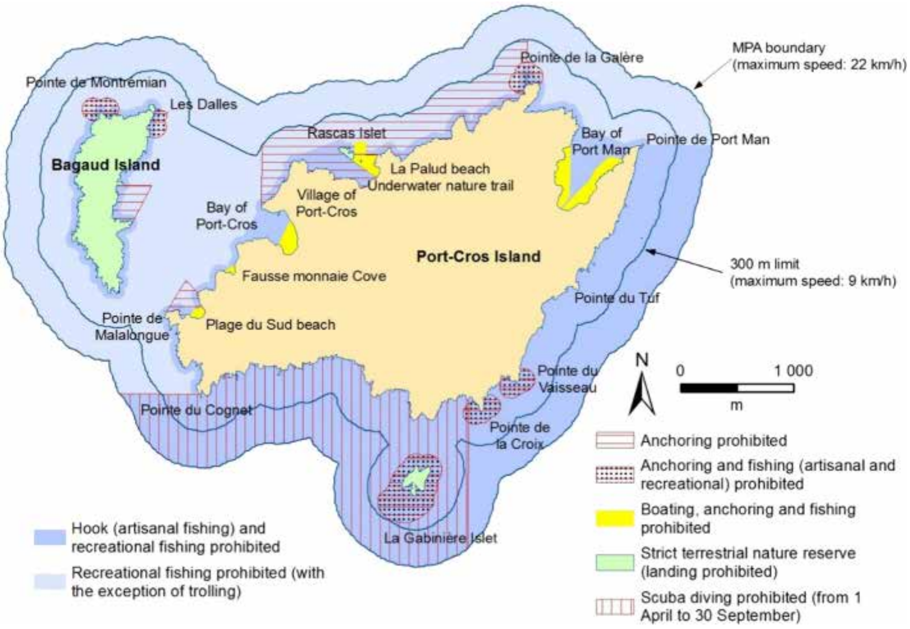

Fig. 7. – Artisanal and recreational fishing regulations within different areas of the Port-Cros Archipelago MPA. Areas where anchoring, artisanal and recreational fishing are prohibited are reserved for diving. 2014 edition.

the current catch by fishers, the complete ban on artisanal fishing (small-scale fishing) could generate an artificial deficit in top predators (Boudouresque *et al.* 2004, Ferretti *et al.* 2008, Astruch *et al.* 2018, Boudouresque *et al.* 2019a). In contrast to the NTZ management, the management of the marine area of the Port-Cros Archipelago MPA can clearly be assigned to the MUM type (Multi-Use Management). It is characterized by complex zoning of the MPA (Fig. 7; see discussion for the complexity of this zoning), based on the uses, conservation goals for the habitats and ecosystems and clearly displayed priorities (artisanal fishing rather than recreational fishing); step by step, recreational fishing has been banned in the whole of the MPA, which favors artisanal fishers and helps them to bear the constraints of the fishing charter (see below) (Le Diréach *et al.* 2018, Boudouresque *et al.* 2019a). To be authorized to fish inside the Port-Cros Archipelago, each fisher has to sign the fishing charter each year and to abide by its rules. The charter is more restrictive than national regulations and the regulations of the local *prud'hommie des pêcheurs* (fishers' guild), *e.g.*, maximum length of the vessel, mesh size, length of fishing nets, maximum soak times, number of hooks and traps. The charter is updated annually, based on scientific monitoring and in consultation with the fishers (Boudouresque *et al.* 2004, Cadiou *et al.* 2009, Robert 2013a, b). It is important to note that these regulations, together with the general regulations of the PCNP and of the N-PCNP (*e.g.*, ban on trawling, spear fishing, angling) are strictly enforced. The MUM strategy, as opposed to the NTZ strategy promoted by conservationist NGOs (Non-Governmental Organizations), is gaining ground today in the world of conservation (see *e.g.*, Burbano *et al.* 2020). Of course, the MUM strategy can include NTZs in the framework of the zoning, such as the '*zone ressource*' of the Porquerolles Archipelago (Rincé *et al.* 2015, Astruch *et al.* 2016, 2017, Barcelo *et al.* 2018) and the six areas of Port-Cros Island surrounding diving spots where artisanal and recreational fishing are banned, so constituting actual NTZs (Fig. 7, Table II, M17).

A mass mortality event has since 2016 been affecting the emblematic fan mussel *Pinna nobilis* Linnaeus, 1758, throughout the Mediterranean (Table II, M19) (Vásquez-Luis *et al.* 2017, Cabanellas-Reboredo *et al.* 2019). It is due to a unicellular haplosporidan parasite, *Haplosporidium pinnae*, probably introduced from the northwest Pacific (Catanese *et al.* 2018). On the basis of a hypothetical assumption that resistance to the parasite would be better at low temperature and therefore at depth, some researchers have proposed transplanting individuals to deep water areas. Some MPAs have viewed this proposal favorably (Parc National des Calanques, western Provence) or even implemented it (Observatoire Marin du Golfe de Saint-Tropez, eastern Provence). The Scientific Council of the PCNP opposed this proposal, which had no scientific basis; in addition, transplantation would increase mortality, by stress or by predation in the new area (De Gaulejac & Vicente 1990, San Martín 1995, Reveret *et al.* 2015; but see Bakran-Petricioli *et al.* 2019). On the other hand, the PCNP has implemented *in situ* physical protection, by means of exclusion zones from anchoring and fishing nets, of surviving individuals, probably resistant to the parasite, and likely to found new populations.

Over-frequentation is increasingly an issue worldwide in protected areas. The impacts of over-frequentation not only concern landscapes, seascapes, emblematic species and the functioning of ecosystems, but also the well-being of inhabitants, visitors and park officials, the overall safety (risk of fire, evacuation of injured visitors, etc.) and the effectiveness of the surveillance by officials of protected areas of environmental offences (Table I, T22) (Bergère & Le Berre 2011). The definition of a carrying capacity is however a complex issue, and the N-PCNP has carefully addressed the question (Wagar 1964, 1974, Deldrève & Michel 2019). As pointed out by Lindberg *et al.* (1997), the question is not '*How many is too many?*' but '*What are the desired conditions?*' The extent to which the local police can, or should, play a role in the protection of natural areas from damage caused by a large number of visitors, in the context of French legislation, has been explored by Jolivet (2018); this author suggested improvements to the present legislation[2]. Improvements in the management of tourism by the local 'business ecosystem' (Porquerolles), or its restructuration, have also been suggested (Van der Yeught 2018).

On October 7th, 2018, the ro-ro ferry *Ulysse* collided with the *CSL Virginia*, 28 km northwest of the Cap Corse (Corsica) (Table II, M20). A maximum of 600 m³ (the capacity of the fuel tanks, if full) of fuel leaked out from the tanks of the *Virginia*. Most of this oil (90 %) was recovered by spill response vessels, according to the French Navy authority (Boudouresque *et al.* 2019b, c). Evaporation further removed about 30 % of the remaining oil (Philippe Cuny, pers. comm.). This oil spill (40 t?) is therefore negligible when compared with historical hazards which have hit European coasts, *e.g.*, *Amoco Cadiz* (1978, Brittany, 223,000 t), *Haven* (1991, Genoa, Italy, 20,000 t), *Erika* (1999, Bay of Biscay, 19,000 t), and *Prestige* (2002, Galicia, 60,000 t) (Marchand *et al.* 1979, Cognetti & Cognetti 1992, Le Moigne & Laubier 2004). Winds and currents pushed remaining oil slicks toward the coasts of Provence (Boudouresque *et al.* 2019b, c). Part of the areas fouled belongs to the core area (Archipelagos of Port-Cros and Porquerolles) or to the transitional area (*Aire Maritime Adjacente*) of the N-PCNP. The Scientific Council of the N-PCNP and its director issued (23rd October 2018) a warning against the risks, for the natural environment and the ecosystems, of an over-reaction and the use of heavy and intensive cleaning methods on the rocks and beaches; they recommended *e.g.*, (i) no use of chemicals (*e.g.*, dispersants, detergents); (ii) no hot water high pressure washing (HWHPW), with the exception of rocky areas accessible to pedestrians, close to the beaches; (iii) no cleaning in areas of high ecological sensitivity such as *Lithophyllum byssoides* (Lamarck) Foslie rims; (iv) on beaches, taking care to remove as little sand as possible and leave as much wood as possible on the spot; (v) *Posidonia oceanica banquettes* of dead leaves, on beaches and rocks, should be preserved, with manual removal limited to surface layers exhibiting oil. The increase of ecological damage due to cleaning has been relatively well documented (*e.g.*, Broman *et al.* 1983, Poncet & Le Bail 2001, De la Huz *et al.* 2005, Jézéquel & Poncet 2011). Unfortunately, the French authorities (*Préfet du Département du Var*) followed the recommendations of the ship's insurers and entrusted to a private company, internationally recognized for its expertise, the task of removing all traces of oil. While the territory of the core area of the PCNP was relatively spared from disproportionate cleaning, thanks to the intervention of the Scientific Council and the decisions of the PCNP's director, this was not the case for the rest of the area, which was intensively and disproportionately cleaned up, both in areas accessible to tourists in summer and in inaccessible areas. Rocks were stripped of the whole of the mediolittoral and supralittoral ecosystem (Cyanobacteria, lichens, *Littorina*, *Euraphia*, *Chthamalus*, *Patella*, etc.) *via* HWHPW, *P. oceanica banquettes* were totally removed, regardless of whether or not oil patches were present on the upper layer of dead leaves, etc. These *banquettes* are of paramount importance for coastal ecosystems and provide a wide range of ecosystem services (Boudouresque *et al.* 2016, 2017b, Rotini *et al.* 2020). The result of this inappropriate over-cleaning is that a very minor oil spill, occurring 9 months before the next tourist season, was transformed by the polluters themselves (*via* the insurers), by carrying out a disproportionately large-scale cleanup operation, into an ecological disaster, for which natural ecosystem restoration will take at least 10 years (Boudouresque *et al.* 2019b, c). Obviously, the N-PCNP strategy can be referred to as ecosystem-based.

The legal policy implemented by the N-PCNP is also an illustration of an ecosystem-based approach. First of all, it should be recalled here that the law is not limited to regulation. For example, the legal policy is also concerned with claims for compensation for ecological damage (Martin 2020). In this regard, for some time now, the National Park has ceased to base its claims on the number of individuals of a species that have been destroyed or disturbed, but instead focuses on the resulting loss of ecosystem services on the one hand and, on the other hand, on the expenditure needed to support natural recovery of damaged ecosystems.

DISCUSSION AND CONCLUSIONS

The species-by-species approach to management is a convenient and recurrently tempting way to try to protect nature. It is convenient, because it is easily understood by the general public, because it does not require the study of the cascading consequences of the action on a given species, because it is far easier to implement than an eco-

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[2] These improvements are underway. The French Senate unanimously voted to amend art.L.2213-4 of the General Code of Local Authorities, in order to authorize the mayor of the municipality to prohibit or regulate access to certain areas if such access is likely to harm the protection of the environment or the character of the site. The National Assembly must in turn soon examine this proposal.

system-based approach, and, last but not least, because it is strongly supported by 'taxonomic lobbies'. It is recurrent, since it dates back to the 19th century and continues to be the dominant practice of international NGOs and administrations, including in France the Ministry of the Environment and, to some extent, its official agency, the OFB *(Office Français de la Biodiversité*), although the EBA is currently gaining ground within the OFB. The requirements of the French Ministry of the Environment have sometimes borne more resemblance to accountancy practices than to scientific ones: how many species (even casual or observed once) are there in your protected area? Is this number on the increase when compared with that of the previous year? Is it higher than in the adjacent non-protected area? (implying: if so, your management is good). Of course, this criticism is something of a caricature: officially, 'and habitats' is always added to the essential inventory of species; but it sometimes seems rather like a kind of ecologically correct polite formula.

This accountants' concept of environmental management is often associated with a simplistic, and even erroneous, vision of biodiversity, where biodiversity is viewed as the number of species. The higher the number of species, the better the status or the health of a habitat is considered to be. Disturbances are wrongly thought to reduce the number of species, while in reality they often increase it (see *e.g.*, Ourgaud *et al.* 2015). In most cases, the highest number of species is reported for intermediate levels of disturbance (IDH – Intermediate Disturbance Hypothesis and DEM – Dynamic Equilibrium Model) (*e.g.*, Lubchenco & Menge 1978, Huston 1979, Valdivia *et al.* 2005, Svensson *et al.* 2009). In fact, biodiversity is a complex multidimensional concept, defined by at least five scales (evolutionary, functional, organizational, spatial and heterogeneity scales) and more than a hundred metrics. These metrics can give apparently contrasting responses, when they are in fact complementary (Sala & Knowlton 2006, Boudouresque 2014, Boudouresque *et al.* 2017a).

The species-by-species approach does not take into account natural fluctuations in the numbers of a species within an ecosystem, as a function of predator-prey cycles, parasite-host cycles, natural inter-annual fluctuations of climate, or threshold effects of physical-chemical and biological parameters that make a population or an ecosystem shift from one state to another (regime shift or phase shift; see *e.g.*, Cury & Shannon 2004, Boudouresque *et al.* 2005a, Litzow & Cianelli 2007). Control of supposed harmful species, in order to increase prey populations, often backfires, with a further unexpected decrease in prey populations (Doak *et al.* 2008). We must not be to hasty to intervene as soon as a species declines or another species proliferates; ecosystems are in constant evolution, contrary to what the old climax concept suggested. This is particularly true for the terrestrial ecosystems of Port-Cros Island, which are slowly recovering after the very strong human impact of the 19th century, that is to say almost yesterday (Médail *et al.* 2013). This is also true for marine ecosystems, even if it seems at first glance less apparent: effluent from a caustic soda plant, relocated from Marseille because it was too polluting, overfishing by artisanal fishers, extirpation of large and emblematic fishes by spear fishers, the extermination by fishers of a top-predator, the monk seal *Monachus monachus*, supposed to be a fish thief, marked the 19th and 20th centuries, before the creation of the PCNP (Marchessaux 1989a, b). The steady but slow recovery of emblematic fish populations, such as the dusky grouper *Epinephelus marginatus* and the brown meagre *Sciaena umbra* Linnaeus, 1758, illustrate a recovery process still in progress (Harmelin & Marinopoulos 1993, Harmelin & Ruitton 2007, Harmelin *et al.* 2010, Harmelin 2013).

Fig. 8. – In the background, the 'bats tower' (Tour du Palmier) in the village of Porquerolles. The half-moon opening at the top of the tower is the entrance to the shelter specially built to house a colony of Geoffroy's bat *Myotis emarginatus*. Photo © Charles-François Boudouresque.

Each 'taxonomic lobby' wishes to favor its group of taxa, to increase the abundance of individuals and its species richness. There is certain logic in increasing the numbers of a 'threatened' species, which also benefits from broad support from the general public. However, 'threatened' species are not always actually threatened: they can even be proliferating thanks to humans; they may be fortunate to belong to an emblematic and lovable taxon. For example, Geoffroy's bat *Myotis emarginatus*, for which a tower with artificial shelter ('Tour du Palmier') was specially built in the village of Porquerolles (Fig. 8, Table I, T13), is far from being a threatened species: its abundance is stable, close to a natural state, and it is classified as LC (Least Concern) in the IUCN Red List (Hutson *et al.* 2007). Taxonomic lobbies also do not consider the possible impact of the increase of their beloved taxon on other taxa. When 'bat lovers' asked the PCNP to promote the abundance of bats through shelters and water supply systems, they were not asked about the effect of their proliferation on other species using the same food

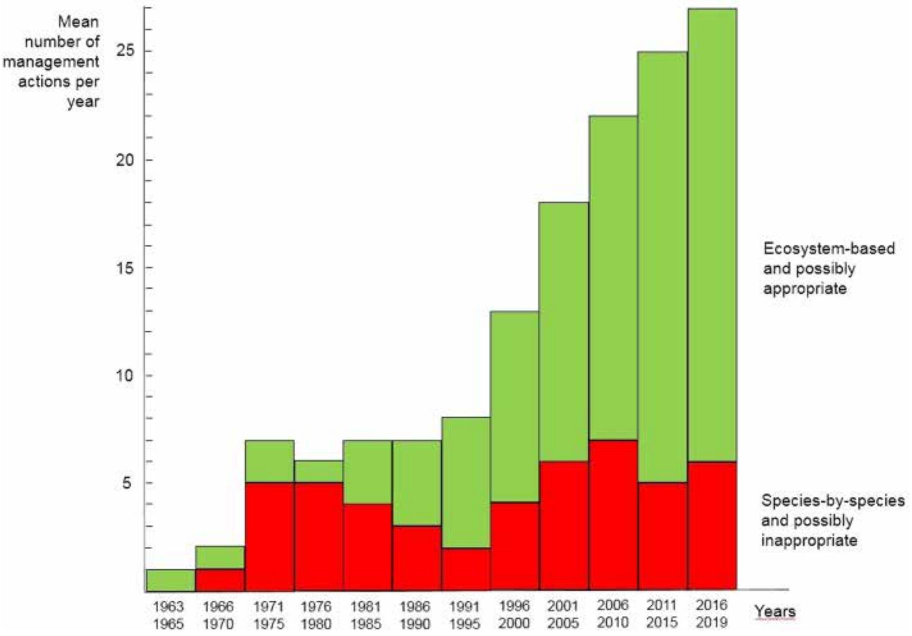

Fig. 9. – Mean number of management actions, species-by-species and possibly inappropriate and ecosystem-based and possibly appropriate, over time. Data from Tables I and II. Many management actions, both appropriate (*e.g.*, T21 and M1) and inappropriate (*e.g.*, T16 and M7), are long-lasting and therefore appear over several time periods.

resource, nocturnal insects (*e.g.*, the Tyrrhenian painted frog *Discoglossus sardus*, shrew, the European leaf-toed gecko *Euleptes europaea*, birds, etc.) (Table I, T15). It is difficult to explain to the general public, and to journalists who have almost always a literary rather than scientific culture, that national parks are neither botanical gardens nor zoos, and that a natural ecosystem can be poor in species. The general public and journalists are often confused and incredulous when they are told that the most effective way to increase the number of species is to open paths and agricultural plots in the forest, to create garbage dumps and to spread fertilizer (see *e.g.*, Landrieu & Gilg 2010, Boudouresque 2014).

In an admirable outburst of anger against taxonomic lobbies and their simplistic view of biodiversity, Landrieu & Gilg (2010) wrote: '*To make us believe that we must cut down forests to open the environment to more species of birds, flowers and butterflies and that, by doing so, we are helping to increase biodiversity, there no, I do not agree! If we let ourselves be guided by the sole objective of the maximum number of species, which leads to increasing the 'species richness' of a site and not its 'biodiversity', the ultimate destiny of the manager could be the maintenance of botanical gardens and zoos! When the environment is naturally forest, it is forest species that constitute biodiversity. Our role is to improve if necessary the functionality of this environment, its 'state of conservation'. It's the ecosystem that decides its biodiversity, based on ecological potential, resources, colonization, dispersal of species. We must beware of 'overselling' biodiversity, by trying, for example, to change a forest environment that is naturally 'poor' in species, into an environment that is artificially rich*' (translated from French by the authors) (see also Pavé 2019). It is worth noting that the so-called species poverty of the 'closed' natural forest environments is an idea propagated by the lobby of flowering plant taxonomists; for insect specialists, the closed forest environments exhibit on the contrary an incredible species richness: saprophagous, saproxylophagous, phytophagous, predators of litter, old wood, old bark, mosses, tree cavities, etc.

Supporters of the species-by-species approach sometimes claim that they are also thinking at ecosystem level, because they consider the habitat of the species. Habitat is generally defined on a descriptive basis, the composition in species, phytosociological in terrestrial environment, fauna and flora in marine environment (*e.g.*, Molinier 1960, Pérès & Picard 1964, Corine 2020). So it is a kind of vicious circle. Of course, habitat is not the ecosystem; habitat and ecosystem are completely different concepts; the habitat is a descriptive concept (physical-chemical characteristics, lists of species), while the ecosystem is a functional concept (the interactions of species between them and with the environment).

Doing nothing is often the best management response, although sometimes difficult to explain to the public at large and policy makers avid for visible actions (Pont 2003, Schnitzler *et al.* 2008). Unfortunately, national authorities, in particular in France (Ministry of the Environment, *Agence de l'Eau*), often favor visible actions (artificial reefs, transplantation, artificial restoration, etc.), excessively expensive, generally ineffective and even seriously destructive of the natural environment, while the use of the budgets for the reduction of impact would have been one or two orders of magnitude more effective.

With the exception of an initial 'teething phase', and of the Island of Porquerolles before it officially became part of the core of the Port-Cros National Park, the PCNP, the N-PCNP and its Scientific Council often avoided the trap of interventionism. Step by step, the National Park has undoubtedly moved away from a species-by-species culture towards an ecosystem-based approach (EBA) (Fig. 9). Possible biases are (i) that ancient inappropriate actions (1963-1985) could have been omitted and (ii) that the effects of ancient inappropriate actions (*e.g.*, the 'bat tower' and the artificial reef, not removed) have persisted to this day; the trend towards an ecosystem-based approach is therefore probably more marked. However, while it is easy to criticize a species-by-species approach to management, with its contradictions and with its absurd, caricatured, sometimes grotesque errors, it is much more difficult to apply an EBA to management. Obviously, it is far easier to describe the ups and downs of a species than to understand the terrifying complexity of the functioning of an ecosystem. In addition, understanding the functioning of an ecosystem requires a multi-disciplinary team, therefore more funding, longer time series, while being handicapped by a lesser understanding on the part of the general public and managers. Finally, if the inappropriate actions of the past are easy to criticize in light of today's ecology and current concepts of nature conservation, it should be recognized that the evidence of today will probably appear ridiculous in 50 years. If our actions or our inaction lead to the disappearance of a species, and if our successors have maintained the cult of the rare and patrimonial species, they will severely criticize our errors of appreciation and management.

The complexity of the marine zoning of the Port-Cros archipelago MPA is a reflection of and the result of the MUM doctrine and almost 60 years of progress in management thinking. Although it has proven to be quite effective, in terms of governance (Sellier 2015, Barcelo *et al.* 2016, 2018, Deldrève & Michel 2019), the protection of ecosystems (Personnic *et al.* 2014, Thibaut *et al.* 2017; but see Astruch *et al.* 2012) and the maintenance of sustainable human activities (artisanal fishing, diving, pleasure boating) (Boudouresque *et al.* 2004, Cadiou *et al.* 2009, Robert 2013a, Le Diréach *et al.* 2018), its complexity makes it difficult for users to read and makes its implementation by the park guards difficult and time consuming.

International literature on the issue is often produced by theorists with no hands-on experience of the reality and governance of a protected area. It is also produced by tricksters who 'sell' paper parks (no reference! It would be inappropriate). It is important here to emphasize that the PCNP and the N-PCNP are among the ~10 % of effective parks (Meinesz & Blanfuné 2015), where the legislation is more or less correctly implemented and respected, taking into account the very many constraints imposed on the manager and the difficulty of convincing the judicial institutions of the importance of protection issues.

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ECOSYSTEM-BASED APPROACH AND RESTORATION OR COMPENSATION FOR ECOLOGICAL DAMAGE

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ECOSYSTEM APPROACH
ECOLOGICAL DAMAGE
RESTORATION
COMPENSATION
IN-KIND REPARATION
METHODS OF ASSESSMENT
DAMAGES

ABSTRACT. – The Law of August 8, 2016 for "the Recovery of biodiversity, nature and landscapes" introduced the principle of compensation for ecological damage into French civil liability law. The definition of ecological damage given by the Law refers to the work of the Millennium Ecosystem Assessment by distinguishing the ecosystem elements and their functions and services. The purpose of this article is to investigate whether the different ways of repairing ecological damage are in line with an ecosystem-based approach. For a very long time, legal doctrine has argued that reparation for ecological damage should be made "in kind". But restoration to return to the *statu quo ante* is not always possible and can sometimes prove dangerous for the ecosystems concerned. It appears that the most appropriate in-kind repair often consists simply in recreating the conditions that will allow new processes and new functions to be expressed. Where reparation in kind is impossible, to what extent are the various methods of assessment currently used by the courts compatible with an ecosystem-based approach? Finally, what are the recommendations that the scientific community can make to the judicial authorities to avoid mistakes being made on this point?

INTRODUCTION

Law n° 2016-1087 of 8 August 2016 on "the Recovery of biodiversity, nature and landscapes" (JORF, n° 0184, 9 August 2016) introduced for the first time in French Law the principle of compensation for ecological damage by providing that "*Any person liable for ecological damage shall be required to make reparation for it*". Until then, the only reparable damage was that which was caused to a human person or to property belonging to a human person. In order to demonstrate that this measure was a new step in the evolution of the classical law of civil liability, the legislator decided to introduce this principle into Article 1246 of the Civil Code (Taylor 2018, Martin 2017, Neyret 2017).

Obviously, the first question raised by this law is how ecological damage is to be defined. The answer to that question is given in the subsequent article (Art. 1247), which states that ecological damage is "*a non-negligible impairment of the elements or functions of ecosystems or of the collective benefits derived by man from the environment*". It should be noted that this definition refers very directly to the work of the Millennium Ecosystem Assessment (MEA 2003) by distinguishing the ecosystem elements and their functions and services (the latter being here referred to as "collective benefits derived by man from the environment").

It should be borne in mind that, prior to the adoption of this text, the courts had already repeatedly condemned those liable for ecological damage to repair it, but they did so without a solid legal basis and, above all, prior to the adoption of any clear definition of ecological damage (Van Lang 2008, Neyret 2010, Neyret & Martin 2012, Jegouzo 2013, Martin 2014, 2015a).

But it is one thing to define ecological damage, and quite another to determine how reparations are to be made for it. On this point again, the 2016 Act provides useful clarification. It stipulates that ecological damage must be repaired "*first and foremost in kind*", but it also provides that if reparation in kind is impossible, for any legal or factual reason, the judge may order the person liable to pay damages, which must be "*allocated to the repair of the environment*" (Article 1249 of the Civil Code).

The purpose of this article is precisely to deal with the reparation for ecological damage, both before and after the adoption of the law, in order to investigate whether the modalities of implementation of such reparation are in line with an ecosystem-based approach.

The first question is relative to reparation "in kind". Is reparation for ecological damage "in kind" always compatible with an ecosystem-based approach? Is there not a risk that it may open the way to manipulations that are dangerous for ecosystems?

Where reparation in kind is impossible, the text provides for the awarding of damages. In this case, an assessment of the ecological damage is required. What are the various methods of assessment currently used by the courts and to what extent are they incompatible or compatible with an ecosystem-based approach?

To conclude, what are the recommendations that the scientific community can make to the judicial authorities to avoid mistakes being made on this point?

IN-KIND REPARATION FOR ECOLOGICAL DAMAGE AND THE ECOSYSTEM APPROACH

For a very long time, legal doctrine has argued that reparation for ecological damage should be made in kind (Prieur *et al*. 2019). The idea seemed to be that, since the damage had been caused to the natural environment, there was no better remedy than "restoring" the damaged or degraded environment. Some texts, for example European Directive 2004/35/EC of 21 April 2004, which established a system of compensation for ecological damage based on administrative policy and not on civil liability, had even excluded any compensation other than compensation in kind (Alt 1995, Martin 2015b). Fortunately, the law of 2016 was more reasonable by simply stating that compensation should be primarily – but not exclusively – in kind.

In fact, a first question arises as to what is meant by "reparation in kind". First of all, one must be aware that in the case of a fairly large number of hypotheses – fortunately less and less frequent nowadays – the judge must decide, in spite of the fact that he has absolutely no expertise in the scientific and technical data of the situation presented to him; most of the time, the lawyers who plead the case as plaintiff or defendant are hardly any more competent. It has often happened – and still happens – that reparation in kind was simply assimilated to the reintroduction of a few individuals of species that had disappeared from the damaged environment, without concern for whether all the balances, processes and functions of the affected ecosystem would have remained as they were before the accident. It is extremely difficult for a magistrate to understand that reintroducing a few eels and crayfish into a river from which they had disappeared as a result of pollution, for example, is not "restoring" or even "repairing in kind". Most of the time this results in something useless because the reintroduced species will not be able to survive and/or will prevent other balances from emerging. It is not surprising, however, that this partial species-specific approach is still sometimes used. Because it is the oldest, it is enshrined in the shared traditional environment culture; because it is the simplest, it allows for a quick solution and gives the illusion of satisfying the plaintiff. The magistrates who deliver rulings, like the lawyers who draft the claims for compensation, have, in most cases, no training in ecology, and the number of competent experts, registered on the lists of experts approved by the Courts of Appeal and who could enlighten the judges, is infinitesimal. For years now, the best-informed legal doctrine has been calling in vain for the creation of a list of accredited experts in the field of the environment and ecology (APCEF 2016).

If we therefore set aside this "false" reparation in kind, the fact remains that the expression can still be subject to various interpretations. In the minds of some authors, reparation in kind must necessarily lead to rehabilitation and thus to the restoration of the *statu quo ante* (Prieur *et al*. 2019). This rather restrictive interpretation of reparation in kind may correspond to certain situations. There are, in fact, hypotheses in which rehabilitation is technically possible, ecologically appropriate and ethically acceptable. This was for example the case in 2004 when *génépi* pickers were caught in the core area of the French Mercantour National Park in possession of more than 6,000 strands of this plant. *Génépi* refers to Alpine plants of the genus *Artemisia* that provide the flavor and the color of an herbal liqueur popular in the Alpine regions of Europe. They had acknowledged the facts and admitted that this harvest enabled them to prepare about 150 liters of liqueur, which they would sell at a good price in Italy, where the plant pickers came from. The Mercantour National Park submitted a request for reparation in kind, explaining to the Court that the offenders had committed a breach of the regulations, but that they had nevertheless more or less respected the site and the *A. umbelliformis* plants, which had not been uprooted or degraded. It was therefore possible to collect seeds on site to ascertain the origin and genetic characteristics of the seeds, then take them to INRA (*Institut National de la Recherche Agronomique* – French National Institute of Agricultural Research) laboratory in Antibes where they were cultivated, before moving the new plants to the original site. Two monitoring visits per year for 3 consecutive years seemed necessary. According to the specialists, a quarter of the seeds collected were lost in the operation and a certain percentage of the new plants did not thrive. This had to be taken into account when trying to find the 6,000 illegally collected strands at the end of the operation. The judge agreed to charge the offenders with the restoration cost. They were condemned to pay more than 18,000 €! (Chevassus-au-Louis 2009). This was possible because it was technically feasible, and the ecosystem had not been disrupted by the harvesting.

However, it is often the case that developments carried out or observed pollution have interrupted or altered some ecosystem processes or functions, making true restoration impossible, strictly speaking. Should it then be considered that, in such a case, the damage is irreversible? No text defines what is irreversible damage. The courts themselves refrain from giving a definition. As a general rule, they consider that restoration is not possible where development, accident or pollution have caused the disappearance of the various elements, processes and functions which made the ecosystem live. From this observation, three kinds of decisions can be identified.

Sometimes, courts may infer that they must forego compensation in kind; they then choose to grant the claimants (often an NGO) financial compensation. This can be seen, for example, in a case decided by the Court of Appeal of Chambéry on 30 June 2011 (reported by Neyret & Martin 2012). Having acknowledged the development of a track dedicated to 4x4 vehicles in the French

Vanoise National Park with ditches encroaching on wetland and resulting in the destruction of the habitat of the protected plant *Cirsium heterophyllum* (Linnaeus) Hill, the Court of Appeal of Chambéry condemned the offenders to compensate for the damages, without ruling out a compensation in kind, which was deemed impossible.

At times, on the contrary, the judges, while recognizing the impossibility of returning to the statu quo ante, consider that it is necessary to adopt a broader interpretation of "reparation in kind". They then accept that the reparation can take the form of a physical intervention aiming at trying to (1) erase what had been done illegally, and (2) either restore the degraded habitat as faithfully as possible, or endeavor to recreate the conditions for natural restoration, knowing that this might span a relatively long period of time.

With regard to work aimed at erasing what has been done illegally, the courts rarely hesitate to order it either as an additional penalty (Brunin & Timbard 2015) or as compensation. Such a decision can often be approved both in legal and ecological terms. On the legal level, it leads to erasing the consequences of an illegal act; on the ecological level, it has no negative consequences when the illegal behavior had already destroyed or very strongly degraded all the ecosystems concerned. Thus, for example, in a case decided by the Bordeaux Court of Appeal (C.A. de Bordeaux, 10 April 2009, reported by Neyret & Martin 2012), illegal dredging work had been carried out on the stream bed, which caused the removal of vegetation and the obstruction of the old watercourse, causing, according to the Court, "*the disappearance of all flora and the damage to the entire living environment*". The Court approved the Criminal Court for having imposed as an additional penalty the "restoration" of the site, that is to say, in reality, the reopening of the old watercourse, which had been filled in. But this is not always the case. Sometimes there is a risk that the intention to erase the consequences of an illegal act aggravate the damage caused by the illegal act itself. Several studies have shown, for example, that the use of heavy methods to "clean" beaches or rocks soiled by oil pollution could have very negative consequences for the environment (Boudouresque *et al*. 2019). Despite this well-documented observation, it is very common for such work to be ordered or spontaneously proposed by polluters and their insurers to minimize some damage, particularly of an economic nature, and to be validated *a posteriori* by the courts.

As for the works intended to reconstitute the degraded environment, they may be approved by the judge without always-adequate examination of their suitability. Advances in ecological engineering sometimes lead applicants to present a project for the "reconstruction of living organisms", which will claim to be able to compete with genuine restoration. Thus, for example, following damage to a *Posidonia oceanica* (Linnaeus) Delile meadow following the sinking of a ship and the attempt to tow the shipwreck away, consideration was given to reconstructing the meadow by "replanting" in the matte of *P. oceanica* cuttings (Martin 2020a). Such an in-kind repair project raises considerable difficulties, both ecologically and ethically. Ecologically, it is unlikely that the destroyed or degraded ecosystem can be reconstructed, since a multitude of factors that are difficult to control (currents, water temperature, occupation of the area by species that will have taken advantage of the damage caused to establish themselves, etc.) may come into play. From an ethical point of view, except the often exorbitant cost of such interventions for a an uncertain result (Boudouresque *et al.* 2012), the question arises as to whether such underwater "gardening" work is compatible with the nature of a *protected* area and with the spirit that must govern its management. That is why it often seems preferable for the judge to take into account the possibilities of natural regeneration of the site and to draw the conclusion that the so-called "restoration" or "repair in kind" works are limited to attempting to reconstitute the conditions of a natural evolution of the degraded site, by organizing the monitoring of the affected environment and by agreeing to consider that part of the damage is not reparable in kind and must be the subject of financial compensation. In the case already cited, which gave rise to a decision by the Bordeaux Court of Appeal in 2009 (C.A. de Bordeaux, 10 April 2009, reported by Neyret & Martin 2012), the Court noted that "*the minutes drawn up by the agents of the 'Conseil Supérieur de la Pêche' (Higher Council for Fisheries) indicate that the readjustment time (sic) for a return to the original profile will in any event be more than 10 years and that risks of a readjustment time of more than a hundred years are not to be ruled out. Consequently, the restoration of the site ordered as an additional penalty by the court has not removed the consequences of the offence and there remains a certain environmental damage*", which the Court makes good by awarding damages to the acting NGO calculated on a lump sum basis (on the methods of monetary evaluation of the ecological damage, see below).

At this point, it is possible to draw some partial conclusions: firstly, that the expression "in-kind repair" covers very diverse realities; secondly, that restoration and attempts to return to the *statu quo ante* are only possible in fairly rare cases and can sometimes may have additional negative effects on the ecosystem concerned; and finally, that the most appropriate in-kind repair often consists simply in recreating the conditions that will allow new balances to be established, and new processes and functions to be expressed.

These conclusions overlap with those that can be drawn after examining the other alternative. Where the judge finds that in-kind compensation is not possible for legal or factual reasons, he may decide to award financial damage compensation to the plaintiff, although the law specifies that these amounts must be "allocated to the repair of the environment”. This option obviously raises the very difficult question of how to assess ecological damage.

FINANCIAL COMPENSATION FOR ECOLOGICAL DAMAGE AND THE ECOSYSTEM APPROACH

It is obvious that financial compensation does not raise any difficulty when the damage is a traditional, material kind of damage, for example, damage to private property or to an economic activity. For a very long time, judges have found and developed methods to assess the personal injury and physical suffering suffered by a person, depending on his or her age, sex, occupation, hobbies, etc., in order to determine the amount of compensation to be awarded. There are even more or less official scales, which make it possible to assess such injuries accurately. It is already more difficult to assess moral damages, but for decades the courts have been willing to repair – and therefore assess – the pain and grief caused by the loss of a loved one, whether a person or an animal, or the harm felt by a person whose honor and reputation have been damaged. The assessment of harm, on the other hand, becomes very delicate when it concerns living and interrelated beings and (eco)systems, which are as far away as possible from the commercial and even emotional universe, at least as it is perceived by humans in occidental culture.

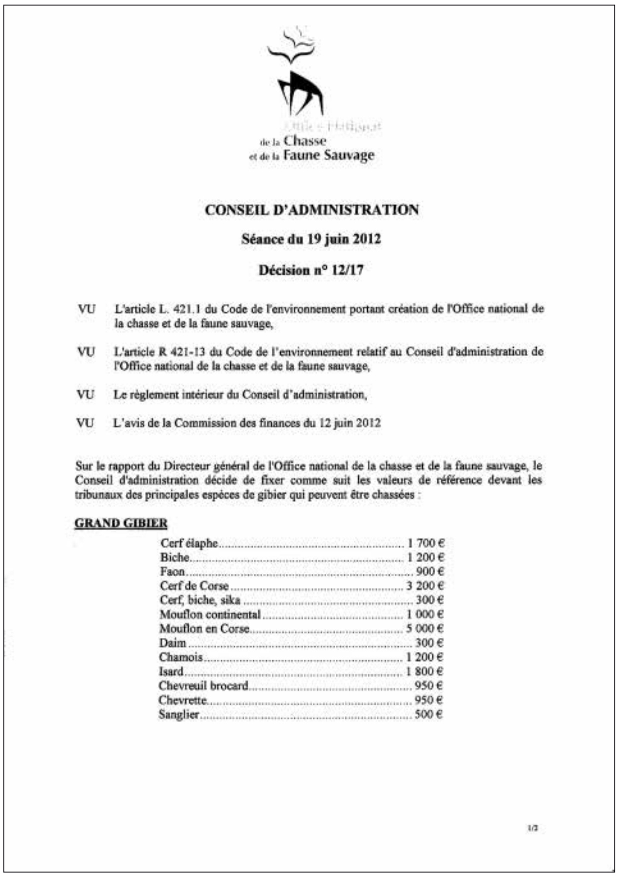
The following discussion examines how the courts are attempting to respond to this challenge, asking whether the valuation methods they use are compatible with an ecosystem-based approach.

If one accepts this schema, five different approaches can be distinguished in case law, noting that these methods are not mutually exclusive and can sometimes be combined.

It is the flat-rate assessment that was used first and which, unfortunately, is still quite often applied: the ecological damage is assessed through an element (*e.g.*, the individuals in a population of animals, one cubic meter of water, one hectare of forest) to which a flat-rate value is given and which is multiplied by the number of units affected by the damaging phenomenon. An illustration of this can be given through the case known as “*the Montedison red mud*” that this company was dumping at sea off Cap Corse (Corsica). As the Bastia fishermen’s guild (*La Prudhomies des Pêcheurs*) had acted to seek compensation for the damage caused to the environment, the *Tribunal de Grande Instance* of Bastia (judgment of 8 December 1976), confirmed by the Court of Appeal of Bastia (judgment of 28 February 1977), proceeded to the following reasoning (on the whole case, see Kiss 1975, Remond-Gouilloud 1979, Huglo 1992): taking into account the tonnage of fish exploited in the waters by the fishermen in that area, the court deduces an average “value” of the m^3 of sea water and, taking into account the extent of the pollution, it multiplies this value by the number of m^3 of polluted water. This way of proceeding combines a plethora of approximations and can be criticized for a variety of reasons. On the one hand, it only takes into account the economic damage as the damage to the environment – and only on a very rough basis; on the other hand, it does so on a flat-rate basis by giving a value per m^3 of water deduced from the supposed average presence of a certain quantity of fish that can be caught; finally, it completely ignores ecosystems that are never mentioned.

This flat-rate method has also been encouraged by the practice of the French *Office National de la Chasse et de la Faune Sauvage* (National Hunting and Wildlife Board) of publishing, at least until 2012, scales determining the “value” of an individual belonging to certain game species (Fig. 1).

The organization claimed, without demonstrating it, that this value represented on average the cost of reintroducing and monitoring an individual of the same species. In any event, a lawyer who presented the judge with the



de la Chasse
et de la Faune Sauvage

CONSEIL D’ADMINISTRATION

Séance du 19 juin 2012

Décision n° 12/17

VU L’article L. 421.1 du Code de l’environnement portant création de l’Office national de la chasse et de la faune sauvage,

VU L’article R 421-13 du Code de l’environnement relatif au Conseil d’administration de l’Office national de la chasse et de la faune sauvage,

VU Le règlement intérieur du Conseil d’administration,

VU L’avis de la Commission des finances du 12 juin 2012

Sur le rapport du Directeur général de l’Office national de la chasse et de la faune sauvage, le Conseil d’administration décide de fixer comme suit les valeurs de référence devant les tribunaux des principales espèces de gibier qui peuvent être chassées :

GRAND GIBIER

Cerf élaphe	1 700 €
Biche	1 200 €
Faon	900 €
Cerf de Corse	3 200 €
Cerf, biche, sika	300 €
Mouflon continental	1 000 €
Mouflon en Corse	5 000 €
Daim	300 €
Chamois	1 200 €
Isard	1 800 €
Chevreuil brocard	950 €
Chevrette	950 €
Sanglier	500 €

1/2

Fig. 1. – The flat-rate method of the French ‘*Office National de la Chasse et de la Faune Sauvage*’: ‘value’ of an individual belonging to certain game species. *Cerf élaphe*: red deer; *biche*: doe; *cerf de Corse*: Corsican deer; *cerf, biche, sika*: sika deer (males and females); *mouflon continental*: continental mouflon; *daim*: fallow deer; *chamois*: Alpine chamois; *isard*: Pyrenean chamois; *chevreuil brocard*: roe deer (male); *chevrette*: roe deer (female); *sanglier*: wild boar.

fee schedule from a specialized public institution was very likely to be followed and could perhaps even see in the judge's eyes a certain glimmer of gratitude for having made his work so much easier! However, the assessment is typically based on a species approach and does not even consider the population density in a given place. For example, the Alpine chamois (*Rupicapra rupicapra*) was "worth" 1,200 € in the entire Alpine region, whether it was shot within a National Park or outside its boundaries, and whether it belonged to a dense or sparse population. The destruction of an entire population was therefore not subject to any coefficient to re-evaluate the damage.

Finally, in the most recent period, it can still be seen that some magistrates continue to assess ecological damage on a flat-rate basis. For example, the sum of two million Euros for the significant damage caused to the Reunion National Park as a result of arson is not considered "*excessive*" by a Court of Assizes, if we relate it to the number of inhabitants of France (judgment of the Court of Assizes of Reunion ruling on civil interests of 2 March 2018, reported by Martin 2020b). Similarly, with regard to the harmful consequences of a prohibited fishing action, the Criminal Court of Toulon (date not communicated, reported by Martin 2020b) recently considered that "*in view of the fundamental role of* Posidonia oceanica *in the coastal ecosystem, the uprooting of* P. oceanica *by Mr B.'s vessel during its fishing activities in an illegal zone necessarily led to the partial destruction of the surrounding ecosystem in the zone, which is characteristic of certain material damage; that damage should be compensated by the allocation of a lump sum, which should be fixed in the amount of € 2,000"*. In this hypothesis, the Tribunal does refer to the "*coastal ecosystem*" and the "*ambient ecosystem*", but it concludes that a lump sum should be awarded, the amount of which is at no time justified. The difficulty in assessing harm cannot be a justification for such 'rule of thumb' approaches. The latter seriously undermine the educational mission of legal action and do nothing to enlighten the person responsible and public opinion on the value of destroyed or degraded environments. "*Moreover, they are experienced by the responsible party only as an additional "fine" and not as compensation for the value of destroyed natural "capital", which must, as far as possible, be restored*" (Martin 2020b).

Another much more satisfactory way of assessing damage is to do so by reference to the cost of rehabilitation, even if such rehabilitation cannot be implemented for technical and/or ethical reasons. As pointed out above, there are cases where it does not seem possible or desirable to carry out reconstruction of living organisms. Nevertheless, it may be interesting to assess what such an operation would cost, in order to give the judge an evaluation of the prejudice by reference to such a cost and thus avoid underestimating the prejudice.

A third way of proceeding is to assess the damage by reference to the budgets spent in pure loss as a result of the occurrence of the harm. A National Park spends a specified budget to manage a species present in its territory and its habitat or a particular ecosystem. Poaching, pollution and accidents have rendered this expenditure useless and it will take several years to return to the initial situation: the damage is equal to the annual invested budget multiplied by this number of years. At first reading, such an accounting and financial evaluation can be considered as having little to do with the environment in general and with degraded ecosystems. This initial reaction must undoubtedly be overcome, especially when the damage has been caused on the territory of a protected area. Indeed, in this hypothesis, taking into account the budget spent in pure loss, multiplied by the number of years necessary for the restoration of habitats and degraded ecosystems or the reappearance of new biological balances, gives an evaluation of the damage that takes into account the management method. If it is based on an ecosystem-based approach, the assessment reflects this reality.

A fourth method of assessing ecological damage is to assess ecological harm by reference to the value of ecosystem services that have been lost as a result of the damage (Doussan 2009, Mongruel *et al*. 2016, Doussan 2017). As is well known, a *Posidonia oceanica* meadow provides several ecosystem services (Boudouresque *et al*. 2016). The value of these services can be assessed by reference to the costs that would have to be incurred to obtain them. If it takes 20 years for the meadow to gradually recover, it is possible to calculate the loss incurred during that time. In the case already mentioned of damage to *P. oceanica* meadows caused by a stranded wreck, where the attempts at towing further aggravated the damage, it was the ecosystem services of *P. oceanica* that were identified and then evaluated by reference to work published in the scientific literature (*e.g.*, Blasi 2009). The National Park chose a bottom-line value of 172 €/m²/year, which is considered to be seriously underestimated. Considering the average rate of recovery of 5 cm/year, from the margins of a living herbarium, the National Park proposes a table establishing the "shortfall" in ecosystem services over 20 years and arrives at the figure of 722,400 € (Martin 2020b).

Finally, the last method, which is often associated with the previous one, adds to the evaluation the investments made necessary to accompany the natural restoration of the environment. The idea here is that it is preferable to accompany a natural restoration, which obviously involves costs of follow-up, monitoring, sometimes development (for example, the installation of substitute moorings, setting up surveillance of the area, etc.). It is in this spirit, for example, that the Port-Cros National Park has attempted to assess the ecological cost resulting from the damage caused to the *Pinna nobilis* by the wreck of a ship and the attempt to refloat and tow it. *Pinna nobilis* is an emblematic and threatened species (Rouanet *et al.* 2015). As the findings established that at least 3 individuals of this shell had been destroyed, the National Park refers to the cost of the moorings it has installed to compensate for the ban on anchoring in certain areas, which are intended to protect the *P. nobilis* present. Given the average density of *P. nobilis* in the areas concerned, the cost is 2,347 € per protected individual. It is deduced from this that expenditure of the same nature and amount will have to be incurred to protect and make possible (although it is not certain that this can be achieved) the recolonization of the degraded area with an equivalent number of individuals belonging to this species. As this case has not yet been tried, it is not possible to know whether this request will be granted by the court.

In any case, the evolution of Court judgments here described – which is obviously not linear and is still in its infancy – is largely driven by the dialogue between lawyers and ecologists: some assessments were – and still are – carried out in a totally arbitrary manner; then came the time when progress in ecological engineering suggested that the relevant reference could be found in active reconstruction and rehabilitation; while today the pendulum is swinging towards taking into account the ecosystem services lost and the cost of the support needed for the natural restoration of degraded sites. This movement follows the gradual emergence of ideas on the need to proceed, as far as possible, to the ecosystem-based management of environments and testifies – which is good news – that the law is not insensitive to it (Martin 2020a).

In conclusion, it remains to call on the entire scientific community to share its knowledge developments and analyses in order to help operational jurists (lawyers, magistrates) to apply the texts of the Civil Code in an appropriate and effective way as possible. It seems that three recommendations inspired by an ecosystem-based approach could be proposed:

– To call on the expertise of academics or researchers working in these fields whenever necessary;

– To order rehabilitation and a return to the *statu quo ante* only after such experts have assessed the suitability and possible counterproductive effects of such intervention;

– To give preference, whenever possible, in the assessment of ecological damage, to direct or indirect reference to the loss of ecosystem services and to the expenditure necessary to accompany and monitor the natural restoration of degraded ecosystems.

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AN ECOSYSTEM-BASED APPROACH TO STUDY THE BIGUGLIA LAGOON SOCIO-ECOSYSTEM (MEDITERRANEAN SEA)

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MEDITERRANEAN LAGOON
ECOSYSTEM-BASED APPROACH
ECOSYSTEM FUNCTIONING
GROUNDWATER
RESTORATION

ABSTRACT. – The Biguglia lagoon, located in the northwestern Mediterranean Sea (Corsica), is a natural reserve, which has been affected by a tremendous territorial change since last decades. An ecosystem-based approach should enable to understand the functioning of an ecosystem, which can be considered, as a socio-ecosystem in order to provide decision support for its conservation and restoration. The objective of this work is to present the research approach carried out on the Biguglia lagoon. The decreasing of the water quality is reflected by a shift from a dominance of aquatic magnoliophytes in the 1970s to varying dominance of phytoplankton and opportunistic macroalgae in the early 2000s. At the same time, it has been observed an ongoing deterioration of the quality of groundwater partly providing the lagoon with fresh water, which can be attributed to the uncontrolled urbanization development all over the alluvial plain surrounding the lagoon. Efforts undertaken to improve the hydraulic management and the reduction of nutrient inputs in the watershed allowed to observe signs of ecological restoration. Nevertheless, the socio-ecosystem functioning study requires a multidisciplinary approach where the natural and social scientists must work together as it is conducted within the framework of the Human-Environment Observatory of the Mediterranean coast (OHM-LM).

INTRODUCTION

Coastal lagoons are among the richest reservoirs of ecological diversity and biological productivity and provide different ecosystem services associated to high economic potential (*e.g.*, Costanza *et al.* 1997, Barbier *et al.* 2011). Owing to their transitional location between continental and marine domains, these ecosystems are naturally subject to environmental variations that induce deep spatio-temporal changes in their physical, chemical and biological properties (Telesh & Khlebovich 2010). Many of these ecosystems have suffered ecological degradation because of human activities, including for example eutrophication, contamination and habitat destruction. This combination of disturbances can dramatically compromise the ecological integrity of these ecosystems, as regularly documented for Mediterranean coastal lagoons (Alvarez-Cobelas *et al.* 2005, Flo *et al.* 2011). To face the huge degradation of lagoons and more generally aquatic ecosystems, European Union (EU) public authorities have designed a new legal managerial framework to reverse adverse anthropic impacts and achieve their good ecological status (Newton *et al.* 2014). In particular, the EU water framework directive in 2000 (WFD, 2000/60/EC) recommends the implementation of management measures to improve the ecological and physico-chemical states of water bodies, in parallel with the monitoring of their ecological and physico-chemical characteristics to evaluate their recovery trajectories.

While the species-centered approach has been adopted for long to meet these expectations, it is now necessary to consider an ecosystem-based approach (Boudouresque *et al.* 2020). An ecosystem-based approach happens to be more adapted to understand the functioning of an ecosystem. This ecosystem displays an organization that includes all the organisms and their interactions, along with the different components of the abiotic environment. The ecosystem functioning can be partly studied through primary producers (phytoplankton, macroalgae and aquatic angiosperms), which react quickly to changes in the environment. Eutrophication can lead to vegetation shifts, described as a transition between alternative states, from pristine slow-growing benthic plants (aquatic angiosperms) to rapidly growing ephemeral plants (macroalgal or phytoplankton communities; Schramm 1999, Pasqualini *et al.* 2017, Le Fur *et al.* 2019). These variations then have major consequences on the overall ecosystem functioning. Another problem that can cause ecosystem disruption is invasive species. In fact, invasive species rarely have an impact on given species, but on entire communities; therefore, understanding their role and impact can only be achieved by considering the whole ecosystem (Boudouresque *et al.* 2011).

The ecosystem-based approach also highlights the importance in tackling the coupling between adjacent

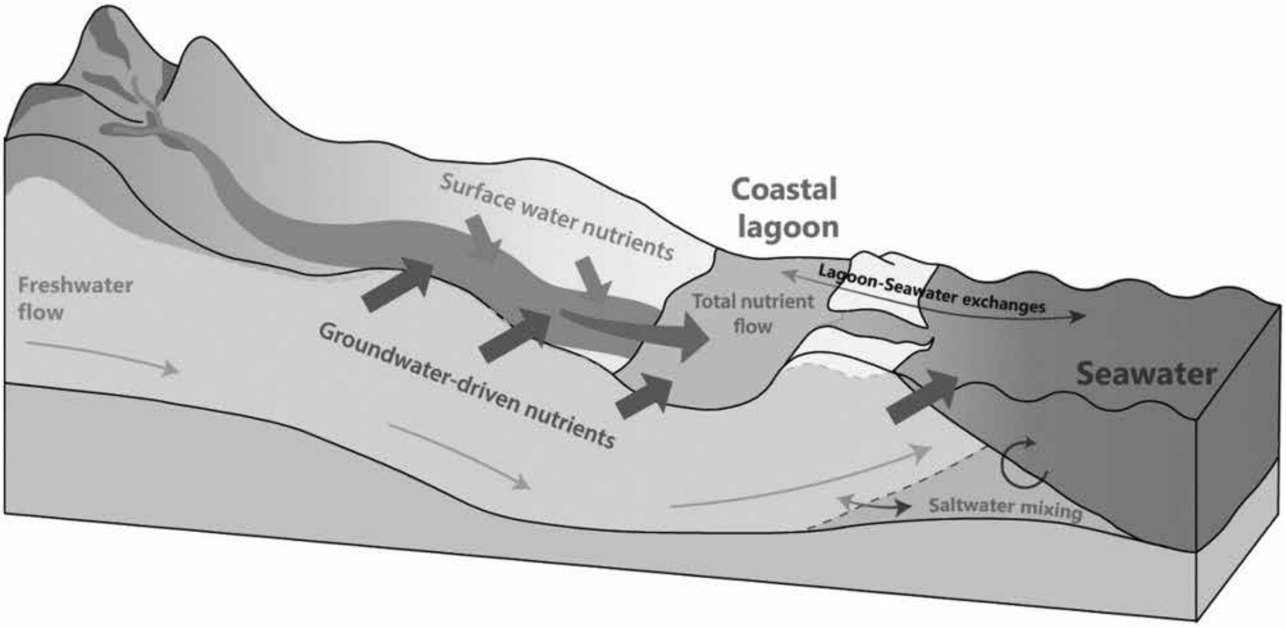

Fig. 1. – Conceptual diagram of the hydrogeological behavior of coastal hydrosystems including a coastal Groundwater Dependent Ecosystem (adapted from Erostate *et al.* 2020).

ecosystems, including groundwater and watershed. In most cases, coastal lagoons are maintained by direct or indirect groundwater supplies, and constitute therefore Groundwater Dependent Ecosystems (GDEs; Krogulec 2016, Menció *et al.* 2017). Since the last decades, several studies have highlighted the importance of groundwater supplies for coastal lagoons (Fig. 1; Malta *et al.* 2017, Erostate *et al.* 2018, Rodellas *et al.* 2018, Correa *et al.* 2019, David *et al.* 2019, Erostate *et al.* 2019). These studies highlight the contribution of groundwater discharge to coastal streams in water and nutrient budgets of coastal zone ecosystems (Fig. 1). During the infiltration, nutrients are leached from the soil and they percolate until the groundwater and migrate directly (groundwater-driven nutrient discharge) or indirectly (via the river) to coastal GDEs (Fig. 1; Rapaglia 2005, Jimenez-Martinez *et al.* 2016, David *et al.* 2019). Human activities in the watershed have an impact on the whole coastal lagoons and within its functional compartments. Depending on the hydraulic behavior of the aquifer, groundwater can thus represent a direct short and/or long-term vector of nutrients/pollutions for coastal GDEs. The efforts undertaken to improve the hydraulic management and the reduction of nutrient inputs in the watershed reduce impacts and can induce ecosystem restoration. However, the ecosystems responses do not result exactly in reversing the processes that occurred during the eutrophication for example (Leruste *et al.* 2019). Moreover, the ecosystem-based approach allows the integration of humans in the ecosystem functioning, thus passing from the notion of ecosystem to that of social-ecological system. In this perspective, an integrated approach for sustainable development in coastal lagoons with a strong partnership among researchers, managers and stakeholders has been proposed (Newton 2012, Newton *et al.* 2014). In this way,

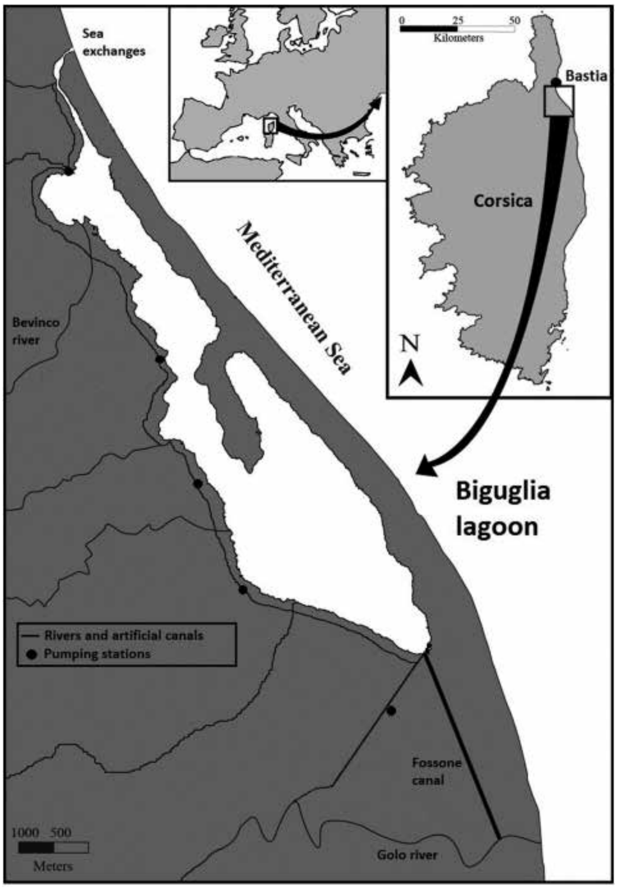

Fig. 2. – The location of Biguglia lagoon (Western Mediterranean Sea) with rivers, artificial canals and pumping stations.

coastal lagoons including their watershed and their coastal zone can be considered as social-ecological systems in which human activity is an integrated part of their own ecology. To better understand the functioning of coastal lagoons as social-ecological systems, organizing multidisciplinary studies and connections between scientists is a prerequisite but also a critical issue (Ostrom 2009).

The objective of this work is to present the ecosystem-based approach carried out on the Biguglia lagoon, located in the northwestern Mediterranean (Corsica; Fig. 2) and affected by a tremendous territorial change these last decades. A diachronic analysis of the ecosystem functioning is carried out on the basis of the primary producers (phytoplankton, macroalgae and aquatic angiosperms) and the ecosystem disturbance observed, coupled with the analysis of groundwater providing the lagoon with freshwater. Studying the hydrogeological component, the dynamics of the aquifer and its capacity to remediate or store pollutants, is indeed essential to assess potential future threats to coastal lagoons. This work is undertaken with regard to the evolution of human activities in the watershed and management actions of the natural reserve. Finally, a multidisciplinary approach on the Biguglia lagoon with the natural and social scientists is conducted within the framework of the Human-Environment Observatory of the Mediterranean coast (OHM-LM) will be presented.

MATERIAL ET METHODS

Biguglia lagoon is a choked, shallow, brackish coastal lagoon located on the Northeast coast of Corsica close to Bastia city (Fig. 2). With a surface of 1,460 hectares, this lagoon is linked to the Mediterranean Sea through a long natural channel to the North (Pasqualini *et al.* 2017). Marine water inputs are limited because the sea channel tends to close (due to the accumulation of sand). Biguglia lagoon receives freshwater from the rivers draining its watershed (180 km²), mostly in the Northwest part (Bevinco River) and from an old artificial channel and pumping stations draining the agricultural plain, sewage plants and rainfall in the West and South parts (Fig. 2). Freshwater inputs dominate the water budget and lagoon renewal is rapid (several weeks or months; Garrido *et al.* 2016). Biguglia lagoon was recognized as a very important site for waterfowl and was included in the RAMSAR list of wetlands of international importance in 1991. Moreover, it was classified as a natural reserve in 1994 and it belongs to the Natura 2000 network since 2006 (Special Protected Areas of the Bird Directive – EU). The property of the whole lagoon is in the hand of the Collectivity of Corsica (local public authority), with its dedicated service that executes the management. The whole lagoon surface and a small part of the fringing wetlands are no-entry zones with the exception of a small number of professional fishermen allowed for this traditional use.

In order to realize the diachronic analysis of the ecosystem functioning, the information is obtained on the basis of the primary producers such as phytoplankton, macroalgae and aquatic angiosperms. For the phytoplankton, the data come from the Lagoon Monitoring Network since 1999 (Souchu *et al.* 2010) and from bibliographic data (Cecchi *et al.* 2016, Garrido *et al.* 2016, Leruste *et al.* 2019). For the macroalgae and aquatic angiosperms, the data are obtained from the Lagoon Monitoring Network since 1999 and from bibliographical data (De Casabianca *et al.* 1973, Frisoni & Dutrieux 1992, Pasqualini *et al.* 2006, 2017). The ecosystem disturbance observed come from by personal observations and from the monitoring carried out by the natural reserve since 2007. The database developed by the Lagoon Monitoring Network can also be used to assess the eutrophication status of lagoons (Souchu *et al.* 2010, Pasqualini *et al.* 2017). For the Lagoon Monitoring Network, sampling was carried once monthly during the summer period from 1999 to 2014. Mean summer lagoon salinity data were obtained from records published since 1978 (Burelli *et al.* 1979, Frisoni & Dutrieux 1992, Lagoon Monitoring Network and monitoring carried out by the natural reserve).

After demonstrating the dependence of the Biguglia lagoon on groundwater (Erostate *et al.* 2019), the assessment of the groundwater quality was carried out in order to understand the potential influence of groundwater in the quality status of the lagoon. To this end, nitrate (NO_3^-) and emergent organic compounds (EOCs) concentrations were measured in groundwater, river water and lagoon water. Data acquisition was carried out through two sampling campaigns in May 2016 and May 2017. The concentration of dissolved major ions was determined at the Hydrogeology Department (CNRS UMR 6134 SPE), University of Corsica, France, using a Dionex ICS 1100 chromatograph. Regarding EOCs, 51 organic compounds were analyzed, including pharmaceuticals and other substances, such as artificial sweeteners, caffeine and so forth were done using positive (ESI+) and negative (ESI-) modes of the electrospray at the "Povodi Vltavy" laboratory, Pilsen, Czech Republic. Finally, tritium (^3H) contents of groundwater were also measured in order to distinguish the contemporary or historical origin of the pollution and thus to assess the aquifer' remediation capacity. Analyses were carried out by liquid scintillation counting (Thatcher *et al.* 1977) after electrolytic enrichment (Kaufman & Libby 1954) at the Hydrogeology Department of the University of Avignon, France.

RESULTS

The diachronic analysis of the ecosystem functioning shows a shift from a dominance of aquatic magnoliophytes in the 1970's to varying dominance of phytoplankton and opportunistic macroalgae in the early 2000's (Fig. 3A). In the early 1970s, four aquatic angiosperms were well developed in Biguglia lagoon: *Zostera noltei* Hornemann close to the sea channel in the north, *Ruppia cirrhosa* (Petagna) Grande and *Stuckenia pectinata* (Lin-

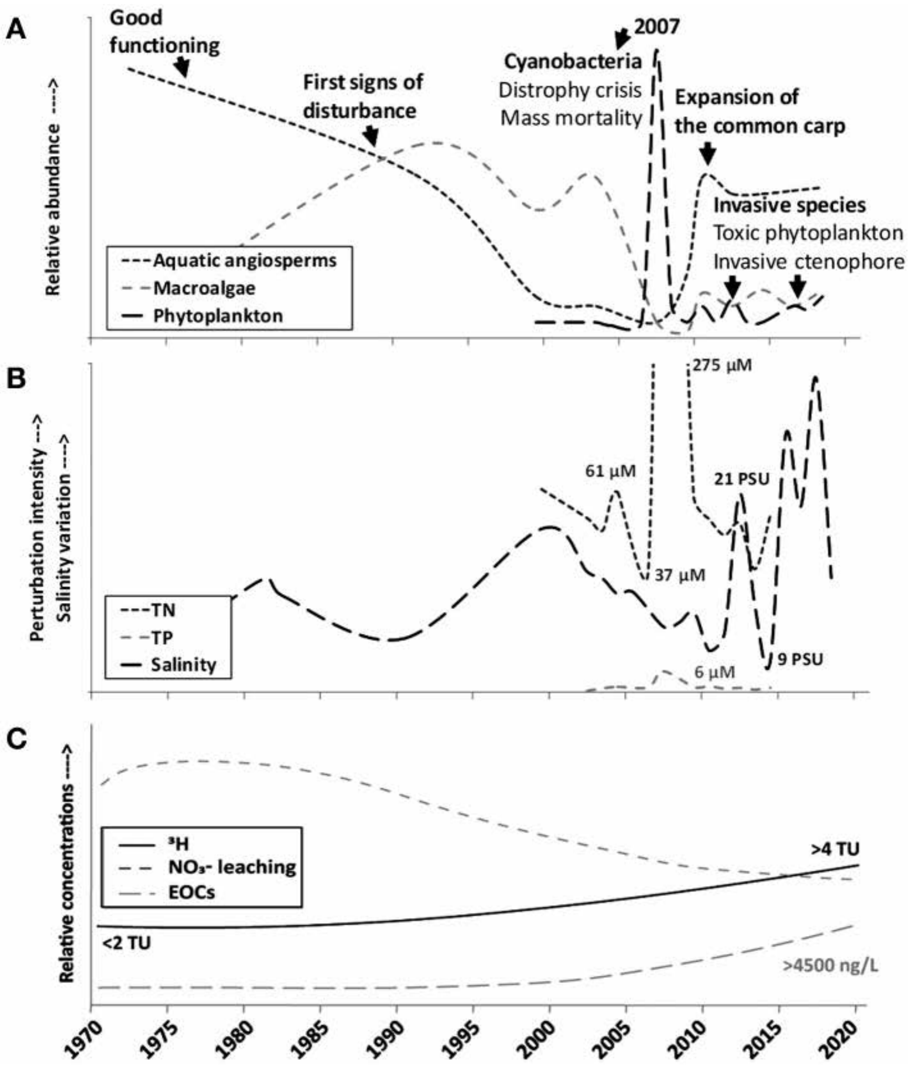

Fig. 3. – Conceptual representation, (**A**) of the relative abundance of aquatic vegetation, phytoplankton communities and ecosystem disturbance observed in Biguglia lagoon (adapted from Pasqualini *et al.* 2017); (**B**) of the eutrophication status and salinity variations in Biguglia lagoon (adapted from Pasqualini *et al.* 2017), (**C**) of the nitrate leaching evolution and EOCs concentrations as a function of water residence time. TN: total nitrogen, TP: total phosphorus.

naeus) Börner throughout the lagoon and *Najas marina* Linnaeus in the southern basin, which testifies to a good functioning of the lagoon. In the early 2000's, *Ulva* sp., *Ulvaria obscura* (Kützing) P. Gayral ex C. Bliding (green algae) and *Gracilaria dura* (C. Agardh) J. Agardh (red algae) occupied a large part of the lagoon. At this period, the concentrations of total nitrogen reached about 60 µM (Fig. 3B). In the summer of 2007, Biguglia lagoon suffered a dystrophic crisis associated with a toxic cyanobacterium bloom, *Anabaenopsis circularis* Woloszynska. The concentrations of total nitrogen reached 275 µM and there was a massive mortality of all organisms in the ecosystem (Fig. 3A, B). After this episode, Biguglia lagoon presented a predominance of aquatic angiosperms, mostly in the southern basin for *Najas marina*, throughout the lagoon for *Ruppia cirrhosa* and/or *Stuckenia pectinata*, and a decrease in macroalgae. The progressive desalination observed in the lagoon at this time has hugely impacted the ecosystem, particularly with the marked development of freshwater magnoliophytes (*Najas marina*), a disappearance of *Zostera noltei* and the expansion of the common carp (*Cyprinus carpio* Linnaeus, 1758), never reported before in the Biguglia lagoon (Fig. 3A, B). The phytoplankton community was strongly influenced by freshwater and nutrients inputs that have led to different physiological and behavioral responses. There ecosystem modifications have dramatically facilitated the successful installation of opportunistic invasive organisms, as a potentially toxic phytoplankton (Fig. 3A; *e.g., Prorocentrum cordatum* (Ostenfeld) J. D. Dodge; dinoflagellate with mixotrophic strategies; Cecchi *et al.* 2016) or an invasive ctenophore (*Mnemiopsis leidyi* A. Agassiz, 1865).

These ecosystem modifications and the variations in total nitrogen concentrations observed in the lagoon (in particular NO_3^-) have raised questions about the NO_3^- origin. The NO_3^- concentrations measured in river water are often very low and cannot explain the NO_3^- concentrations in the lagoon. The good correlation of the dynamics between groundwater flow and NO_3^- concentrations in the lagoon suggests that groundwater has a predominant impact on the NO_3^- supply to the lagoon (Erostate

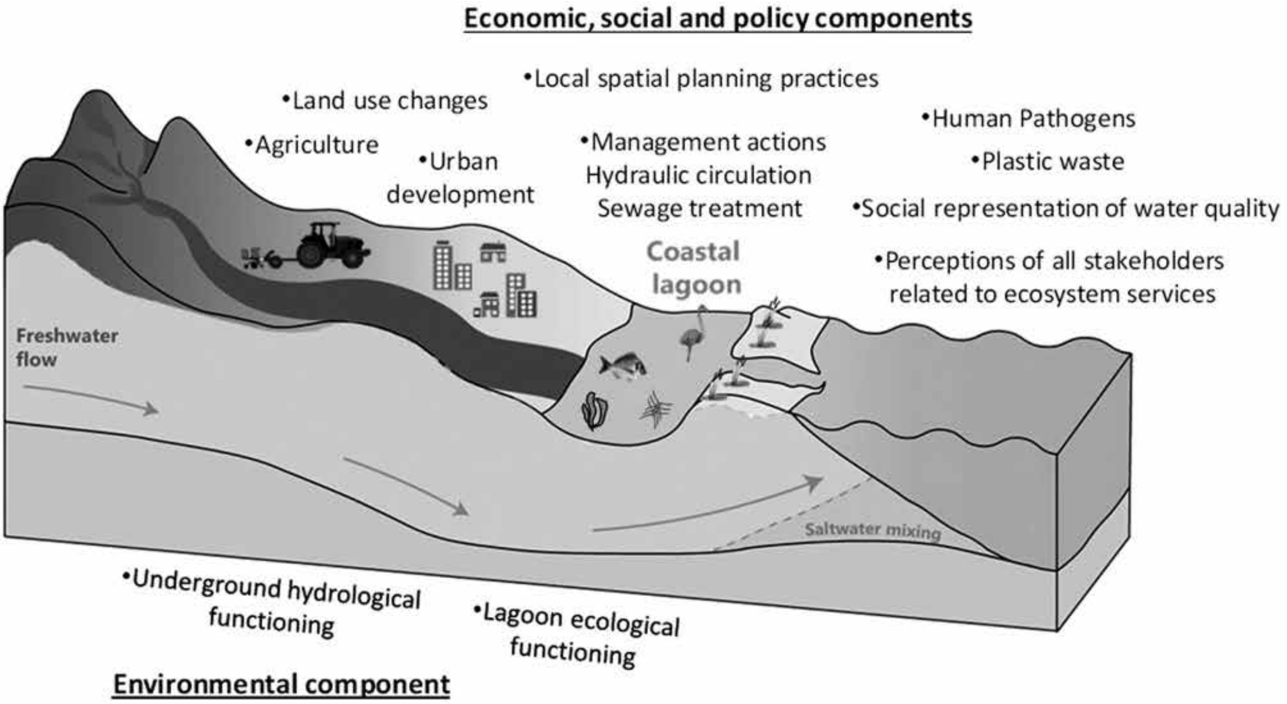

Fig. 4. – Schematic representation of the Biguglia socio-ecological system including hydrological and ecological functioning and the aspects covered by the scientific projects supported by the Human-Environment Observatory of Mediterranean Coasts (OHM-LM).

et al. 2018). The correlation between NO_3^- concentrations and ^3H contents made it possible to assess the temporal dynamic of NO_3^- in groundwater (Fig. 3C). The groundwater with the longest residence times ($^3H < 2$ TU; TU = Tritium Unit) has the highest NO_3^- concentrations (between 40 to 70 mg/L or 650 to 1,100 µM; Fig. 3C). As decrease the residence time (2 TU < ^3H < 4 TU), NO_3^- concentrations also decrease (> 15 mg/L on average; Fig. 3C). High NO_3^- concentrations in the aquifer are therefore the result of accumulated historical pollution inherited from the 1960s-1970s. Groundwater, by flowing towards the lagoon, contributes to a NO_3^- enrichment. However, this phenomenon is tending to decrease due to the observed decrease in NO_3^- concentrations in groundwater (Fig. 3C).

This recently observed improvement in groundwater quality with respect to NO_3^- concentrations is consistent with current observations in the lagoon. At present, the salinity is higher and the concentration of total nitrogen seems to decrease in the lagoon, which has led to a reduction in carp, a reappearance of *Zostera noltei*, and which demonstrates the beginning of the Biguglia lagoon restoration. However, a persistence of dinoflagellate blooms is observed, which can reflect the vulnerability of the whole ecosystem. In addition, the vulnerability of the hydrosystem is also underlined by the detection of EOCs in groundwater and lagoon water, EOCs was not detected in the river (Fig. 3C). These pollutants, entirely human-made, illustrate a strong anthropic pressure on the watershed.

DISCUSSION

An ecosystem-based approach must enable to consider the global functioning of an ecosystem, in particular the interactions between organisms, along with the different components of the abiotic environment. The originality of our study lies in a) considering the lagoon ecosystems with groundwater since these are ecosystems dependent on groundwater, and b) indicators of human activities in the watershed, which have an impact on the whole coastal lagoons and within its functional compartments (Fig. 4). Biguglia lagoon has been affected by a tremendous territorial change these last decades, as is the case with a large number of Mediterranean lagoons due to their location in the coastal zone (La Jeunesse *et al.* 2002, Alvarez-Cobelas *et al.* 2005, Serrano *et al.* 2006, Shili *et al.* 2007, Flo *et al.* 2011). In the Biguglia watershed, agricultural practices have experienced strong modifications while the whole area has been facing the increasing development of urban settlements related to the extension of Bastia city (Fig. 4; Département de la Haute-Corse 2013). Over the last century, the urbanized areas of the catchment area have grown five-fold in size, mainly on the alluvial plain (Fox *et al.* 2012). The strong urban pressure (Robert *et al.* 2015), the lack of urban planning (Prévost & Robert 2016), and the construction of isolated residential areas have amplified the risk of localized pollution and leakage from the extensive sanitation network (*i.e.*, leakage from water pipelines and septic tanks). Agricultural land-use has decreased by almost 40 %. Over the last decades, orchard and vineyard farming was progressively replaced by cattle breeding and vegetables production. Consequently, ecosystem functioning of Biguglia lagoon has been profoundly altered, with changes in the aquatic vegetation, from a predominance of aquatic angiosperms to macroalgae and phytoplankton during a dystrophic crisis in 2007. Massive soil remodeling due to urban developments and culture type changes has led to a significant soil N mobilization in the 1970/1980's, resulting in high nitrate inputs to the aquifer (as indicated by the relatively high nitrate concentrations in the oldest groundwater samples; Daum 1997).

To face the huge degradation of lagoons, the Water Framework Directive in 2000 (WFD, 2000/60/EC) recommends the implementation of management measures to improve the ecological and physico-chemical states of water bodies. In this context, the managers of the Biguglia natural reserve led various remedial measures, such as cleaning the Fossone Canal between 2009 and 2012 to decrease the confinement of the southern basin. This action, as a consequence, causes significant freshwater inputs from the Golo River and desalination of the lagoon (Fig. 4; Département de la Haute-Corse 2013, Garrido *et al.* 2016). Other interventions were also carried out, such as periodically opening up the channel connecting the lagoon with the sea by mechanical means, which led to significant salt-water input and an increase in the salinity of the lagoon afterwards. Nevertheless, ecological restoration efforts realized to improve hydraulic management must be accompanied by the reduction of nutrient inputs in the watershed with an awareness of regional authorities that goes beyond the boundaries of the nature reserve. Substantial efforts have been made recently to improve sewage treatment in the watershed under the *Schéma d'Aménagement et de Gestion des Eaux* (SAGE; Fig. 4).

Within the studied hydrosystem, the resilience capacity of water bodies seems to be strongly contrasted. For the lagoon, such management measures improved the quality of the water column, favoring the resettlement of aquatic magnoliophytes and the decrease of macroalgae and phytoplankton over a relatively short period of time (4-5 years; Pasqualini *et al.* 2017). The observed shift in communities suggests that Biguglia lagoon is resilient and that the transition may be reversible, even if the whole ecosystem remains vulnerable. The alterations in the ecosystem have drastically facilitated the successful installation of invasive and tolerant organisms, which weakens the whole ecosystem. Such modifications can weaken the entire ecosystem and have significant impacts on fish resources, but certainly also on bird populations, which have a key role in the conservation of the Biguglia lagoon. Such responses of lagoon ecosystems to a disturbance have already been observed in other Mediterranean lagoons, but ecosystem restoration depends on the intensity and persistence of the disturbance (Souchu *et al.* 2010, Leruste *et al.* 2016, Kermagoret *et al.* 2019, Le Fur *et al.* 2019). For groundwater, management measures are struggling to restore the qualitative degradation. While the relatively low nitrate contamination in modern groundwater (groundwater with short residence time) underlines that ongoing management practices to reduce surface nitrate pollution produce their effect, progressive nitrate contamination in groundwater with long residence time indicate the poor self-remediating capacity of the aquifer. The time lag of several decades between pollution and groundwater contamination indicates that even a complete halt of anthropogenic nitrate inputs to the groundwater would not result in an immediate improvement of the groundwater quality. In addition, the occurrence of EOCs in groundwater is an indicator of an ongoing groundwater degradation related to wastewater infiltration (Erostate *et al.* 2019, Vystavna *et al.* 2019). Once infiltrated, the pollutants follow the groundwater flow and can migrate to the lagoon (Knee & Paytan 2011, Jimenez-Martinez *et al.* 2016, David *et al.* 2019). Thus, legacy and current pollution in groundwater threaten the resilience and long-term protection of the lagoon, which directly rely on groundwater supply.

The ecosystem-based approach allows the integration of humans in the ecosystem functioning, with the notion of social-ecological system. Social-ecological systems are linking people and nature, emphasizing that humans are a part of, and not apart from, nature (Berkes *et al.* 1998). An integrated approach for sustainable development in coastal lagoons with a strong partnership among researchers, managers and stakeholders has been proposed (Hopkins *et al.* 2012, Newton 2012, Newton *et al.* 2014). Lagoon conservation requires an integrated and multidisciplinary approach where the natural and social sciences collaborate together. Since 2012, the Biguglia lagoon is the subject of particular attention in this direction through the Human-Environment Observatory of the Mediterranean Coasts (OHM-LM, under the CNRS and LabEx DRIIHM). This research initiative has contributed to a better understanding of Biguglia lagoon within its watershed, through multidisciplinary research projects (Fig. 4; *e.g.*, land use changes, local spatial planning practices, coastal aquifers, social representation of water quality, perceptions of all stakeholders related to ecosystem services; Robert *et al.* 2015, Prévost & Robert 2016, Sy *et al.* 2018, Audouit *et al.* 2019, Jaunat *et al.* 2019, Leruste *et al.* 2019, Robert *et al.* 2019; see details in http://www.ohm-littoral-mediterraneen.fr/spip.php?rubrique8). Data acquisition is still continuing notably in the natural sciences (ecological integrity) and must intensify in the social sciences. The OHM promotes communication between the scientific community and coastal zone stakeholders. The originality of its approach lies in integration of social and natural sciences and in the analysis of the interactions between Humans and the Environment, which are necessary for the study of a socio-ecosystem. In the frame of ecosystem-based approach, this integrated multidisciplinary approach is essential to optimize the conservation of coastal lagoons subject to anthropogenic pressures and climate change.

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DESCRIPTION OF A THEORETICAL SOCIAL-ECOLOGICAL APPROACH TO MANAGE ARTIFICIAL REEFS

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ARTIFICIAL REEF
ACTOR NETWORK THEORY
TROPHIC NETWORK
SOCIAL-ECOLOGICAL NETWORK

ABSTRACT. – Network analysis is used to address diverse ecological, social, economic and management questions, but few studies combine social and ecological issues in a single analysis. Understanding the links between social and ecological networks helps in establishing coastal management strategy for the sustainable use of marine resources. The aim of this research study is to apply this approach to artificial reefs (AR) along the French metropolitan coasts. For fifty years, AR has been deployed in France with a single main goal: to sustain artisanal fisheries by enhancing resources. Assessing the effectiveness of this tool requires considering each stakeholder's initial intentions and comparing them to the actual results (social and ecological). Network analysis provides a holistic view of the relations between all the actors of the system that offers a basis for suggesting a suitable management strategy for each objective identified.

INTRODUCTION

Nowadays coastal ecosystems are strategically important in French marine areas. As the number of marine activities and their pressure on biodiversity and resources increase, there is increasing necessity to protect and manage the coastal environment. Achieving a balance between the ecological protection and the economic development of coastal areas is a complex matter. The drastic decline of biodiversity and marine resources and the increasing level of pollution are a threat to the major ecological functions, the health of ecosystems and human activities.

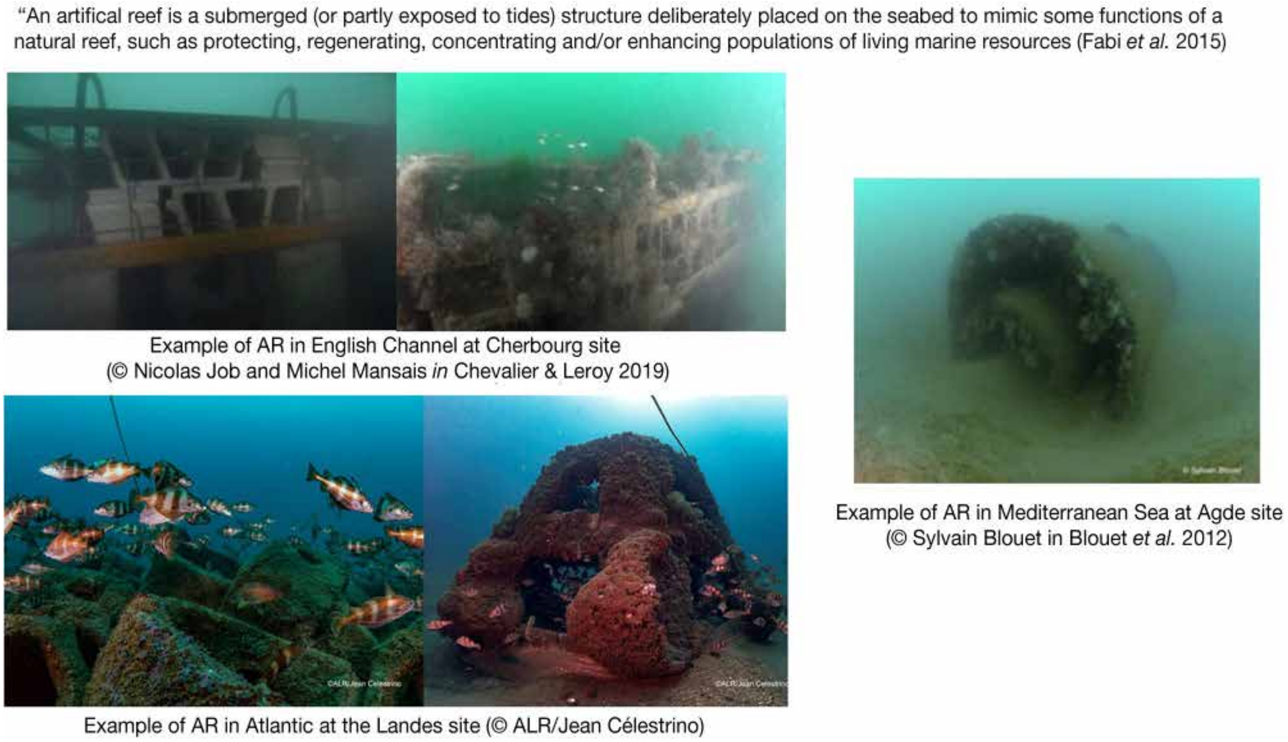
For more than fifty years, Artificial Reefs (AR) have been deployed in France (Fig. 1) to respond to the decline in fish stocks (Tessier *et al.* 2015). The main aim of these structures is to sustain artisanal fisheries and enhance fish stocks (Fabi *et al.* 2011). Assessments of their effectiveness, when they have been carried out, have been focused only on certain ecological components such as commercial fishes, for example (Véron *et al.* 2008). In 2012, despite a regional strategy for the Languedoc-Roussillon Region regarding the management of AR, there are still almost 10 areas of AR with no monitoring (Cépralmar, Région Languedoc-Roussillon, 2015). The lack of feedback raises questions regarding the real ecological and social efficacy of these structures.

A social-ecological study seems to be an appropriate research axis to understand the functioning of AR on the basis of a holistic approach. The study begins with an understanding of expectations of each of the territorial actors and stakeholders regarding the deployment of AR. Then ecological results are assessed with modeling of the food web before and after immersion of AR. In the third part, the social network will be analyzed on the basis of all the available data (such as report files, legal authorization) and the patterns of change in the social-ecological system will be highlighted. Finally, the comparison of the objectives identified with the social and ecological results will provide a basis for the assessment of the overall functioning of AR. The aims of this work are to understand how the social-ecological system could help to extend the integrated approach, and find solutions for better management of the coastline.

MATERIALS AND METHODS

Study areas: For each of the three French metropolitan coasts, two types of AR areas have been selected where possible: areas of active management and areas of soft management. To define the soft management of AR, we refer to the Mediterranean strategy for Languedoc-Roussillon (Cépralmar, Région Languedoc-Roussillon 2015), which recommends monitoring every three to five years and other criteria such as type of funding, communication and management (Table I).

In the English Channel, only two places have immersed AR: in Cherbourg harbor in the North Cotentin and off the coast of Etretat in the eastern basin (Fig. 2). On the Atlantic coast, more numerous sites with AR means that three of them could be selected: the island of Yeu in the north of the Bay of Biscay for the soft management group, Capbreton and Oléron Island representing active management for the southern and northern part of the Bay of Biscay (Fig. 2). On the Mediterranean coast, 26 AR have been established since the first deployment in 1968.



"An artifical reef is a submerged (or partly exposed to tides) structure deliberately placed on the seabed to mimic some functions of a natural reef, such as protecting, regenerating, concentrating and/or enhancing populations of living marine resources (Fabi *et al.* 2015)

Example of AR in English Channel at Cherbourg site (© Nicolas Job and Michel Mansais *in* Chevalier & Leroy 2019)

Example of AR in Mediterranean Sea at Agde site (© Sylvain Blouet in Blouet *et al.* 2012)

Example of AR in Atlantic at the Landes site (© ALR/Jean Célestrino)

Fig. 1. – Pictures of Artificial Reefs' structures (J. Salaün).

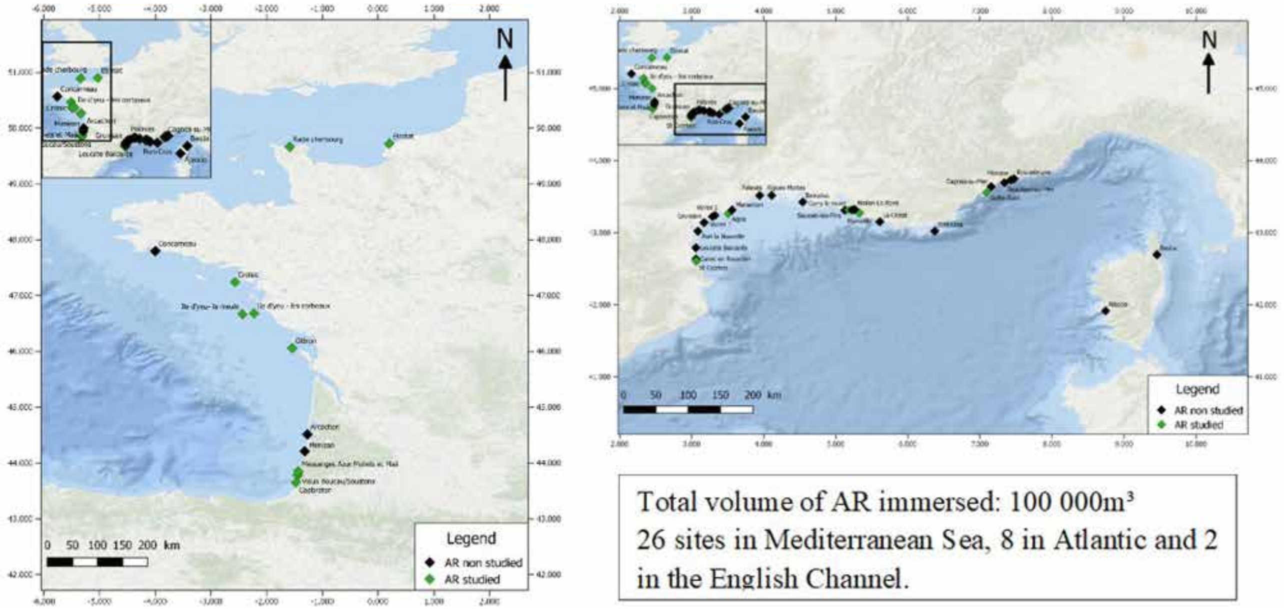


Fig. 2. – Maps of Atlantic, English Channel and Mediterranean locations of Artificial Reefs along the metropolitan coast of France (J. Salaün).

The latest AR project is the Prado reef in the bay of Marseille (the largest AR in the Mediterranean and in Europe, with a gross volume of 27,300 m^3; Charbonnel *et al.* 2011) and will be studied as the active management group with the Agde AR. For the soft management group, AR sites such as Gruissan or Niolon in the Côte Bleue Marine Park have been identified (Fig. 2). Other sites with mixed management could also be studied such as Vallauris/Golfe-Juan or Leucate.

Translation process method: The four-stage method of analysis within the Actor Network Theory (ANT) framework was used (Crozier & Frieberg, 1977). This frame was developed by Callon (1986) and Latour (1987) to understand the process of innovation and how scientific facts become constructed (Latour 1987). The translation process method consists in describing each step of the network construction and finding the Required Crossing Point that gathers all actors in order to achieve a mutually desired out-come (Jeacle 2017). In the ANT framework, actors can be human or non-human. The translation process method is described on the basis of four stages (Callon 1986, Reverdy 2013, Lombard-Latune 2018): (i) 'problematization', is the initial step that defines the individual issues and proposes common solutions; (ii) 'interest', is the step when the common project begins and each actor defines their motivation; (iii) 'enrolment' corresponds to the role played by each actor; (iv) 'mobilization' is a stage that makes it possible to extend and consolidate the actor network around the common project.

The data sources for this study derive from qualitative open-ended interviews of each actor for each study area (Alami *et al.* 2013). Actors are identified by project documentation and by the interviewed actors themselves. At the beginning of the interviewing survey, a social network is built, which is then completed by the other interviewees and so on, until that the last interview does not provide any new information to the constructed network (Kaufmann 2016).

Trophic modeling approach: The trophic network has been constructed by using the Ecopath with Ecosim software (Polovina 1984, Christensen & Pauly 1992, Christensen *et al.* 2008).

The Ecopath model was parameterized with two main equations. The first one describes the production for a group *i* and its predator *j* (Eq. 1) whereas the second describes the energy balance within a group i (Eq. 2):

$$B_iP / B_i = j = 1nB_jQ / B_jDC_{ij} + Y_i + E_i + BA_i + PB_i \text{ x } B_i(1 - EE_i) \quad \text{(Eq. 1)}$$

$$Q_i = P_i + R_i + U_i \quad \text{(Eq. 2)}$$

where the parameters are biomass (B, gCm^{-2}), production rate (P/B, $year^{-1}$), consumption rate (Q/B, $year^{-1}$), proportion of *i* in the diet of *j* (DC_{ij}, diet composition), total fishery catch rate (Y_i, gCm^{-2}), net migration rate (E_i, $year^{-1}$), biomass accumulation (BA_i, $year^{-1}$), ecotrophic efficiency (EE_i), respiration (R_i, gCm^{-2}) and unassimilated food (U_i).

Functional groups were defined by using biological and ecological characteristics of species. Functional groups have been chosen to be the same on both models for the Atlantic and the English Channel. For Mediterranean AR, a model already exists (Cresson *et al.* 2014). Twenty-three groups have been made: plunge and pursuit diver seabirds, surface feeder seabirds, marine mammals, representing top predators, benthopelagic cephalopods, benthic cephalopods, Gadidae, piscivorous fish, benthos feeder fish, Labridae, Sparidae, flatfish, planktivorous fish, commercial decapods, benthic invertebrates as Predators, Scavenger/Omnivorous, Filter, surface deposit feeder, subsurface deposit feeder, meiofauna, Zooplankton, Bacteria, Phytoplankton and Detritus. For all those groups, data was collected from the literature and/or from field measurements.

Table I. – Criteria to define managerial type.

Criteria	Differences	Managerial type
Objectives	Production	–
	Protection	–
Survey	Scientific survey	1
	Dissemination of results (report, publication, oral communication)	2
Management	Limited access	1
	Supervised activities	2
	Supervised sites	2
Authorization	Upload legal concession	1
	New immersion	2
Communication	Press articles	1
	Press articles and public awareness campaign	2
Funding	Occasional	1
	Annual	2

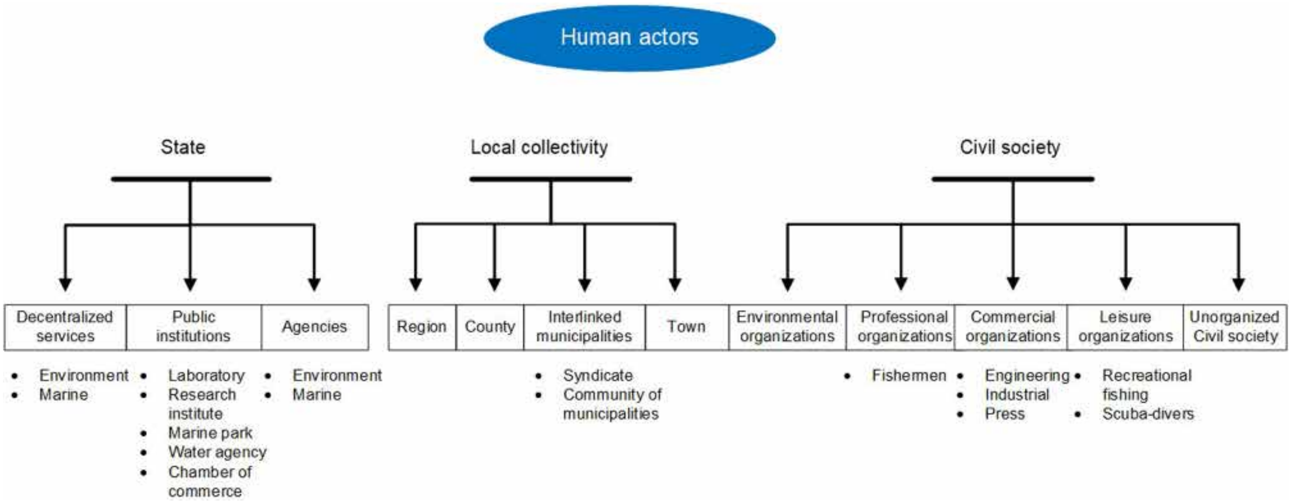


Fig. 3. – Type of social structures of artificial reef actors (J. Salaün).

Graph theory and network analysis: In the social-ecological network, nodes represent social actors of AR and links between interactions. Human actors have been grouped within three types of social structure (Fig. 3): state, local collectivity or civil society (Meur-Ferec 2006). They are also characterized by the type of actions that they carry out with regard to AR (financial support, technical support, governance, management, monitoring, users, etc.). All relations between actors can be divided into four general groups: information flow, technical material flow, monetary flow or human and skills flow. Links are directed from donor to receiver and are unweighed. Quantifying their strength or intensity will require a high degree of data that are not always available, such as the frequency of information flow. The network is analyzed using the iGraph package in R v3.5.2 (R Core Team 2019).

RESULTS

The revealed goals of AR

Each territorial actor has its own goal to achieve. The sociology of translation makes it possible to reveal them. Generally, the territorial actors of AR are:

– French State with DREAL, DIRM, DDTM, AFB and scientists from IFREMER or CNRS or universities;

– Local authority of Regional, *Département* or municipal territories;

– Civil society including environmental associations, professional associations, professional organizations, leisure associations and commercial activities.

They have different goals: developing marine activities, protecting their professional activity, improving knowledge of marine systems, enforcing regulation, promoting their business, their political group or territory, developing tourism, etc. The translation of their own goals in relation with the AR immersion project will reveal their real objectives.

Analysis of ecosystem organization

Ecological Network Analysis (ENA) is one well-known method to quantify how species interact with and influence their environment (Haak *et al.* 2017). For example, some of these indicators are:

– The Total System Throughput: it is the overall flows of the network;

– The effective Trophic Level (TL) indicates the effective position of species in the trophic network of AR;

– The Omnivory Index (OI) is a measure representing the diversity of the trophic level prey of a predator (Christensen & Walters 2004). This indicator indicates the selective predators that are fully dependent on their prey.

The Mixed Trophic Impact (MTI) routine is used to assessing the effect that biomass modification of one group could have on the biomass of other groups in the network (Ulanowicz & Puccia 1990).

The keystones index completes the analysis by assessing the effect that a minimal biomass variation will have on the biomass of another group (Libralato *et al.* 2006). From this assessment it is possible to understand the relative importance of the top-down or bottom-up trophic control in this AR ecosystem.

Analysis of social system

As the social systems are built before, after 5 years of immersion and nowadays, comparison of the architecture of those systems should indicate the key steps of their evolution. Some indicators such as density, degree centrality and betweenness centrality will be calculated to analyze models.

The density indicator measures the connectivity of a global web, by dividing the total number of connections present by the total number of possible connections (Kong *et al.* 2019). In AR networks, the increase or decrease in density indicates the involvement of territorial actors. The degree centrality represents the interconnection of network nodes, corresponding to the nodes' relation activities (number of neighbors of a node). Articulation points or betweenness centrality is a node that when it is gone, separates the network into pieces. This node plays a key role like a bridge between two distinct groups of actors. All these indicators make it possible to reveal key actors of the network and enable us to understand their connection within the AR network. This analysis helps in the design of effective management strategy and facilitates the comprehension of their functioning (Kluger *et al.* 2019).

DISCUSSION

AR ecosystem model

Trophic network modeling has been developed over decades and has been applied to various marine ecosystems around the world. This approach has been particularly used to understand the effect of fisheries on the entire ecosystem (Trites *et al.* 1999, Coll *et al.* 2006, Lassalle *et al.* 2011, Banaru *et al.* 2013, Moullec 2015, Bentorcha *et al.* 2017, Bentley *et al.* 2018). Recently, this approach has been extended to other research domains such as management of Marine Protected Area (Valls *et al.* 2012, Hermosillo-Núñez *et al.* 2018), rehabilitation measures (Espinosa-Romero *et al.* 2011, Guan *et al.* 2016) or to simulate the effect of wind farming on the ecosystem (Pezy *et al.* 2017, Raoux *et al.* 2018). Like wind farms, AR are mostly deployed on soft bottom habitats. They create hard substrate that will be colonized by different populations. Using ENA on this AR ecosystem makes it

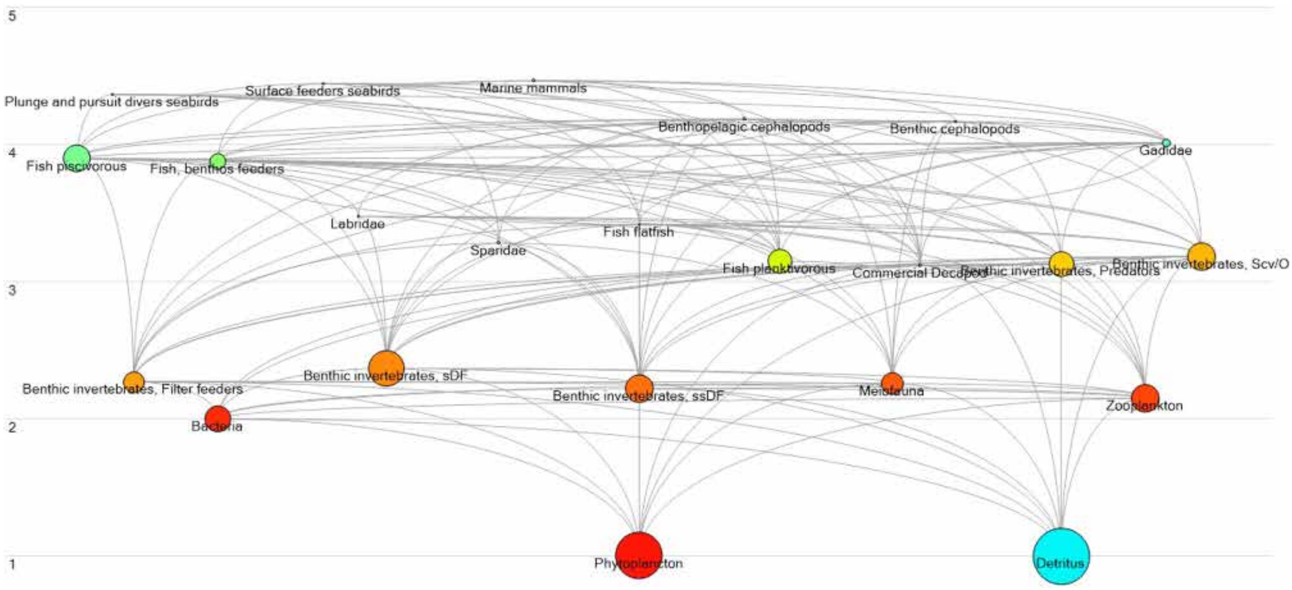


Fig. 4. – Trophic modeling of Cherbourg before installation of Artificial Reefs (adapted from A. Raoux).

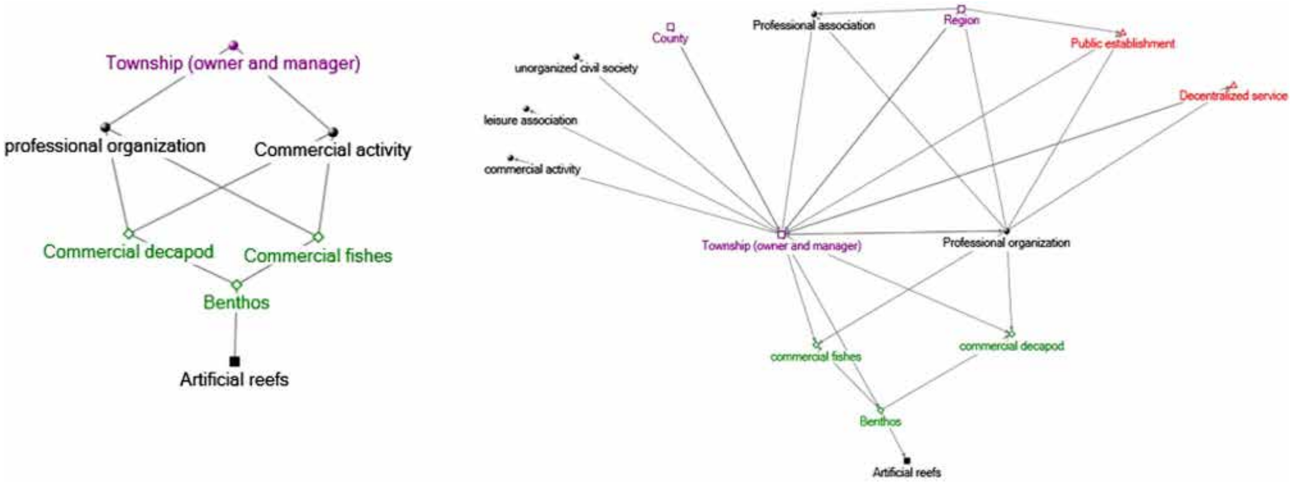


Fig. 5. – Example of social-ecological system of two Artificial Reefs areas: soft management on the left and active management on the right (J. Salaün).

possible to highlight the trophic modification linked to the introduction of hard substrate on soft habitats (Fig. 4).

Social-ecological model

Modeling before AR, after five years of immersion, and nowadays, offers the means to assess the social-ecological efficacy of AR. The assessment compares the identified goals to the actual results from a holistic point of view. These assessments show that, depending of the structure of the social-ecological network (Fig. 5) but also in function of the social type of the manager, results (social and ecological) are different. Furthermore, some actors such as fishers' organizations are essential and crucial at the beginning of the project but less interested during the "exploitation" step. In contrast, some actors could appear in the network only in this phase, such as leisure or environmental associations. Analyses of these assessments reveal the best social-ecological organization for each AR goal. This result can be used for future projects as an example of a social-ecological project adapted to the objectives of AR.

CONCLUSION: MANAGEMENT OF COASTAL AREAS

The uses of AR in function of new goals, such as ecological restoration (functionalities: nursery, spawning or protection) are a valuable help for the management of coastal activities (tourism, diving, artisanal and recreational fisheries). Recent AR deployments have targeted fisheries production, development of coastal activities (recreational, eco-mooring) and ecological restoration to offset human negative impacts (water filtering, nursery). They are innovative with regard to their ecological objectives but also materials and their proposal of new territorial dynamic governance goals to manage coastal areas (Pioch & Léocadie 2017). They try to address issues linked to diving for recreational activities, innovative biomimetic AR dedicated to targeted species production such as the spiny lobster *Palinurus elephas* (Fabricius, 1787) or the common dentex *Dentex dentex* (Linnaeus, 1758), the association between ecological restoration and mooring systems, or to enhance water filtration at the Marseille urban sewage treatment plant (Pioch *et al.* 2019). As the purposes and actors around AR are very diverse, the social-ecological approach is of interest to develop an overall management strategy for coastal areas under multi-use pressures.

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CHANGES IN THE ECOLOGICAL STATUS AND NATURAL CAPITAL OF *POSIDONIA OCEANICA* MEADOWS DUE TO HUMAN PRESSURE AND EXTREME EVENTS

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SEAGRASS MEADOWS
EMERGY ANALYSIS
MARINE PROTECTED AREAS
DISTURBANCES
LIGURIA (ITALY)

ABSTRACT. – Littorals represent highly dynamic and complex systems which undergo changes imposed by several environmental factors and human-induced disturbances. Among coastal ecosystems, seagrass meadows represent a key habitat and, according to the European Water Framework Directive, the endemic Mediterranean *Posidonia oceanica* is considered as biological indicators thanks to their susceptibility to pressures and changes. In this work, four *P. oceanica* meadows in the Portofino Marine Protected Area (NW Mediterranean) are investigated to evaluate the potential effects of different disturbances (*i.e.*, anchoring and a severe sea storm) on them through the study of their conservation status (measured with the Conservation Index, CI) and their natural capital (NC). Results obtained for CI and NC are not always consistent: meadows with high conservation status often showed low NC values. A link between the ability of ecosystems to store NC and develop a complex functioning and the meadows conservation is evident only when ecosystems are subjected to strong disturbances like the sea storm that hit the Ligurian coast on October 2018. The two indices should thus be integrated in monitoring activities because they account for different and complementary aspects of the meadow status.

INTRODUCTION

Seagrasses support complex marine food webs and provide essential habitat for many coastal species, playing a critical role in the equilibrium of coastal ecosystems and human livelihoods (Short *et al*. 2011).

They are present in all coastal areas of the world, except for Antarctic shores, forming meadows that have important ecological functions. In fact, seagrass beds are highly productive ecosystems, provide habitat and nursery areas for a variety of invertebrates, fish and mammals (Francour 1997), and enhance water quality by stabilizing sediments, removing nutrients and concentrating and retaining toxic chemicals in their tissues (Lewis & Devereux 2009).

An iconic example of seagrass is represented by the endemic Mediterranean *Posidonia oceanica* (Linnaeus) Delile, whose meadows are able to protect the coast, buffering waves and currents (Terrados & Duarte 2000).

Posidonia oceanica meadows have been listed as priority natural habitat to be included in the Sites of Community Interest (SCIs), for which special plans of management and conservation must be designated (EEC 1992). Due to its wide distribution, long-life and susceptibility to changing environmental conditions, *P. oceanica* is considered a good biological indicator of water quality and health (Pergent-Martini *et al.* 2005), in accordance with the Annex V of the Water Framework Directive (WFD, 000/60/EC) (Foden & Brazier 2007).

Since the early 20th century, seagrasses have been experiencing a global crisis, as highlighted by decreasing coverage and associated biodiversity loss worldwide (Orth *et al.* 2006, Telesca *et al.* 2015, Thomson *et al.* 2015). Loss of meadows has been attributed to the combined effects of direct human activities (*i.e.*, habitat fragmentation, eutrophication, pollution, overfishing and biological invasions) and global climate change, both challenging their adaptability (Waycott *et al.* 2009).

An alarming decline of the *P. oceanica* meadows has been reported in the Mediterranean Sea and mainly in the north-western side of the basin (Ardizzone *et al.* 2006, Montefalcone *et al.* 2007a, Boudouresque *et al.* 2009, Montefalcone *et al.* 2010), where many meadows have already lost their original extension during last decades (Marbà *et al.* 1996, Bianchi & Morri 2000, Leriche *et al.* 2006, Montefalcone *et al.* 2007b, Burgos *et al.* 2017).

Disturbance is a key factor influencing the structure of ecological assemblages and evolution of species within ecosystems (Dornelas 2010, Ponge 2013). The degradation of habitats such as *P. oceanica* meadows means losing a series of ecosystem functions and services useful for the maintenance of the coastal marine system but also for human activities. There is an urgent need to quantify and estimate the ecological effects of natural and human disturbances to guide conservation efforts and the management of ecological resources.

This decline has been proved in response to human impacts that produce changes in water quality (Delgado *et al.* 1997, 1999, Dimech *et al.* 2000, Ruíz *et al.* 2001, Cancemi *et al.* 2003), mechanical erosion (Sánchez Lizaso *et al.* 1990, García Charton *et al.* 1993, Martín *et al.* 1997, Francour *et al.* 1999, Milazzo *et al.* 2002, 2004) or burial (Manzanera *et al.* 1998, Fernández Torquemada & Sánchez Lizaso 2005, González Correa *et al.* 2008), but also because meadows of *P. oceanica* are often affected by hasty environmental alterations resulting from natural phenomena. There is an urgent need to quantify and estimate the ecological effects of natural and human disturbances to guide conservation efforts and the management of ecological resources.

Ecosystem services have been defined as the benefits people obtain from ecosystems (Millennium Ecosystem Assessment 2005, TEEB 2010). In the last decades it became clear that ecosystem services provision completely depends upon ecosystems and their natural capital. As a consequence it is increasingly being emphasized that the measurement of the status of natural capital stocks, and not just the marginal valuation of flows of services and benefits, is vital to ensure that services can be provided in the future (HM Treasury 2018). Therefore, efficient management of natural resources and environmental assets requires adequate assessment of natural capital (Azad *et al.* 2020).

This study was carried out in the context of the Interreg GIREPAM (Integrated management of ecological networks through parks and marine areas) project for the Marine Protected Area (MPA) of Portofino (NW Mediterranean). The aim was to evaluate the effects of disturbances, such as anchoring and a severe sea storm on *P. oceanica* meadows.

Covering the seabed from the surface down to about 40 m depth, meadows of *P. oceanica* are often affected by direct mechanical damages caused by boat anchoring and mooring activities (Francour *et al.* 1999; Montefalcone *et al.* 2008). Boat anchoring can lead to seagrass meadow fragmentation and to formation of patches, thus modifying the seascape configuration (Meinesz & Lefèvre 1984, Kiparissis *et al.* 2011, Okudan *et al.* 2011). In fact, the impacts of anchoring systems on *P. oceanica* have been shown to impose huge stresses on the meadow, pulling up leaves and rhizomes (Walker *et al.* 1989, Hastings *et al.* 1995, Ceccherelli *et al.* 2007) and reducing shoot density and cover of the meadow (Francour *et al.* 1999). Notwithstanding, protection measures undertaken by the European Community for their conservation, *P. oceanica* meadows keep on being affected by this kind of impacts, which are hardly controlled within marine protected areas (La Manna *et al.* 2015).

Similarly, extreme storm events cause significant ecological shifts, and their occurrence is likely to increase due to climate change (IPCC, 2019) and it is considered a major environmental concern (Easterling *et al.* 2000, Harley *et al.* 2006). In fact, among natural factors, water movement, such as that associated with waves and currents, appears to be a main factor influencing the *P. oceanica* meadow structure at both within-meadow and seascape scales (Abadie *et al.* 2018).

Pace *et al.* (2017) showed that at shallow depth (6-11 m), high-energy wave climate leads to an increase of meadow patchiness and a decrease in architectural complexity. At greater depths also, even if negligible, currents derived from wave energy result in a decrease of meadow cover provoking the generation of patches of different bottom type (bare matte or sandy bottom) (Vacchi *et al.* 2010, Gobert *et al.* 2016, Abadie *et al.* 2017). Species often have a lower capacity to adapt to sudden events rather than to gradual changes (Wernberg *et al.* 2012, Smale & Wernberg 2013).

Two metrics were employed to assess effects of these two disturbances on *P. oceanica* meadows: the evaluation of conservation status through the application of Conservation Index (CI) (Moreno *et al.* 2001, Montefalcone 2009) and the Natural Capital (NC) evaluated through the emergy analysis (Odum 1988, 1996).

The CI was used to get information about meadows conservation status and their potential to recover.

Emergy analysis was applied to quantify the value and the changes in Natural Capital (NC) stock due to disturbances. NC is composed by all biophysical elements and it is an economic metaphor for the limited stocks of physical and biological resources (Costanza & Daly 1992). NC includes land, air, water, sea and ecosystems therein: a tight link exists between ecosystem services provision and NC since only if NC is preserved intact the supply of services in the future and at the actual level can be guaranteed (De Groot *et al.* 2012). Emergy analysis can be classified as a donor side approach since it accounts for the environmental effort, in terms of resources used, required to generate a certain product or service. For this purpose, emergy evaluates the convergence of matter and energy from several inputs to a system on a common basis: the equivalent solar energy required to maintain a process. The NC stocked within an ecosystem is then assessed as the environmental resources spent in space and time to create it (Vassallo *et al.* 2017).

The comparison between information obtained through CI and NC allowed making considerations about the ability of these two measures to record changes imposed by nature or humans.

MATERIALS AND METHODS

Study area: The MPA of Portofino (Liguria Region, NW Italy) hosts many tourist activities, especially during the summer period, that have the potential to damage coastal habitats. Nonetheless, some restrictions on tourist activities are imposed by the MPA. In particular, the MPA is divided into three zones: A, B and C, each with a different degree of accessibility. In the zone A (integral reserve) the system is fully preserved, being forbidden recreational and professional activities and permitted only rescue and scientific research activities. The zone B (general reserve) is characterized by wider constraints: recreational fishing is allowed (regulated) only to residents, scuba diving is allowed to diving centers and authorized individuals, while free bathing is allowed. The zone C (partial reserve), renowned for its large meadows of *P. oceanica*, has less restrictions allowing. Other activities are allowed, as underwater activity, recreational fishing, bathing, mooring and anchoring, considering limitations for the protection of the area. For management issues, the MPA is divided into 19 sectors moving from East to West as shown in Fig. 1.

Four *P. oceanica* meadows along the Portofino promontory have been investigated: Niasca (44°18’05.35”N; 9°12’44.98”E), Cervara (44°19’05.56”N; 9°12’43.74”E), Punta Pedale (44°19’12.92”N; 9°12’47.41”E) and San Rocco (44°20’01.07”N; 09°09’13.08”E) (Fig. 1). All meadows are within the zones C of the Portofino MPA.

Field activities: Surveys have been done during summer months, and particularly in 2005, 2011, 2017, 2018, but also after the severe sea storm of 29 October 2018. In each site, two divers moved along a transect perpendicular to the coast, collecting data from the lower to the upper limit of the meadow. The two operators independently estimated the percentage cover of living *P. oceanica* (on sand and on rock), dead matte and mosaic of *P. oceanica* and dead matte (henceforth mosaic) along each transect and recorded the linear occupancy (in meters) of each descriptor.

Conservation index: To evaluate the conservation status and the degree of alteration that the *P. oceanica* meadows have undergone, the Conservation Index (CI) was calculated (Moreno

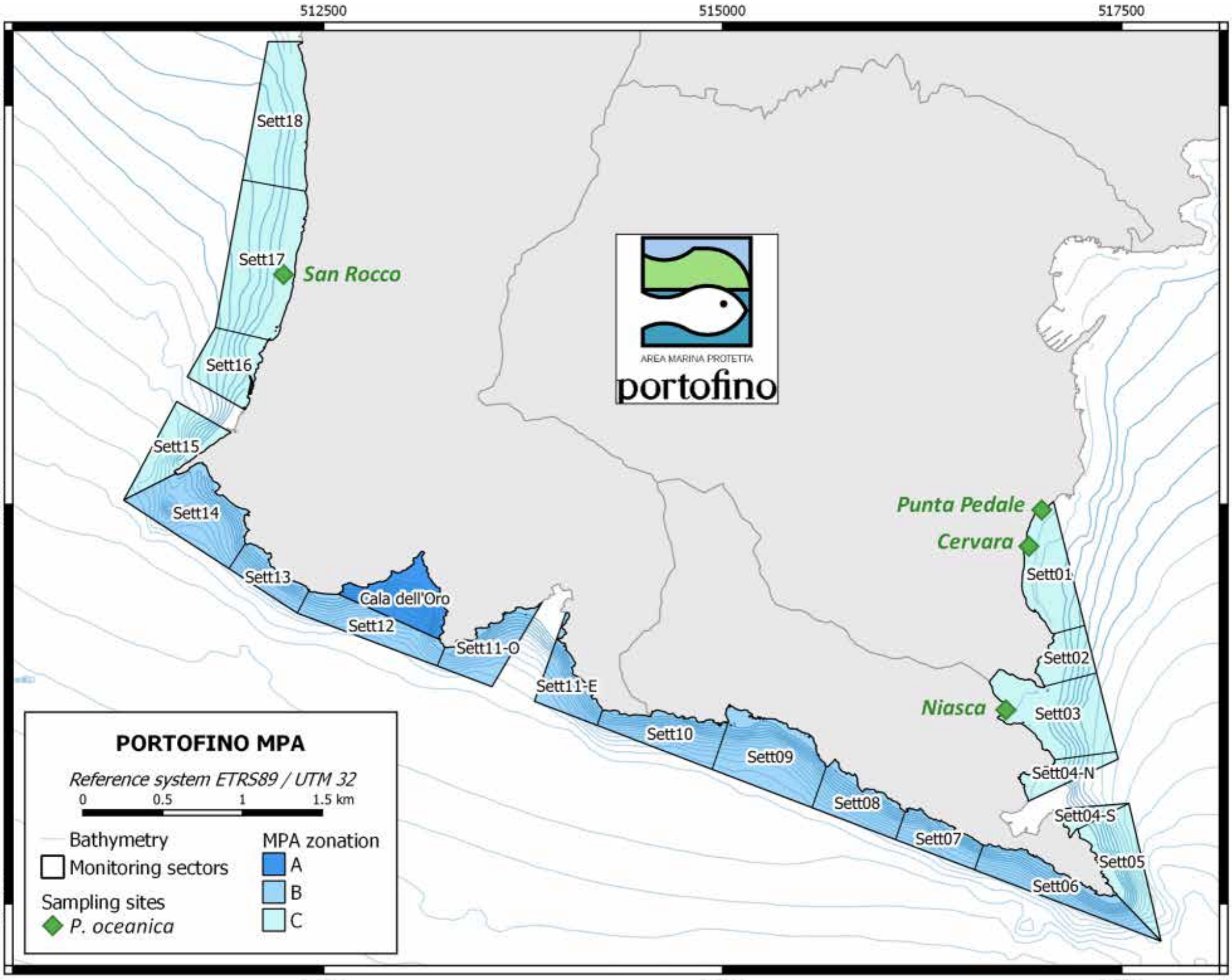

Fig. 1. – The Marine Protected Area of Portofino (Italy), with the four meadows investigated: San Rocco, Niasca, Cervara, and Punta Pedale.

et al. 2001) basing on the following formula:

$$CI = P / (P + D)$$

where P is the percentage cover of living *P. oceanica* (on sand, on rock and living part of mosaic), D is the percentage cover of dead matte determined for each transect.

The index ranges between 0 (maximum state of alteration or minimum state of conservation, where only dead matte is present), and 1 (maximum state of conservation, where no dead matte is present).

Natural capital through time: NC had been already assessed by emergy analysis (Brown & Ulgiati 1999, Odum & Odum 2000, Pulselli *et al.* 2011, Franzese *et al.* 2015, Vassallo *et al.* 2017, Paoli *et al.* 2018). Emergy is a thermodynamic method (Odum 1983, 1996) able to analyze the overall functioning of a system and to ascribe a value to it.

In particular, emergy converts the effort made by the environment (measured as resources, space and time invested) to produce biomass stock (donor side approach) into a monetary value. The value calculated with emergy corresponds to the amount of natural resources (*e.g.*, nutrients, rain) used (directly and indirectly) to build up and maintain the biomass of all the organisms within the habitat (Vassallo *et al*. 2017).

These resources are then represented in a single unit of measurement (solar emergy joules – sej) and later expressed in monetary terms (emergy-Euros – em€) using an appropriate conversion factor. The emergy baseline 15.20E+24 sej (Brown & Ulgiati 2010) was used to calculate emergy in this study. Here the 9.60E+11 sej/€ ratio is employed (Pereira *et al.* 2013).

Solar energy is used up, directly and indirectly, in transformations chains happening in the biosphere: this energy is a measure of the work done to provide a flow or a service and of the investment made by nature and can be considered a proxy of NC value as production cost (Odum 1996, 2000, Odum & Odum 2000, Pulselli *et al.* 2011).

In this work, the surface that the studied descriptors (*P. oceanica* on rock, *P. oceanica* on sand, mosaic, dead matte) covered in 2005 was calculated from the map of Ligurian marine habitats classified by Diviacco & Coppo (2006). The surfaces of descriptors in years 2011, 2017 and 2018 were calculated from field activities data. In particular, the percentages of *P. oceanica* on sand, on rock, mosaic and dead matte were calculated as the ratio between the length of each descriptor and the length of the entire transect. The percentages variation obtained for a specific time frame (*e.g.*, 2005-2011) were applied to the 2005 cartographic surface to obtain the extent of descriptors in each year. NC values per unit of surface of the descriptors were accounted in previous studies (Vassallo *et al.* 2017, Paoli *et al.* 2018). Multiplying the calculated surfaces by biophysical values in emergy terms (sej m^{-2}) and monetary terms^{-2}, the overall values were evaluated to detect the changes of NC in the analyzed period of time, looking for effects due to different restriction/protection actions or caused by the severe storm.

In addition, in each site and year, the variation in the total value of NC was calculated, thus taking into account all the descriptors (*P. oceanica* on rock, *P. oceanica* on sand, mosaic and dead matte).

Anchoring pressure: This study was carried out in the zones C of the Portofino MPA where anchoring was allowed in the considered period (2005-2018). During the summer of each year, a monitoring of boats presence in zones C of the MPA was carried out on an annual basis. In order to easily identify and count boats and to make comparisons over the years, the number of small and medium sized boats was counted for each sector in which the Portofino MPA is divided for monitoring purposes (Venturini *et al.* 2016).

The damage of anchoring was evaluated as removed surface of *P. oceanica* on sand and mosaic, excluding *P. oceanica* on rock that is not suitable for anchorage. The calculation of these surfaces was based on the number of shoots removed by a single anchor considering previous studies (Francour *et al.* 1999, Milazzo *et al.* 2002, 2004, Lloret *et al.* 2008). The quantity of shoots removed was converted in the corresponding degraded surface and multiplied by the corresponding number of anchorages in the considered sector.

The effect of anchoring on the meadows was evaluated assessing the NC in three sectors of the MPA: sector 17 (San Rocco), sector 3 (Niasca) and sector 1 (Cervara e Punta Pedale) (Fig. 1).

The values obtained through this calculation represent the amount of NC subtracted from the overall values estimated as described in the previous paragraph.

To obtain the percentages of surface and NC removed by the impact of anchoring the following formula was used:

$$P = La_{(i-j)} / O_i \qquad (1)$$

where:

P = percentage of surface or NC loss;

$La_{(i-j)}$ = loss of surface or NC due to anchorages from the year i to the year j;

O_i = overall surface or NC in the year i.

Sea storm pressure: The sea storm of 29 October 2018, that hit the Ligurian coast, could be considered the cause of important damages on marine habitats. During this event gale, torrential rainfalls and extremely rough sea caused catastrophic consequences on the anthropic coastal structures, particularly along the coastal area surrounding the Portofino Promontory (Betti *et al*. 2020). For example, the parapet surrounding the Portofino cape lighthouse, placed 30 m over the sea level, was widely destroyed and the littoral road from S. Margherita Ligure to Portofino (SP227) completely collapsed for about 200 m. This storm, with SE winds exceeding 130 km/h and generating 10 m high waves, changed the coast morphology, due to the fall out of large rocky boulders.

At the purpose of evaluating storm effects on the four investigated *P. oceanica* meadows in Portofino MPA, two sampling campaigns, before and after the sea storm, were carried out through the application of CI and NC evaluation.

Finally, by comparing the difference between the NC values (as the sum of all descriptors) before and after the storm and the

Table I. – Linear occupancy (in meters) and cover (in %) of the meadow descriptors (PR: *Posidonia oceanica* on rock, PS: *P. oceanica* on sand, MOS: mosaic of *P. oceanica* and dead matte, DM: dead matte) and CI values in the four meadows investigated in each sampling period. * data obtained after the severe sea storm of October 2018.

Sites	Years	Transect length (m)	PR (m)	PS (m)	MOS (m)	DM (m)	PR (%)	PS (%)	MOS (%)	DM (%)	CI
San Rocco	2005	205	45	100	40	20	21.95	48.78	19.51	9.76	0.81
	2011	230	10	150	15	55	4.35	65.22	6.52	23.91	0.66
	2018	280	41	145	64	30	14.64	51.79	22.86	10.71	0.69
	2018*	290	20	140	90	40	6.90	48.28	31.03	13.79	0.66
Niasca	2005	40	0	30	5	5	0.00	75.00	12.50	12.50	0.61
	2018	32	0	10	2	0	0.00	83.33	16.67	0.00	0.77
	2018*	30	0	20	0	10	0.00	66.67	0.00	33.33	0.61
Cervara	2005	185	80	40	20	40	43.24	21.62	10.81	21.62	0.57
	2011	140	75	20	35	10	53.57	14.29	25.00	7.14	0.75
	2018	159	95	23	8	33	59.75	14.47	5.03	20.75	0.68
	2018*	235	10	0	110	115	4.26	0.00	46.81	48.94	0.47
Punta Pedale	2005	265	20	55	65	75	7.55	20.75	24.53	28.30	0.44
	2017	260	20	80	30	110	7.69	30.77	11.54	42.31	0.39
	2018*	240	0	0	80	160	0.00	0.00	33.33	66.67	0.27

initial value, it was possible to assess which sites were the most damaged due to the heavy event and the corresponding percentage of loss.

RESULTS

Conservation index

The conservation status of the four meadows showed high spatial and temporal variability (Table I).

In San Rocco the *P. oceanica* meadow showed a worsening of the conservation status between 2005 and 2011, which stabilized in the following years, also after the sea storm of October 2018.

The Niasca meadow showed a little improvement in the conservation from 2005 to 2018. Following the storm the status declined and returned to the condition detected in 2005.

Also the Cervara meadow underwent an improvement of values from 2005 to 2011, then a decline in 2018 and a sharp worsening after the sea storm.

Finally, Punta Pedale meadow maintained a low conservation status through time, with a further decrease after the sea storm.

Results reported a general CI decrease in all sites after the sea storm, which varies between 3 % (San Rocco) to 31 % (Cervara and Punta Pedale).

Natural capital through time

The NC values were calculated from information obtained from cartography and transects data. Value per unit area are reported in Table II, expressed in biophysical and monetary units (sej and em€). NC values showed different trends through time in the four sites. The percentage of variation between sampling years is shown in Fig. 2.

From 2005 to 2011, the meadow in San Rocco had a decrease in the NC value for *P. oceanica* on rock and mosaic of about 17.6 % and 12.9 %, respectively. These values recovered in 2018, partly for *P. oceanica* on rock and completely for mosaic. Concerning *P. oceanica* on sand, a 16.4 % increase in its NC value was recorded between 2005 and 2011. However, since 2011 inconstant losses were assessed over time.

The *P. oceanica* meadow in Niasca develops only on sand. Between 2005 and 2018 the NC value increased by 8.3 % for *P. oceanica* on sand and 4.2 % for mosaic, while the storm caused a NC loss for both descriptors (16.7 %) and a significant increase of dead matte (33.3 %).

NC of *P. oceanica* on rock in Cervara increased by 10.3 % from 2005 to 2011 and further by 6.2 % from 2011 to 2018. Instead, *P. oceanica* on sand decreased from 2005 to 2011 (7.3 %), remaining then stable until 2018. Mosaic in Cervara had an increase in NC values between 2005 and 2011 and a decrease in 2018.

In Punta Pedale site the meadow showed a stable condition of NC values for *P. oceanica* on rock and an increase for *P. oceanica* on sand and dead matte in the period between 2005 and 2017. Instead NC value associated with mosaic, decreasing from 2005 to 2017, resulted increased in 2018.

Anchoring pressure

Starting from the dataset on boats monitoring provided by the MPA, the estimation of the impacts (expressed

Table II. – Values of natural capital expressed in surface (m²), emergy (seJ) and monetary equivalents (em€) in the four meadows investigated in each sampling period. PR: *Posidonia oceanica* on rock, PS: *P. oceanica* on sand, MOS: mosaic of *P. oceanica* and dead matte, DM: dead matte.

Sites	MPA sectors	Years	Surface (1E+02 m²)					Natural Capital (1E+15 seJ)					Natural Capital (1E+03 em€)				
			PR	PS	MOS	DM	Total	PR	PS	MOS	DM	Total	PR	PS	MOS	DM	Total
San Rocco	17	2005	49.00	124.94	61.54	0.00	235.48	301.45	767.52	215.94	0.00	1284.92	314.02	799.50	224.94	0.00	1,338.46
		2011	40.37	145.48	53.55	0.00	239.40	248.39	893.68	187.89	0.00	1329.96	258.74	930.92	195.72	0.00	1,385.37
		2018	44.53	125.94	62.30	0.00	232.76	273.96	773.64	218.58	0.00	1266.19	285.37	805.88	227.69	0.00	1,318.94
		2018*	41.08	121.52	67.39	0.00	229.99	252.74	746.49	236.46	0.00	1235.68	263.27	777.59	246.31	0.00	1,287.17
Niasca	3	2005	0.00	9.01	0.82	9.37	19.19	0.00	50.30	2.40	15.10	67.81	0.00	52.40	2.50	15.73	70.63
		2018	0.00	9.76	0.85	8.20	18.80	0.00	54.50	2.50	13.21	70.21	0.00	56.77	2.61	13.76	73.14
		2018*	0.00	8.13	0.71	10.93	19.77	0.00	45.41	2.09	17.62	65.12	0.00	47.31	2.17	18.35	67.83
Cervara	1	2005	5.80	56.34	4.69	27.77	94.60	32.43	314.74	13.80	44.76	405.73	33.78	327.85	14.37	46.63	422.63
		2011	6.39	52.21	5.36	23.75	87.71	35.78	291.65	15.76	38.28	381.47	37.27	303.80	16.41	39.87	397.36
		2018	6.79	52.30	4.29	26.98	90.36	37.99	292.17	12.61	43.49	386.26	39.58	304.34	13.13	45.30	402.36
		2018*	3.02	44.74	6.08	34.59	88.42	16.91	249.91	17.88	55.75	340.44	17.61	260.32	18.62	58.07	354.63
Punta Pedale	1	2005	5.80	56.34	4.69	27.77	94.60	32.43	314.74	13.80	44.76	405.73	33.78	327.85	14.37	46.63	422.63
		2017	5.80	61.98	4.08	31.66	103.53	32.48	346.26	12.01	51.03	441.77	33.83	360.68	12.51	53.16	460.18
		2018*	5.36	42.91	4.97	39.37	92.61	29.98	239.72	14.62	63.46	347.78	31.23	249.70	15.23	66.10	362.27

as shoots, surface and NC losses) due to anchorages on *P. oceanica* meadows, for the investigated period between 2005 and 2018, is shown in Table III. The ratio between the losses reported in Table III and the initial overall NC values of the period reported in Table II gives the percentages of removed NC.

Considering the formula 1 reported in materials and methods, from 2005 to 2018, the San Rocco meadow reported a loss of *P. oceanica* on sand of 911.5 m² ($La_{(i-j)}$) (Table III) that correspond to the 0.7 % (P) of the initial overall surface (O_i). Concerning mosaic, the loss is 478.4 m² that corresponds to 0.8 % of the initial overall surface. These damages are equal to a 0.7 % decrease in NC values in both descriptors, that amount, in monetary terms, to 5,304.3 em€ for *P. oceanica* on sand and 1,465.7 em€ for mosaic.

Anchoring monitoring was not carried out in Niasca, because it is an area where anchoring was banned since the establishment of the MPA.

On the other hand, the meadow in Cervara showed a small loss of NC for both descriptors. The decrease of surface was 0.1 % of the initial overall surface for both *P. oceanica* on sand and mosaic during the entire period from 2005 to 2018. As consequence of the impact due to anchoring in this site NC diminishes of 2,458.4 em€ for *P. oceanica* on sand and 144.5 em€ for mosaic.

In Punta Pedale the decrease of surface was lower than 1 % for both descriptors. This decrease corresponds to a NC loss of 2,228.8 em€ for *P. oceanica* on sand and 131.0 em€ for mosaic.

Sea storm pressure

After the severe sea storm of October 2018 the meadow at San Rocco suffered a 7.8 % loss of NC for *P. oceanica* on rock, 3.5 % for *P. oceanica* on sand and a 8.2 % increase in mosaic compared to data collected in the same year during the summer season (Fig. 3).

These percentages expressed a loss of 22,106.0 and 28,285.2 em€ for *P. oceanica* on rock and on sand, respectively, and an increase in mosaic of 18,618.9 em€.

Niasca meadow experienced a 16.7 % loss for both *P. oceanica* on sand and mosaic, which correspond to 9,461.2 and 434.8 em€.

The meadow in Cervara reported a high loss of NC for *P. oceanica* on rock (55.5 %) but also for *P. oceanica* on sand (14.5 %).

Losses were therefore 21,962.2 em€ for the former and 44,024.7 em€ for the latter. On the contrary, the mosaic showed a 41.8 % increase of the NC, which in monetary equivalents corresponds to 5,487.3 em€.

Similarly, at Punta Pedale the meadow showed a decrease in *P. oceanica* both on rock (7.7 %) and on sand (30.8 %), representing losses of 2,602.6 em€ and 110,979.2 em€, respectively.

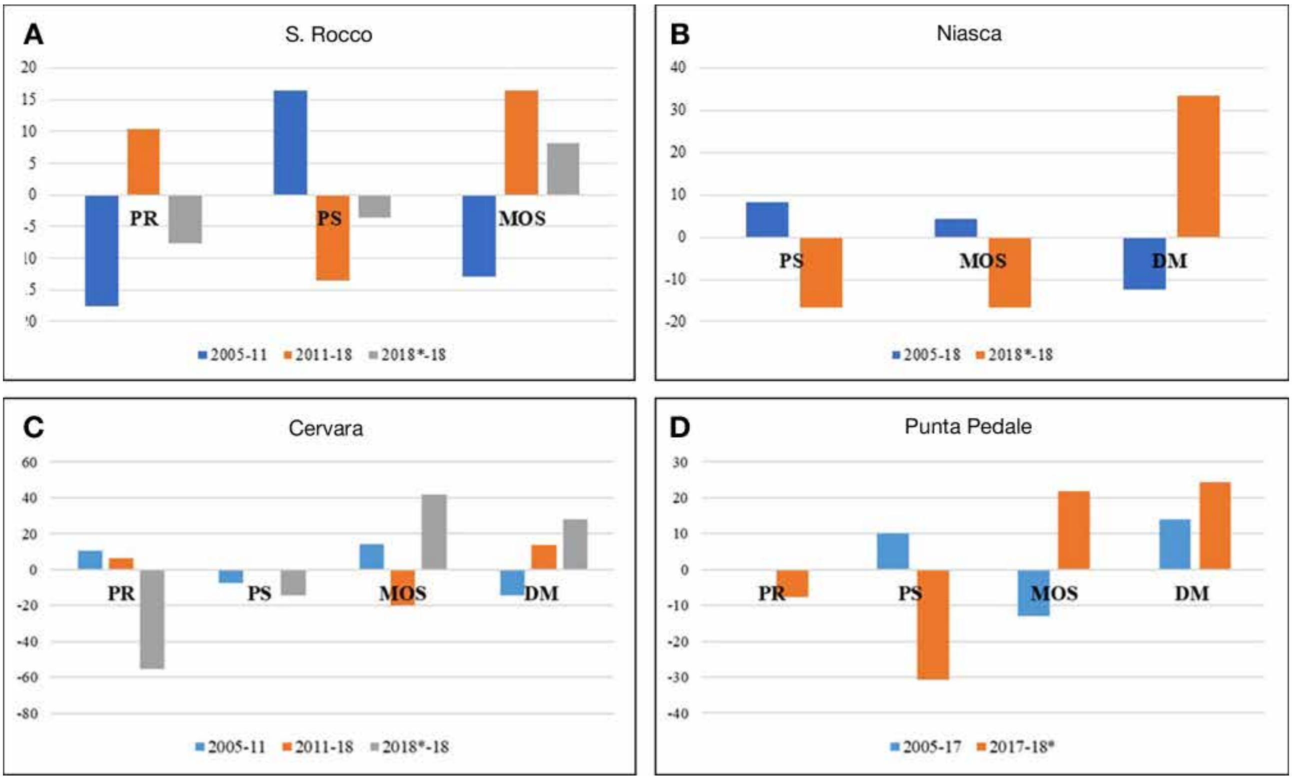

Fig. 2. – The percentage of NC variation between sampling years in each site. **A**: San Rocco; **B**: Niasca; **C**: Cervara; **D**: Punta Pedale.

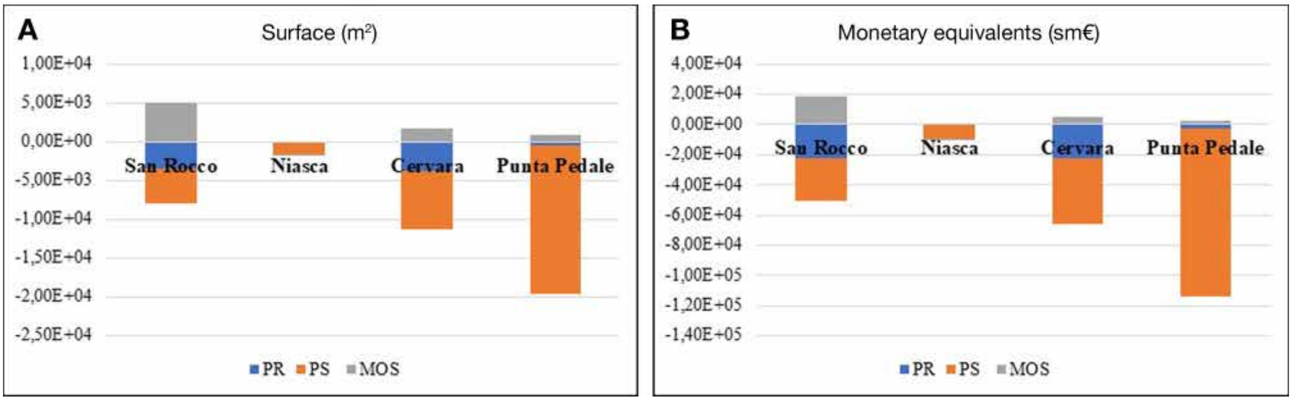

Fig. 3. – Changes in natural capital due to the sea storm of October 2018 expressed as (**A**) surface (m^2) and (**B**) monetary equivalents (em€**)** in the four meadows investigated in each sampling period.

Table III. – Changes in natural capital due to anchoring expressed as shoots (no), surface (m^2), emergy (sej) and monetary equivalents (em€**)** in the four meadows investigated in each sampling period. PR: *Posidonia oceanica* on rock, PS: *P. oceanica* on sand, MOS: mosaic of *P. oceanica*

Sites	MPA sectors	Years	Shoot (1E+03 no)			Surface (m²)			Natural Capital (1E+12 sej)			Natural Capital (em€)		
			PR	PS	MOS	PR	PS	MOS	PR	PS	MOS	PR	PS	MOS
San Rocco	17	2005-2011	ND	–17.3	–29.1	ND	–455.9	–231.6	ND	–363.8	–97.3	ND	–379.0	–101.4
		2011-2018	ND	–20.9	–31.0	ND	–455.7	–246.8	ND	–363.6	–103.7	ND	–378.8	–108.0
		2005-2018	***ND***	***–38.1***	***–60.0***	***ND***	***–911.5***	***–478.4***	***ND***	***–5092.1***	***–1407.1***	*ND*	***–5304.3***	***–1465.7***
Niasca	3	2005-2018	No anchoring zone											
Cervara	1	2005-2011	ND	–17.3	–1.8	ND	–24.9	–2.6	ND	–152.7	–9.0	ND	–159.0	–9.4
		2011-2018	ND	–20.9	–2.1	ND	–30.0	–3.1	ND	–184.5	–10.8	ND	–192.1	–11.3
		2005-2018	***ND***	***–38.1***	***–3.9***	***ND***	***–54.9***	***–5.6***	***ND***	***–2360.0***	***–138.7***	*ND*	***–2458.4***	***–144.5***
Punta Pedale	1	***2005-2017***	***ND***	***–34.6***	***–3.6***	***ND***	***–49.8***	***–5.1***	***ND***	***–2139.6***	***–125.8***	***ND***	***–2228.8***	***–131.0***

The increase of mosaic was instead 21.8 % compared to the period before the storm, corresponding to 2,725.6 em€.

Results reported a overall NC decrease in all sites after the sea storm which vary between 2 % (San Rocco) to 21 % (Punta Pedale).

DISCUSSION

Over the last three decades the interest in landscape ecology has grown and spread from land to marine ecosystems (Bell *et al*. 2006). In the marine realm, *P. oceanica* meadows are among systems analyzed to assess the conservation status of coastal areas (Montefalcone *et al*. 2013).

In Europe, the umbrella regulations for addressing the ecological quality of the coastal and marine systems are the Water Framework Directive (WFD, 2000/60/EC) for lakes, rivers, transitional and coastal waters and the Marine Strategy Framework Directive (MSFD, 2008/56/EC) for marine waters. To manage human pressures on marine environments, recent and worldwide-approved legislative instruments address the need to assess a system's condition (Borja & Dauer 2008). Both health status and natural capital assessment provide a system-wide analysis fulfilling to this requirement.

Therefore, in this work, human-induced and natural impacts that affect *P. oceanica* ecosystem conditions were investigated applying two methodologies based on the study of different properties: conservation status (using the Conservation Index) and the natural capital (using emergy analysis).

The case study of the Portofino MPA was examined because recently the *P. oceanica* meadows were affected by an overlap of several natural and anthropogenic disturbances. Among them, the impacts of boat anchoring and of a severe sea storm were analyzed to evaluate ecosystem changes.

Comparing MPA sectors, the impact of boats appeared 2.7 times greater in terms of NC in the western sector (*i.e.*, San Rocco) than in the eastern one (*i.e.*, Cervara and Punta Pedale) of the Portofino promontory. The southeast direction of the wind, which usually affects coasts of Portofino MPA, causes more turbulence on the eastern side. Therefore, boats tend to anchor in areas sheltered by the promontory on the western side, consequently reducing the impact due to anchoring on the east coast.

Only in eastern sector where anchoring is forbidden (*i.e.*, Niasca) the positive effect due to the ban is recorded by a 8.3 % increase in NC value for living *P. oceanica* and a 12.5 % decrease for dead matte between 2005 and 2018, with a consequent improvement in meadows conservation status.

Table IV. – Comparison between trends obtained through the Natural Capital (NC) and the Conservation Index (CI). The symbol "+" indicates a situation of increase, the symbol "–" a situation of decrease. * data obtained after the sea storm of October 2018.

Sites	Years	NC	CI
San Rocco	2005-2011	+	–
	2011-2018	–	+
	2018-2018*	–	–
Niasca	2005-2018	+	+
	2018-2018*	–	–
Cervara	2005-2011	–	+
	2011-2018	+	–
	2018-2018*	–	–
Punta Pedale	2005-2017	+	–
	2017-2018*	–	–

On the contrary, the loss of NC due to the severe sea storm of October 2018 was 12.7 % greater in the eastern side of the promontory than in the western side.

The CI confirmed this evaluation, showing a major decrease of status in meadows located in the eastern side. This was due to the main direction of the wind generating the storm: it formed in the southwest of Corsica and then released its force on the Ligurian coast, first along its eastern part and then along its western one (Betti *et al.* 2020). From October 26th, in fact, the wind direction was initially from south-east until the afternoon of October 29th when suddenly it increased its intensity reaching a speed of 130 km.h^{-1} and producing waves up to 10 m high (Betti *et al*. 2020). This intense storm lasted until the early morning of October 30th, when the wind turned to southwest.

Comparing damage of anchoring and storm, reduction of NC per unit area due to a single day of sea storm was about 32 times greater than the impact of anchoring over 13 years. Therefore, the force of the storm caused heavy consequences on meadows, already brittle because of the chronic impact due to seasonal anchoring.

This study allowed pointing-out different aspects of CI and NC for the evaluation of the environmental status. The comparison of the two indices in the same time frame, in fact, showed that they were consistent only in the 50% of the situations (Table IV). However, excluding the decrease in values obtained for both CI and NC due to the heavy sea storm at all sites, the indices showed discordant signals. This uncoupling can be explained considering the different nature of the two metrics.

The CI is a widely used index for the assessment of the conservation status of *P. oceanica* meadows because of its simple formulation and the ease of data collection on field. However, CI has some limitations, especially related to the fluctuating nature of soft substrates. Distribution of these substrates can vary as a function of both time and hydrodynamic conditions: dead matte areas might be buried and hidden by sand or, *vice versa*, waves and currents may remove sand and expose dead matte areas or create sand corridors due to a natural constant erosion activity on seagrass meadow (Pasqualini *et al.* 2001, Gobert *et al.* 2016, Vacchi *et al.* 2016). Dead matte surfaces might decrease (when buried), while live *P. oceanica* remains constant. CI is an intensive measure being a ratio between the cover of living *P. oceanica* and that of dead matte: an increase in the CI value might thus result without a real improvement in the meadow conservation status.

From an operational perspective, it is advisable to avoid the use of the "mosaic" notation during field activity whereas note more accurately the percentage of *P. oceanica* on sand cover and dead matte. This is expected to avoid misinterpretations and to lose some precision. It is also sometimes difficult to understand when a CI variation can be attributed to a specific human-induced or natural disturbance. Commonly the presence of dead matte has been misinterpreted as an unequivocal sign of human impact. Nonetheless, occurrence of dead matte can be due to natural events (Boudouresque *et al.* 2006), such as hydrodynamics that can alter the meadow status.

The NC, on the contrary, represents an extensive measure, because it considers the surface of each descriptor (*P. oceanica* on rock, *P. oceanica* on sand, mosaic and dead matte) and their biophysical value. When the surfaces of these descriptors change, the damage or the improvement of the system is then assessed as the sum of the lost or acquired NC values over the years. The status of the meadow is dependent by the intrinsic value of the descriptors and the surface they occupy. Indeed, dead matte, corresponding to an undesirable condition in comparison with living meadow, has a halved value compared to *P. oceanica* on sand and on rock (Paoli *et al*. 2018).

Moreover, the mosaic, consisting of 50 % living *P. oceanica* and 50 % of dead matte, represents a lower NC value than living *P. oceanica* on rock or on sand. Over time, the *P. oceanica* on sand can become mosaic and be largely replaced by dead matte losing its value. For example, in the case of the sea storm in San Rocco and Cervara, an increase of mosaic is associated with a decrease of *P. oceanica* on sand. Thus, even when the dead matte or mosaic decrease without changes of living *P. oceanica* surfaces, NC diminishes and better represents the effective meadow status.

However, in this study the surfaces of the meadows, useful to assess NC, have been drawn from the digital cartography, but one of the major problems of image processing when applied to the marine environment is the impact of water column (of variable quality and thickness) (Pasqualini *et al*. 2001). The complexity of the investigated areas in term of topography, bathymetric range and water turbidity, can also alter the perception of the data and consequently the reliability of the result. It is thus important to dispose of suitable criteria for assessing reliability of maps (Pasqualini *et al*. 2001).

Therefore, the cause of the partial inconsistency between CI and NC can be attributed to the fact that changes in CI are dictated by variations of dead matte. This was verified in Cervara and Punta Pedale meadows, where an increase of dead matte occurred, without a decrease of *P. oceanica* on sand and on rock, thus causing an overall decrease of the meadow conservation status. On the contrary, NC was able to weigh losses and rises of the considered descriptors reporting a general increase of the meadows status.

Moreover, thanks to NC, it is possible to quantitatively assess how much a single disturbance affects the meadow: variations in NC due to anchoring and the severe sea storm were measured separately.

Despite all above, results showed that in the case of strong disturbances, such as the sea storm of October 2018, CI and NC were consistent. The sea storm caused a considerable reduction of *P. oceanica* surface, which resulted in a conservation decreased and in a loss of NC.

This decline was recorded at both meadow and sites scale due to the local worsening for all the descriptors taken into account.

As a conclusive remark it can be stated that it is not enough to use a single ecological index to identify the status of seagrass meadows. It is necessary, instead, to flank qualitative information with quantitative system and extensive indicators.

Combining ecological indices with the study of NC can be a potential effective approach, also considering that nowadays it is widely accepted that the load from human activities should not exceed the carrying capacity of the environment and that the NC must be kept intact.

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HOW TO STUDY THE SOCIO-ECOLOGICAL IMPACTS OF INVASIVE SPECIES: THE CASE OF THE CTENOPHORE *MNEMIOPSIS LEIDYI* IN BERRE LAGOON (SOUTHEAST FRANCE)

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MEDITERRANEAN LAGOONS
INTERDISCIPLINARITY
COMB JELLYFISH

ABSTRACT. – Berre Lagoon is an ecosystem historically disturbed by strong industrial discharges and significant freshwaters inputs from both natural and anthropogenic (hydroelectric powerplant) sources. While rehabilitation policies initiated in 1994 are already showing some success, the introduction and proliferation of *Mnemiopsis leidyi* since 2005 could limit their effectiveness. The originality of this study is to associate oceanography and sociology to estimate the impact of *M. leidyi* on the functioning of this socio-ecosystem. This article presents an overview of a broad range of results obtained using interdisciplinary approach. It appeared that *M. leidyi* has an impact on plankton community functioning and on human activities, particularly on fishing. Therefore, in the context of biological invasions, the prospects for the rehabilitation of Berre Lagoon remain subject to the unpredictability of nature.

INTRODUCTION

In the Mediterranean Sea, more than a thousand marine species are considered non-indigenous species (NIS) (Zenetos *et al*. 2010) and their numbers have been increasing exponentially since the beginning of the 20th century (Katsanevakis *et al*. 2009, Boudouresque & Verlaque 2010, Zenetos *et al*. 2010). In the Mediterranean Sea, there are several vectors of species introduction. The main vector is the opening of the Suez Canal between the Red and Mediterranean Seas allowing the entry of Lessepsian species into the Eastern Mediterranean basin (63 % of NIS observed in the Mediterranean; Boudouresque & Verlaque 2005). This canal has made the Mediterranean Sea a shortest maritime route for global maritime trade, linking America to Asia, leading to a high level of maritime traffic likely to introduce species (ballast water, fouling). Aquaculture and shellfish farming are also efficient vectors for the introduction of species accompanying fish and other mollusks (Boudouresque & Verlaque 2005, 2012).

While the majority of introduced species do not pose a direct risk to humans; however, 86 Mediterranean NIS are considered to have a strong socio-ecological impact (Katsanevakis *et al*. 2014), particularly in the coastal zone, the area most studied. Because of the various advantages they present for a range of industries (tourism, trade, agriculture, fishing), coastal environments are the most impacted by human pressures. At the same time, certain of the marine compartments are developing, such as the gelatinous zooplankton (Re *et al*. 2014, Halpern *et al*. 2015, Gibbs *et al*. 2017, Visbeck 2018).

In the Mediterranean Sea, eight species of gelatinous zooplankton are exotic. Five species of cnidarians entered the Mediterranean via the Suez Canal (*Phyllorhiza punctata* von Lendenfeld, 1884, *Cassiopea andromeda* (Forsskål, 1775), *Carybdea marsupialis* (Linnaeus, 1758), *Marivagia stellate* Galil & Gershwin 2010, *Rhopilema nomadica* Galil, Spanier & Ferguson 1990), while *Gonionemus vertens* A. Agassiz, 1862 was certainly introduced in polyp form with shellfish for shellfish farming (Marchessaux *et al*. 2017). The ctenophore *Mnemiopsis leidyi* A. Agassiz, 1865 has been transported in the ballast water of commercial ships and *Beroe ovata* Bruguière, 1789 has been accidentally introduced into the Black Sea (Bordehore *et al*. 2014, Brotz & Pauly 2016). The impact of these alien gelatinous species on the environment and on human activities has been relatively poorly documented. Katsanevakis *et al*. (2014) reported on the impact of some of them in Europe.

Mnemiopsis leidyi is a ctenophore (class Tentaculata, order Lobata) endemic to the American Atlantic coasts: South American coasts, Gulf of Mexico and Chesapeake Bay (Purcell *et al*. 2001, Mianzan *et al*. 2010, Costello *et al*. 2012) and which mainly colonizes estuaries, lagoons or inland seas (Purcell *et al*. 2001). Areas invaded by *M. leidyi* are generally subject to anthropogenic pressures such as eutrophication or overfishing, the latter favoring the success of *M. leidyi* (Bilio & Niermann 2004, Daskalov *et al*. 2007). *Mnemiopsis leidyi* was ranked among the 100 most invasive marine species in the world (Lowe *et al*. 2004).

Mnemiopsis leidyi appeared in the Black Sea in 1982 (Fig. 1) (Vinogradov *et al*. 1989, Reusch *et al*. 2010, Bolte *et al*. 2013, Ghabooli *et al*. 2013) and it has since spread to the Caspian Sea (Vinogradov *et al*. 1995), the Sea of Marmara (Isinibilir & Kideys 2004) and the Mediterranean Sea (Aegean Sea, Galil *et al*. 2009; Adriatic Sea,

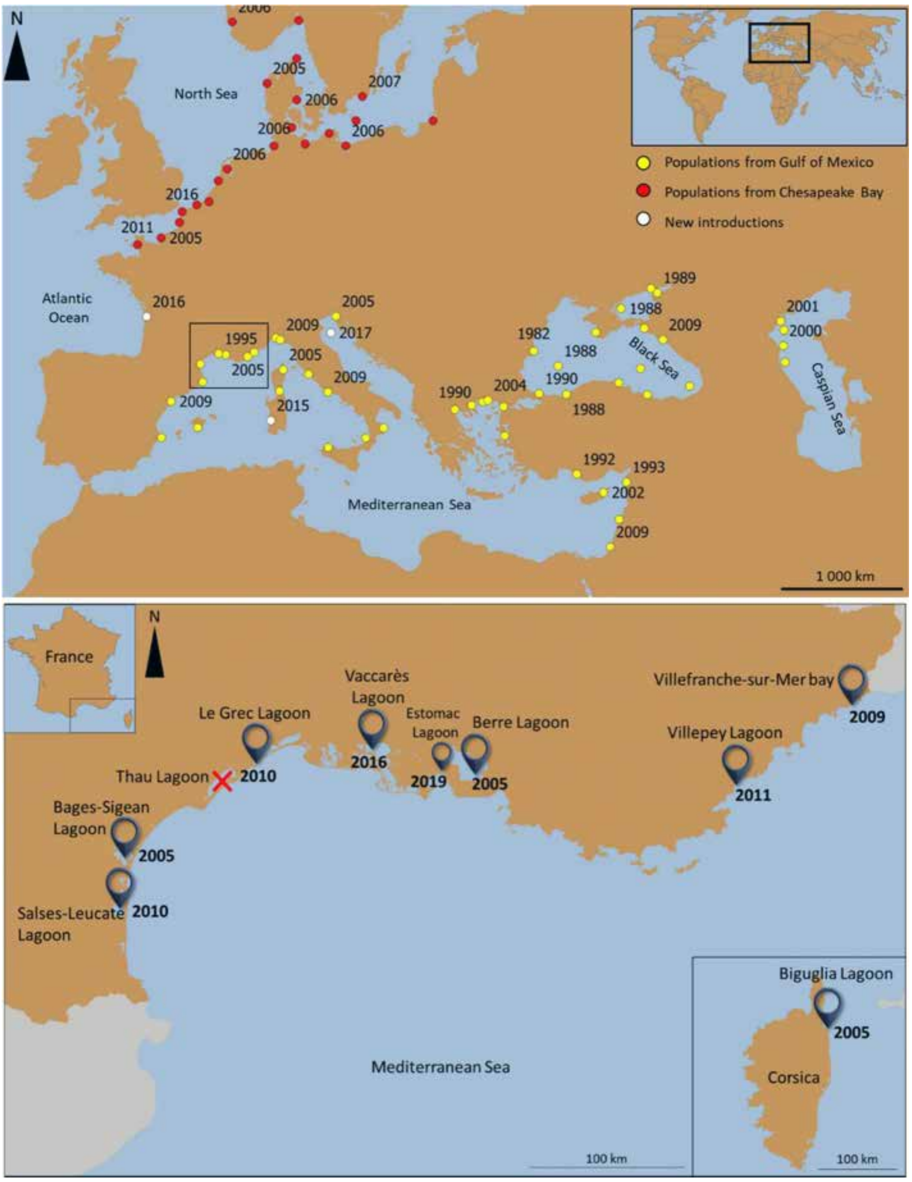

Fig. 1. – Distribution map of the introduction areas of *Mnemiopsis leidyi* in Europe (top) and on French Mediterranean coasts (below). Red cross: no *M. leidyi*.

Shiganova & Malej 2009; North-Western Basin, Boero *et al*. 2009, Fuentes *et al*. 2010, Bolte *et al*. 2013, Ghabooli *et al*. 2013, Marchessaux *et al*. 2020a, b) following the hydrodynamic circulation or again by means of shipping. More recently, new invasions were observed in northern Europe, in 2005 in Norway and the Baltic Sea (Javidpour *et al*. 2006, Oliveira 2007, Javidpour 2008). The organisms in this case were thought to originate from the east coast of the USA (Chesapeake Bay) (Reusch *et al*. 2010), while the Mediterranean populations were thought to originate from Mexico. On the French Mediterranean coastlines, *M. leidyi* was first formally identified in the Berre Lagoon in 2005 (Fig. 1, Table I, Marchessaux *et al*. 2020a), and has been episodically present in six other Mediterranean lagoons (Fig. 1, Table I) but the first arrival of *M. leidyi* along French Mediterranean coastlines could have occurred much earlier. *Mnemiopsis leidyi* has been observed in widely varying conditions of temperature (0 °C-31 °C), salinity (0.1-40) and chlorophyll *a* concentration (0.02-9.7 µg L^{-1}) reflecting its high ecological tolerance.

From the beginning of the 19th century, the rapid development of modern science led to the increased specialization of scientists. The (re)establishment of dialogue between disciplines is recent. From the 1970s onwards, in the field of the environment, it has been encouraged, if not driven by social demands that submit to researchers' questions that escape their disciplinary divisions (Jollivet 1992, Henry & Jollivet 2002, Claeys-Mekdade 2003, Gerini 2005). Thus, the overall functioning of an ecosystem cannot be explained in a compartmentalized approach alone (Legay 2004, Boudouresque *et al.* 2020). Neither the life sciences on the one hand nor the human sciences on the other are capable of separately providing

Table I. – Hydrological conditions of invaded French Mediterranean lagoons.

Date	Location	Latitude	Longitude	References
2005	Berre Lagoon	43.4592	5.1059	Marchessaux *et al.* (2020a)
2005	Bages-Sigean Lagoon	43.1054	2.9920	Delpy *et al.* (2016)
2005	Biguglia Lagoon	42.6263	9.4649	Etourneau (2011)
2009	Villefranche-sur-Mer bay	43.6994	7.3161	Fuentes *et al.* (2010)
2009-2010	Salses-Leucate Lagoon	42.8584	2.9956	Delpy *et al*. (2016)
2010	Le Grec Lagoon	43.5379	3.9433	Marchessaux & Belloni (2021)
2011	Villepey Lagoon	43.4047	6.7171	Marchessaux *et al.* (2020b)
2016	Vaccarès Lagoon	43.5353	4.6370	Marchessaux *et al.* (2020b)
2019	Estomac Lagoon	43.4445	4.9536	Marchessaux *et al.* (2020b)

complete answers to these questions. Thus, the researcher becomes a "*border-crosser*" (Jollivet 1992) bridging the life sciences and the human sciences in order to grasp the complexity of interactions between ecological and social processes.

Currently, the opening up of research is characterized by a process of knowledge dialogue (Jollivet & Legay 2005). Interdisciplinarity is one of the methods for sharing knowledge and fields (Jollivet & Legay 2005). Interdisciplinarity responds to the dual objective of an articulation between the disciplines involved while preserving the theoretical and methodological specificities of each. The different forms of collaboration between disciplines need to be clearly defined. Claeys & Thian-Bo-Morel (2015) proposed clarification of attempted to elucidate the differences between three possible forms of collaboration between disciplines: multidisciplinary, interdisciplinary and transdisciplinary: "*multidisciplinary can be defined as the aggregation of different disciplines, interdisciplinarity by interpenetration between several disciplines and finally transdisciplinary by challenging disciplinary boundaries that can produce meta-disciplinary*" (Claeys & Thian-Bo-Morel 2015). Interdisciplinarity responds to the dual objective of close interaction between the disciplines involved while preserving the theoretical and methodological specificities of each. Thus, each specialty retains its own study methods (protocols, concepts) but, within the framework of interdisciplinarity, it is essential for its success to create links to build bridges between these specialties (Charaudeau 2010).

This interdisciplinary approach is essential for studying biological invasions. The human aspect must be taken into consideration in the study of biological invasions for a global understanding of the ecology, the impact, and the societal challenge this phenomenon involves (Dalla Bernardina 2010, Javelle *et al*. 2010, Atlan & Darrot 2012, Pimentel 2014). Indeed, the accidental introduction and invasion of *M. leidyi* in Europe has had a strong impact on invaded ecosystems and human activities. From a sociological point of view, this study deals with a major contemporary issue, relating to what Giddens (1990) calls "*the consequences of modernity*" and its environmental paradoxes. The case study chosen, relating to the anthropogenic causes and consequences of the development of *M. leidyi*, is an innovative research theme in sociology that is part of a broader reflection on "*the place of the animal*" (Staszak *et al.* 2002) and "*the proliferating nature*" (Claeys & Sirost 2010).

The notion of invasive alien species proposed by Charles Sutherland Elton in his book "*The ecology of invasion by animals and plants*" (Elton 1958) really became widespread in the scientific literature from the 1980s onwards. It has been the subject of passionate academic controversies tending to oppose the human sciences and life sciences. In the United States, this debate was particularly crystallized around the publications of the philosopher Sagoff (1999, 2005), Simberloff (1998, 2003) and Simberloff *et al*. (2013), founder of the journal Biological Invasion. Sagoff was concerned about the lexical and potentially ideological similarities between the vocabulary of biological invasion ecology and that of xenophobic nationalist policies. In response, Simberloff (2003) defended a precautionary principle, advocating a posture of generalized suspicion towards all new alien species. It is worth noting that the book of Elton (1958) was based upon ecological concepts that were considered as widely outdated by modern ecologists (*e.g.*, Boudouresque & Verlaque 2012).

In France, the beginning of the 2000s was marked by the controversy over the invasive green macroalga *Caulerpa taxifolia* colonizing the *Posidonia oceanica* seagrass meadows of the Mediterranean. In this case, the oceanographers involved played the role of high-profile whistleblower (Meinesz *et al*. 1993, Boudouresque *et al*. 1995, De Villèle & Verlaque 1995, Meinesz *et al*. 2001, Humair *et al*. 2014). On the other hand, social scientists analyzed these scientific discourses and their media uptake their diffusion in the media as the fabrication of a modern myth (Dalla Bernardina 2010).

At the same time, sociological studies led by naturalist sponsors, notably the International Union for Conservation of Nature (IUCN), were developed (McNeely 2002). These studies were part of a social engineering approach using non-critical perception surveys aimed at formulating awareness campaigns to combat biological invasions. They have been well received in the journal Biological

Invasion, which expresses a cautious opening towards the human sciences (Simberloff *et al.* 2013). Nevertheless, this was a situation of "*science under influence*" (Jollivet 1992), depriving the social sciences of their critical power.

In between these two extremes, a dialogue that is both appeased and critical has emerged, often a little on the fringes of the academic power base. For example, in France, the hydrobiologist ethnologist were collaborating, producing constructive focused on around the invasive aquatic plant *Ludwigia peploides* (Menozzi & Dutartre 2007). Other experiments followed on the same principle of critical and peaceful co-construction (Kalaora 1998, Charpentier & Claeys-Mekdade 2006). What these different experiences have in common is that they have opted for "interdisciplinarity from below" (Zuindeau 2006). This mode of interdisciplinarity proposed an entry through objects and fields of study to calm epistemic tensions between the human and social sciences and the life sciences. It is a question, to use the expression of Brendon Larson (Larson 2005), of demilitarizing the problem of biological invasions, this "*war of the roses*".

This study on Berre Lagoon is in line with this "interdisciplinarity from below" approach (Zuindeau 2006). It involved submitting to investigation by natural and sociological sciences a complex subject, the proliferation of *M. leidyi*, which they could only fully elucidate through a co-understanding of the interactions between biological and anthropogenic processes in the socio-ecosystem of the Berre Lagoon. In this article an overview of the main results obtained based on a socio-ecological approach are presented to determine the potential impact of the invasive ctenophore *M. leidyi* on the socio-ecosystem of the Berre Lagoon.

MATERIALS AND METHODS

The creation of a common study protocol linking up the methodological tools of the different disciplines involved was a fundamental step in the implementation of interdisciplinary research. The interdisciplinary approach adopted in this study combined *in situ* oceanographic data, laboratory analysis and measurements with sociological investigations and citizen science (Fig. 2). This linkage provided a better understanding of the dynamics and possible impacts of the invasion of *Mnemiopsis leidyi* on the socio-ecosystem of the Berre Lagoon.

The conceptual framework of risk was used following formula: Risk = hazard × vulnerability. Risk resulted from the combination of hazards and vulnerability. Risk was defined as the result of damage caused because of interactions between disturbance factors (hazards) and vulnerability factors (use). The ctenophores represented the disturbance or hazard elements, and vulnerability was characterized by the more or less high exposure of humans (the practice of users) to the hazard (Veyret & Reghezza 2006, Gilbert 2009). In this study, hazard was measured by oceanography (presence, abundance of ctenophores, etc.) and vulnerability was analyzed by sociology (observation of lagoon frequentation patterns, behavior of users, etc.). Risk is also an object of discourse for individuals. In this case, the sociological survey records the levels of risk expressed by social actors and their ways of naming and qualifying these risks. In common parlance, when faced with the proliferation of an animal species, social actors tend to talk more about discomfort (Mieulet 2015). From this point of view, discomfort can be defined as the negative consequences experienced by social actors, in this case the users of the Berre Lagoon, when an event occurs, in this case the proliferation of gelatinous zooplankton.

A bimonthly monitoring (temperature, salinity, oxygen, chlorophyll *a*, abundances of zooplankton and ctenophores) was performed in the Berre Lagoon between October 2015 and September 2017 (Marchessaux *et al.* 2020a). Laboratory experiments were performed to study the physiology of *M. leidyi* (respiration, excretion, ingestion, digestion, reproduction, larval growth, and survival in starvation conditions) to determine its invasive potential. The human experiments were characterized by semi-directional interviews with different actors of the lagoons (fishermen, bathers, naturalists, yachting clubs, etc.). Finally, a citizen science was also performed to complete *in situ* monitoring and study the evolution of nuisance caused by cteno-

Fig. 2. – Diagram illustrating the strategy for interdisciplinary study of the dynamics and impact of *Mnemiopsis leidyi*. In blue: oceanography; orange: sociology.

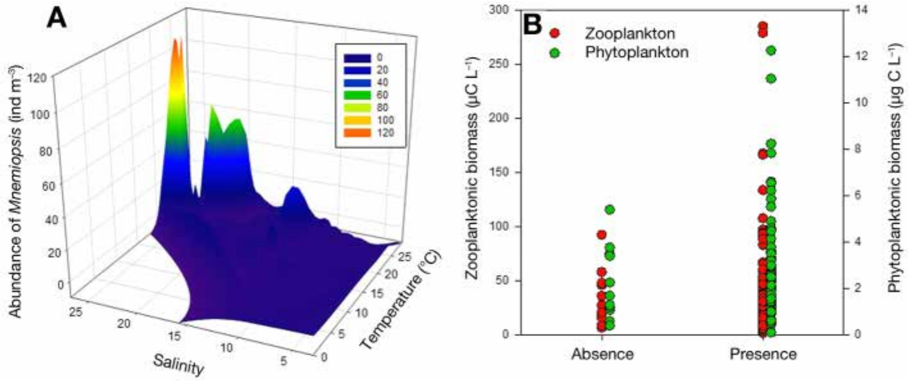

Fig. 3. – **A**: Three-dimensional representation of the abundance of *Mnemiopsis leidyi* (rainbow colors, ind m^{-3}) as a function of temperature (°C) and salinity in the Berre Lagoon; **B**: Zooplanktonic and phytoplanktonic biomasses (µg L^{-1}) during periods of absence or presence of *M. leidyi*. Data from 2010 to 2017 extracted from Marchessaux *et al*. (2020a).

phores proliferations. More details on the *in situ* monitoring are available in Marchessaux *et al.* (2020a) and, data presented in these articles were mainly extract from this article and presented differently.

RESULTS AND DISCUSSIONS

The main thread of this work was the perspective between human experiences and the measured. The human experiences concerned the experiences and testimonies of the social actors encountered. This human experience can be sensory, involving the body of individuals and their five senses, sight, touch, smell, taste, and hearing. This experience may be social, taking shape in the relationships between social actors and/or maybe economic, relating to the actors' losses and gains. Thus, it was a question of analyzing how different physical and socio-economic experiences could lead to a different understanding, definition, and quantification of the same phenomenon, in this case, the causes and consequences of the proliferation of gelatinous zooplankton.

What is the success rate of Mnemiopsis leidyi *invasions in the Berre Lagoon?*

Native to estuarine and lagoon environments, *Mnemiopsis leidyi* was able to maintain itself over a wide range of temperatures (1 °C to 32 °C; Kremer & Reeve 1989, Purcell *et al*. 2001, Lehtiniemi *et al*. 2012) and tolerated significant variations of salinity (0 to 40; Shiganova *et al*. 2004, Grove & Breitburd 2005). In the Berre Lagoon, *M. leidyi* was observed for temperature and salinity comprised between 3 °C and 28 °C and, 0 to 25, respectively (Fig. 3A) with highest abundances, 43 ± 24 ind m^{-3} (data range: 27 ind m^{-3}-95 ind m^{-3}), comprised between 19-25 for temperature and 22-28 for salinity. *Mnemiopsis leidyi* can also survive in areas with very little oxygen (> 1 mg O_2 L^{-1}; Decker *et al*. 2004, Grove & Breitburd 2005). Under unfavorable dietary conditions, *M. leidyi* can self-digest from its reserves and can thus survive for weeks (Ivanov *et al*. 2000, Yousefian & Kideys 2003, Anninsky *et al.* 2005).

The success of *Mnemiopsis leidyi* in Berre Lagoon was certainly due to a sufficient supply of carbon (Fig. 3B). According to Kremer (1994) the quantity of carbon limiting the development of *M. leidyi* was estimated at 3 µg C L^{-1}. Long-term monitoring data showed that the amount of particulate organic carbon (POC) did not fall below this threshold for the survival of *M. leidyi* (Marchessaux *et al.* 2020a). The sufficient supply of carbon in the Berre Lagoon therefore allowed *M. leidyi* to produce enough eggs (Reeve *et al*. 1989), and to have an optimal growth rate (Rapoza *et al*. 2005, Sullivan & Gifford 2007, Marchessaux 2019).

The other success of *M. leidyi* in native and introduced areas was possible thanks to the presence of refugia areas during winter (Costello *et al*. 2006, Marchessaux *et al.* 2020a). In the Berre Lagoon, the Vaine sub-basin provided favorable conditions (low temperature and salinity variations, sufficient carbon content, low currents) to maintain *M. leidyi* in critical environmental conditions (*i.e.*, cold winters; Fig. 4) (Marchessaux *et al*. 2020a).

Impact of Mnemiopsis leidyi *on the Berre Lagoon socio-ecosystem*

The notion of socio-ecosystem refers to the relationship between Society and the Environment and the effects that one can have on the other. Considering the links

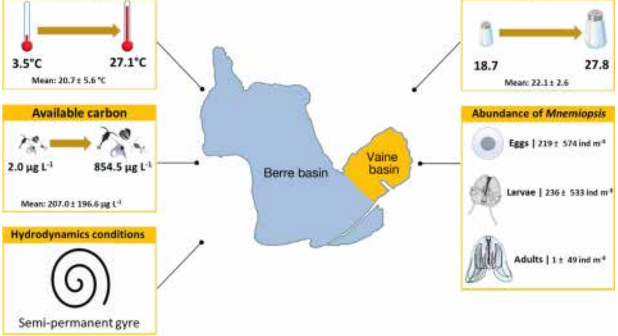

Fig. 4. – Assessment of the conditions measured in the refugia area of *M. leidyi* in the Berre Lagoon. Redrawn from Marchessaux *et al*. (2020a).

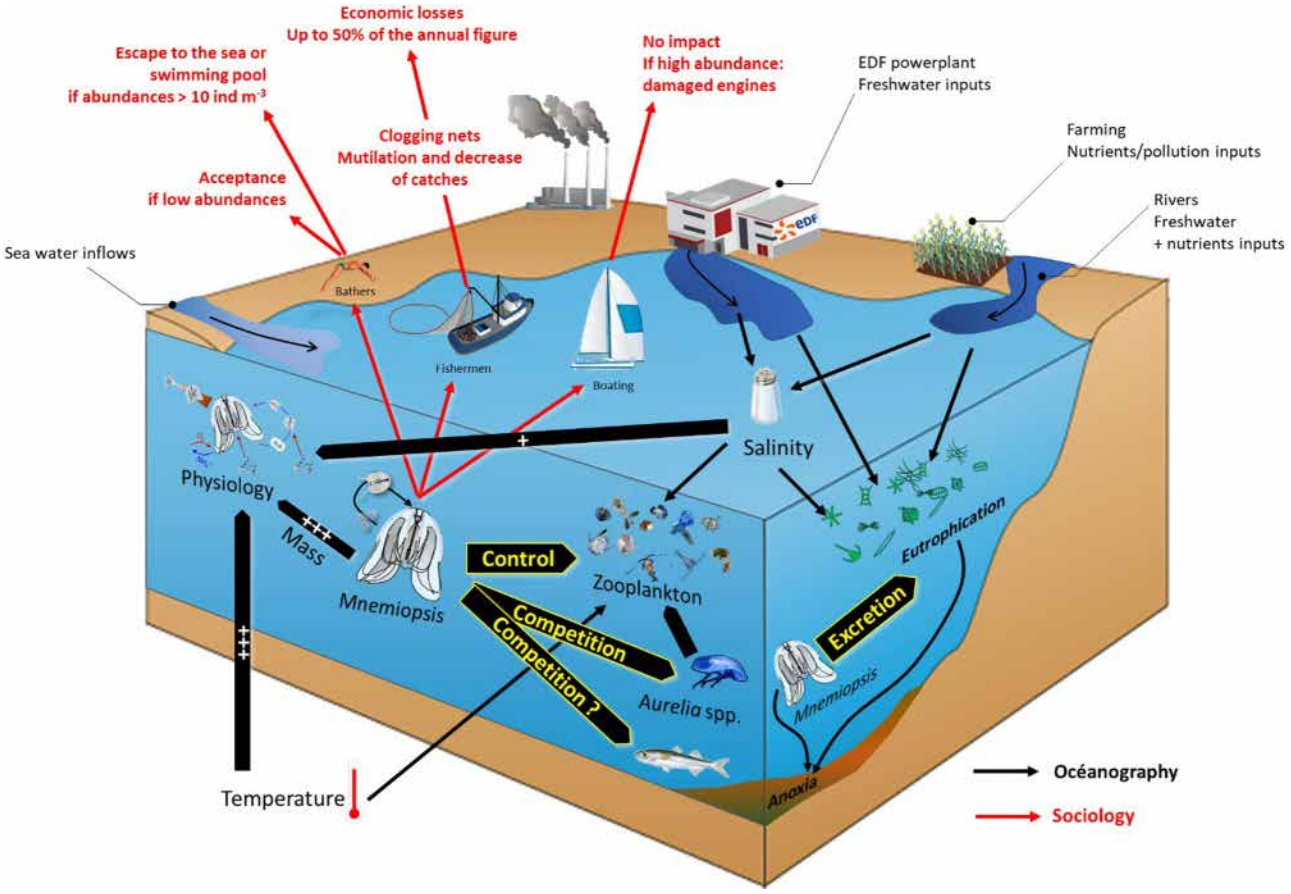

Fig. 5. – Overall summary of the interactions identified between *Mnemiopsis leidyi* and the socio-ecosystem of the Berre Lagoon.

between society and the environment, the framework of our interdisciplinary study allowed us to establish a complete inventory of the interactions and impacts of the invasion of *M. leidyi* (Fig. 5).

A voracious predator, *M. leidyi* had a significant predation pressure on the zooplankton community, whose grazing on phytoplankton remained limited in this context. The presence of *M. leidyi* contributed to the eutrophication of the lagoon on the one hand by controlling the zooplankton and on the other hand by contributing to phytoplankton development through the excretion of ammoniacal nitrogen (~ 3 %; Marchessaux 2019) as observed in Chesapeake Bay (3 %; Nemazie *et al*. 1993) and in York River (4 %; Condon *et al*. 2009). These hypoxic crises, which generally occur in summer, are fatal for populations of benthic organisms.

The potential competitiveness of *M. leidyi* for the resource had certainly contributed to the decline of the autochthonous cnidarian *Aurelia* spp. (jellyfish) whose observation in the environment became limited in time. In the light of this potential competition between the two gelatinous species, the hypothesis of competition with planktonophagous fish in the lagoon can be proposed.

This has not been tested in our study, but if there was competition, a decrease of commercial fish species could be observed, which could have catastrophic economic consequences for the professional fishery. Professional fishing was the human activity most impacted by gelatinous species blooms. The damage caused by *M. leidyi* was significant (clogging of nets, mutilation of catches, damage to equipment, increased workload) and involved losses of up to 50 % of the annual revenues. Bathing activity was little affected. The presence of *Mnemiopsis leidyi* was acceptable for bathers because this species is not stinging. On the other hand, this study has determined a threshold of bathers' acceptability (10 ± 8 ind m^{-3}). Beyond this threshold of acceptability, the swimmers deserted the beaches of the lagoon. Boating was not affected by the proliferation of *M. leidyi* but, in the case of strong proliferation, damage to the cooling systems of boat engines has been observed.

Mnemiopsis leidyi *invasion and rehabilitation efforts*

This interdisciplinary study has demonstrated the extent to which current European rehabilitation efforts were potentially counteracted by the presence of *Mnemiopsis leidyi*. To address this issue, based on the data collected in oceanography (abundances of ctenophores and zooplankton, ctenophores gut contents, physiology) and sociology (impact on different activities, discomfort of users, citizen science), we propose two scenarios (Fig. 6).

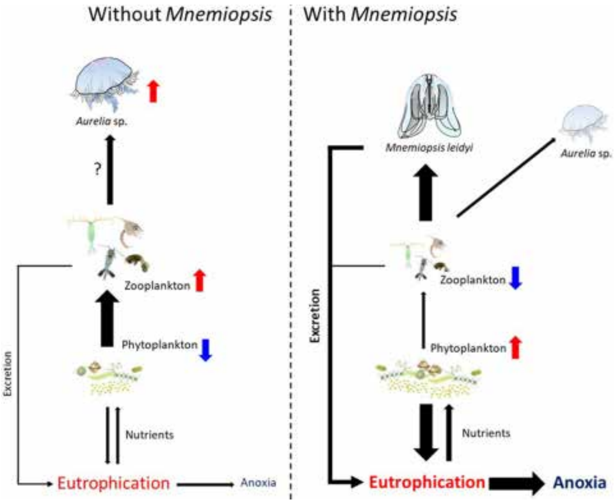

Fig. 6. – Conceptual diagram of the possible scenarios of changes in the functioning of the pelago-benthic food web in the Berre Lagoon in the absence or presence of *Mnemiopsis leidyi*.

Scenario 1 – No Mnemiopsis leidyi *(before introduction)*: Without *M. leidyi*, zooplankton exerts a "top-down" control over phytoplankton, decreasing in synergy with the decrease in freshwater and nutrient inputs inducing significantly low eutrophication of the lagoon and limiting hypoxic/anoxic crises. Without ctenophores, *Aurelia* spp. could recover in the lagoon.

Scenario 2 – With Mnemiopsis leidyi *(current configuration)*: The presence of a large population of *M. leidyi* counters rehabilitation efforts and helps to maintain eutrophication. *Mnemiopsis leidyi* controls the zooplankton community that can be ingested at a rate of up to 80 % of its abundance per day. This directly results in a reduction in zooplankton grazing pressure on phytoplankton communities. In addition, the N-NH_4 excretion of *M. leidyi* contributes 3.8 % of the ammonium inputs to the lagoon, which also promotes phytoplankton growth via regenerated production. Thus, by a "top-down" and "bottom-up" effect, the population has a favorable effect on the maintenance of eutrophication in the Berre Lagoon.

The issue of combating the proliferation of *M. leidyi* in the Berre Lagoon was raised by more than 80 % of the users encountered. Opinions suggest that controlling the proliferation of *M. leidyi* is one of the priority issues for future management of Berre Lagoon. The contribution of our interdisciplinary study enabled us, with knowledge of the users' wishes in terms of management, to report on the factors favoring the maintenance of *M. leidyi* and the interactions within the socio-ecosystem. Based on this observation, management measures will be proposed (anti-jellyfish nets, risk maps, etc.).

CONCLUSION

The convergence of life sciences (LS) and human and social sciences (HSS) responds to a social and political demand, both of which are growing in the context of the development of environmental policies and their application. This *rapprochement* of LS and HSS helps to overcome the problem of the life sciences considering pristine ecosystems in isolation from human influence and it is therefore necessary to consider the human factor in ecological studies. The combination of LS and HSS can give rise to misunderstandings, particularly from the point of view of methodology, which are mitigated by the interdisciplinary approach that consists in the interpenetration of disciplines aimed at the co-construction of scientific protocols and the coproduction of new knowledge that goes beyond the aggregation of disciplinary knowledge.

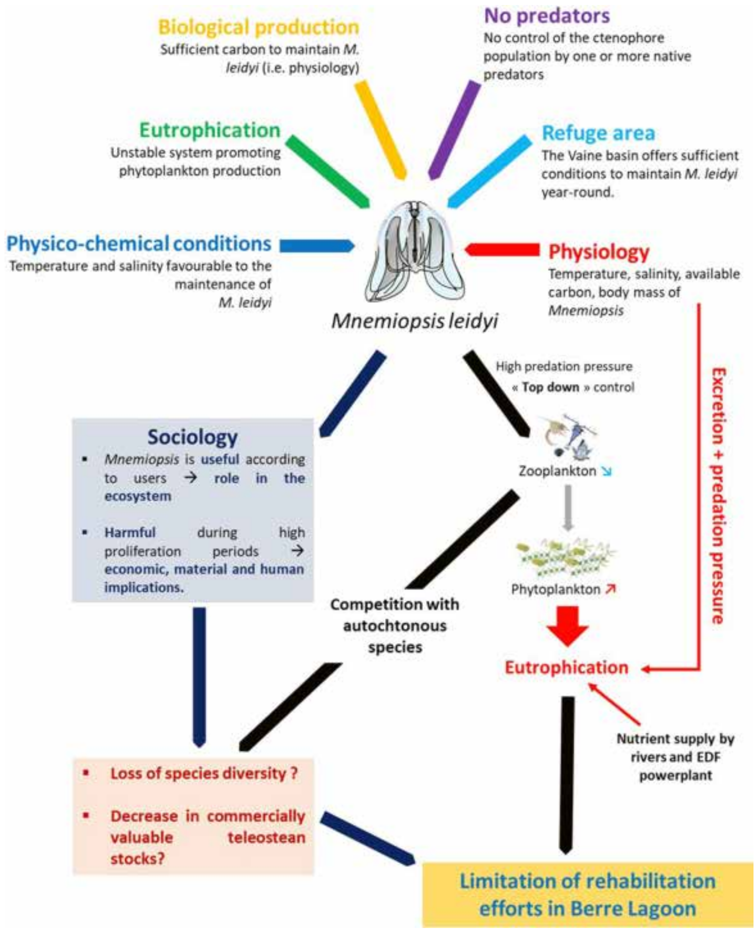

Fig. 7. – Summary diagram of the parameters effecting on the physiology of *Mnemiopsis leidyi* and its role in the socio-ecosystem of the Berre Lagoon.

In fact, the interests and objectives of each discipline are brought together in a reciprocal and mutually beneficial comprehension by excluding the false hierarchy between LS and HSS. The success of interdisciplinarity lies in the acceptance by the life sciences of the critical posture of the human and social sciences and, conversely, the recognition by HSS of the materiality of life as analyzed by the life sciences. This collective practice does not exclude the skills and methodologies of each discipline, but resides in an acculturation over time, respect, and mutual trust from a human and scientific point of view.

This study is in line with this bottom-up interdisciplinary approach, where the two disciplines involved had to deal with a complex subject, the proliferation of *M. leidyi*, which they can only fully understand through combined understanding of the interactions between biological and anthropogenic processes in the Berre Lagoon (Fig. 7). The interdisciplinary approach has fully proved its worth in this work. The testimonies of users have sometimes led us to seek/question ecological aspects that would not have been explored in the case of a life science study alone. Moreover, the contribution of sociology in this work has represented a real added value in that thanks to the network developed with the users of the lagoon, we were able to place able to obtain complementary data on the distribution and the dynamics *in situ* of the ctenophores and on the other hand the proliferation of *M. leidyi* was placed within the societal requirements concerning this socio-ecosystem.

In Berre Lagoon, the installation of *M. leidyi* in conjunction with the implementation of rehabilitation efforts in this case illustrates the fragility of the ecosystem historically disturbed by human activities. Heavy freshwater discharges from the EDF (*Électricité de France*) hydroelectric powerplant and industrial and urban discharges have considerably altered the hydrological functioning and have thus destroyed a large part of the lagoon's habitats. The anthropogenic pressures have allowed the arrival and development of invasive species (*i.e.*, *M. leidyi*, *Gonionemus vertens*, *Acartia tonsa* Dana, 1849, *Ulva* sp., etc.). Thus, Berre Lagoon represent anthropized place where ecological history has conditioned its management but also the fluctuations of nature.

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SMALL-SCALE CONNECTIVITY OF CORALLIGENOUS HABITATS: INSIGHTS FROM A MODELING APPROACH WITHIN A SEMI-OPENED MEDITERRANEAN BAY

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HABITAT CONNECTIVITY
LARVAL DISPERSAL
CORALLIGENOUS ASSEMBLAGES
BENTHIC ORGANISMS
OCEANIC CIRCULATION
LAGRANGIAN MODELLING
POPULATION GENETICS
COMMUNITY COMPOSITION

ABSTRACT. – Habitat connectivity is central to several key evolutionary and ecological processes, having implications for the spatial structuring of marine populations. For benthic species that have no or little mobility as adults, connectivity is evaluated by analyzing the dispersal of propagules across the seascape. We investigate fine-scale connectivity of coralligenous habitats in Marseille Bay (located in the north-western Mediterranean Sea) using high-resolution cartography and a particle-tracking model ran over a range of parameters derived from the biological traits of potential target species. We present annual and seasonal means of various connectivity diagnostics measuring the retention and exchange of propagules among coralligenous patches. A synthetic view is obtained by clustering individual patches into "coralligenous provinces" (ensemble of patches tightly connected by ocean currents). When discussing our results against historical observations, we highlight some genetic breaks, sharp community changes and ecological clusters that fit well our simulated connectivity patterns. Consistent findings include reduced biodiversity along Côte Bleue, high biodiversity at Planier Island and the presence of various dispersal barriers, which evolve with dispersal durations and provide the backbone of habitat connectivity. Our results help to apprehend and test hypothesis on marine population structures, providing useful information for ecologists and conservationists.

INTRODUCTION

The spatial structure of marine populations is a fundamental pre-requisite to understand, manage and protect marine ecosystems (Botsford *et al.* 2009, Dubois *et al.* 2016). At first order, the distribution of marine organisms is thought to be determined by the availability of habitats, which are heterogeneously distributed across the seascape, and by the movements of organisms among favorable areas. The latter process, which is referred to as "connectivity" (Rossi *et al.* 2016), influences key evolutionary (*e.g.*, speciation, long-term persistence, genetic structure, local adaptation) and ecological (invasion, colonization, demography, resilience to external perturbations) processes (Duputie & Massol 2013). Hence, population connectivity has also been related to biodiversity (Jones *et al.* 2007). Finally yet importantly, habitat connectivity is a key feature to optimize when prioritizing specific areas for protection and conservation (Olds *et al.* 2012).

Here we evaluate the small-scale connectivity of one emblematic rocky habitat, the coralligenous, which are endemic of the Mediterranean Sea and highly fragmented at all scales (Martin *et al.* 2015). Our study particularly applies to benthic species (sessile and demersal): since their adult phases have no or very low motility, the interrelationships among separated coralligenous patches are essentially controlled by the dispersal of propagules. Historical research (*e.g.*, ecological monitoring, population genetics, artificial reefs, etc.) as well as numerous information about the physical environment (models and observations of hydrography and hydrodynamics) make the Mediterranean Sea an ideal "natural laboratory" to study fine-scale habitat connectivity. With more than 1500 species, sessile or associated (about 315 algae, 1241 invertebrates and 110 fishes, according to Ballesteros 2006), coralligenous habitats contribute greatly to the Mediterranean biodiversity hotspot (Costantini *et al.* 2018), and are present along most Mediterranean shorelines (Martin *et al.* 2015). While some species inhabiting coralligenous habitats can be found elsewhere, there is likely more diversity per unit surface in coralligenous than in any other marine habitats as space is saturated by species and epiobiosis is frequent (Ballesteros 2006, Çinar *et al.* 2020). Due to their great diversity and accessibility to coastal populations, coralligenous habitats provide ecosystem services of all types (Thierry de Ville d'Avray *et al.* 2019). Species tightly depending on these habitats (*e.g.*, red coral, spiny lobster, grouper, etc.) are of high patrimonial and commercial values. Indeed, coralligenous landscapes are appreciated by divers and scientists for the potential of new discoveries, as many species remain understudied. Moreover, the calcareous algae forming bio-concretions, the basis of coralligenous habitats, have potential for carbon sequestration (Martin *et al.* 2013a). Last, the main ecosystem services provided by coralligenous (*e.g.*, habitat, nursery, refugee, food) are crucial to sustain marine Mediterranean populations (Thierry de Ville d'Avray, 2018).

Coralligenous reefs appear particularly vulnerable in the "Anthropocene". The coralline algae builders as well as the main animal bio-constructors have slow growth (Garrabou & Ballesteros 2000, Torrents *et al.* 2005, Rodriguez-Prieto 2016) and narrow thermal tolerances (adapted to relatively stable temperatures found below the thermocline). Hence, coralligenous habitats are particularly sensitive to marine heat waves, which are likely to become stronger and more frequent with global warming (Pairaud *et al.* 2014). It was shown by a few historical mortality events well documented in the North-Western Mediterranean (*e.g.*, Garrabou *et al.* 2009, Crisci *et al.* 2011) that resulted in a loss of both habitat complexity and biodiversity (Piazzi *et al.* 2012). As the unique limestone formation of biogenic origin (Ballesteros 2006), coralligenous bio-constructions are also threatened by ocean acidification since calcareous algae use dissolved inorganic carbon for photosynthesis and calcification (Martin *et al.* 2013b). Other noticeable threats are sedimentation, nutrient enrichment (from wastewater or river outflow) and biological invasions (mainly from the algae *Caulerpa cylindracea* [Sonder] and *Womersleyella setacea* [Hollenberg, Norris], Piazzi *et al.* 2012). When a coralligenous patch is negatively affected by mortality outbreaks, connectivity is the main process to ensure population persistence, favoring recolonization and/or access to refugee (Gerber *et al.* 2014).

We investigate fine-scale connectivity of coralligenous habitats and its potential impacts on benthic populations focusing on the bay of Marseille, a semi-opened embayment (about 40 km by 25 km, from the Gulf of Fos to the Riou archipelago, Fig. 1A) located at the eastern entrance of the Gulf of Lion in the North-West Mediterranean Sea. The coastal circulation is influenced by various forcing conditions including complex bathymetry, strong surface winds, the Rhône River outflow and the large-scale circulation (cyclonic Northern Current NC) producing intermittent hydrodynamic patterns (Millet *et al.* 2018). Both dominant wind regimes, that are North-westerlies (Mistral) and South-easterlies, respectively induce upwelling and downwelling processes (Pairaud *et al.* 2011, Fraysse *et al.* 2013). While Marseille bay can be affected by episodic arrival of eutrophic (cold, fresh and nutrient-rich) waters from the Rhône River plume (Fraysse *et al.* 2014), the southern sector is usually under the influence of oligotrophic waters brought by the NC (Fig. 1A). Oligotrophic conditions may also reach the northern sector on rare occasions when, instead of its usual westward offshore flow, the Northern current intrudes the continental shelf of the Gulf of Lion (Ross *et al.* 2016). Due to their high fragmentation (Fig. 1B) and the chaotic circulation of the bay, the connectivity of coralligenous habitat is difficult to apprehend and mostly unknown to-date.

Previous multi-specific studies of coralligenous in Marseille Bay suggested that genetic diversity differs among sites and that barriers to gene flows occur within the bay. For instance, Cahill *et al.* (2017) highlighted the presence of genetic differentiation for many species between the Côte Bleue and the northern bay (Fig. 1) but the presence of a physical dispersal barrier has not been investigated yet. De Jode *et al.* (2019) also found community differentiation between crustose algae (builder) communities located apparently in the same area, somewhere between Côte Bleue and Calanques. Only few studies already investigated the hydrodynamic connectivity of Marseille Bay to compare against the genetic structures of seaweed, which is not a coralligenous dweller (Thibault *et al.* 2016), and to track the fate of wastewater among a few specific sites (Millet *et al.* 2018). An exhaustive evaluation of connectivity at small-scale and considering the precise distribution of such specific and emblematic habitats is, to our knowledge, still missing in this region.

This paper presents a statistical description of fine-scale habitat connectivity for several time scales and a range of parameters that should be relevant for some benthic species typical of coralligenous. We model the retention and exchange of propagules (*e.g.*, eggs, larvae, spores, body fragments, rafts, etc.) driven by ocean currents among habitat patches in the bay of Marseille. We provide annually- and seasonally-averaged estimates of three complementary connectivity metrics, namely Local Retention, Self-Recruitment and Source-Sink. Furthermore, to give a synthetic view of the interlinked network of coralligenous habitats, we cluster all individual patches into a set of "provinces" (*e.g.*, ensemble of patches, possibly disconnected geographically, which are tightly connected by ocean currents) and report mean statistics for each of them. By comparing observed spatial structure with the set of simulated spatial structures for a corresponding set of dispersal traits, one may infer the most likely dispersal traits of a target species whose dispersal abilities are unknown. We finally discuss our results against the literature, highlighting that some dispersal predictions are plausible explanations for observed genetic structures or biodiversity patterns.

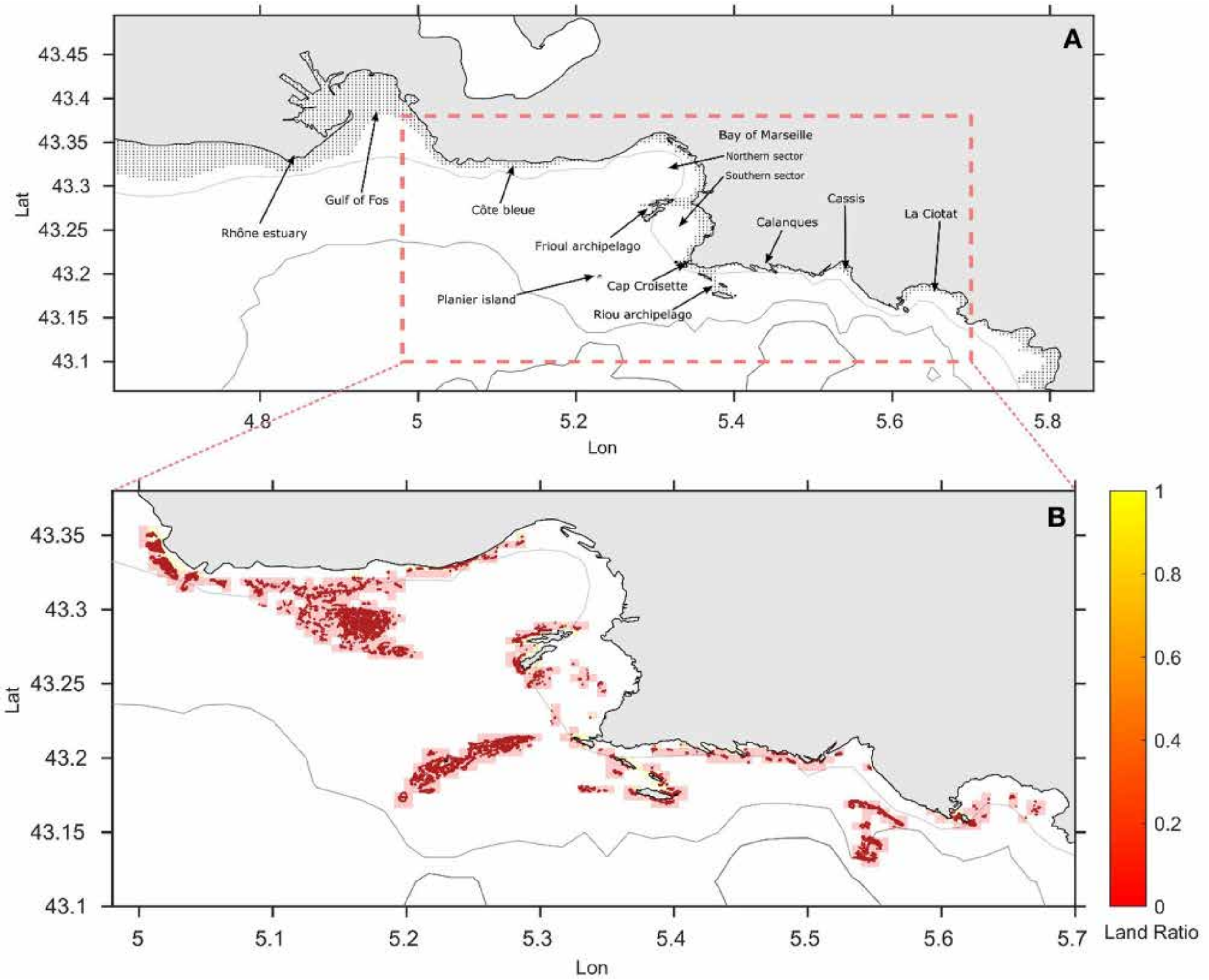

Fig. 1. – Maps of the study area. Panel (**A**) represents the domain covered by the RHOMA hydrodynamical model (black dots symbolize the coastal mask at 20 m); annotations indicate the names of a few specific locations recalled in the main text; the red dotted rectangle highlights the core region of study (including, from west to east, “Côte Bleue”, “bay of Marseille”, “Calanques” and “La Ciotat”). Panel (**B**) displays the CARTHAM map of coralligenous habitats (dark red dots) with the LFN grid superimposed (semi-transparent rectangles colored according to their land-ratio, reddish colors indicate full ocean nodes while yellowish colors stand for nodes partially covered by land). In both panels the grey-scaled lines represent the 40 m (clearest), 100 m and 200 m (darkest line) isobaths.

MATERIALS AND METHODS

Potential target species: Our ecosystem-based approach to dispersal consists in analyzing habitat connectivity for a range of “Pelagic Larval Durations” (PLDs; *i.e.*, the time during which propagules drift with ocean currents) and spawning periods, which are relevant for several species inhabiting coralligenous. By reviewing some information (often partial and uncertain) of their early-life traits, we intend to make a non-exhaustive list of benthic organisms to which our results could apply. The main builders of Mediterranean coralligenous habitats are calcareous red algae belonging to two families: Peyssonneliacae and Corallinaceae. In Marseille bay, species of genus *Mesophyllum* and *Lithophyllum* are the main builders of these habitats above 20 m (below 20 m, respectively). The dispersal abilities of these calcareous red algae are thought to be very low (confirmed by population genetics results, De Jode *et al.* 2019) but are mostly unknown (Norton 1992, Opazo & Otaiza 2007). *Myriapora truncata* (Pallas, 1766) dispersal propagule is a lecithotrophic larva that can survive only up to 24 hours in aquarium (Ferretti *et al.* 2007). The gorgonian *Paramuricea clavata* (Risso, 1826), one of the most abundant structuring species of coralligenous habitats, also has lecithotrophic larvae suggesting low to moderate dispersal abilities, also supported by genetic structuring at small scales (Mokhtar-Jamaï *et al.* 2011 and references therein). Despite the fact that supposed early-life traits would point toward high dispersal potential (*e.g.*, Martínez-Quintana *et al.* 2015), other species typical of coralligenous present a clear genetic structure at small scales in the bay: the red coral *Corallium rubrum* (Linnaeus, 1758) (Ledoux *et al.* 2010, Pratlong *et al.* 2018) and *Eunicella cavolini* (Koch, 1887) (Cánovas-Molina *et al.* 2018). Finally, other coralligenous species whose dispersal traits are unknown but which could have low dispersal abilities both through sexual or asexual reproduction include: *Caryo-*

phyllia inornata (Ducan, 1878) (Goffredo *et al.* 2012), sponges such as *Oscarella lobularis* (Schmidt, 1862) (Ereskovsky 2010, Ereskovsky *et al.* 2013). Animals like bivalves and polychaetes, which also contribute in building bioconcretions, may have high dispersal abilities but no information is available to confirm this (Costantini *et al.* 2018). Note that some polychaetes brood larvae in their tubes suggesting very low dispersal, if any. Metazoan groups typical of coralligenous outcrops are bryozoans, tunicates, cnidarians and sponges whose propagules would suggest lower dispersal as compared to planktotrophic and lecithotrophic larvae.

Cartography of coralligenous habitats: The cartography of coralligenous habitats was extracted from the dataset CARTHAM 2010-2012 by aggregating the sub-dataset #12 (Astruch *et al.* 2011) and #13 (Astruch *et al.* 2012), while some pixels were completed later by Andromède Océanologie (2014). This dataset is available online on the Medtrix platform (www.plateforme.medtrix.fr, accessed in Sep. 2015) hold by the “Agence de l’Eau” and “Andromède Océanologie”. It is a fine mapping (resolution of about 20 m²) of marine habitats carried out using side scan sonar, a multibeam sounder, including confirmation in the field by divers. High-resolution habitat maps were interpolated on our model grid and then transformed into presence/absence data, resulting in 423 discrete patches widespread across the study area (Fig. 1B).

Hydrodynamic model: The ocean model MARS3D (Lazure & Dumas 2008) was implemented in the RHOMA (RHOne-MArseille) configuration extending westward from the Rhône River mouth till the Cap Sicié, eastward of Marseille bay. This configuration was set up, validated and exploited by numerous previous studies (Pairaud *et al.* 2011, Fraysse *et al.* 2013, 2014, Ross *et al.* 2016, Millet *et al.* 2018). The horizontal grid mesh resolution was of 400 m with 30 vertical sigma levels. The model run used in this study covers the period of the years 2009 to 2011 with the same model setup (forcing, parameterization, and boundary conditions) described in Pairaud *et al.* (2011). In particular, the model was forced by the outputs from a high-resolution (3 km; 3 hours) atmospheric model allowing the simulation of short-lived wind-forced upwelling and downwelling events. To take into account the influence of the large-scale general circulation, and especially the Northern Current intrusions in the bay, the model was forced at the western and southern boundaries by the 3-hourly results of a regional hydrodynamic model under its MENOR configuration (Nicolle *et al.* 2009). Hourly current outputs of the RHOMA model are provided on sigma vertical levels and Arakawa-C horizontal grid. Hourly flow fields on an Arakawa-A grid at a given depth (20 m) were obtained by vertical interpolation followed by horizontal interpolation to feed in the offline dispersal model.

Off-line dispersal model: The Lagrangian Flow Network (LFN) methodology combines network theory tools and particle-tracking model to investigate transport and dispersal induced by ocean currents. As most off-line Lagrangian models, it may be coupled to any gridded velocity fields, returning dispersal diagnostics as realistic as is the input flow field. Extensive description can be found in Rossi *et al.* (2014), Ser-Giacomi *et al.* (2015) and Monroy *et al.* (2017). Here LFN simulates the dispersal of passively drifting propagules as horizontal Lagrangian trajectories obtained after integration of the high-resolution 2-dimensional flow field generated by the RHOMA configuration. Parameters are tuned in accord with both biological and numerical knowledge from the literature. Only the most relevant elements are summarized hereafter while the design of our numerical experiments are reported in Table I.

To encompass all possible circulation schemes affecting dispersal, we simulate about 1090 starting dates representing successive spawning events occurring every day at midnight over years 2009-2011. Despite undetectable influence on our results (not shown), we retained this release time instead of midday as numerous benthic organisms spawn at dusk to limit mortality of their propagules by predation (Rasmuson *et al.* 2014). While the daily periodicity of spawning is higher than the prescriptions of Monroy *et al.* (2017), it provides large statistics to compute annual and seasonal means (see below). The tracking times mimic three different PLDs: 6, 24 and 48 hours. These PLDs would correspond to the dispersal traits of low-dispersing species inhabiting coralligenous grounds (see above). Despite the fact that some benthic species have planktotrophic larvae, supposedly surviving several days to a few weeks in the water column, the short time-scales of our study could still be relevant for these organisms as Cahill *et al.* (2017) reported genetic discontinuities in the bay for several species, including those dispersing via planktotrophic larvae. Note that PLDs longer than 48 hours are technically out-of-reach with this flow field as the mean residence time of water masses within the bay are usually of the same order of magnitude (ranging 1 to 5 days). In other words, tracking dispersal for durations longer than 48 hours with RHOMA would return biased results due to the limited extent of the domain (preventing the consideration of the influences of more distant patches).

The resolution of each quasi-squared node, representing a patch of coralligenous, is about 600 m (0.005°). Each node has the same area and contains approximately the same initial number of particles (proportional to the land-ratio); as prescribed by Monroy *et al.* (2017), 100 particles are seeded per full-ocean node. The time step of the Runge-Kutta algorithm is 10 min, fulfilling the Courant-Friedrichs-Lewy condition. We retained the vertical layer at 20 m, as it compromises the depth at which larvae of the target species are more likely to be released while it

Table I. – Design of the numerical experiments performed with the LFN.

PLD	Depth of dispersion	Time period	Spawning periodicity
6 h	20 m	Jan. 1st 2009 to Dec. 31st 2011	24 h, at midnight
24 h	20 m	Jan. 1st 2009 to Dec. 31st 2011	24 h, at midnight
48 h	20 m	Jan. 1st 2009 to Dec. 31st 2011	24 h, at midnight

gives a good spatial overlap between the LFN grid and the coralligenous map. More specifically, the coastal mask of RHOMA is not consistent with the delimitation of coralligenous habitats of CARTHAM; as such, selecting a deeper layer of the model would force us to exclude many of the near coastal patches.

Connectivity metrics and statistical analyses: For each experiment, LFN builds a connectivity matrix that encodes the movements of particles resulting from dispersal among all habitat patches. The diagonal elements represent the number of particles, which stayed or returned to the same location after dispersal; the non-diagonal elements represent the particles exchanged between each pair of nodes, after dispersal during a given PLD, considering the origin (destination) node as a row (column, respectively) of the matrix. Thanks to various computations, including normalization procedure to get probabilities ranging from 0 to 1, we compute three connectivity diagnostics for each coralligenous patch following the formulation of Dubois *et al.* (2016). More specifically, Local Retention (LR) is computed as the ratio of particles retained in their origin node to all particles released from that node. Self-Recruitment (SR) is the ratio of particles retained in their origin node to all incoming particles (those that were released from this node as well as from the surrounding ones). It measures the proportion of local recruits that originated from the source population. SR differs from LR, as it is a function of the number of propagules arriving from elsewhere while LR depends on the number of those leaving. Finally, we evaluate the relative importance of propagules export versus import by calculating the Source-Sink (SS) metric. It is the ratio of the ingoing number of propagules to the sum of both ingoing and outgoing propagules. To read more about the interpretation of such connectivity metrics and their robustness, the readers are referred to Dubois *et al.* (2016) and Monroy *et al.* (2017). The LFN methodology provides robust estimations of dispersal also for open fluid domains, as we study here (Ser-Giacomi *et al.* 2017).

Following Rossi *et al.* (2014) and Ser-Giacomi *et al.* (2015), we finally identify clusters of coralligenous patches as several groups of tightly connected nodes. So-called "coralligenous provinces" are obtained by applying a state-of-art community detection algorithm called *Infomap* (Rosvall & Bergstrom 2008). The partition is based on random walkers moving through the network with transition probabilities encoded in the connectivity matrix, an equivalent of network adjacency matrix. By exploiting the properties of information compression when describing such probability flow, *Infomap* finds the optimal network partition when minimizing the mean size of the codeword that describes inter- and intra-community transitions. This methodology is especially suited to partition dispersal networks analyzed here as (i) it takes into account both "direction" and "fluxes" of all links, (ii) it does not constrain a-priori the number of communities, and (iii) it detects communities of different sizes, relieving the usual "resolution limit" typical of other clustering algorithms (Fortunato & Barthelemy 2007). All in all, *Infomap* decomposes the network of coralligenous patches into an optimal number of communities, defining "coralligenous provinces" that are well connected internally but weakly connected among them. It allows us to analyze statistics of retention and exchanges of particles among synthetic provinces (Fig. 5 and Table II).

We compute multi-annual means of connectivity diagnostics (LR, Fig. 2; SR, Fig. 3; SS, Fig. 4) and coralligenous provinces (Fig. 5) by averaging about 1090 daily experiments over 2009-2011, encompassing a total of more than 40 million Lagrangian trajectories. Seasonal means of LR and SS are obtained by averaging about 280 daily experiments (aggregating all experiments occurring during a given season across 3 years), encompassing more than 10 million Lagrangian trajectories. Annual means could be seen as relevant for modes of asexual (*e.g.*, colony/algal fragmentation, sponge budding) or sexual reproduction that would occur all year long. Seasonal means is more relevant when the release of sexual or asexual propagules occurs during a specific season. Note that large statistics allow considering other manners to aggregate experiments in the future, for instance for species that have well-defined spawning period or whose spawning is triggered by predictable physical clues.

RESULTS

Values of Local Retention (LR) are highly dependent on the tracking time, with retention rates decreasing as the PLD increases (not shown). Total means (*i.e.*, spatial average for 423 patches considering the multi-annual means) of LR are 10.2 % for PLD = 6 hours, 2.3 % for PLD = 24 hours and 1.5 % for PLD = 48 hours. When mapping annual mean of LR for a PLD of 6 hours (Fig. 2A), it reveals contrasting spatial patterns. Most coralligenous patches situated in the vicinity of the shoreline, *i.e.*, within the 40 m isobaths, are usually characterized by retention rates ranging from 30 to 100 %. Conversely, the most offshore patches (*e.g.*, around Planier Island and off Cassis) are characterized by low retention, spanning 0-10 %. Note that retention around Planier islands is higher in its northwestern side. LR exhibits substantial seasonal variability (Fig. 2B-E) with highest total means simulated in spring (LR = 12.6 %) as compared to other seasons (ranging 8.8 % to 9.6 %). Minimum values are obtained for autumn and winter, suggesting more efficient and homogenous dispersion during these seasons. A few noticeable seasonal changes can be highlighted in Marseille bay: the retention along "Côte Bleue" is maximized in spring while showing local minima in summer; offshore patches (around Planier islands and off Cassis) are more retentive in winter and spring than in summer and autumn.

Self-Recruitment (SR) also weakens when the PLD increases (Fig. 3) but to a lesser extent than LR: total means of SR are 17.5 % for PLD = 6 hours, 5.2 % for PLD = 24 hours and 3.3 % for PLD = 48 hours. Spatial patterns also reveal relatively high SR in shallower patches (ranging from 10 to 80 %) as compared to most offshore locations where SR spans 0-10 %. Conversely to

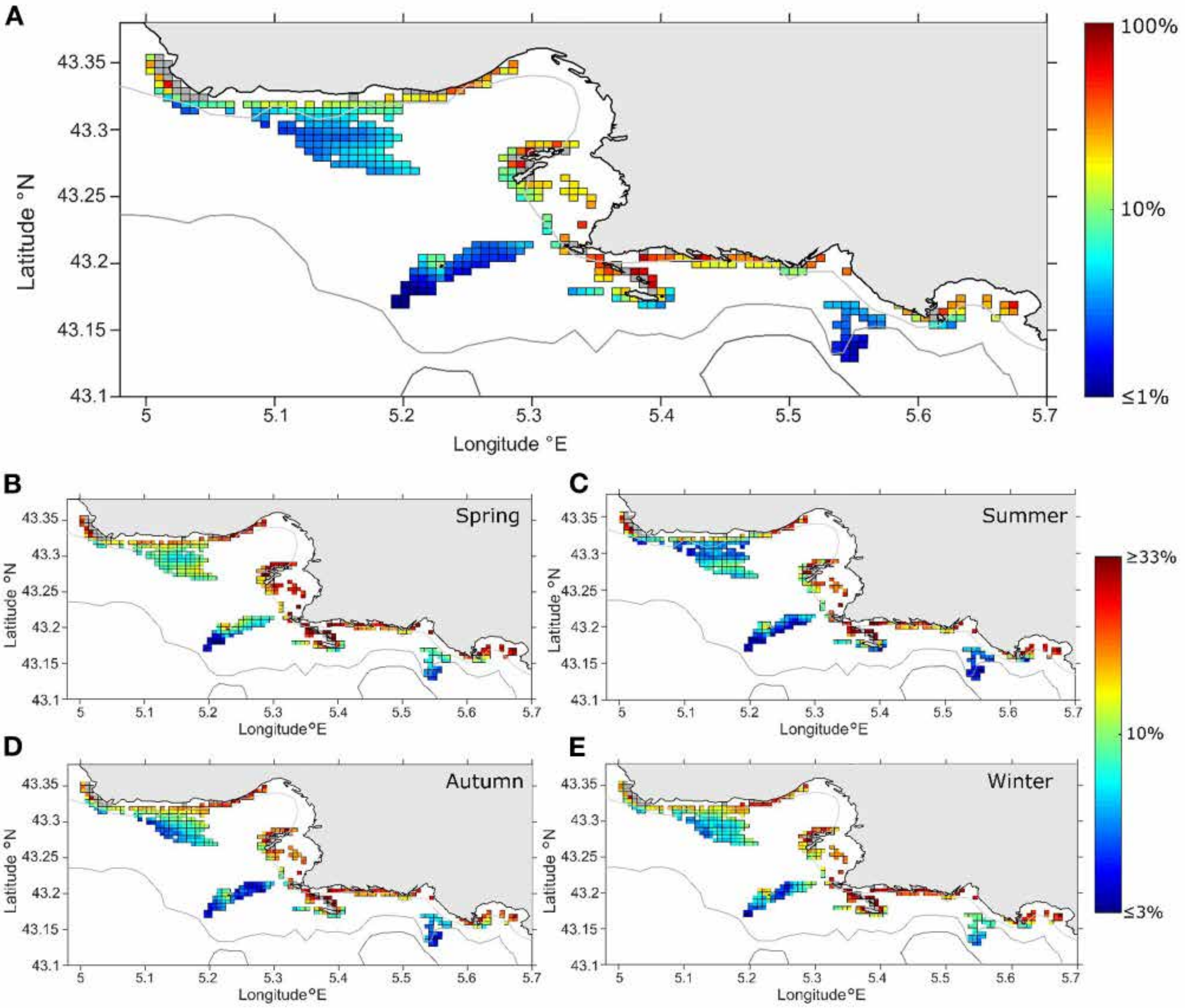

Fig. 2. – Maps of Local Retention (LR, in % using log-scale) for a PLD of 6 hours. Panel (**A**) represents the multi-annual mean (over 2009-2011); lower panels are seasonal averages: (**B**) spring, (**C**) summer, (**D**) autumn, (**E**) winter. The grey-colored nodes have been disregarded as they are partially- or fully-covered by land, resulting in a poor reliability of LR values in these nodes. The grey-scaled lines represent the 40 m (clearest), 100 m and 200 m (darkest line) isobaths.

LR, mean SR values along the Côte Bleue (from 5.05° to 5.2°E) and, to a lesser extent along the Calanques (from 5.4° to 5.5°E), tend to be weaker than along other coastlines for all PLDs. While LR values of all individual patches decrease when PLD increases, this does not hold for SR: shallower nodes tend to maintain moderate levels of SR even when the PLD increases (Fig. 3C).

Source-Sink (SS) is less sensitive to the PLD than LR and SR (Fig. 4A, B). For all PLDs, patches off the western shorelines of Côte Bleue (from 5.05° to 5.18°E) are consistently characterized as sinks. Those located to the east and within the southern bay (from 5.18° to 5.3°E) are instead identified as sources. Most patches on the eastern side of the domain (Riou archipelago, Calanques, off Cassis and la Ciotat) are also mainly categorized as sources. Coralligenous patches around Planier islands show a bimodal pattern: sinks are identified on the offshore flank (*e.g.*, south-westward) of the island while sources are found on its inshore side (north-eastward). A few seasonal patterns can be distinguished (Fig. 4C, D): the marked sources and sinks previously described along Côte Bleue are emphasized during summer while the upstream sources (Riou archipelago, off Cassis) are weak in summer but maximized during winter.

Clustered patches are highly dependent on the tracking duration: 24 coralligenous provinces are captured for a PLD of 6 hours while only 9 and 8 provinces were obtained for PLD of 24 and 48 hours, respectively (Fig. 5). The mean areas of these provinces scale with the PLD (Table II): it is about 3.4 km^2 for 6 hours, 9.1 km^2 for 24 hours and 10.2 km^2 for 48 hours. Regional local retention (*i.e.*, similar to LR but computed at the "province" scale) is maximized by *Infomap*, reaching 90 % or more for most provinces. Their exchanges with other provinces range several orders of magnitude (Table II): for each province, a couple of links with neighboring provinces make up to 99 % of the exchanges (plotted on the inserts of Fig. 5), while a few other links are weak and rare but do occur sometimes to time with more distant provinces (reported in Table II). Focusing on the prominent links only, we find several subgroups of provinces, which are disconnected for PLD of 6h. When the PLD increases, the provinces are less numerous and become larger; the network of coralligenous patches is more and more connected. Longer tracking times allow the development of bridging links among these provinces that were disconnected at short time-scales. For instance, the province around Planier Island is connected with the Côte Bleue province for PLD of 24 hours, while they were isolated at 6 h. For 48 h hours, the Planier/Côte Bleue connection remains while both Frioul and Riou archipelago act as intermediate hubs connecting relatively well the most distant patches located at both western (Côte Bleue) and eastern (Calanques, Cassis, La Ciotat) limits of the domain.

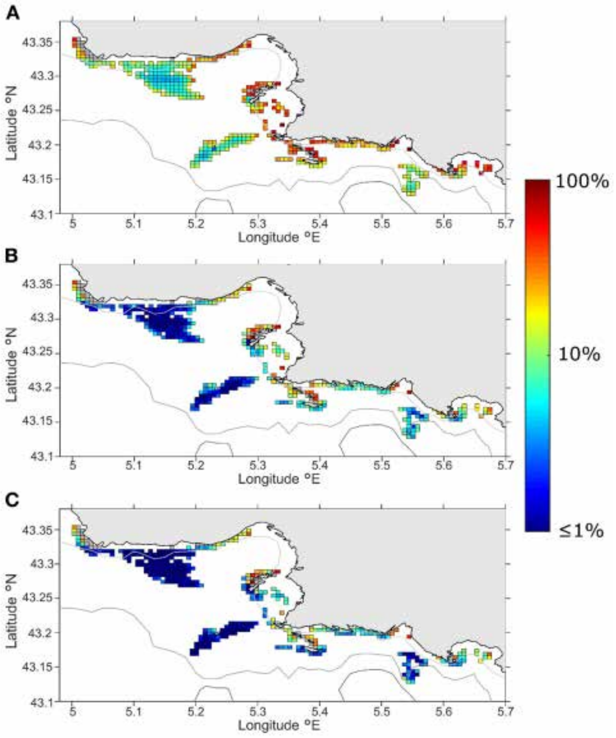

Fig. 3. – Maps of Self-Recruitment (SR, in % using log-scale). Multi-annual mean (over 2009-2011) are displayed for various PLDs: panel (**A**) is 6 hours, panel (**B**) is 24 hours and panel (**C**) is 48 hours. The grey-colored nodes have been disregarded as they are partially- or fully-covered by land, resulting in a poor reliability of SR values in these nodes. The grey-scaled lines represent the 40 m (clearest), 100 m and 200 m (darkest line) isobaths.

GENERAL DISCUSSION & CONCLUSIONS

Relating mean patterns of habitat connectivity to hydrodynamics

Our results suggest that retention processes are highly dependent on the bathymetry, *e.g.*, elevated retention at

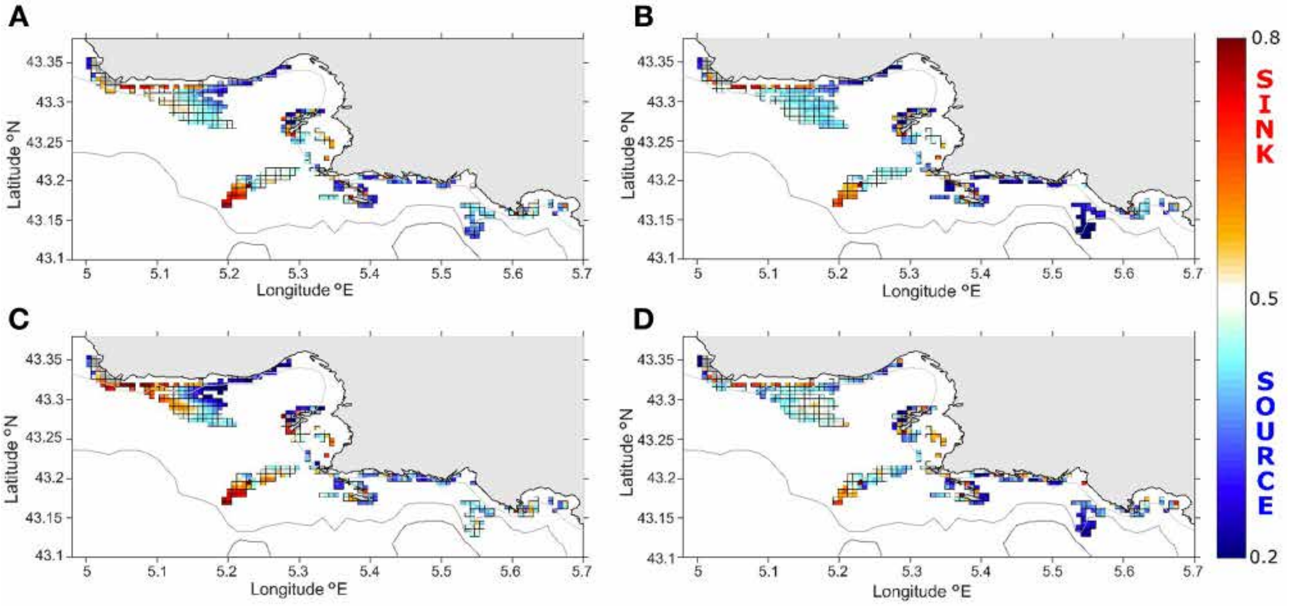

Fig. 4. – Maps of Source-Sink (SS, near-zero values indicate "sources"; values approaching 1 represent "sinks"). Multi-annual mean (over 2009-2011) are displayed for PLD = 24 hours in panel (**A**) and for PLD = 48 hours in panel (**B**). Seasonal averages for PLD = 24 hours are displayed for summer in panel (**C**) and for winter in panel (**D**). The grey-colored nodes have been disregarded as they are partially- or fully-covered by land, resulting in a poor reliability of SS values in these nodes. The grey-scaled lines represent the 40 m (clearest), 100 m and 200 m (darkest line) isobaths.

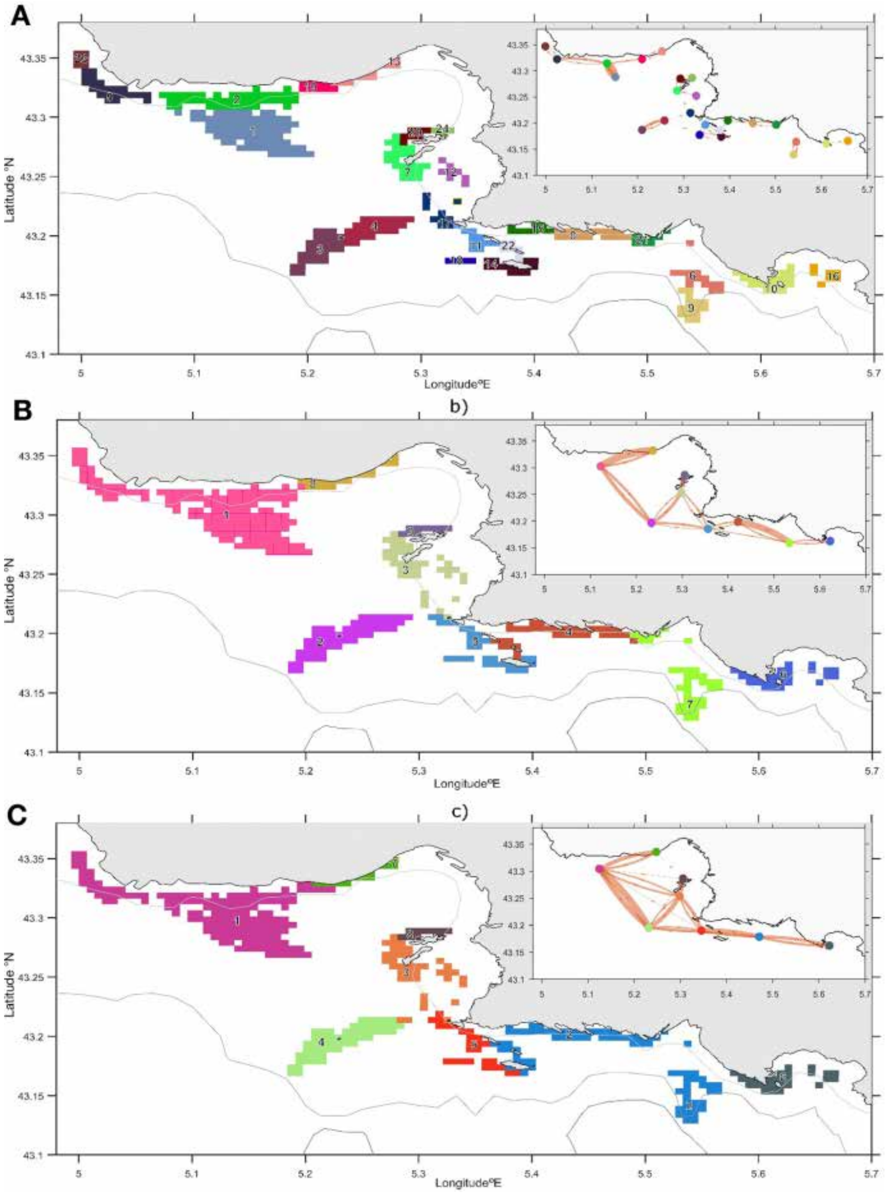

Fig. 5. – Maps of clustered coralligenous patches. Synthetic provinces are derived from the multi-annual mean connectivity matrices for PLD = 6 h in panel (**A**), for PLD = 24 h in panel (**B**) and for PLD = 48 h in panel (**C**). In all panels, each province is identified with a specific color (randomly chosen) and a number (consistent with those reported in Table II). Upper right inserts symbolize the averaged network of provinces (colored dots) and their main directed links (reddish arrows; width is proportional to the fluxes of particles). It excludes both the exchanges lower than 1 % and the regional local retention (reported in Table II). The grey-scaled lines represent the 40 m (clearest), 100 m and 200 m (darkest line) isobaths.

sites where water depth is less than 20-40 m, due to weaker currents as compared to the more offshore locations (coastal currents tend to weaken by dissipative processes such as bottom friction). Intuitively, the longer dispersal time, the less retention of propagules (Dubois *et al.* 2016, Monroy *et al.* 2017). While this interpretation for LR is straightforward, it is not so valid for SR which combines together retained particles (numerator) and ingoing fluxes (denominator): so high values of SR can be caused by both high retention and/or low incoming fluxes, highlighting the counter-intuitive effect of recirculation processes in this semi-opened bay (which would not be prominent along straight coastlines). Indeed, LR and SR scale differently with the PLD probably because longer dispersal may allow some particles to leave a given patch and eventually return to it later on, emphasizing the importance of recirculation processes, especially in the southern bay.

Wind-induced circulation may be associated with the seasonal variations of LR off Côte Bleue: maxima occur in spring when SE/SW winds prevail, favoring coastal convergence explaining the coastal sinks, while minima are simulated during summer when NW winds and associated sporadic upwelling (*i.e.*, coastal divergence, favoring coastal sources, as shown by Dubois *et al.* 2016) are prominent. The SS patterns described off Côte Bleue are emphasized during summer, probably due to the more frequent occurrence of wind-forced events during this season. More generally, sources are overall located at the eastern side of our domain and sinks at the western side (Fig. 4) due to the westward (cyclonic) basin-scale circulation.

The relationship between the NC and our connectivity diagnostics is more puzzling. Earlier research suggested that the maximum transport of the NC, associated

Table II. – Statistics of exchange and retention among the synthetic coralligenous provinces (the numbers identifying each province are consistent with those reported on Fig. 5).

PLD (hours)	Province identity	Area (km^2)	Regional Local Retention (%)	Identity of connected provinces		
				< 0.1 %	0.1 ≤ x < 1 %	≥ 1% (inserts Fig. 6)
6	1	15.6	95	5	–	2
	2	9	88.8	–	19	1; 5
	3	5.8	95.4	–	–	4
	4	5.4	87.8	1; 18	7; 17	3
	5	4.4	94	–	1	2; 23
	6	2.8	94.6	8	10; 21	9
	7	4.8	94.8	4; 24	17	12; 20
	8	3.6	92.7	–	–	15; 21
	9	3	89.2	21	–	6
	10	4.2	98.5	–	6; 16	–
	11	2.2	95.6	–	–	14; 17; 18
	12	1.6	98.8	–	7; 17	–
	13	2.8	93.3	2	–	19
	14	2.8	89.1	4; 8; 15; 17	22	11; 18
	15	1.8	97.8	–	8	22
	16	1.4	96.4	–	–	10
	17	2	94.3	12	11; 18	4; 7
	18	0.8	82.1	–	3; 17	4; 11; 14
	19	1.2	76.8	–	–	2; 13; 19
	20	1.4	86.7	–	–	7; 24
	21	1.6	74.5	–	6	8
	22	1.4	96.2	–	15	14
	23	1.2	83.2	–	–	5
	24	0.8	95.5	–	–	20
24	1	30.4	96.7	5; 9	3	2; 8
	2	11.2	76.9	8	9	1; 3; 5
	3	7.8	90.6	4; 8	1	2; 5; 9
	4	7.2	92.5	2; 6	–	5; 7
	5	6.4	79.9	1; 9	7	2; 3; 4
	6	5.6	93.6	–	4; 5	7
	7	7.2	56.7	–	2	4; 5; 6
	8	4.4	64.7	2; 3; 9	–	1
	9	2	74.5	8	1; 2	3
48	1	31.6	92.8	2; 5	8	3; 4; 7
	2	15.8	84.6	7; 8	1; 3	4; 5; 6
	3	8	78.4	–	2; 7	1; 4; 5; 8
	4	10.8	41.5	–	2; 7	1; 3; 4; 5; 8
	5	5.2	65.6	6; 7	8	1; 2; 3; 4
	6	5.6	82.1	1; 3	4; 5	2
	7	3.2	59.9	4	3; 8	1
	8	2	68.4	5	4	1; 3; 7

to a narrower jet flowing closer-to-shore, occur in winter (Alberola *et al.* 1995). The NC becomes baroclinically unstable especially in winter, producing large mesoscale meanders, which may penetrate into the Gulf of Lion (Barrier *et al.* 2016) and Marseille Bay (Pairaud *et al.* 2011; Ross *et al.* 2016). This knowledge is consistent with the sinks located to the southeast flank of Planier Island and the weak sources associated with relatively high retention in the wake of the island (northwest side). The southeast sinks are likely receiving propagules from the identified upstream sources, *e.g.*, Riou archipelago and patches off Cassis, which are weak in summer but maximized during winter. It is however contradictory with the seasonal minimum of LR simulated in summer/autumn for the coralligenous habitats off Cassis, suggesting that NC intrusions may not impinge on these patches.

As the PLD increases, the provinces become larger. This is consistent with the monotonic increase of provinces areas with advection time already documented by Ser-Giacomi *et al.* (2015). The northern sector of Marseille Bay is weakly connected to Côte Bleue due to higher residence time (Pairaud *et al.* 2011). Currents at 20 m depth are globally westwards along the Côte Bleue coastline (Fig. 4 and Fig. 5A). For PLD of 24 h and 48 h, Planier is connected with all the other provinces. Under the influence of all the above-mentioned major hydrodynamical processes (*i.e.*, up/downwelling, NC meanders and intrusions and mesoscale eddies), Planier Island acts as a major “connecting hub”. The transit times and pathways reported by Thibault *et al.* (2016), derived from the same RHOMA model using another methodology, are consistent with our analyses. For instance, they found that the connections between Planier and the Côte Bleue need a minimum of 12 h-18 h to be realized, which explains why the corresponding provinces are disconnected for PLD = 6 h but become connected for 24 and 48 h. Note that our synthetic coralligenous provinces could also help identifying community similarities. Lagrangian particles could equally represent passive propagules or small water parcels; as such, pair of sites that connected at short time-scales (sufficiently short so that hydrographic conditions can be assumed to remain relatively constant) would also experience relatively similar abiotic conditions. Both processes could homogenize species diversity among local communities.

In fact, all hydrodynamical processes contribute altogether to the patterns of connectivity reported here; it is thus difficult to link a specific pattern with a given process. For instance, SR values along the Côte Bleue (from 5.05° to 5.2°E) and, to a lesser extent along the Calanques (from 5.4° to 5.5°E), tend to be weaker than other coastlines. It may be because local propagules spread offshore due to upwelling events while recirculation processes related to NC intrusions on one hand, and to the occurrence of the "Marseille Eddy" on the other hand, would bring particles from elsewhere into Calanques and Côte Bleue, respectively (Schaeffer *et al.* 2011, Fraysse *et al.* 2014). Moreover, as hydrodynamics exhibit high spatio-temporal variability, large statistics are required to get robust diagnostics of connectivity encompassing all processes. The good ability of RHOMA to reproduce realistically the complex circulation of Marseille Bay over 2009-2011 reported by Fraysse *et al.* (2013) suggest that our average patterns are robust while taking into account environmental variability. Note however that inter-annual variability is important in the NW Mediterranean (Hidalgo *et al*. 2019). Indeed year 2009 was characterized by unusual conditions: a long-lasting summer warming event occurs down to 40 m, inducing mortality for *P. clavata* populations (Pairaud *et al.* 2014) and possibly exceptional circulation patterns. Nevertheless, the exact dispersal pathways of water parcels originating from the Calanques studied by Millet *et al.* (2018) for two very distinct NC intrusion events (in June 2008 and October 2011) are consistent with our mean patterns. As the circulation of the bay depends on short wind-forced events (dominated by north-west and south-east winds, Millet *et al.* 2018) occurring episodically at all time of the year, future work could aim at producing climatology of "wind-forced events". By categorizing all events characterized by similar wind-forcing conditions across the 3-year period and by aggregating matrices accordingly (independently of the season and the year), one could describe extreme connectivity. It would contrast the mean patterns analyzed here and could be relevant for species whose spawning is triggered by specific physical clues such as wind-forced upwelling and associated coastal cooling. Note however that wind-driven currents are often associated with strong vertical velocities that are neglected in our "horizontal" assumption. Other perspectives could be to analyze other dispersal depth and to perform 3-dimensional simulations.

Discussing the biological implications of our simulated results

According to Dubois *et al.* (2016), the integrated analyses of our metrics inform on the connectivity of coralligenous provinces. When both LR and SR are high, as in both northern and southern sectors of Marseille Bay, it symbolizes mostly "closed" populations for which one could expect high differentiation with external sites and low diversity. The Côte Bleue also displays consistent patterns of retention: both LR and SR are large along both eastern and western domains (from 5° to 5.05°E and from 5.2° to 5.3°E) and moderate in-between (5.05-5.2°E). SS shows a discontinuity at about 5.2°E, with sinks westward and sources eastward. Altogether, spatial patterns of LR, SR and SS suggest (i) low diversity along Côte Bleue and (ii) a sharp change of connectivity at about 5.2°E, which was also the presumed north-west limit of the Calanques influence (Millet *et al.* 2018).

Interestingly, a multi-specific population genetics study along French Mediterranean shorelines (especially detailed in Marseille Bay) confirms the prediction of a low genetic diversity for Côte Bleue: for most species, it has a significant negative contribution to global genetic diversity (Cahill *et al.* 2017). While they invoked anthropogenic activities, such as habitat degradation or pollution, to explain variations in local genetic diversity, the present study suggests that physical connectivity among patches could alone explain their observations. For the builder algae *Lithophyllum stictiforme*, the Côte Bleue population (Carry-le-Rouet and Couronne) actually exhibits the lowest genetic diversity as compared to the populations sampled around Frioul and Riou archipelago as well as Cassis (De Jode 2018). Monospecific population genetic studies comparing sites across the ~5.2°E discontinuity tend to support its effect as a barrier to gene flow. For *Cystoseira amentacea*, an algae of the shallowest infralittoral not restricted to coralligenous habitats, Thibaut *et al*. (2016) reported genetic structuring concordant with our results: among the Côte Bleue samples, Niolon population (5.25°E) appear well differentiated from the most western sampling sites (Carro and Ponteau). For the red coral *C. rubrum*, Ledoux *et al*. (2010) found significant differentiation between the population situated at about 5.2°E and the other populations sampled at Carro (5.04°E) and Ponteau (~5°E).

Conversely, when both LR and SR are low, as for Planier Island and Cassis provinces, it suggests largely "open" population with high genetic diversity. Furthermore, the offshore flank of Planier province is a sink (*i.e.*, net importer of external propagules with little retention of its own propagules), suggesting it could house higher diversity than Cassis, which behaves as a source (*i.e.*, net exporter). While these conclusions seem robust for Planier Island, the behavior of Cassis could change if the model domain was larger, hence allowing us to consider larval influences from upstream patches located eastward (*i.e.*, from "côte Varoise"). The high diversity expected at Planier Island, also supported by its central position in the network (Fig. 5), is consistent with Thibault *et al.* (2016) who found that Planier Island is a mix of all origins with high genetic diversity. This central hub ensures the links between the most western and eastern patches at 24 h and 48 h; it could also ensure such connection at 6 h thanks to stepping-stone processes.

The networks of coralligenous provinces obtained at different time-scales provide insights about the kind of biological clusters one should expect if habitat connectivity is the main factor structuring population. Thibaut *et al.* (2016) found a strong and significant genetic structure within Marseille Bay for *C. amentacea*, whose dispersal is ensured by a zygote with survival time spanning 4-18 h. Pairs of sites characterized by low genetic differentiation are those tightly connected by ocean currents, as indicated by the synthetic provinces obtained for 6 h (Fig. 5A). Cahill *et al.* (2017) also found that the most important barriers of genetic differentiation separate Côte Bleue from the rest of the Bay. The crustose coralline algae of the genus *Lithophyllum* (De Jode *et al.* 2019) or the bryozoan *M. truncata* (De Jode, 2018), which are both engineering species of coralligenous, exhibited genetic differentiation in accord with our results. These species have propagules dispersing from a few hours to a day, explaining why their genetic structures match well our model results for 6 h and 24 h (Fig. 5A, B). Observed structures of the red coral documented significant genetic differentiations between Côte Bleue, Marseille Bay and Calanques (Ledoux *et al.* 2010, Pratlong *et al.* 2018), suggesting effective dispersal closer to 6 h than to 24 h. Note also that our provinces and their typical spatial-scales could be used to guide future sampling efforts (Dubois *et al.* 2016), *e.g.*, by targeting disconnected provinces while avoiding duplicate sampling in each province.

An important aspect of the present approach is that it constrains current-driven dispersal by considering habitat patchiness in order to estimate biological connectivity. Although population genetics stipulate that gene flow is proportional to the product of 'migration rate' and 'effective population size' (genetic drift), most studies interpreting genetic structures invoke the duration of dispersal solely, without discussing the effective population size. The latter should be, at first order, proportional to the overall extent of the preferential habitat of the species. Although gorgonians have similar dispersal traits, genetic differentiation in the bay was lower in *Eunicella cavolini* than in *P. clavata*, and lower in *P. clavata* than in the red coral *C. rubrum* (Cánovas-Molina *et al*. 2018). The respective spatial extension of their actual habitats may explain the different levels of genetic differentiation: *E. cavolini* is widespread over various habitats (not restricted to coralligenous) while *C. rubrum* displays the less extended populations, exclusively developing on coralligenous patches. So even if the species of interest is restricted, or not, to coralligenous habitats, our synthetic provinces incorporating both habitat patchiness and current-driven connectivity may bring interesting insights into population structure.

Species community differentiation (*e.g.*, beta-diversity) could be affected by connectivity in a similar way as genetics: well-connected communities would show more similarities. In Marseille bay, species diversity varies from site-to-site but, contrary to intra-specific studies, there is no clear spatial structuring within the bay (Doxa *et al.* 2016, De Jode 2018, Çinar *et al.* 2020). Local abiotic factors seem to control community composition at small scales more likely through natural selection (De Jode 2018), reflecting favorable ecological niches rather than dispersal limitations (conversely to "within-species" structuring). Nevertheless, the present study could help explaining community composition under certain circumstances, such as when mortality events affect a population. In such cases, their transient spatial distributions would follow recolonization routes, which result from both ocean currents and favorable conditions for settlement. After the *P. clavata* mortality outbreak in summer 2009, Mokhtar-Jamaï *et al.* (2011) observed significant genetic structure in Marseille bay, including a genetic break around Cap Croisette. It matches best with our coralligenous provinces defined for 6 hours, despite its lecithotrophic larvae potentially dispersing longer. Investigating population structures of genus *Echinocardium* (sand-dweller sea urchins that have planktotrophic larva dispersing for about 3 weeks), Egea (2011) observed that *E. cordatum* is prominent within Marseille bay till the western flanks of Riou archipelago (~5.4°E); it is then replaced by *E. mediterraneum* in "Calanques" and along eastern coastlines. Only the 6-hour synthetic provinces reveal a connectivity break at 5.4°E, while it vanishes for longer dispersal. It might indicate that the biogeography of *Echinocardium* is primarily controlled by abiotic factors instead of dispersal. Overall, it suggests that short connectivity (< 12 h) in this region tend to be associated with relatively constant abiotic conditions whereas longer connectivity (≥ 24-48 h) would overcome stronger environmental gradients, potentially preventing propagules to survive along dispersal routes and/or to settle at final destination.

General conclusions

We investigated fine-scale connectivity of coralligenous habitats in Marseille bay for a range of parameters relevant for many benthic species using dispersal modeling and high-resolution cartography. We analyzed annual and seasonal averages of three connectivity metrics measuring the retention and exchange of propagules driven by ocean currents within and among habitat patches. Our synthetic coralligenous provinces and associated statistics for several time-scales provide broader views of the ecological clusters structuring benthic populations. By discussing our results against genetic and ecological studies, we emphasized previous findings that are well supported by our simulated habitat connectivity. Altogether, it suggests that current-driven dispersal of propagules and habitat patchiness control to a certain degree the spatial structure of benthic populations.

It illustrates the potential of such interdisciplinary studies to reveal counter-intuitive and non-straightforward relationships among hydrodynamics and population structures that could worth further investigations. Moreover, our simulated connectivity patterns could be used by other researchers to design future sampling and to help explaining observed population structures. Last but not least, model outputs are available upon request and can be aggregated at other levels, such as monthly/weekly time-scales or as climatology of wind-forced events.

Among the conclusions backed up by both modeling and observations, our study suggests that low diversity is expected along Côte Bleue whereas Planier Island, which acts as a key connecting hub for all surrounding coralligenous patches, would be characterized by high diversity. In combination with an index evaluating the conservation state of coralligenous habitats (*e.g.*, Sartoretto *et al.* 2017), these areas could be considered as good candidates for constant monitoring and protection (Ods *et al.* 2012). Future work may focus on how habitat connectivity will evolve with climate change (Gerber *et al.* 2014) and with the constant development of anthropogenic structures such as harbors, wind farms and offshore platforms (Henry *et al.* 2018).

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COUPLING THE MAP OF MARINE HABITATS AND FISH ACCUMULATION ZONES: A THREE-DIMENSIONAL SPATIAL APPROACH FOR THE MANAGEMENT OF HALIEUTIC RESOURCES

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MAPPING
MARINE HABITATS
3D
MULTIBEAM ECHO SOUNDER
WATER COLUMN
HALIEUTIC
MEDITERRANEAN

ABSTRACT. – The spatialization of halieutic data is an essential element to define and create efficient protected and managed areas. Moreover, the distribution of fish schools is not homogeneous in the water column and is strongly linked with marine habitats. It is thus necessary to develop techniques allowing a spatial evaluation of halieutic resources. Multibeam echo sounders (MBES) provide acoustic data of the seafloor and the water column with a high accuracy and resolution. A single acquisition gives the bathymetry, a backscatter mosaic of the sea bottom and an acoustic imagery of the water column. The bathymetric data processing highlights the seafloor rugosity using several metric indices. A semi-automated classification including depth, rugosity indices and backscatter values provides maps of marine habitats, which are finally validated with ground truth. Data from the water column are analyzed using an algorithm that detects acoustic targets corresponding to fishes. A georeferenced scatter graph of fish schools is thus automatically created. The 3D model of the seafloor obtained from the bathymetry is textured with the map of marine habitats. Points corresponding to fish detection are then added on the 3D model to provide a complete map. Through this process managers can access to a clear visualization of fish accumulations and the key marine habitats within their areas of interest.

INTRODUCTION

Marine ecosystems are among the most rich and complex biological systems of our planet but remain difficult to study when compared with terrestrial habitats (Appeltans *et al.* 2012). To integrate and assess marine ecosystems' complexity in governmental efforts to protect them against anthropogenic influences, an ecosystem-based management theory was produced (Slocombe 1993). More and more applications based on this approach are currently developed to consider the resilience and robustness of marine systems (Curtin & Prellezo 2010). These approaches rely on the understanding of ecosystems functioning by associating marine species in functional groups according to their ecosystemic roles (Buchmann & Roy 2002). They are especially suitable to respond to the European directives, such as the Marine Strategy Framework Directive (MSFD), aiming to reach a good ecological status of marine areas.

Due to complexity and the difficult access to the marine environment, most of the survey techniques classically used are based on discrete samplings and observations focusing on a single or few functional compartments. A good example of the paradigm evolution concerning the survey of marine ecosystems is the study of *Posidonia oceanica* (L.) Delile seagrass meadows that previously focused on the plant morphology (Pergent-Martini *et al.* 2005) while, nowadays, ecosystem-based approaches are developed (Personnic *et al.* 2014). Although this shift in paradigm provides a better evaluation of ecosystem status, the data on which it relies are still discrete and random measurements upscaled to large areas. Complementary spatial data are thus required to fulfill an effective ecosystem-based management of the marine environment.

The most effective tool to acquire spatial information on the seafloor and the water column above – with a high resolution and positioning accuracy – is currently the multibeam echo sounder (MBES) (Abadie & Viala 2018). This type of acoustic probe is able to provide simultaneously bathymetric data, backscatter images and the water column imagery (WCI) on a large swath (increasing with the depth). Recent signal processing methods for bathymetric data allow to generate maps of marine habitats using various rugosity indices (Abadie *et al.* 2018). Likewise, innovative algorithms are able to detect various acoustic targets on the WCI among which the fish schools and the individuals composing them (Lamouret *et al.* 2019).

In order to pave the way for a spatial approach of the ecosystem-based management, we investigated the capacity of MBES to provide exploitable two and three-dimensional information on several functional compartments of key marine ecosystems of the Mediterranean Sea. With this main target in mind, we studied the inter-seasonal and inter-annual characteristics of fish accumulations on various habitats. We also experimented different environmental indicators in an attempt to describe the link between marine habitats' features and fish distribution.

MATERIALS AND METHODS

Study site and data acquisition: This study took place in the Bay of La Ciotat in the south of France off the Ile Verte (Fig. 1) on a site renowned for the richness of its habitats and biodiversity. The site covers an area of 0.83 km². An exhaustive acoustic data acquisition was performed two times in June and August 2016 to consider the increase of the water temperature linked with fish observations. These two acquisitions were realized in similar conditions: a two hours work done in the morning to collect the data along the same lines north-south oriented. Another acoustic dataset was obtained in June 2019, allowing an inter-annual comparison.

Acoustic data were acquired by using a R2Sonic 2022 MBES fixed on the hull of a 6 m long survey boat. Position and attitude were recorded by an Applanix I2NS, an inertial system equipped with a RTK GNSS positioning device providing 0.015° roll/pitch precision as well as a horizontal accuracy of 1 cm and a vertical one of 1.5 cm. Acoustic data were acquired at a frequency of 450 kHz with an individual beam width of 0.9° × 0.9° for a maximum swath sector of 160° and 1024 soundings per swath. Transects were defined prior to the data acquisition and the navigation was operated by a Raymarine ACU 200 autopilot synchronized with the RTK GNSS using ViewMap, a Geographic Information System (GIS) and navigation software developed by Viala (2015a). The underwater sound velocity was constantly checked using a Valeport Ltd miniSVS sound velocity sensor mounted on the MBES. Additional underwater sound velocity profiles were performed with another miniSVS to detect the possible presence of a thermocline or fresh water layers impacting the sound propagation. Water temperature profiles were computed from the sound velocity data. Ground truth data were performed by scuba diving to validate the seafloor classification.

Acoustic data treatment and generation of habitat maps: R2Sonic 2022 bathymetric soundings were processed using the ViewSMF computer program developed by Viala (2015b) for the visualization and processing (automatic or manual) of MBES acoustic data and metadata. False echoes were removed using filters to isolate one or several soundings. A rugosity index, named Bathymetric Automated Treatment for the Classification of the Seafloor (BATCLAS), is then computed from bathymetric soundings to highlight the underwater landscape according to the method developed by Abadie *et al*. (2018). The noise on the backscatter imagery was reduced using a time variable gain and snippets. A digital elevation model (DEM) encompassing a bathymetric map, the treated backscatter imagery and the BATCLAS index was generated. Finally, data from the DEMs and ground truthing were computed in ViewMap using a decision tree to classify marine habitats and build polygons exploitable in Geographical Information System (GIS) for further analysis. The final maps take the shape of 2D and 3D maps of marine habitats.

Water column processing and environmental indicators: The WCI was analyzed following the technique developed by Lamouret *et al*. (2019). This method utilizes an algorithm that automatically detects and identifies acoustic targets corresponding to fishes on the WCI. For each detection, the localization, the dimensions and the energy are computed and stored in computer files. The halieutic data set is finally exported under the shape of a scatter plot for 2D and 3D analyses, as well as for investigating the relationships between fish distribution and marine habitats.

In order to compare the density between the different dataset, the fish density is computed from the scatter plot for the whole area and for several sub-areas of interest of the marine habitat map. The density is given in fishes/m², corresponding to the number of fishes in a column of 1 m² and of height given by the bathymetry. The density by subzone is then easily comparable

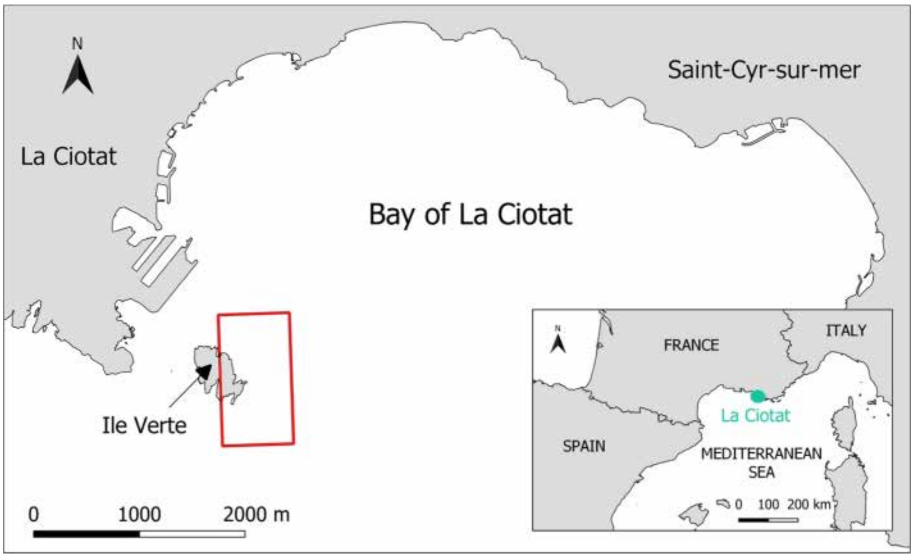

Fig. 1. – Study site (red frame) in the Bay of La Ciotat.

through times on 2D maps. The study area was divided in three main sub-areas: (1) the Ile Verte walls, (2) the rocky reefs, (3) the sedimentary plains. Moreover, the rocky reefs were subdivided one by one.

Table I. – Areas covered by each marine habitat and their proportion.

Habitat	Area (m²)	Proportion (%)
Coralligenous communities	27,838	3.9
P. oceanica meadows	7,225	1.0
Algal cover on rocky substrate	39,151	5.7
Soft sediments	624,502	89.4
Total	**698,716**	**100.0**

RESULTS

Depths varied from a few meters (< 10 m) on the shallowest cost of the Ile Verte to 66 m in the south-east corner (Fig. 2A). In the extension of the island towards the south-east an uneven seascape was clearly visible, composed of vertical walls and plateaus. The backscatter imagery (Fig. 2B) highlights the relief seen on bathymetric data. Apart from these irregularities, the seabed was even with two ranges of backscatter intensity: a high value on the northern part and a lower one in the south, indicating two types of sediments with contrasted granularities. The

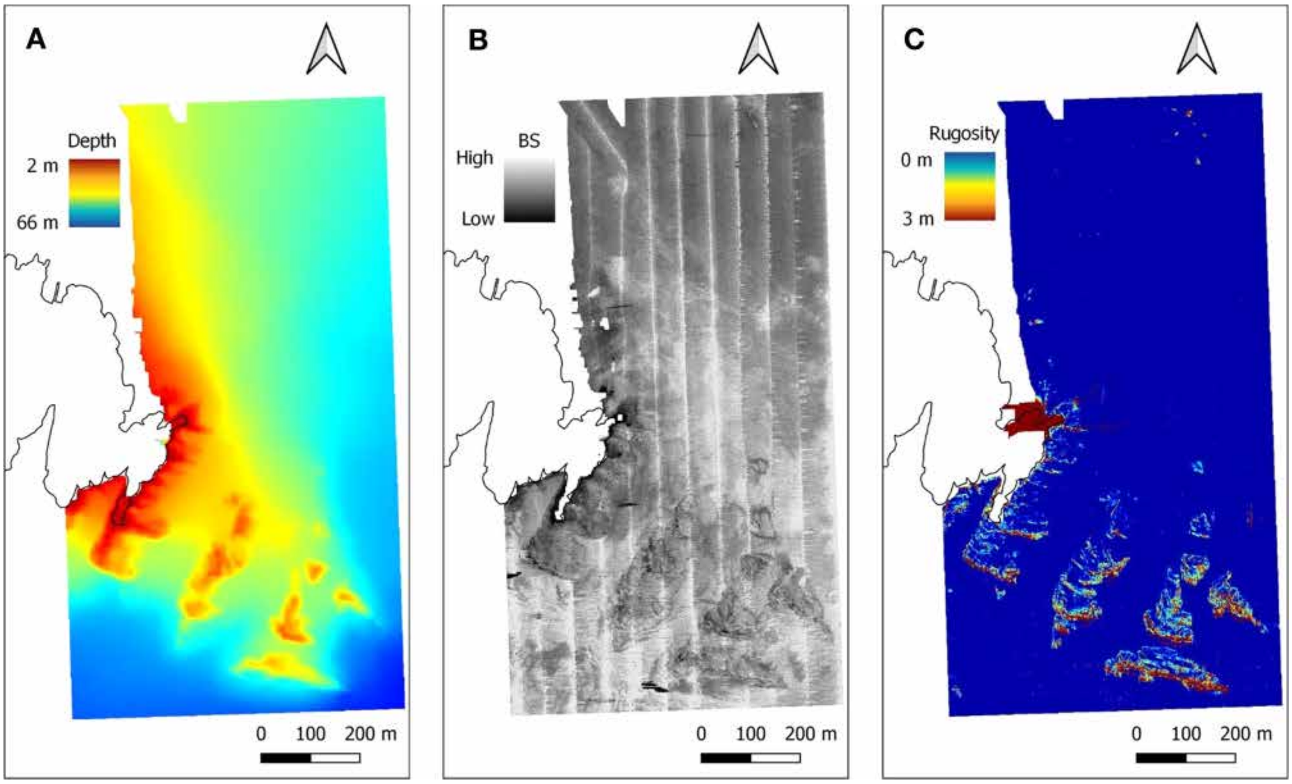

Fig. 2. – Multibeam echo sounder data products. **A**: Bathymetry; **B**: Backscatter imagery; **C**: Rugosity.

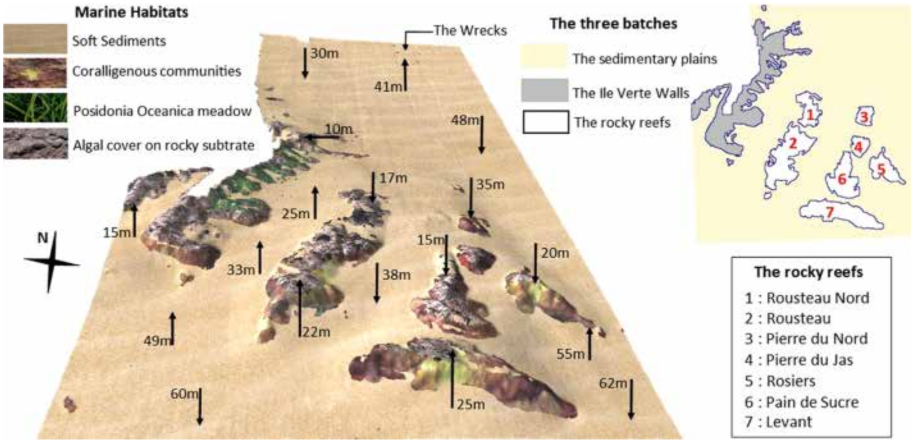

Fig. 3. – 3D representation of the seabed topology with marine habitats and details of the three batches studied with reefs names for fish density studies.

unevenness were highlighted again and well delimited on the rugosity index map (Fig. 2C). The rest of the seabed appeared smooth, except for small spots in the north east of the zone corresponding to known ship wrecks.

Ground truthing confirmed the existence of the spots that were three very dilapidated wrecks – a wooden trawler and two sailing ships – playing the role of artificial reefs on the sandy plain. Ground truths also established the main habitats, *i.e.*, rocky substrate with algal cover (39,151 m²), *P. oceanica* (L.) Delile meadows (7,225 m²), coralligenous communities (27,838 m²) and soft sediments (624,502 m²; Table I). The *P. oceanica* meadows were not found on sediments, but rather on hard substrates (Fig. 3). Sparse meadows were observed on the top of the rocky reefs too; however, they were not represented due to their small size. The coralligenous communities were present on each rocky substrate from around 25 m depth while rocks covered by algal communities were found above this limit.

The temperature profiles of the two June acquisitions did not present a clear thermocline, but rather two main temperature gradients (Fig. 4). The greatest temperature decreasing of 1.5 °C was found in the twenty-first meters in June 2016 (from 19.0 °C to 17.5 °C), and in the ten first meters in June 2019 (from 23 °C to 21.5 °C). Beyond 20 m depth, the temperature decreased more slowly to stabilize at 17.2 °C and 19.0 °C in June 2016 and June 2019, respectively. The August 2016 profile showed a thermocline at 5-10 m depth where temperature dropped from 25 °C to 21 °C and then declined more slowly to 19 °C deeper (Fig. 4).

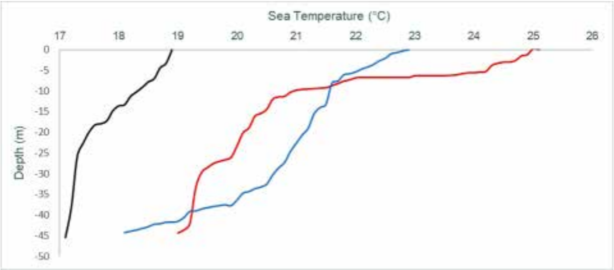

Fig. 4. – Temperature profile in June 2016 (black); August 2016 (red); June 2019 (blue).

For each acquisition, fish accumulations were well focused on the Ile Verte walls and on the rocky reefs (Fig. 5). They appeared as large continuous and dense schools rather than numerous medium schools. The accumulation on the wrecks was worth more noteworthy than the surroundings, except in August 2019 where a large fish school was observed (Fig. 5C). With the exception of the north-west boundary of the site, there were no major fish schools on the sedimentary plains (Fig. 5). Numerous single detections were pointed on this habitat, however. Fish detection were more numerous in June 2016 (38,498 detections) than in August (23,268 detections), while the number of fishes in August 2019 reached 57,472 detec-

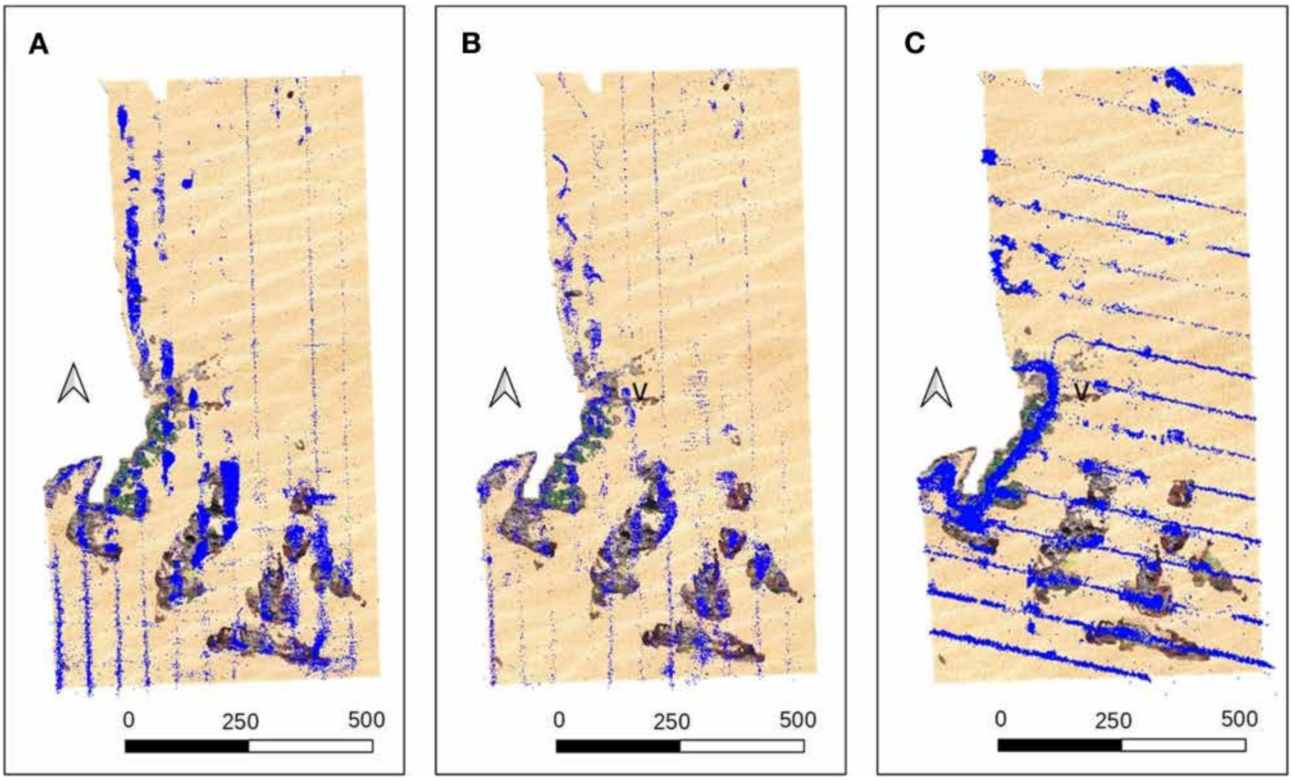

Fig. 5. – Scatter plots representing fishes in **A**: June 2016; **B**: August 2016; **C**: June 2019. Each blue dot represents a detected fish whatever its size.

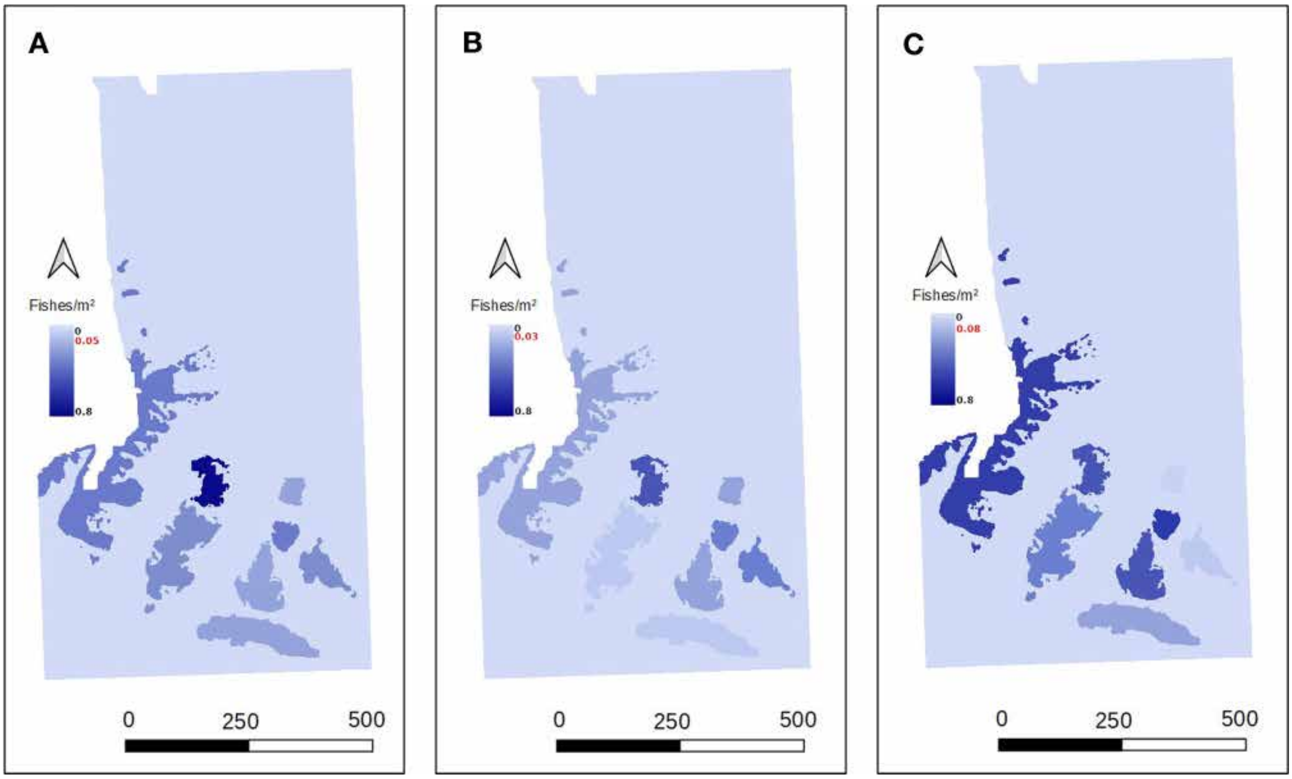

Fig. 6. – Local fish density in fishes/m² in **A**: June 2016; **B**: August 2016; **C**: June 2019. The red value corresponds to the mean number of fishes/m².

Table II. – Number of fish detections and proportions per depth categories.

Position in the water column and depth range (m)	June 2016		August 2016		June 2019	
Near surface [0, 20[	25,695	66.8 %	14,316	61.5 %	29,590	51.5 %
Mid-depth [20, 40[	10,892	28.3 %	6,979	30.0 %	23,653	41.1 %
Deep [40, max]	1,903	4.9 %	1,971	8.5 %	4,253	7.4 %
Total	**38,492**		**23,268**		**57,472**	

tions (Table II). Concerning fish densities, soft sediments showed low fish densities (from 0.01 fishes/m² in June and August 2016 to 0.03 fishes/m² in June 2019), however close to the average value (Fig. 6). On average over the three acquisitions, the Ile Verte walls had a higher fish abundance (0.31, 0.14 and 0.38 fishes/m² in June, August 2016 and June 2019, respectively) than the rocky reefs (0.25, 0.15 and 0.24 fishes/m² in June, August 2016 and June 2019, respectively; Fig. 6). Among the reefs, Rousteau Nord and Pierre du Jas showed the highest fish abundances, while the largest reefs – Rousteau and Levant – were among the poorest (Fig. 6).

Whatever the time of acquisition, fishes were mainly detected in the twenty-first meters, representing more than 50 % of all detections in general and up to 66.8 % in June 2016 (Table II). By averaging the three acquisitions, about a third of detections were located in the mid-depth waters, between 20 and 40 m. Finally, less than 10 % of the fishes detected were found in the deepest waters of the study site (Table II).

DISCUSSION

This work was aimed at studying the capacity of a compact MBES to provide a precise map of marine habitats along with fish accumulations, with the final purpose of providing a spatial ecosystem-based approach to managers and stakeholders.

MBES advantages and operability

The main original outcome of this research effort is to put into light the possibility to have a global spatial review of an area with a single acoustic acquisition. Modern MBES are able to collect both bathymetry, backscatter and WCI without compensation of data quality and quantity. The WCI alone is used in various fields of research, such as biology, archeology, physical oceanology (Colbo *et al.* 2014), and reveals its full potential when combined with the other MBES outputs. The versatility of the MBES used in this study (a R2Sonic 2022) is an advantage, on the condition of having the capacity to properly collect, store and process all the acoustic data, without forgetting GPS and Navigation information. However, this MBES is commonly dedicated to bathymetry and seafloor imagery acquisition, it is not a fishery-dedicated tool. Moreover, the system is adjusted and calibrated for the seafloor measurement and not for the water column observation. This implies several consequences: (1) acoustic noises can alter the WCI and hinder their processing; (2) only the pelagic fishes can be seen, the demersal and benthic ones are mingled with the seabed noise; (3) When compared to fishery MBES, the one used in this work is installed under the hull at mid-length of the boat and not in the bow as a forward-looking MBES. Thus, the WCI is a vertical cut of the fish schools, and not a fish school seen as a whole; (4) as the MBES is used to map the seabed, the acoustic signals are emitted towards the nadir. There is also a good cover of the seabed, but the water column is not entirely scanned. This explains why the scatter plot is composed of bands North-South oriented.

The place on the hull and the looking direction of the MBES become important when the avoiding-boat behavior of schools is considered. In a previous study on fish behaviors regarding MBES acquisition, Soria *et al*. (1996) explained that a fish school feels the vessel coming far away. From this moment, a first part of the school avoids laterally the vessel and is not seen. Then, when the disturbance, *i.e.*, the boat, arrives above the school, another part of the school also avoids it laterally and can only be seen on the edge of the WCI. What remains of the school is the little part recorded by the MBES while fishes are avoiding the disturbance by diving. That is why Soria *et al*. (1996) and Paramo *et al.* (2010) used a MBES with a 45° tilt from the nadir. Nevertheless, it is hard to say how many fishes are missed in the detection process.

Processing of the WCI

Although the algorithm developed to extract fish information from the WCI is able to automatically detect fish targets without human intervention, this type of processing has currently several drawbacks. In this line of thought, wherever in the area, several fishes are not detected because they do not pass the filters. Some false alarms exist too. This is not disturbing the scatter plot and it does not influence so much the local density as well. However, it is more troublesome on the vast soft sedimentary seafloor where it seems that too much detections were performed. Thus, all these points are lonely, close to the seabed and around the nadir and could correspond to some acoustic noise. However, when looking the WCI at great depths, these points look like fishes for the algorithm as well as the human eyes. On the one hand, should these detections might be noises, then the sedimentary seafloor is really deserted by pelagic fishes. On the other hand, they could be true detections and these vast areas might be more populated than expected, while remaining very sparsely populated. We decided to display them, at least so that the reader may view the boat trajectories and assess the difficulty to validate fish detections.

Despite all these drawbacks, the processing method is fast enough to provide quick results, meaning with a mid-powerful computer, one hour of acquisition is processed in one hour of computer calculation. Moreover, improvements are under study in order to obtain a higher precision in target detection while decreasing the processing time. Another enhancement under progress concerns the pre-processing of the WCI to reduce the noise or calibrate the background noise.

Contribution to the ecosystem based management

This research work suggests a new approach to evaluate the ecological status of an area of interest with different levels of analysis that can be adapted according the characteristics of the managed zone (*e.g.*, large areas, complex patchwork of marine habitats, extensive seagrass meadows), and the final aim of the study (*e.g.*, MSFD, seascape analysis, halieutic research). It also allows to obtain spatial data for several functional compartments of the Mediterranean ecosystems *P. oceanica* meadows, algae-dominated rock reefs and coralligenous communities (Table III), according to the conceptual representa-

Table III. – Functional compartments of the Mediterranean ecosystems investigated by coupling marine habitat maps and fish detection in the water column.

Ecosystem	Functional compartments	Type of data
Posidonia oceanica meadows	*Posidonia* leaves	Area covered
	Planktivorous teleosts Piscivorous teleosts Predatory teleosts Herbivores 1	Fish number
Algae-dominated rock reefs	Multicellular photosynthetic organisms	Area covered
	Herbivorous teleosts Piscivorous teleosts Omnivorous teleosts Invertivorous teleosts Planktivorous teleosts	Fish number
Coralligenous communities	Builders	Area covered
	High-level predators Predatory teleosts Planktivorous teleosts	Fish number

tion of Personnic *et al.* (2014), Thibaut *et al.* (2017) and Ruitton *et al.* (2014), respectively. Obviously, this spatial approach does not replace the qualitative and quantitative assessment made by scuba diving but rather intervenes as a complementary tool allowing cross-validation. It may also be used to produce a first investigation of an area with few data on fish accumulations and benthic habitats for a more efficient scuba diving evaluation later.

One of the limits of a spatial approach through discrete acquisitions relies on the instantaneousness of the maps produced. Moreover, the pelagic fish distribution depends of numerous biotic and abiotic parameters that are virtually impossible to wholly assess such as marine habitats, sea temperature, salinity currents, day period, light intensity (Saraux *et al*. 2014). This difficulty is illustrated in this study where the combination of sea temperature, seascape and marine habitats are not sufficient to explain the fish distribution observed. In fact, if the fish biomass is increasing with sea temperature, the August 2016 acquisition should have been the survey with the most detections in total and especially within the twenty first meters where temperatures were the highest. On the contrary, this acquisition shows about two times less fish detections than in June 2016 and 2019 (Table II). The same observation is made for the June acquisitions, due to the sea surface temperature, the 2019 one should have presented a higher fish abundance than the 2016 one.

Although this first application is promising for an effective evaluation in a management purpose, further developments are still required for a deeper investigation of the link between marine habitats and fish accumulations. Moreover, the anthropogenic impacts and abiotic factors must be included in the analysis for a sharper ecological assessment. For instance, the fish densities need to be calculated per habitat and not only by area. Likewise, the vertical dimension should be more exploited with volumetric analysis rather than the only study of the vertical repartition of fish schools. An important research effort is also required to link the size of the WCI acoustic targets with the one of actual fishes to produce an evaluation of the biomass per surface and/or volume (even with a relatively large margin of error). At last, an ecological index can be built on the comparison between different sites (impacted and protected) at various seasons to link anthropogenic impacts with the ecological status.

This research effort clearly highlights the capacity of spatial acoustic data obtained with a MBES to provide quantitative information on the marine habitats and fish distribution. This work aimed at paving the way to further developments to provide managers with effective spatial tools to evaluate the ecological status of key Mediterranean marine ecosystems. If coupled with *in situ* underwater observations, this spatial approach has the potential to give a complete view of underwater key biological systems as never been before.

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A SPATIAL DECISION SUPPORT SYSTEM FOR THE SUSTAINABLE MANAGEMENT OF FISHING IN MARINE PROTECTED AREAS

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ENVIRONMENTAL IMPACT
EMERGY ANALYSIS
STRONG SUSTAINABILITY
GIS
SPATIAL DISTRIBUTION

ABSTRACT. – Recreational and professional fishing are spread all over the world, generating revenues and opportunities but impacting on limited fish stocks. Even if stocks can recover, over-exploitation and permanent damages to ecosystems must be avoided. The aim of this work is to provide a Spatial Decision Support System (SDSS) to develop sustainable management strategies for fishing in marine protected areas (MPAs). The system, based on a generalized and iterable procedure and tested on the Portofino MPA (Italy), can be applied to any area. Starting from the assessment of the natural capital, the SDSS allows an ecosystem-based evaluation of fishing impact based on information (namely the harvest per species) gathered during the authorization procedure, mandatory for fishermen in MPA. The system assists MPA managers to understand if the MPA is able to support fishing or if there is an over-fishing status, by subtracting the fishing annual catches flow from the fish biomass production flow for each species. Being fishing spatially distributed, the SDSS generates results (*e.g.*, values, maps) at different spatial level: MPA, protection zones and even smaller. The implementation of fishing management procedures through the SDSS allows the quick update of results and the simulation of alternative management scenarios.

INTRODUCTION

Recreational and professional fishing

The marine coastal environment and its natural resources offer goods and services from which humans benefit, both to satisfy their necessities for life (*e.g.*, fishing for fish stocks needed for food) and to improve their well-being (*e.g.*, recreational activities). Fishing is one of the main and traditional activities performed at sea. Along seacoast, we can find small-scale professional fishing, focused on an economic sphere, and recreational fishing, focused on a ludic and social sphere.

In many parts of the world, including the Mediterranean Sea, there is a high level of exploitation of the sea and its resources. The degenerative phenomenon of the system is determined by over-fishing, where the intensity of the catches is greater than the ability to reproduce natural stocks, with a consequent reduction in the stocks themselves. A reduction in natural stocks inevitably leads to the collapse of the fishing activities themselves. In order to obtain a sustainability that is environmental and economic alike, it is therefore necessary to avoid this collapse and to ensure that fishing activities do not damage the coastal marine environment (https://ec.europa.eu/fisheries/; FAO 2018).

The Italian fish production in 2016 saw a reduction of about 60 % compared to catches in the mid-1980s (about from 400,000 to 170,000 tons; ISTAT 2003, Pauly *et al.* 2014, http://dati.istat.it/ precisely in “Pesca serie interotte” folder under “Agricoltura”, “Foreste, caccia, pesca”).

Recreational fishing is very widespread in the Italian coastal seas and, unlike professional fishing, it involves people of all ages and gives the opportunity to stay in the open area in contact with nature, relax, socialize and, in case of successful fishing trips, consume extremely fresh fish products (Cappanera *et al.* 2010, 2012). Despite its small size, for some time now, environmental associations, scientific community, professional fishermen and part of the political world have considered recreational fishing a real problem. This is because this activity addresses limited, exhaustible and often over-exploited resources and so many stakeholder categories, although driven by different interests, agree that recreational fishing must be properly managed (Cappanera *et al.* 2010, Radford *et al.* 2018). For a correct conservation of the ecosystem it should not be neglected that, often, the stocks from which sport and professional fishermen draw are the same (Campodonico 2010, Prato *et al.* 2016), exerting both a pressure on the marine coastal environment and entering into conflict each other. Moreover, even if the methods employed by recreational fishing are usually considered as having a low environmental impact, the cumulative impacts of recreational fishing, have been assessed as comparable to or even greater than those generated by the professional sector (Cooke & Cowx 2004, West *et al.* 2015, Brown 2016).

To effectively manage these two activities, it is necessary to know their magnitude. This has always been a complex task in Italy for recreational fishing, even if, since December 2010, the Ministry of Agricultural, Food and Forestry Policies issued a decree obliging sport fishermen to register and take part in a census, with the aim of quantifying their number and determining the degree of competition with professional fishing (Campodonico 2010, Cappanera *et al.* 2012). Despite this regulation, there is a lack on qualitative and quantitative data on real fishing days and fishing catches (Cappanera *et al.* 2010, Radford *et al.* 2018). The less rigorous monitoring of recreational fishing, in comparison with professional one, is worldwide spread and there is the need to incorporate recreational fishing data into stock assessments and coastal zone management plans, especially where it is particularly important, as in the Mediterranean Sea (Cooke *et al.* 2006, Lloret *et al.* 2008).

Marine protected areas and fishing management

The fishing management is important everywhere, but it is even more important in Marine Protected Areas (MPAs) where it must be a priority. According to the objectives dictated by the Italian framework law on Protected Areas (Law 394 of 6/12/1991), protection and conservation of environment and its resources must be guaranteed, compatibly with existing traditional activities. Especially in areas where fishermen have a strong socio-political weight or where it is a traditional activity.

In the context of ecosystem-based fisheries management, MPAs have often been identified as an appropriate tool to address a variety of fisheries management problems related to the conservation of exploited stocks, biodiversity conservation, exploitation of fishery yields and other social objectives (Dugan & Davis 1992, Costanza *et al.* 1997, Roberts *et al.* 2001, Gerber *et al.* 2003, Halpern 2003, Murawski 2007). MPAs, in fact, can help in the protection of fish stocks and to manage traditional small-scale professional and recreational fisheries. In order to pursue a sustainable development, it is important to adopt a system view, considering the synergistic and conflicting action of professional and recreational fishing. Indeed, even if these are competitive activities, they have interacting ecological effects, which are difficult to understand as the whole (Prato *et al.* 2016).

Natural capital

Costanza & Daly (1992) elaborated the natural capital (NC) concept in relation to human and manufactured capital. NC is defined as the stock of natural resources generating valuable flows of different types of ecosystem goods and services. Human capital comprises individuals' capacities, while manufactured capital includes material goods generated through economic activity and supply chain (UNU-IHDP & UNEP 2012).

Recently it has been widely accepted the human well-being is tightly linked to NC. From NC stocks ecosystem functions arise, representing the potential to generate services, and ecosystem services arise from functions in turn. Ecosystem services (*e.g.*, harvest of resources, such as fish, for food and recreational purposes) represent benefits that ecosystems directly and indirectly generate for the mankind and from which well-being arises (De Groot *et al.* 2010, De la Fuente *et al.* 2019). Since in this pathway ecosystems generate well-being, it is important to know and to evaluate each step of this so-called "pathway from nature to well-being" (De Groot *et al.* 2010, Paoli *et al.* 2017). Specifically, it is important to being aware and measure NC since in absence of it the pathway cannot start. This means that only if NC is preserved the supply of services in the future and at the actual level can be guaranteed (De Groot *et al.* 2012). With this supply it is also assured the chance to access to the ecosystem services and to the economic benefits generated by their exploitation.

As a consequence, in terms of conservation, the goal of MPAs should be a "very strong" sustainability. The strong sustainability theory, developed in the last decades, claims that natural and human capital are not mutually replaceable, so each of them must be kept constant, since the production of the second depends on the availability of the first (Chiesura & De Groot 2003). The "very strong" sustainability takes a step forward on ecological aspect and implies that every component or subsystem of the natural environment must be preserved (Van den Bergh 2010). In a precautionary approach and in absence of clear evaluation of NC depletion consequences, the "very strong" sustainability theory should be embraced.

Spatial decision support system for fishing management in MPA

For an effective management of an MPA and its activities, policies need to be based on informed decision-making processes. The development and implementation of innovative systems that facilitate this process are increasingly necessary, especially in marine-coastal environments. Furthermore, this responds to national, regional and international request of territorial planning and integrated and sustainable management of the coastal zone (2014/89/EU, 2008/56/CE, 2030 Agenda for Sustainable Development, Strategic Plan for Biological Diversity 2011-2020, Mediterranean Strategy for Sustainable Development 2016-2025, ICZM protocol).

Spatial Decision Support Systems (SDSS) are designed to facilitate the decision process for complex problems, improving the consistency and the quality of these decisions, also taking into account the spatial dimension of the problem (Malczewski 1997, Rizzoli & Young 1997, Cortés *et al.* 2000, Poch *et al.* 2003, Dapueto *et al.* 2015).

A SDSS for the identification of sustainable management strategies in marine-coastal areas, in particular for fishing activities in MPAs, is here proposed. It consists of an information system that supports decision-makers in choosing between alternative solutions, integrating artificial intelligence methods, GIS components, mathematical-statistical techniques and environmental ontologies. Taking advantage of modern information technology and software, the entire system is computerized, both storing alphanumeric and spatial data and implementing the procedures developed, in order to optimize and speed up the decision-making process.

Specifically, the SDSS is a tool that helps to evaluate the impact on the environment due to human activities, namely in this research fishing. In particular, the impact has been considered as the fish harvest that corresponds to the removal of biophysical resources, potentially affecting the NC. It allows to understand if the MPA system is able to support fishing activities or if the system is in an over-fishing condition.

In particular, here the harvest of all fishing activities is considered, in order to understand the overall state of the system. Depending on the management needs, the SDSS, due to its plasticity and adaptability, could be used to evaluate the effect of the individual activities (professional and recreational), which, being very different, could require different management strategies.

MATERIALS AND METHODS

Study area – Portofino MPA: The Portofino MPA (Fig. 1) is an area of great interest for the conservation of Mediterranean biodiversity surrounding the Portofino Promontory (north-western Italy). It achieved the status of SPAMI (Specially Protected Areas of Mediterranean Importance) in 2005. In particular, the area is mainly characterized by two priority habitats, *Posidonia oceanica* (Linnaeus) Delile and coralligenous. The Portofino MPA, as others Italian MPA, is divided in 3 zones with different

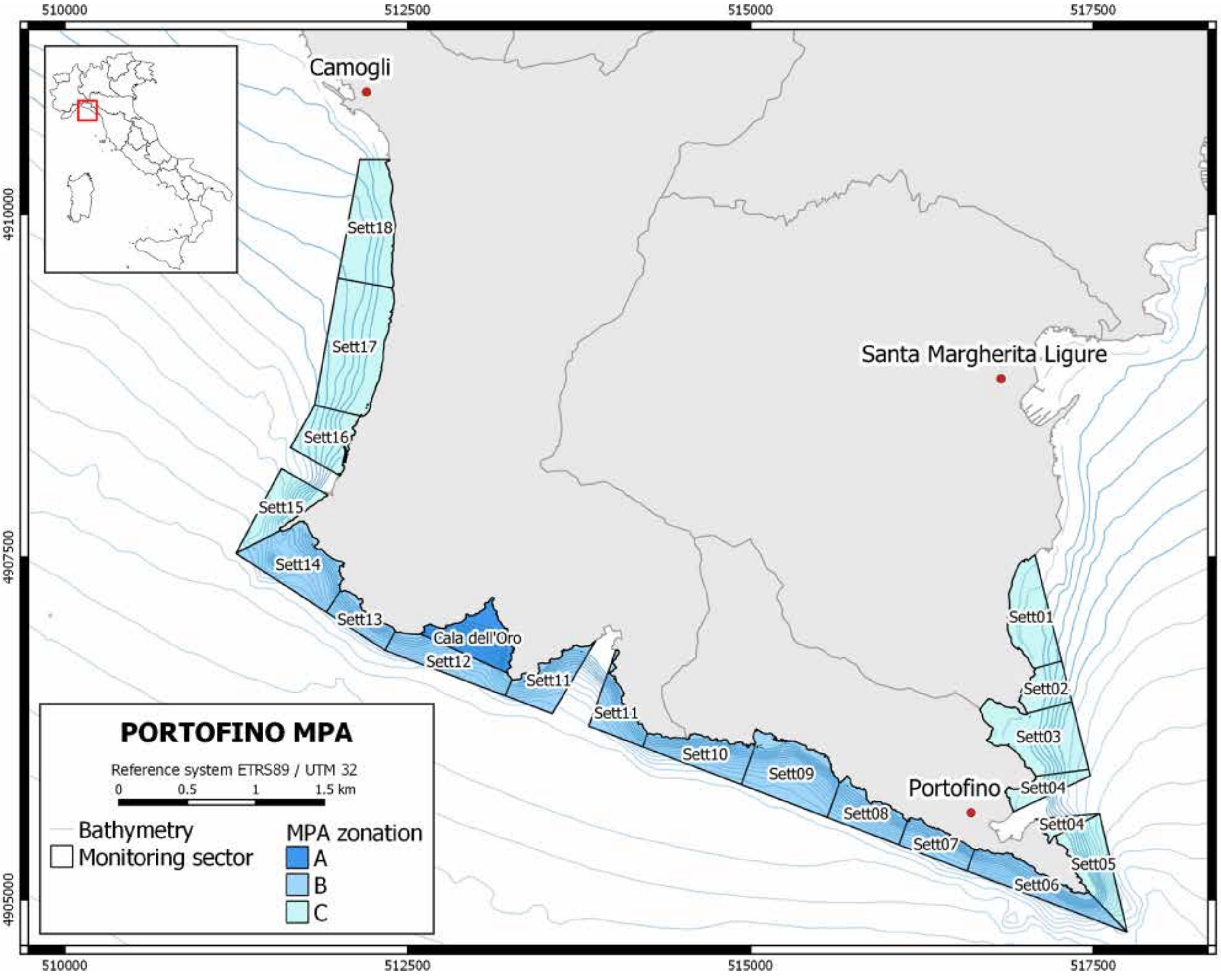

Fig. 1. – Marine Protected Area of Portofino, Italy (realized with QGIS software, version 3.10.1).

level of protection: zone A (integral reserve, “no entry no take zone”), zone B (general reserve) and zone C (partial reserve).

The high environmental relevance of the Portofino MPA is associated with a high human presence and several activities bringing socioeconomic benefits but, at the same time, also logistical and environmental problems (Cappanera *et al.* 2009, 2010).

Recreational and professional (mainly artisanal with small boats, less than 10 meters long) fishing play a relevant role among these activities, also entering into conflict with each other and exercising a synergistic pressure on stocks (Venturini *et al.* 2017, 2019).

As in many MPAs, in Portofino the application of a prohibition rule on recreational and professional fishing is not possible because they are well-established realities: although the consensus process is going on, it’s far away to be closed. In particular, professional fishing represents a craft activity rooted in tradition that historically has always been the major source of food, employment and economic benefit for the MPA municipalities (Cappanera *et al.* 2010, 2012).

Therefore, the MPA managing body issued an authorization procedure that allows fishing in the MPA.

Fishing access rules and control: The regulation of the Portofino MPA defines where and how recreational and professional fishing activities are allowed inside the MPA (art. 20 and 21). Specifically, for these activities an authorization is mandatory. Users request the managing body to carry out the activity; the managing body, verified the compliance with the requisites foreseen by the MPA regulation, issues the authorization against a payment.

Recreational fishing is permitted both in zones B and C to residents, only in zone C to not residents. There are some general limitations (*e.g.*, species, maximum catch weight, minimum catch size) and some specific ones depending on the area and type of user (*e.g.*, fishing gears, shore or boat fishing).

In the Portofino MPA, professional fishing is allowed both in zones B and C only to resident fishermen and to fishing enterprises and cooperatives with registered office in the MPA at the date of entry into force of the MPA regulation. For this type of fishing there are general limitations too.

For monitoring purposes, the managing body provides recreational fishermen with a logbook on which must be noted information about catches: date and time, fishing site, caught fishes (species, biomass and length), fishing technique and tools (Venturini *et al.* 2017, 2019). The authorization renewal can only occur after the fisherman has delivered the compiled logbook of the previous year.

Authorized professional fishermen must communicate annually to the managing body the periods, tools used and fishing methods within the MPA for monitoring purposes. The quantification of fishing effort of professional activity, instead, comes from landings evaluation, carried on by monitoring of the catches with an MPA operator on the quayside at the time of landing. This type of monitoring follows the protocol applied under the EU Interreg MED project FishMPABlue2 (Di Franco *et al.* 2018).

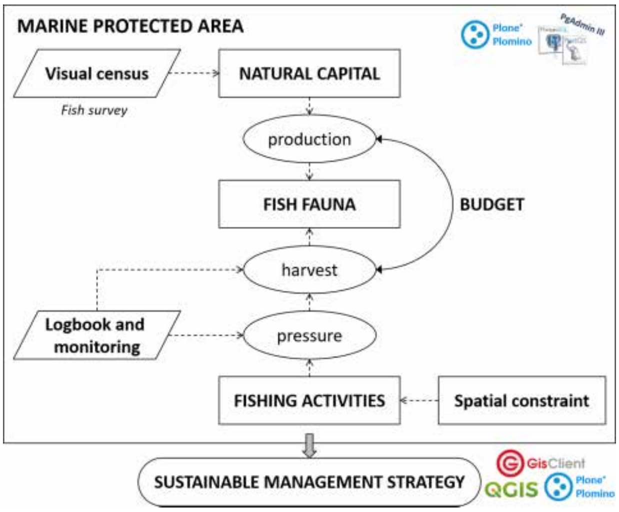

Fig. 2. – Spatial Decision Support System (SDSS) schema.

For any violations, the Portofino MPA regulation (art. 32) provides for the application of the Italian Framework Law on Protected Areas (no. 394/1991, art. 29 and 30). If the violation involves a modification in the environment status, the immediate suspension of the harmful activity is ordered to return to the condition before the damage or to reconstitute the species, at the expense of the offender. The violation can lead to suspension or revocation of the authorizations.

In order to improve and focus the monitoring of fishing and other activities, the MPA is divided into 19 sectors (Fig. 1) and the SDSS is designed to reach the spatial resolution of a single monitoring sector.

SDSS architecture: The system procedure (Fig. 2) allows the evaluation of the direct environmental impact of fishing activities (fish catches). Basically, starting from information gathered during the authorization and the monitoring procedure (mandatory for each fisherman in the MPA), and applying an ecosystem approach, the use and production of NC are assessed. Subtracting the annual biomass catches flow from the fish secondary production flow, it is possible to assess the status of the MPA system. The procedure is generalized and iterable, so that it can be applied to any area, but it was tested on the Portofino MPA.

To facilitate the collection and management of MPA data, coming from both MPA users and managers, and to share information and results of the decision-making system with managers, the SDSS include a user-friendly MPA website (https://www.portofinoamp.it). This website was realized basing on MACISTE (MArine Coastal Information sySTEm), an information system able to manage in an integrated way cartographic and alphanumeric information of the marine environment (Povero *et al.* 2010). In particular, the Portofino MPA website was developed using the content management system Plone (version 4.31.18) and the application Plomino (version 1.19). Collected data are stored into a geodatabase, linked with the website and realized with PostgreSQL/PostGIS (version 9.6).

This structure could facilitate both the users and managers and the relationship between them. Specifically, the authorization system has been implemented. In particular, the website has two sections: one for users and one for the MPA managing body. In the first section, each user, once registered on the website, can access his private area and apply for authorization for different activities (*e.g.*, fishing, diving, boating). Once obtained the authorization, the user can manage his activities performed in the MPA (*e.g.*, communication of daily exits in MPA, compilation of logbooks). In the MPA section the managers can view and administer the activities of all users. Thanks to this system fishermen can apply for recreational or professional fishing and fill in logbooks (when required). Thanks to the responsiveness of the website, fisherman can fill in logbook directly from its mobile phone at the time of capture, thus registering the precise harvest point thanks to GPS, and can upload photos of the caught species.

The modules within these sections have been designed to be, on one hand, general and applicable to the different MPAs and, on the other, adaptable to specific needs. From the authorization system detailed information about fishing activities is collected.

Being fishing spatially distributed, the SDSS generates results, as well as values and maps, at different spatial level (MPA, protection zones and even smaller), based on the MPA management needs. In particular, a WebGIS on GisClient (version 4) was created for displaying maps (Povero *et al.* 2010).

The SDSS procedure, specifically, considers as input: the areas in which fishing is allowed (spatial constraint), the fish species and their distribution in the environment (fish fauna), the distribution of recreational and professional fishing (pressure), and the variety and distribution of catches (harvest). Information about pressure and catches are gathered from fishermen logbooks and monitoring activity. The outputs generated are the estimation of the impact due to each captured species, the overall impact in the MPA (number and biomass) and the loss of associated NC (in ecological terms and in monetary equivalents). Results are generated both as number and spatial distribution (impact maps).

Evaluation of natural capital: To check whether fishing activities in MPA erode NC or not, a comparison between the annual flow of harvest and the annual net flow of NC generated by the MPA is performed, following an ecosystem-based approach. The net flow is obtained from the difference between the biomass secondary production and the mortality rate. This information allows to understand if the MPA is able to sustain the losses or if the NC is eroded.

The fishing activities impact is here accounted as the catches of fish species both in terms of removed biomass and NC decrease.

The assessment of production (*i.e.*, secondary production) and loss (*i.e.*, biomass subtracted by fishing) is carried out in biophysical terms.

The evaluation of secondary production is based on information gathered from visual census campaigns. The visual census is a non-destructive widely adopted technique for the study of the littoral fish communities, particularly in protected habitats, due to its minimal impact (Brock 1954, Harmelin-Vivien *et al.* 1985, Guidetti 2002, Azzurro *et al.* 2007). These field activities allowed to obtain data on species abundances and size distribution then converted into fish biomass by using parameters obtained from FishBase (http://www.fishbase.org/).

The biomass subtracted is extrapolated from logbooks and monitoring reports, where the number and the size or weight of caught species are reported.

The calculation of biomass has been obtained using the Von Bertalanffy equation (Baker *et al.* 1993):

$$W_j = a_j \times L_j b_j$$

where W_j is the weight of the single individual, L_j its length and a_j and b_j are the constants of Von Bertalanffy specific for the species j.

Considering the production/biomass ratio and the mortality rates of each species, biomass is converted into net secondary production according to the following equation:

$$NSP_j = \sum_{j=1}^{n} W_j \cdot \left(\frac{P_j}{B_j} - M_j \right)$$

where NSP_j is the net secondary production, W_j the biomass of the single individual, P_j/B_j is the production/biomass ratio and M_j is the mortality rate for the species j. The production/biomass ratio for each species is obtained from literature (Pinnegar & Polunin 2004, Coll *et al.* 2007, 2008, Diaz Lopez *et al.* 2008, Barausse *et al.* 2009, Heymans *et al.* 2009, Piroddi *et al.* 2010, Lassalle *et al.* 2011, Bănaru *et al.* 2013, Bayle-Sempere *et al.* 2013, Prado *et al.* 2013, Torres *et al.* 2013, Corrales *et al.* 2015). The mortality is obtained by using the following equation:

$$M_j = 1 - e^{(-k_j \times 1.63)}$$

where M_j is the mortality rate and k_j the Von Bertalanffy growth coefficient for the species j.

In order to make comparable fish production and subtraction, only species detectable by visual census are considered in the estimation of harvest (*e.g.*, cryptic species, species hidden because of presence of operators, pelagic species are omitted).

The intrinsic value of the considered good or service (fish) is assessed as both biophysical and monetary value. At this purpose, the methodology proposed by Vassallo *et al.* (2017) for MPAs is applied. This method is based on the emergy analysis, an environmental accounting method introduced by Odum (1988, 1996) that follows an ecocentric approach aimed at assessing the environmental performance and sustainability of processes and systems on the global biosphere scale (Vassallo *et al.* 2017). Emergy is a donor side approach since it measures the nature's investment, in terms of resources used, to create and maintain a good or a service (namely to fish species, produced or subtracted). The investment is evaluated in ecological terms as emergy value. According to emergy, all inputs are accounted as solar equivalent Joules (sej), calculated as the total amount of solar available energy directly or indirectly required to make a given product or flow. The emergy required to generate one unit of input is named Unit Emergy Value (UEV) or emergy intensity (sej/J, sej/g, sej/€). Raw data on mass, energy, labour, and money input flows are converted into emergy units, and then summed into a total amount of emergy used by the investigated system. In this work 15.20E+24 sej emergy baseline (Brown & Ulgiati 2010) was used for emergy calculation.

This value can be converted into monetary equivalents, expressed in emergy-euro (em €), using an emergy-to-money ratio (Brown & Ulgiati 2004a, b) to better convey the importance of NC to policy makers and other stakeholders.

Vassallo *et al*. (2017) methodology has already been employed in the Italian national project "The environmental accounting of Italian MPA (EAMPA)" (Franzese *et al.* 2017, Picone *et al.* 2017, Paoli *et al.* 2018) and in the EU Interreg Maritime project "Integrated management of ecological networks through parks and marine areas (GIREPAM)" (Paoli *et al.* 2019).

Very strong sustainability approach: As highlighted above, the management of fishing in MPAs must be addressed with a view to sustainability, *i.e.*, the catches must be controlled so that the environment can counterbalance this loss with an appropriate production of new resources. Therefore, the exhaustion of natural resources must be taken into account and their uncontrolled exploitation must not be encouraged. Otherwise, an unsustainable use of resources could make impossible to use them in the future at the current level or could completely exhaust them (Gowdy & O'Hara 1997). From all these considerations it is clear that a declining NC is an indisputable sign of non-sustainability (Vitousek *et al.* 1997).

The assessment of the environmental impact due to fishing can be approached in different ways: considering the overall species or individual species in a given area.

The SDSS allows to carry out a very strong sustainability analysis producing as results a sustainability map that permits to see the status of the system:

– sustainable: all species are in surplus, *i.e.*, the production is greater than the subtraction (green);

– balance: all species are in balance, *i.e.*, the production is the same of the subtraction (blue);

– not sustainable: at least one species is in deficit, *i.e.*, the subtraction is greater than the production (red).

If only one species is in deficit, all the system is considered in deficit since the very strong sustainability is not maintained.

RESULTS

Evaluation of natural capital: fish production

The total annual net secondary production of fishes in Portofino MPA is 24 645.87 kg/y, corresponding to 1.38E+18 sej/y and 1,432,599.15 em €/y. The species with highest biomass values in MPA are *Diplodus vulgaris* (Saint-Hilaire, 1817; 33.58 %) and *Diplodus sargus* (Linnaeus, 1758; 24.07 %), followed by *Sarpa salpa* (Linnaeus, 1758; 11.62 %). As emergy and monetary value the major production is for *Dentex dentex* (Linnaeus, 1758; 35.23 % *vs* 5.41 % of biomass), *D. vulgaris* (20.27 %) and *Epinephelus marginatus* (Lowe, 1834; 20.22 %).

Considering the comparison among different sectors, the most productive ones, both as biomass and emergy, are Cala Oro corresponding to zone A, 17 and 18 in zone C west (37.42 % of biomass and 46.50 % of emergy overall). Comparing species, the species more representative of the overall MPA are the same that generate the greater production of these sectors (*D. vulgaris*, *D. sargus* and *S. salpa* for biomass; *D. dentex*, *D. vulgaris* and *E. marginatus* for emergy), except for sectors 3 and 5 where *Pagellus erythrinus* (Linnaeus, 1758) is the fish originating the greatest production.

Results by sector are reported in Table I and Fig. 3.

Evaluation of natural capital: fishing harvest

Data on catches are extrapolated from the geodatabase, after they have been put into online logbooks on the web-

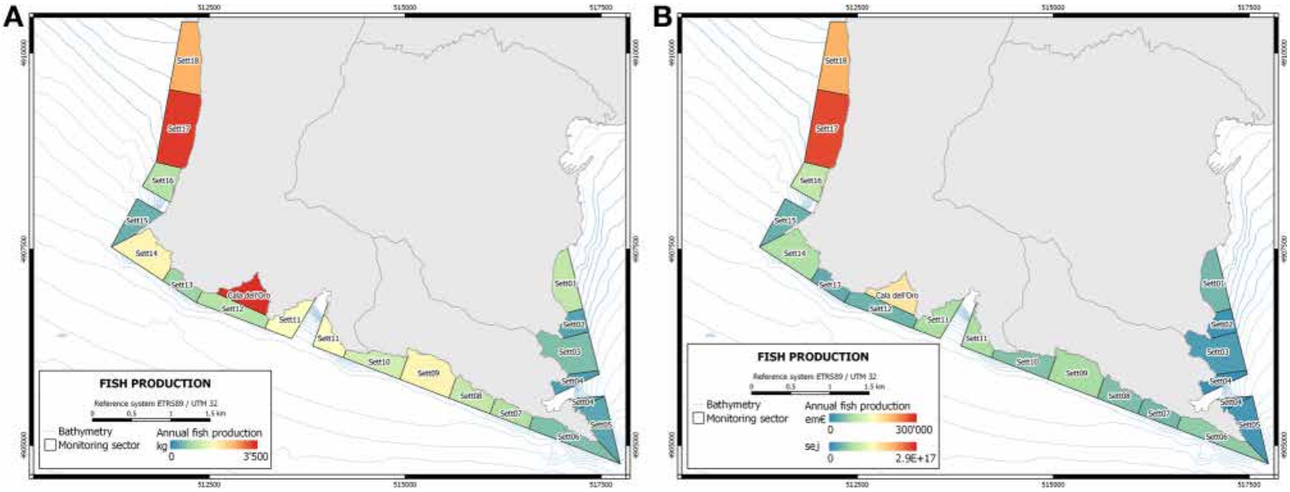

Fig. 3. – Fish production in Portofino MPA at sector level. **A**: Biomass (kg/y); **B**: Emergy (sej/y) and monetary values (em €/y) (realized with QGIS software, version 3.10.1).

Table I. – Annual secondary production, pressure and harvest due to recreational and professional fishing in Portofino MPA.

Sector	**Secondary production**			**Pressure**		**Recreational fishing harvest**			**Professional fishing harvest**		
	Biomass (kg/y)	**Ecological value (sej/y)**	**Monetary value (em€/y)**	**Recreational fishing (day/y)**	**Professional fishing (day/y)**	**Biomass (kg/y)**	**Ecological value (sej/y)**	**Monetary value (em€/y)**	**Biomass (kg/y)**	**Ecological value (sej/y)**	**Monetary value (em€/y)**
Cala Oro	3,357.55	1.66E+17	172,505.06								
01	1,117.56	4.21E+16	43,878.48	150	9	20.94	3.82E+14	397.45	0.53	6.69E+13	69.70
02	340.60	1.33E+16	13,893.74	119	24	5.35	5.95E+14	619.61	0.60	7.49E+13	77.98
03	574.91	2.34E+16	24,386.00	173	15	3.96	3.52E+14	366.19	0.05	6.44E+12	6.70
04	272.26	1.19E+16	12,363.04	73	15	1.37	5.83E+13	60.69	0.05	6.44E+12	6.70
05	361.50	1.77E+16	18,485.15	120	15	2.16	2.70E+14	281.57	0.05	6.44E+12	6.70
06	579.16	5.83E+16	60,732.19	130	17	14.82	4.11E+15	4,277.89	0.47	6.81E+13	70.89
07	974.02	4.19E+16	43,668.66	60	24	14.82	4.11E+15	4,277.91	0.73	7.78E+13	81.05
08	1,165.86	4.75E+16	49,463.69	90	24	15.84	3.86E+15	4,017.63	0.73	7.78E+13	81.05
09	1,865.79	7.32E+16	76,249.71	93	24	25.10	2.67E+15	2,780.23	0.73	7.78E+13	81.05
10	1,283.38	5.18E+16	53,930.49	79	24	18.81	1.29E+15	1,346.36	0.73	7.78E+13	81.05
11	1,732.44	7.42E+16	77,287.21	103	27	32.81	2.18E+15	2,271.34	14.95	1.59E+15	1,655.31
12	1,018.42	3.94E+16	41,086.63	42	3	1.68	3.46E+14	359.97	14.22	1.51E+15	1,574.25
13	802.22	3.37E+16	35,099.72	79	3	14.29	2.39E+15	2,491.65	14.22	1.51E+15	1,574.25
14	1,867.66	7.60E+16	79,178.32	242	72	47.97	1.06E+16	11,066.36	269.84	2.87E+16	29,868.91
15	454.32	3.71E+16	38,651.29	165	72	12.95	2.26E+15	2,351.15	172.03	2.51E+16	26,125.02
16	1,012.76	9.40E+16	97,965.71	77	24	2.06	1.70E+14	177.01	95.80	1.40E+16	14,548.27
17	3,305.84	2.64E+17	275,192.27	117	54	13.71	3.15E+15	3,286.29	68.67	1.00E+16	10,427.86
18	2,559.61	2.10E+17	218,581.79	66	54	0.94	1.46E+14	151.96	68.67	1.00E+16	10,427.86
Total	***24,645.87***	***1.38E+18***	***1,432,599.15***	***1,979***	***500***	***249.58***	***3.90E+16***	***40,581.25***	***723.08***	***9.29E+16***	***96,764.63***

site of Portofino MPA.

The fishing pressure (entity and spatial distribution) is here estimated as the number of fishing days in MPA. The average annual total pressure due to recreational fishing is given by 1,979 fishing days, with the greatest pressure exerted on sectors 14, 3 and 15 (Table I). Instead, professional fishing exerts an average annual pressure of 500 fishing days, mostly on sectors 17 and 18.

Fishing direct environmental impact corresponds to a subtraction of 972.66 kg/y, corresponding to 1.32E+17 sej/y and 137,345.87 em€/y: 249.58 kg/y by recreational fishing, corresponding to 3.90E+16 sej/y and 40,581.58 em€/y, and 723.08 kg/y by artisanal fishing, corresponding to 9.29E+16 sej/y and 96,764.63 em€/y.

Overall, the most caught species is *E. marginatus* with 47.98 % of biomass and 55.73 % of emergy, followed by *D. dentex* (12.91 % of biomass and 34.62 % of emergy), *Sparus aurata* (Linnaeus, 1758; 8.87 % of biomass and 2.74 % of emergy) and *D. sargus* (8.38 % of biomass and 1.65 % of emergy). Sectors 14 and 15 are the ones with the greatest harvest (more than 50 % overall), mainly due to *E. marginatus*.

Results are reported by species in Fig. 4 and by sector in Table I and Fig. 5.

Very strong sustainability approach

The overall annual harvest of fishing activities in the Portofino MPA is smaller than the production (3.95 % of the biomass and 9.59 % of the emergy value). The same result is obtained by analyzing each sector and each species in MPA. This would seem to demonstrate that there is no erosion of NC.

Analyzing, instead, the single species within each sector (very strong sustainability) it results that in sectors 4, 14, 15 and 18 there are species for which fishing harvest is greater than what that sector produces. For example, *D. dentex, E. marginatus, Ser-*

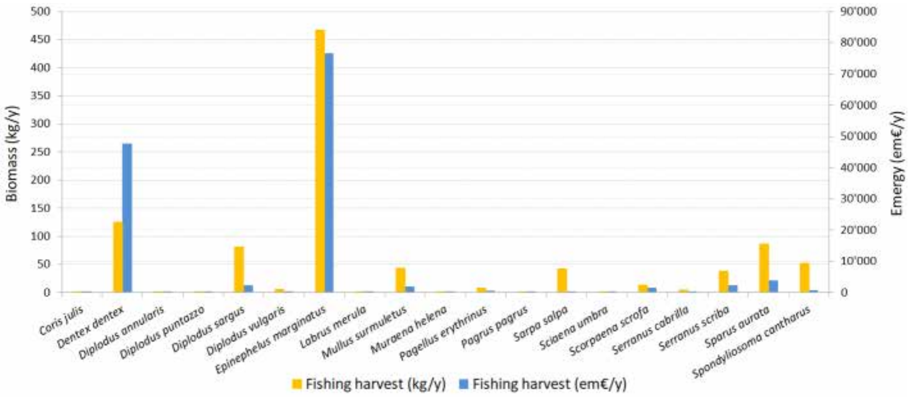

Fig. 4. – Annual direct environmental impact of recreational and professional fishing (only *species of visual census*) in Portofino MPA: biomass (kg/y) and emergy (em€/y).

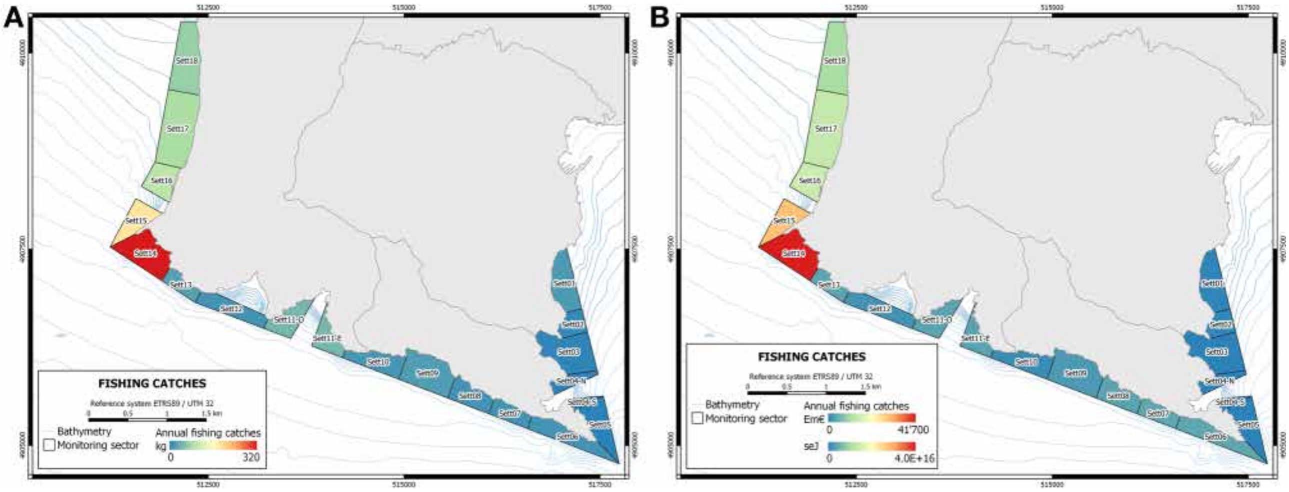

Fig. 5. – Distribution map of fishing catches in Portofino MPA at sector level. **A**: Biomass (kg/y); **B**: Emergy (sej/y) and monetary values (em€/y) (realized with QGIS software, version 3.10.1).

ranus scriba (Linnaeus, 1758), are overfished in sector 14. Basing on this latter budget, the SDSS produces the very strong sustainability map shown in Fig. 6.

DISCUSSION

Fishing is a worldwide activity impacting on limited stocks. Stocks can recover but it is necessary to avoid over-exploitation and permanent damages to ecosystems. MPAs have a fundamental role in the protection of the marine environment and, at the same time, in promoting the enhancement of a sustainable socioeconomic development of local community.

Here the SDSS proposed is a tool to assist MPA managing body in planning a sustainable management of fishing activities, starting from the assessment of the NC supporting the fish stocks and its production. The SDSS provides an environmentally focused accounting model in order to keep the NC at least intact in the framework of a strong or very strong sustainability.

The estimation of fish stock is based on information gathered from visual census campaigns, the most non-destructive widespread and used one. Nevertheless, this technique does not allow to detect the stock as a whole (*e.g.*, cryptic species, species not detected because they hide in the presence of operators, pelagic species; Brock 1982). In particular, many harvested species reported in logbooks are not detected by visual census. For this reason, for the sake of this analysis, only species identifiable by visual census are considered in the estimation of fishing harvest. This inevitably leads to an underestimate of the real impact generated by fishing on NC. With a view of further improvement, a sampling method that allows to evaluate the entire fish stock of the MPA or an integration of visual census with other techniques is therefore needed to get to a more consistent assessment of the sustainability of the MPA.

In order to fine-tune the system, the case of the Ligurian MPA of Portofino was examined, located in the context of a highly anthropic coast and lobbied by strong social pressures from local communities. The analysis is carried out at sector level in order to meet MPA managers needs and to better identify the areas where there is greater pressure and, in turn greater environmental costs. This punctual analysis allows to highlight any critical issues

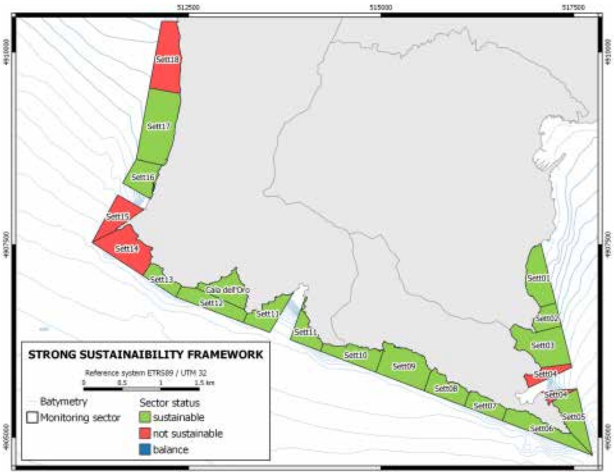

Fig. 6. – Very strong sustainability map for Portofino MPA fishing (realized with QGIS software, version 3.10.1).

not necessarily noticeable at overall level (MPA level) and is expected to better address protection and conservation strategies.

If a very strong sustainability approach is followed, according to each species must be preserved as it is, and the analysis is conducted at sector and species level, it appears that some species are being caught more than produced. Indeed, the overall budgets at MPA, sector or species level proved that, apparently, the NC is not eroded. This happens because, in the first two levels, the loss of one or more species is compensated by the surplus of other species or, in the third one, the loss of a specific species is compensated by other sectors in which it is less caught. For example, an overall analysis of sector 14 shows that it is in a good condition with a production surplus of almost 1 thousand of kilograms per year (4.18E+16 sej/y, 4.36E+04 em€/y). Analyzing instead single species in this sector, 3 species (*Dentex dentex*, *Epinephelus marginatus* and *Serranus scriba*) appear to be in a suffering condition with a production deficit.

According to Prato *et al.* (2016), the data reliability and availability are important for an accurate account. In this context, an in-deep analysis about the truthfulness of data declared by the fishermen in the logbooks, which often fill in hastily and incompletely, should be done. The consequence is a further underestimation of pressure exerted, subtraction and impact on NC. Measurements in MPA can be biased due to catches limitations (*e.g.*, species, maximum catch weight, minimum catch size). For this reason, even if fishermen caught more than they could, they wouldn't declare these catches in order not to incur in penalty provided by Portofino MPA regulation (art. 32). However, logbooks represent most of the data on fishing in MPAs. To deal with this underestimation, monitoring at sea is necessary in order to compare data declared with the actual catch.

Despite these, results confirm previous researches realized in the area, that highlighted how, although the Portofino MPA was able to recover fish biomass (Guidetti *et al.* 2008), the overlap of catches among artisanal and recreational fisheries, causes strong fishing losses on high trophic level predators (Prato *et al.* 2016). Moreover, according to Prato *et al.* (2016), results show that at the current exploitation level, the ecosystem is far away from its carrying capacity and fishing within the MPA borders should be reduced to pursue the MPA conservation objectives.

The SDSS is a tool that can be exported and applied to wider realities than the national and local context. In fact, the European Union, within the EU Biodiversity Strategy to 2020 (COM/2011/0244), called Member States to map and assess the state of ecosystems and their services while promoting the integration of these values into national accounting systems by 2020. Moreover, according to the strategy, the ecosystem services and its NC should be protected, valued and appropriately restored by 2050, also taking into account their essential contribution to human well-being and in order to avoid catastrophic changes. This highlights how much urgent is to define and apply methodologies able to assess NC and changes that our activities impose to it with the aim of its preservation or restoration (UN *et al.* 2014).

This is more and more important in those areas where a protection regime is established (such as in the case of MPAs) to assess the efficacy of undertaken conservation strategies (Vassallo *et al.* 2017). Fishing activities are an ecosystem service that must properly managed within MPA borders providing managers with operational decision-making tools which allow to make informed and aware decisions (Cortés *et al.* 2000, Poch *et al.* 2003, Pérez *et al.* 2005), mostly where the consensus process struggles to well-end.

The proposed SDSS hits this target since it plays an important role supporting the MPA in reducing the risks arising from the interaction of human societies with natural environments (Cortés *et al.* 2000). Indeed, SDSSs are widely used in environmental field (Latteman 2010, Stewart & Purucker 2011, Garrido-Baserba *et al.* 2015, Zhang *et al.* 2015) and in particular for protected areas management (*e.g.*, MARXAN, see Stewart *et al.* 2003). The SDSS allows, through the production of a map of very strong sustainability, to identify where fishing activities affect the NC. In such cases, it may be necessary for the managing body to review accessibility to individual sectors or to change the species that can be caught, giving the environment the opportunity to recover the depleted NC.

The iterability and implementation of the SDSS procedures in a computerized system allows 1) to quickly insert a large amount of data that is stored in a geodatabase, 2) to update the results fast and easily and 3) to produce different management scenarios, responding to manager need of having information in real time. At this purpose, the development of new technologies such as apps and responsive websites easily usable with a mobile phone can help more and more (Papenfuss *et al.* 2015, Venturelli *et al.* 2017, Joly *et al.* 2018). For this purpose, a responsive website linked to geodatabase is integrated in the SDSS in order to collect data on authorizations of fishing activities and logbooks and to overcome problems of excessive time and inefficiency related to handwriting. Moreover, filling logbook directly on the boat it is possible to register the precise harvest point thanks to GPS and to upload photos of the caught species.

An additional advantage of the SDSS is the ability to visualize spatial results within a WebGIS reserved for MPA managers. WebGIS are currently the most advanced and used systems for the visualization and diffusion of geographical information and represent a fundamental aid for activities in the field of environmental management. By accessing the WebGIS, the cartographic and/or alphanumeric data can be viewed, consulted and downloaded. This WebGIS fully responds to the problems of integration, dissemination and use of data, as it is an easy-to-use tool, which allows a quick update and easy access to data, without having to install any software on computer.

The SDSS potentially also would allow to spread the results to a large audience at reduced costs. At the moment results are accessible to MPA managers. Moreover, user-friendly interface for data sharing and information spreading (*e.g.*, diagrams and pictures) to fishermen and other MPA users, with respect for privacy, are under discussion and development, thanks to potentialities and applications of used software (Plone/Plomino and PostgreSQL/PostGIS).

A constant relationship between MPA and users and a better management of activities, *e.g.*, through a dedicated website, is expected to increase the level of loyalty with respect to the MPA by the users themselves. Fishing activities in MPAs, if properly managed, can constitute a sustainable activity to maintain the sociocultural and economic structure of the regions (UNIMAR 2001). Moreover, in recent years in some Italian MPAs fishermen seem to be able to change their relationship with sea and to be available to operate in harmony with the rules of environmental protection, even if it is a very slow and difficult process (Cattaneo-Vietti & Tunesi 2007).

Inadequate public and stakeholder involvement and communication/education about the MPA decision-making process undoubtedly lead to conflicts and disapproval by locals about the establishment of marine reserves, do not increase the perceived legitimacy of decisions, and lowers compliance with restrictions (Guidetti *et al.* 2008).

The SDSS is a step towards solving this problem and can be a useful tool for the citizen science. In the last decades, citizen science is increasingly used in biology, conservation and ecology. Concerning marine environment, citizen science projects rely on fishermen and target nearshore habitats (Changeux *et al.* 2020). The SDSS not only allow to collect mandatory data for carrying out the activities, but also further volunteer data (*e.g.*, in logbooks and monitoring reports there is the possibility to write down any information that may be useful for the MPA management).

A system for MPA managers to insert updated data for the calculation of the value of NC has also been developed, through the implementation of the methodology. Thanks to a specific interface on the website, managers will be able to enter new input data, such as fish fauna.

To make the SDSS even more efficient and supportive to the MPA managing body, a simulation system for developing real-time forecasting scenarios is under construction. It will allow to see the changes that would occur following management strategies alternatives to those currently in place. Managers, again through the website, will be able to perform simulations by changing the pressure exerted by users on the various sectors and see how the direct environmental cost and the reduction of NC associated would change.

From the management point of view, the availability of data, results and impact maps, together with the possibility to design different scenarios allows for the planning of multiple management interventions aimed at regulating human activities, such as those involving marine areas at risk and to implement appropriate policies for the conservation of biocenosis.

In conclusion, the described SDSS procedures allow to obtain a reliable result, both in numerical and spatial form opening up new potential perspectives. In particular, the maps generated by the SDSS allow both experts and managers to identify and characterize MPA areas at different levels of detail and to provide the results needed to operationalise a strong sustainable management.

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A STRATEGIC APPROACH TO ASSESS THE BUNDLE OF ECOSYSTEM SERVICES PROVIDED BY *POSIDONIA OCEANICA* MEADOWS IN THE BAY OF MARSEILLE

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STATE-AND-TRANSITION MODELS
TRIAGE METHOD
DELPHI PROCESS
ECOSYSTEM SERVICE ASSESSMENT
ECOSYSTEM BASED QUALITY INDEX

ABSTRACT. – The aim of the European program Life Integrated Project Environment (IPE) MarHa (2018-2025) is to restore and sustain the favorable conservation status of marine natural habitats in French Natura 2000 sites. In this context, Ecosystem Service Assessments (ESA) are carried out at various French sites including the Bay of Marseille (Provence, France). First, we applied the TRIAGE methodology: a strategic assessment of the issues with experts of the area (local MPA managers and scientists). TRIAGE raises two main concerns: (i) the intensification of recreational uses (by both residents and visitors), and (ii) the conservation of *Posidonia oceanica* (Linnaeus) Delile, 1813 seagrass meadows. In order to address both issues, we selected two adapted methodologies based on the strategic assessment: (i) a study oriented toward understanding the demand for ecosystem services (focused on recreational activities), and (ii) a study of the capacity of *P. oceanica* meadows to deliver ecosystem services using state-and-transition modeling. The objective of this work is to present the ESA process, from the strategic assessment to the analysis results. We focus on the study of the capacity of *P. oceanica* seagrass meadows to provide ecosystem services. State and transition models consist in defining alternative states of the habitat based on ecological indicators, identifying the bundle of services associated with each state and identifying transition vectors capable of explaining the shifts between each state. State-and-transition models can be very powerful frameworks for integrating multiple functions and services delivered by ecosystems while accounting for their temporal dynamics.

INTRODUCTION

Anthropogenic activities and climate change can deeply alter ocean productivity and food web dynamics, reduce the abundance of habitat-forming species, shift species distributions, and lead to a higher incidence of disease (Hoegh-Guldberg & Bruno 2010, Ceballos *et al.* 2015). The European Union (EU) established directives to designate strictly protected areas. An assessment of the conservation status of habitats is carried out periodically within the framework of the Habitat Directive (HD) on the conservation of natural habitats and of wild fauna and flora (HD, 92/43/EC). In 2012, the conservation status of most of the Natura 2000 marine habitats was assessed as "unfavorable" throughout the French Atlantic and Mediterranean biogeographical regions (Bensettiti & Puissauve 2015, Meinesz & Blanfuné 2015). Consequently, restoring and maintaining the favorable conservation status of marine habitats now appears as: (i) a moral duty, consisting in managing the common good for present and future generations, (ii) a legal duty, with regard to European and international biodiversity commitments, and (iii) a socio-economic duty, which takes into account and sustains the maintenance and restoration of Ecosystem Services (ES).

The ES concept seeks to account for the dependence of human societies on ecosystems and is commonly defined as the contributions of ecosystem structures and functions to human well being (MA 2005). The ES conceptual framework was initiated in the 1970s by the conservation biology movement (SCEP 1970) and is perfectly aligned with the biodiversity conservation paradigm. The publication of the Millennium Ecosystem Assessment report (MA 2005) consolidated this conceptual framework and represented the culmination of an institutionalization process of the concept by science, politics and law (Mongruel *et al.* 2016). This report opened the door to other initiatives at the international (*e.g.*, TEEB 2010), regional (*e.g.*, Maes *et al.* 2013) and national scales (*e.g.*, EFESE program in France[1]).

ES science is an interdisciplinary field, mainly resulting from the meeting of ecological and economic approaches. It is also operational and effective, since it supports and facilitates biodiversity management policies. The Ecosystem Service Assessment (ESA) is now a common method used in public environmental policies and is relatively well known by environmental stakeholders. Nevertheless, ESA is far from being a unified set of scientific practices. The epistemological foundations that support our economic approaches are those of strong sustainability[2]. The choice of this paradigm led us to reject systematic and large-scale monetary valuation. In this perspective we rely on a strategic approach to valuation. We consider that, since we cannot assess everything, the best strategy is to choose "what" and "how" it deserves to be assessed in terms of ES.

The objective of this paper is to present the process implemented to produce the ESA in the Bay of Marseille, with the objective of providing the managers of the Marine Protected Areas (MPA) of this sector with a useful and effective tool to identify appropriate levers to restore and maintain the favorable conservation status of natural marine habitats in the site they manage. This work was carried out at the scale of the Bay of Marseille, which includes two Natura 2000 sites designated under HD 92/43/EC. The strategic approach used for the ESA in this study allowed us to use different methods: the TRIAGE approach to ESA (Pendleton *et al.* 2015) combined with a Delphi process (Rowe & Wright 1999) and then a state-and-transition model (Lavorel *et al.* 2015).

MATERIALS AND METHODS

Case study: the Bay of Marseille: The Bay of Marseille (Provence, France) has been a good model to illustrate the complex interactions between ecological and socio-economic issues since antiquity. Located on the northwestern Mediterranean seafront, its coastline has been greatly impacted by anthropic activities, mostly associated with urbanization[3], the harbor[4] and both freight and passenger traffic. Despite this accumulation of pressures, the bay shelters several marine habitats listed in HD, notably *Posidonia oceanica* (Linnaeus) Delile, 1813 seagrass meadows, coralligenous reefs, and sea caves. The Bay of Marseille is bounded by two MPAs which are Natura 2000 Special Areas of Conservation[5] (SAC): the Côte-Bleue Marine Park (CBMP) and the Calanques National Park (CNP). In these two MPAs, the *Posidonia* meadows cover a surface area of 1,198 ha in CBMP and 1,186 ha in CNP.

In the northwest, the Côte-Bleue Marine Park (CBMP) is a public structure created in 1983, resulting from the merging of five coastal municipalities. It extends over an area of 98.7 km². It includes notably two no-take-zones (2.95 km² in total surface area) established at the initiative of local fishermen and their representatives[6]. The CBMP is managing the Natura 2000 SAC 'Côte Bleue Marine' (FR9301999) since 2009.

In the southeast, the Calanques National Park (CNP) was created in 2012 and includes the Natura 2000 SAC "Calanques et îles marseillaises – Cap Canaille et massif du Grand Caunet" (FR9301602). The Park covers both terrestrial and marine areas, and includes several peri-urban sectors. At sea, a marine surface area of about 435 km² makes up the "core" area, in which the objective of protection is stricter and human activities are regulated in order to ensure the efficient conservation of fauna, flora, the natural environment and landscape. The sea core extends 10 nautical miles from the coast and includes seven No-Take zones, accounting for almost 11% of its surface area (46.5 km²). The "adjacent marine area", which is the part of the park's territory where activities are not subject to specific regulations (although they must conform to a reference in terms of sustainable development), extends over a marine surface area of 977 km².

In the centre of the Bay of Marseille, in front of the Prado beaches, a 2-km² reserve managed by the municipality of Marseille has been equipped with more than 400 artificial reefs in 2007-2008 (Cresson *et al.* 2019). All activities are banned inside the reserve, with the exception of scientific experiments and monitoring with the aim of supporting artisanal fisheries and sustaining and improving the ecological quality of adjacent natural habitats (*via* fish biomass exportation). With a gross volume of 27,300 m^3, the Prado reef is the biggest artificial reef in Europe and the Mediterranean Sea (Charbonnel *et al.* 2011).

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[1] https://www.ecologique-solidaire.gouv.fr/levaluation-francaise-des-ecosystemes-et-des-services-ecosystemiques

2 In economic analysis the key question regarding sustainable development is whether natural and human capitals can be substituted by manufactured capital (weak sustainability), or whether each should be maintained (strong sustainability). Under the strong sustainability paradigm, economic analysis goes beyond optimizing the substitution of capitals to examine the means of achieving their conservation (Dietz & Neumayer 2007).

[3] The collectivity of Aix-Marseille-Provence has a population of more than 1.8 million.

[4] The harbor of Fos-Marseille is the most important commercial harbor in France with traffic amounting to 78 million tons of cargo and 3,276,902 passengers in 2014 (Bas & Kalaydjian 2018)

[5] Special Areas of Conservation is a designation for natural sites representing a strong interest regarding the restoration or maintenance of a favorable state of conservation of the habitats or species of the HD.

[6] The regional committee of fisheries and marine cultures of Provence-Alpes-Côte d'Azur (CRPMEM PACA) and the '*Prud'homie des Pêcheurs*'.

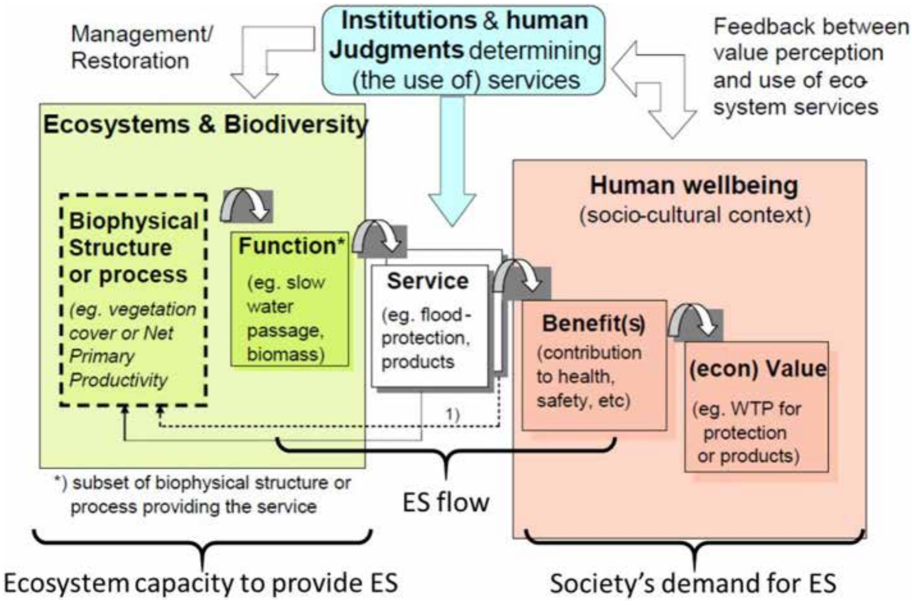

Fig. 1. – The Ecosystem Service "cascade" (Source: TEEB (2010) adapted from Haines-Young & Potschin (2010)).

Ecosystem services assessment: ES are defined as the benefits society acquires from the functioning of ecosystems. They allow establishing a link between the functioning of ecosystems and their real benefits to society (Fig. 1). We can highlight three dimensions to assess ecosystems (Villamagna *et al.* 2013, Burkhard *et al.* 2014): (i) the capacity of an ecosystem to provide an ES or its potential defined as the hypothetical maximum production of a given ecosystem; (ii) the flow of ES is the bundle of ES and other outputs effectively selected at a given place and period; and (iii) the demand for ES is defined as the quantity of ES required or desired by society. The ESA method and ES indicators differ according to the dimension we want to focus on. In this study, this choice is based on a strategic analysis of the management issue conducted with a TRIAGE process.

The triage method: The TRIAGE method guides the ESA by identifying and specifying the objective, scale, methods and tools required for its implementation. It is a preliminary step to the ESA and allows perceiving the latter in strategic (the scope is circumscribed as it is not realistic to assess everything) and operational (according to the availability of knowledge and means, and consistent with the site management framework) ways. The TRIAGE method is composed of a three-stage sequence. The first step defines the scope of the ESA: (i) identify the purpose of the ESA: is it to provide information on the ES (informative use), to contribute to the implementation of a measure (technical use) or to help the manager to carry out trade-offs (decisive use)? (ii) identify the management issues that will most influence the ES in order to situate the ESA in local reflections related to the issues at stake; (iii) identify the components of the socio-ecosystem (ecological compartments, functions, ES, actors) that will be the most influenced by these management issues. The second step selects the key ES based on their prioritization regarding three criteria: (i) the importance of the different ES; (ii) the exposure of ES to drivers of change; (iii) the possibility for local managers to act on the ES (based on a ranking between 1 and 5). The more an ES is considered important, the more it is exposed to changes and it is possible to act on it, then the more it will be useful to evaluate the ES (Pendleton *et al.* 2015). The third step consists in: (i) choosing the type of indicators (biophysical, economic and/or based on social perceptions) that are most usable by the MPA manager; (ii) choosing the methods and tools that can be used to inform these indicators; (iii*)* assessing the resources required and the availability of data for the ESA.

The exchanges with MPA managers began in January 2019, while the TRIAGE process was implemented in May 2019 during a workshop bringing together 11 local MPA managers and scientists, experts of the Bay of Marseille.

The Delphi process: the Delphi process is an approach that reveals and refines the judgment of a group and whose core principle is the fact that the judgment of a group is more relevant when uncertainty is high (Kaynak & McCauley 1984). As a result, it is perfectly coherent with the TRIAGE method, which aims to prioritize assessment issues based on stakeholders' expertise. We used a limited number of iterations to maximize response rates, considering the number of experts interviewed.

The Delphi process was applied in the TRIAGE method. The local MPA managers and scientists, experts of the Bay of Marseille, were contacted by email following their participation in the workshop in May 2020. We sent the first report of the workshop allowing them to return to the concepts and questions related to ES assessment, to step back from the first results of the ES prioritization process and to allow them to review their judgments and therefore modify the ES prioritization (step 2 of the TRIAGE). The results of the TRIAGE are thus the fruit of an individual and collective reflection that leads to consensus.

State-and-transition models: State-and-transition models provide an operational and conceptual framework for organizing and providing information about ecosystem dynamics and management outcomes, describing how communities respond to pressures and management (Briske *et al.* 2005, Bestelmeyer 2015). They were developed by Westoby *et al.* (1989) for rangeland ecological sites in southern Arizona (United States of America). While their scientific application is widespread for certain terrestrial habitats (*e.g.*, McIntyre & Lavorel 2001, Quétier *et al.* 2007, Tarrasón *et al.* 2016), they have very rarely been applied to marine environments.

In this study, the evolution of *P. oceanica* seagrass meadows was described at the scale of the French Mediterranean with respect to different pressures. The first step has consisted in identifying and describing the different ecological states of the *P. oceanica* seagrass meadows in the French Mediterranean and then describing the transition drivers (*e.g.*, natural and anthropogenic pressures) that make *P. oceanica* seagrass meadows switch from one state to another. This exercise was initiated during workshops in May and December 2019, supplemented by bilateral meetings with several local MPA managers and scientific experts of *P. oceanica* meadows.

RESULTS

A triage of management issues and key ES for ESA

The first step of TRIAGE is the definition of the scope of the ESA. We reviewed 28 social-ecological issues that were hierarchized consensually by the participants to the workshop. We identified two primary management issues at stake: (i) the intensification of recreational activities and (ii) the conservation of *P. oceanica* meadows. We also identified four secondary issues: fishing (both recreational and professional), tourism, chemical pollution and governance. We then conducted a preliminary definition of the socio-ecosystem that led to identifying a list of 24 relevant ES in connection with these issues. We observed a small decrease in ranking after the second round of the Delphi process (mean decrease of 0.35, max 0.78 and mean 0.02). Table I presents the final results of the ES hierarchization process. The final scores are relatively homogeneous, the main ES is the provision of fish and to a lesser extent the maintenance ES of shelter, the provision ES of sea urchins and octopuses and the recreational ES of observation, education and recreational fishing. However, we can see bigger differences in the ranking of each criterion. Then, we conducted a Principal Component Analysis to help visualizing the different groups of ES (Fig. 2). The two first axes explain 92.25 % of the variance of data. First, on the positive part of axis 1 that explains 56.26 % of the variance of data, we find the services more at stake regarding the global score. Second, on the positive part of axis 2 are positioned the important ES for which a low possibil-

Table I. – Result of the hierarchization of the ES provided by the ecosystems of the bay of Marseille.

Ecosystem services		A : Importance	B : Exposition	C : Possibility of action	SCORE (A+B+C)/3
Maintenance and support services	Trophic network	4.33	3.78	2.22	3.44
	Nursery	4.00	3.67	2.78	3.48
	Shelter	3.67	3.89	3.11	3.56
	Reproduction	3.67	3.56	3.00	3.41
Provision services	Urchins	3.11	4.11	3.78	3.67
	Octopuses	3.00	4.22	4.00	3.74
	Fishes	4.44	4.33	3.67	4.15
	Crustaceans	2.13	3.88	3.50	3.17
	Shellfish (mussels and limpets)	2.33	3.44	3.33	3.04
Regulation services	Climate regulation	3.56	3.78	1.89	3.07
	Purification capacity	3.50	2.63	2.00	2.71
	Coastal protection	3.56	3.78	3.00	3.44
Cultural services	Marine landscape	4.33	3.22	2.56	3.37
	Sub-marine landscape	4.22	3.89	2.33	3.48
	Acoustic landscape (feeling of well-being)	2.78	3.00	2.78	2.85
	Wrecks (abiotic)	3.22	2.00	2.33	2.52
	Observation of flagship species	4.22	4.33	3.00	3.85
	Education	3.44	3.50	3.89	3.61
	Research	3.00	2.63	2.89	2.84
	Recreational fishing	3.89	3.78	3.67	3.78
	Arts (*e.g.*, photography)	2.56	2.38	2.25	2.39
Heritage dimensions	Historical dimension	3.11	2.13	2.50	2.58
	Natural Capital (common good)	3.67	3.33	2.67	3.22
	Cultural anchoring (*e.g.*, toponymy)	2.67	2.13	2.00	2.26

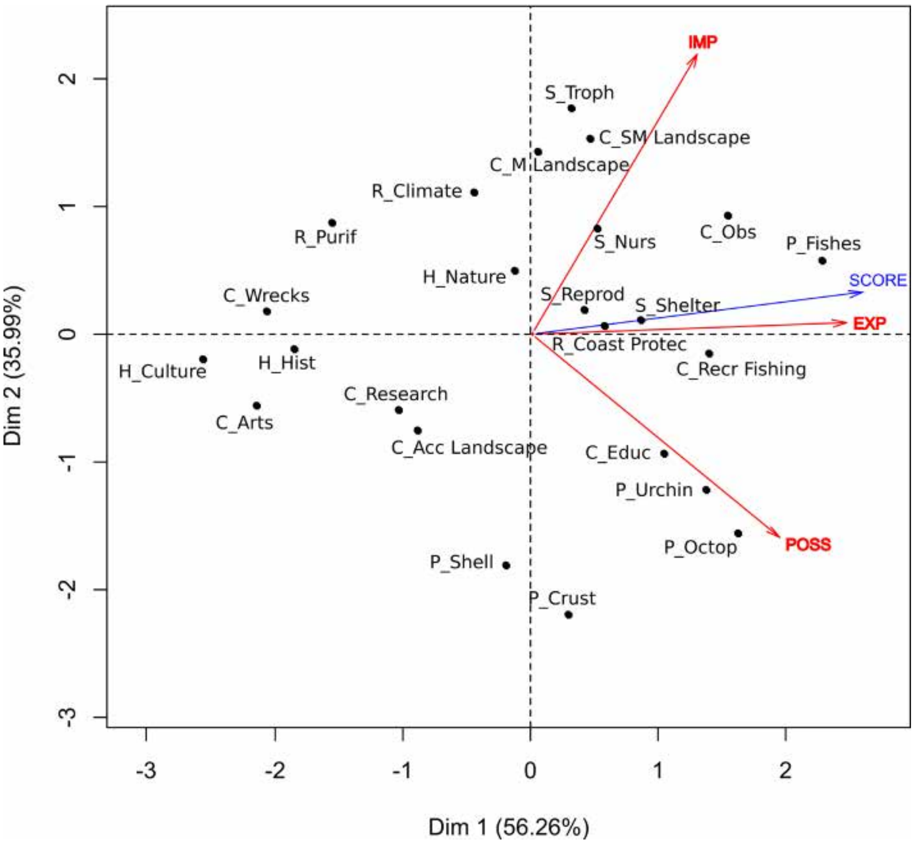

Fig. 2. – Result of a PCA conducted on the ranking of the three criteria of Importance (IMP), Possibility of action (POSS) and Exposure (EXP) as explicative variables and the global score (SCORE) as illustrative variable.

ity of action exists (*e.g.*, landscapes and seascapes, nursery role, etc.) in the top right square (1st category) and on the negative part of axis 2, the ES that are less important but for which a high possibility of action exists (*e.g.*, education, urchins, octopuses, etc.) (2nd category).

This ES ranking can be linked to the two main issues revealed by the TRIAGE. The question of the conservation of *P. oceanica* is more associated with the services of the 1st category (notably support services and coastal protection) while the question of the massification of recreational activities is more transversal as cultural services are present in the two categories. In terms of management purposes, a low possibility of action implies that the ESA should be focused on an informative use. In this perspective the ESA is built to widen the scope of possibilities for future action. On the other hand, when the possibility of action is high, the ESA can be useful to support decision-making in the arbitration between different management measures or in their design. The TRIAGE method led us to choose two different ESA methods. First, an assessment focused on the ecological importance of the *P. oceanica* ecosystem in the provision of ES based on the state-and-transition model. This method is adapted to the question regarding the conservation of *P. oceanica*; it will highlight its importance in the provision of ecosystem services (notably support services). In addition, it presents a good opportunity to integrate existing knowledge on ecology in the operational framework of the ESA. Second, an assessment of the demand for ES associated with the evolution of recreational and touristic practices. This method will allow us to investigate the question of the intensification of recreational activities with retrospective and prospective objectives. The current work is focused on the first analysis.

A state and transition model for **P. oceanica** *meadows*

The first result of the workshop and bilateral interviews we conducted with experts on *P. oceanica* was to adjust the list of ES from the TRIAGE (relevant at the scale of the entire Bay of Marseille in all its dimension, *i.e.*, diversity of habitats, uses and management objectives) to a list that is more adapted to deal with *P. oceanica* in the French Mediterranean. We identified a list of 18 ES[7].

Fig. 3 presents the different states (S) of the *P. oceanica* seagrass meadows identified in the model and the dif-

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[7] During the TRIAGE, 24 ES were identified for the Bay of Marseille (Table I), with the focus placed on the *P. oceanica* ecosystem. We refined this list to 18 ES: 6 support ES (primary production, secondary production, nursery, shelter, reproduction, biomass exportation); 3 provision ES (sea urchins, fish and cephalopods, crustaceans); 4 regulation ES (climate regulation, contaminant sequestration, coastal protection, production and sequestration of sediment), and 5 cultural ES (landscape and emblematic species, preservation of archeological resources, education, research, recreational fishing).

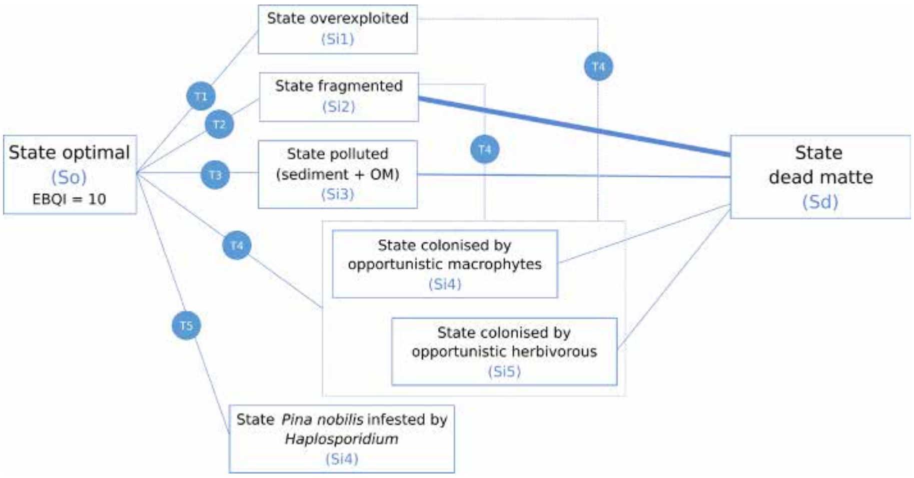

Fig. 3. – Ecological states of the *P. oceanica* meadows and transition factors.

ferent transition factors (T) capable of explaining the shift from one state to another. The optimal state (So) represents the reference state of the *P. oceanica* seagrass meadows. In this state, the functioning of the habitat is optimal. From this state, *P. oceanica* seagrass meadows can reach different intermediary states (Si1 through Si6) that are described as transition factors (T1 through T5) regarding the pressures on the ecosystem. The decline of the *P. oceanica* meadow is generally indicated by the appearance of areas of ‘dead matte’. The matte is the structure composed of live and dead parts of rhizomes and roots, together with the sediment, which fills the interstices. ‘Dead matte’ areas are areas where *P. oceanica* dies, leaving the matte uncovered by shoots of living leaves (Mateo *et al.* 1997, Boudouresque *et al.* 2016). The matte (living matte beneath the shoots of leaves and dead matte) may persist for decades and even centuries. The dead-matte patches are occupied by macroalgae that do not at all play the same ecological role as *P. oceanica*, which is an ecosystem engineer.

First, in the overexploited state (Si1), fish stocks are depleted (especially predatory and piscivorous teleosts). This depletion can lead to the proliferation of the herbivorous species such as sea urchins *Paracentrotus lividus* (Lamarck, 1816) and *Arbacia lixula* (Linnaeus, 1758) and salema porgy *Sarpa salpa* (Linnaeus, 1758) and to overgrazing of the seagrass meadow (Ferrari *et al.* 2008, Boudouresque *et al.* 2017). *P. oceanica* can be dragged into this state from the good state due to overfishing (T1). In the French Mediterranean Sea, 80 % of fishing activities are performed in coastal ecosystems (with one third in coastal lagoons), with a wide range of trades (Mongruel *et al.* 2019). At the scale of *P. oceanica* seagrass meadows of the Côte-Bleue Marine Park, artisanal small-scale fishing (*i.e.*, boats smaller than 12 m length) mainly target gilthead seabream *Sparus aurata* Linnaeus, 1758, European seabass (*Dicentrarchus labrax* (Linnaeus, 1758), red mullet *Mullus surmuletus* Linnaeus, 1758 and scorpion fish *Scorpaena* spp. (Leleu *et al.* 2014) while coastal trawling (*i.e.*, boats between 12 m and 19 m length) also catch non targeted species (Boudouresque *et al.* 2017). Current knowledge does not allow us to define a maximum sustainable yield (MSY) beyond which we would shift from sustainable exploitation to over-exploitation.

The second intermediate state is a fragmented state (Si2) in which the surface area and connectivity of *P. oceanica* seagrass meadows is interrupted by patches or extensive areas of dead matte. The corresponding transition factor (T2) is the qualitative or quantitative degradation of habitats due to fishing gears (active or lost), boats anchoring and extreme events as well as to the natural dynamics of the meadow (Boudouresque *et al.* 2009). First, *P. oceanica* is particularly vulnerable to trawling (Boudouresque *et al.* 2009). A standard trawler can uproot between 99,000 and 363,000 plants per hour (Martin *et al.* 1997). In order to limit this impact, trawling within the 3 nautical miles (5.6 km) zone is forbidden. However, this regulation is often not observed. Trawling is responsible for the loss of 12 % of the surface area of *P. oceanica* seagrass meadows of Corsica (Pasqualini *et al*. 2000). The time needed for *P. oceanica* to recover from trawling is estimated at 100 years (González-Correa *et al.* 2005), but depends on the surface area destroyed. The second is the impact of anchoring. Major threats come from large ships, notably cruise ships. For example, close to Porquerolles Island, anchoring is responsible for scars between 1 and 2 meters wide and up to 296 meters length that has generated 4.2 hectares of dead matte (Ganteaume *et al.* 2005, Montefalcone *et al.* 2006, Boudouresque *et al.* 2009). To a lesser extent, the anchoring of small pleasure boats can also lead to uprooting *P. oceanica* (*e.g.*, 68,000 shoots per hectare in a 1.4 ha zone in Corsica). However, impacted seagrass meadows can still produce new leaves in the next year if some shoots stay alive. Sustainable pressure from small boat anchoring is theoretically estimated to 2 moorings per hectare per day (as a yearly mean) and should not exceed 10 moorings per hectare per day (Boudouresque *et al.* 2012). Finally, on coasts exposed to extreme events such as strong winds and storm surges, sediment movements can bury sprouts, expose roots and rhizomes and even uproot entire plants (Gera *et al.* 2014). Infantes *et al* (2011) estimated that *P. oceanica* sprouts need to have more than half of their roots anchored in the sediment.

The third intermediate state is the polluted state (Si3): *P. oceanica* seagrass meadow is disturbed by terrigenous inputs, hypersedimentation, eutrophication and turbidity. We identified two drivers that can induce the transition factor associated with this state (T3): pollution from urban and industrial activities or from aquaculture (Boudouresque *et al.* 2009, 2012, 2020). A loss of *P. oceanica* meadows is observed close to Mediterranean urban centers and sewage outlets, notably in the Bay of Marseille (Boudouresque *et al.* 2009). Increased eutrophication, hypersedimentation and turbidity explain a large part of this loss. This pollution has been decreasing since the 1970s thanks to the improvement of wastewater treatment (Jackson *et al.* 2006). As a result, a *P. oceanica* seagrass meadow in recovery has been identified in the Bay of Marseille (Pergent-Martini *et al.* 1995); however, the recovery process is very slow (a few centimeters per year when good environmental conditions are restored). In the French Mediterranean Sea as elsewhere, aquaculture also generates pollution associated with food leftovers and fish feces that accumulate under cages (Boudouresque *et al*. 2020). In addition, shade generated by cages and turbidity leads to a decrease in the light intensity necessary for *P. oceanica* development (Boudouresque *et al.* 2009, 2020).

In the fourth and fifth intermediate state (Si4 and Si5), the transition factor is the presence of non-indigenous invasive species (NIS) (T4) that compete with *P. oceanica* and other species of the ecosystem. The impacts of invasive species can be different depending on whether NIS competes for space or if they are predatory/herbivorous species (Giakoumi *et al.* 2015). In this way, we distinguished two intermediary states: a state colonized by NIS macrophytes (Si4) and a state colonized by the herbivorous rabbitfishes *Siganus luridus* (Rüppell, 1829) and *S. rivulatus* (Forsskål, 1775) (Si5). Macrophytes (Si4) such as *Caulerpa* species (*Caulerpa taxifolia* (Vahl) C. Agardh and *C. cylindracea* Sonder)[8] compete with *P. oceanica*. When meadows are vulnerable, this competition can lead to (i) chlorosis, (ii) necrosis, (iii) decrease in the number, length, thickness and longevity of leaves, and (iv) the death of sprouts (Klein & Verlaque 2008). Colonization by macrophytes is more difficult within a dense and healthy meadow. *Siganus* spp. (Si5) has not yet settled on the French Mediterranean coasts, however it is now become very common in the eastern Mediterranean where it strongly interacts with native herbivorous fish species through competition for food resources and habitat (Bariche *et al.* 2004, Boudouresque *et al.* 2017). It also modifies marine vegetation when it is abundant. One individual of *S luridus* was fished for the first time in the Côte Bleue Marine Park in 2009 (Daniel *et al.* 2009). A second species of rabbit fish (*S. rivulatus*) was also caught by artisanal fishers close to CBMP in 2018 (Iglésias *et al.* 2020). These fish species constitute a potential threat in French seas.

In the last and sixth intermediate state (Si6), the meadows are infected by *Haplosporidium pinnae* sp. nov. (Si6), a pathogenic protozoan of the noble pen shell (*Pinna nobilis* Linnaeus, 1758), inducing mass mortality of this species with no impact on the other compartments of the ecosystem (Catanese *et al*. 2018). *Pinna nobilis* is a large bivalve endemic to the Mediterranean, threatened with extinction and providing important ecosystem services linked to its high heritage value.

Finally, when pressures are too high, the *P. oceanica* seagrass meadow shifts into the dead matte state (Sd) (Montefalcone *et al.* 2007). The dead matte is the ultimate degraded facies of the *P. oceanica seagrass* meadow on which can develop certain macroalgae and invertebrates although it has lost most of its ecological functions. The substrate is composed of an entanglement of roots and rhizomes clogged by sediments of various grain sizes, that are particularly compact and favorable to the establishment of a relatively specialized fauna.

DISCUSSION

In view to producing an ESA, the TRIAGE method is a useful process for increasing stakeholders' involvement, allowing their appropriation of the concepts and of the results obtained which in turn increase the chance of identifying scientific knowledge that fit into their management policies. This is important as there is a risk of discrepancy between the production of 'more knowledge' and its actual utilization by decision makers (Jordan & Russel 2014). Moreover, a strategic analysis prior to ESA allows us to be sure that knowledge is produced, integrated and aligned with the needs of the users and their perception (Honey-Rosés & Pendleton 2013). However, TRIAGE implies a shift in the construction of projects regarding ESA as it makes work planning difficult. Finally, ESA is the outcome of a trade-off between the management needs and the capacity (in terms of skills and means) of the assessors.

In the state-and-transition model, we have not selected climate change as a transition factor that can lead to

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[8] Five species of macrophytes have been identified as competitors of *P. oceanica* (Boudouresque *et al*. 2009): two species of Australian chlorophytes (*Caulerpa taxifolia* et *Caulerpa racemosa* var. *cylindracea*) and three species of Indo-Pacific rhodophytes (*Acrothamnion preissii*, *Lophocladia lallemandii* and *Womersleyella lallemandii*). We will focus on the two caulerpas as they have strongly impacted French Mediterranean coasts (Verlaque & Fritayre 1994, Piazzi *et al.* 2003).

a well-identified intermediate state. Indeed, most of the consequences of climate change on coastal and marine ecosystems are still uncertain. Moreover, its effects are multiple and associated with different factors (marine water warming, acidification, sea level rise, increase in the frequency and intensity of extreme events, etc.) (Pergent *et al.* 2014). For example, different hypotheses can be made regarding the evolution of *P. oceanica* due to water heating. First, other Mediterranean seagrasses are good candidates in the succession, notably *Cymodocea nodosa* (Ucria) Asch and, to a lesser extent, *Zostera noltei* Hornemann (Montefalcone *et al.* 2007, Boudouresque *et al.* 2012). Second, *P. oceanica* could also be threatened by NIS such as *Halophila stipulacea* (Forsk.) Asch (Pergent *et al*. 2014), generally less structuring species that can trigger deep changes in associated communities and ecological functioning. Thirdly, *P. oceanica* could also adapt by modifying its thermic optimum, a phenomenon already observed for terrestrial plants (Koch *et al*. 2013). These hypotheses highlight the uncertainty associated with the impact of climate change on *P. oceanica* ecosystems and the concomitant difficulty regarding the definition of intermediate state. In this study, climate change is considered as an aggravating factor as it can increase the intensity of transition factors: (i) increasing the occurrence of extreme events (Lejeusne *et al.* 2010) and their associated fragmentation impacts (Gera *et al.* 2014); (ii) increasing exposure to NIS that are more resilient to changes in environmental conditions whether they are thermophilic NIS such as rabbit fish *Siganus* spp. or non-calcareous NIS such as *Caulerpa* spp. that are more resilient to water acidification (Hall-Spencer *et al.* 2008). But on the other hand, the rise in temperature can benefit the plant, promoting flowering and sexual reproduction, as was observed everywhere in autumn 2018.

In order to establish a link between states of *P. oceanica* and the potential bundle of ES it provides, we propose to move the research agenda forward by relying on the analysis conducted and by calculating the EBQI (Ecosystem-based Quality Index, Personnic *et al*. 2014). EBQI is an index based on a set of representative functional compartments, the weighting of these compartments and the assessment of the ecological status of each compartment by comparison to a supposed ‘good’ baseline. We propose to adopt a similar approach to determine the potential bundle of ES by assessing the contribution of each compartment to the provision of each ES. Using this approach, we need to follow several of the steps summarized in Table II. For step 1, we’ll ask experts to determine the potential ES bundle. Steps 1 and 2 are based on expert judgments gathered during workshops and bilateral interviews, for step 1 we’ll ask experts of Mediterranean habitats to assess the bundle of ES provided by *P. oceanica* meadows. Step 3 is based on the identification of existing meadows illustrative of each state for which a measure of EBQI is available. This work is still ongoing.

Thus, in the ESA process, we propose to base the assessment of the *P. oceanica* ecosystem’s capacity to provide ES by establishing a linear relation with EBQI. Such a relation is useful as EBQI is used in ecological monitoring to assess the conservation status of seagrass meadows in the framework of several European Directives. EBQI allows the accumulation of large amounts of data that will allow comparison between *P. oceanica* seagrass meadows. In addition, EBQI is a well-known indicator that reinforces the appropriation of the ESA by decision makers. However, this may raise several problems. First, EBQI may not be a good indicator of the ES. For example, regarding the climate regulation service, a better indicator could be the mean thickness of the matte as it is where carbon is stored (up to 1,500 t/ha/y, Pergent *et al.* 1997, 2014). Secondly, the relation between ES and the quality of the ecosystem may be not linear. For example, we can discuss the question of the provision of sea urchins. According to EBQI, sea urchins (*P. lividus* and *A. lixula*) are part of the functional compartment called “herbivorous”, based on the density of sea urchins. The highest EBQI rank for this compartment is 4 when the density of urchins is between 1 and 5 individuals per m², EBQI is decreased to 2 and to 0 if the number of urchins exceeds 5 and 10, respectively, as it shows a dysfunction of the ecosystem. However, in the perspective of the ES corresponding to the provision of sea urchins, a higher score should be given to a higher density of urchins. Assuming that ES and EBQI follow the same dynamic tends towards the demand assessment as we are presuming that we should give higher ranking to ES that are provided sustainably regarding the global state of the *P. oceanica* ecosystem. Thirdly, the weight attributed

Table II. – Summary of the step associated to the assessment of the ecosystem services bundle of *Posidonia oceanica* ecosystem.

Step	Method
1 – Determination of the potential bundle of ES associated to the optimal state of the ecosystem	Expert judgment
2 – Identification of the compartments of *P. oceanica* ecosystem involved in the provision of each service	Expert judgment with bibliographic support
3 – Establishment of a direct link between EBQI and the ES bundle	Expert judgment with bibliographic and field data support
4 – Determination of the EBQI of *P. oceanica* ecosystem in each state	Expert judgment and field data
5 – Calculation of the ES bundle of *P. oceanica* in each state	Calculation

to each compartment to take into account its importance in the good functioning of the *P. oceanica* ecosystem may be different when it comes to assessing the provision of ES. In order to overcome these issues, we have initiated a discussion with experts in order to identify the best compromise.

Finally, linking the state-and-transition model to EBQI data from specific sites allows changing the focus we made, *i.e.*, to move from a general model (at the French Mediterranean scale) towards a smaller scale, consistent with an existing management framework (*i.e.*, the Bay of Marseille).

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TOWARDS STRONG SUSTAINABILITY: A FRAMEWORK FOR ECONOMIC AND ECOLOGICAL MANAGEMENT OF MARINE PROTECTED AREAS

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NATURAL CAPITAL
BENEFITS
COST BENEFIT ANALYSIS
EMERGY ANALYSIS
SYSTEM ANALYSIS
BIOPHYSICAL METHODS
ECOSYSTEM SERVICES MAPS

ABSTRACT. – In this study, an operational tool for environmental accounting is proposed to assess natural capital value and to obtain a budget encompassing ecosystem services, putting into practice the notion of "ecosystem services cascade". The accounting system is based on a previous framework (EAMPA project) developed for MPAs. The tool implements some additional features bringing significant improvement of the framework in order to achieve sustainability. At this purpose two parallel accounts, ecological and economic, are developed and then compared aiming at achieving a net benefit from both domains. The economic approach considers, together to financial revenues and costs, all the direct and indirect benefits from MPA's ecosystem services fruition. The ecological approach is formulated so that results are obtained under a strong sustainability perspective and direct impacts on MPA natural capital, associated with ecosystem services fruition by MPA's customers, are taken into account and included into the budget.

INTRODUCTION

Haines-Young & Potschin proposed in 2011 the "ecosystem services cascade", a framework that highlights the feedback from the socio-economic system on ecosystems. This feedback is due to impacts generated not only by ecosystem services (ES) fruition but also by management strategies (Fig. 1). The cascade can be interpreted as the pathway from ecosystem structure and processes to human well-being, a framework where the ES are the link between ecosystems and economics. From biophysical components, that make up the natural capital (NC), all the functions potentially useful for mankind are originated. Ecosystem functions represent the potential to generate ES from NC stock and they exist independently from humans' behaviour (TEEB 2010). When humans find some utilities in a function, this function enters the ES domain. Benefits to humans are originated from ES fruition.

Based on these theoretical foundations, in 2013 the Italian Ministry of the Environment and Protection of Land and Sea launched the Environmental Accounting in Marine Protected Areas (EAMPA) project. EAMPA is a 4-years research programme aimed at implementing an environmental accounting system across Italian Marine Protected Areas (MPAs).

The main goal of EAMPA was the calculation of a budget taking into account the ecological and economic value of the MPAs, with particular reference to ES generated in each protected area (Franzese et al. 2015) and the aggregated net benefit returned to the economy.

The program aimed at the achievement of a standardized assessment of NC as well as environmental costs and benefits in all Italian protected areas by means of two parallel pathways and six operational phases (Fig. 2, Table I). The detailed methodology and steps of the project are described in Franzese *et al.* (2015) and Vassallo *et al.* (2017).

Following EAMPA, in the context of the EU Interreg project *Integrated management of ecological networks through parks and marine areas* (GIREPAM), an upgraded version of the framework has been realized and it is here illustrated. The updated framework aims: 1) to obtain two different budgets (with the corresponding net benefit), the economic and the ecological accounts; 2) to highlights the interactions between the ecological and the human domains, according to the ES cascade theory.

The ES cascade clearly shows how human economies are constrained by the availability of natural stocks.

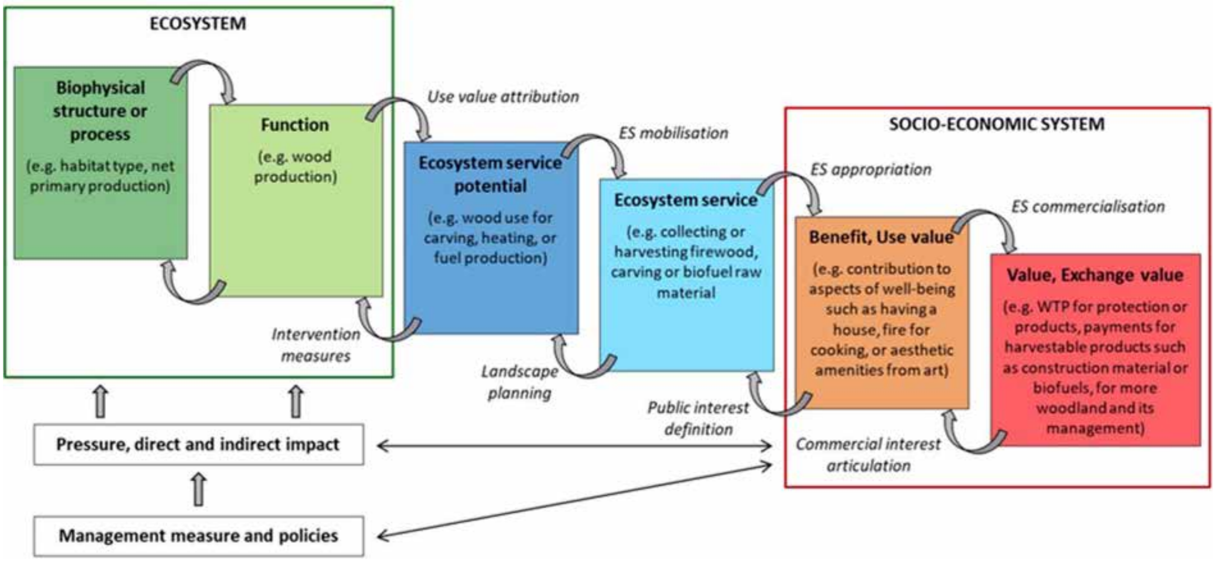

Fig. 1. – A schematic representation of the ecosystem services cascade (Spangenberg *et al*. 2014).

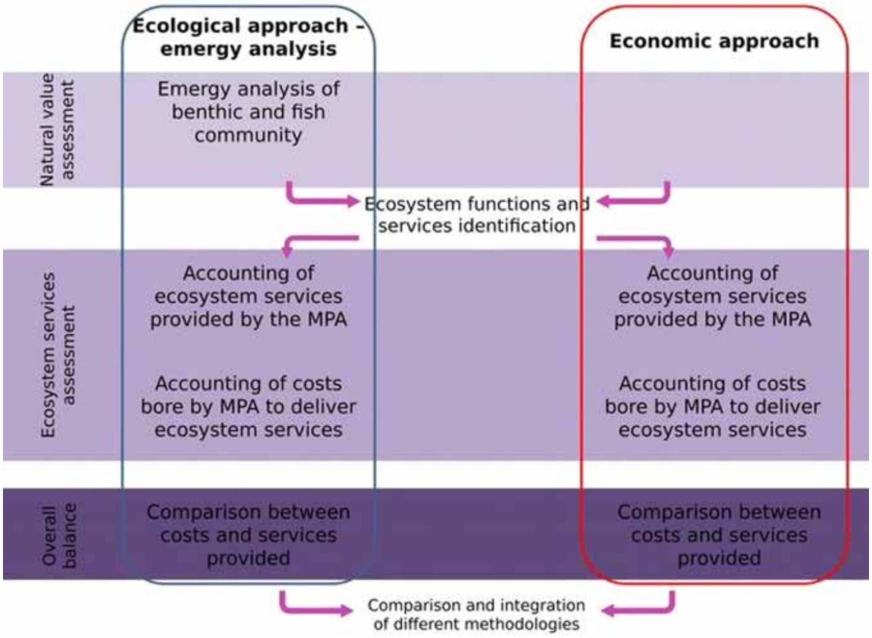

Fig. 2. – Flowchart of the EAMPA project: environmental and economic research pathways.

Table I. – Main steps of the EAMPA project and calculation methods employed for the fulfilment of the different project phase.

Activity description	Ecological pathway calculation method	Economic pathway calculation method
Data gathering: analysis of existing data and new data collection		
Assessment of the ecological value of the MPAs by means of emergy analysis	Emergy analysis (Odum 1996, Vassallo *et al.* 2017, Paoli *et al.* 2018)	
Identification of ecosystem functions and services	Haines-Young & Potschin 2011, CICES	
Assessment of environmental and economic costs and environmental impacts	Emergy analysis (Odum 1996)	Carbon footprint with social cost of carbon (Visintin *et al.* 2016)
Assessment of environmental and economic benefits		Willingness to pay, financial statement analysis (Visintin *et al.* 2016)
Overall costs-benefits balance	Franzese *et al.* 2015	
Implementation of an operational GIS platform		

Indeed, ES flows that keep our economies working originate from NC (Sukhdev *et al.* 2010). Costanza and Daly (1992) introduced the concept of NC, associated with human capital and manufactured capital. NC is the economic metaphor for the limited stocks of physical and biological resources and it includes land, air, water, sea and ecosystems themselves. Human capital comprises all individuals' capacities for work, while manufactured capital encompasses material goods generated through economic activity and technological change (UNU-IHDP & UNEP 2012). Under the perspective of strong sustainability, NC is irreplaceable with manufactured capital (de Groot *et al.* 2002, 2012) since the current level of ES supply can be ensured only if NC is maintained constant.

It is important to distinguish financial and environmental accounting. Financial accounting is designed to convey information to external shareholders and financial authorities by means of standardized procedures that generate comparable data. The main goal of financial accounting is to assess the economic performance of a company or institution in accordance with national laws and international accounting standards (Jasch 2003). Environmental accounting, instead, is based on material flow budgets. These budgets are realized through the quantification of material and energy flows within a defined system boundary and expressed in physical units. Biophysical evaluation methods, able to quantify physical features and developed to be integrated with the preference-based assessments of natural resources (Jørgensen 2010, Müller & Burkhard 2012, Odum 1996, Wackernagel *et al.* 1999), are particularly suitable for environmental accounting. Biophysical methods usually use a cost of production approach or the so-called *donor-side* perspective. To understand this perspective nature can be represented as an input-state-output system (Pulselli *et al.* 2011). A *user-side* approach focuses on outputs and on the identification of users that exploit them; a *donor-side* one focuses on inputs. The ES theory is a typical *user-side* approach (Costanza *et al.* 1997, TEEB 2010) based on an anthropocentric viewpoint (de Groot *et al.* 2002), while biophysical methods are founded on the assessment of taken resources and are thus classified as *donor-side* approaches.

To gain a real sustainability, integrating both economic and ecological approaches, it is fundamental to connect the two sides of the coin (donor/user) in order to set up efficient management strategies. Making this connection a paradigm shift is needed, from weak to strong sustainability theory. The weak sustainability theory presupposes the full substitutability of NC with the manufactured capital and aims to maintain their sum constant over time, compensating for the decrease of one with the increase of the other. According to this theory, an economy can be considered sustainable even if it impoverishes the NC on which is based upon. Moreover, if the weak sustainability concept is embraced, there is no contradiction between sustainability and continuous economic growth, since the NC can be replaced by a same amount of manufactured capital (Gowdy & O'Hara 1997) blinding the loss of intrinsic value of the NC that so occurred. On the contrary, according to strong sustainability theory, natural and manufactured capital is not mutually replaceable, so each component must be kept constant (Chiesura & de Groot 2003). The unsubstitutability lays on several reasoning among which the existence of NC "critical" components contributing to welfare in a unique way (Chiesura & de Groot 2003). Furthermore, according to the laws of thermodynamics, the transformation of NC into artificial is an irreversible process. A decrease in NC is, therefore, a sign of non-sustainability (Vitousek *et al.* 1997).

As a consequence, to embrace the strong sustainability theory, two parallel budgets should be realized aiming at preserving the NC intact (Chiesura & de Groot 2003, Vitousek *et al.* 1997). Moreover, a net benefit in both of them must be obtained to manage the ecological and the economic components in a sustainable way. These two budgets can be named ecological and economic: the ecological approach measures the biophysical effort made by nature to create the exploited resources (*e.g.*, sun, wind, rain, materials, fuels, manpower) and it gives information about the environmental sustainability, while the economic one assesses the financial flows derived from this exploitation. The main goal of this research is to provide synthetic indicators of ecological and economic sustainability dealing with the ecological issue in a strong sustainability perspective and to describe the operational results of the framework application to the Portofino Marine Protected Area (NW Italy).

MATERIALS AND METHODS

Portofino MPA: The Portofino MPA was established in 1999 in the Northwestern part of Italy. It is 363 ha wide and it is widely recognized as a high natural value area, worldwide known for its emerged and submerged landscapes as well as for the rich biodiversity hosted with the remarkable presence of several endemic and endangered species. The MPA of Portofino is included in the European Natura 2000 Network as Site of Community Importance (SCI IT1332674: Fondali monte di Portofino). Since 2005 the Portofino MPA is a SPAMI (Specially Protected Area of Mediterranean Interest) according to the decision of the RAC/SPA Office (UNEP 2005).

Within the MPA many activities such as diving, fishing and recreational boating are carried out and regulated with different protection levels, from more severe in zone A to less severe in zone C. Despite the protection regime, activities, mainly tourism oriented, are very common and the pressure on local environment is very high: for instance, recreational boating reaches even around 200 units per days (Venturini *et al.* 2016) and diving activities count over 40,000 annual dives (Betti *et al.* 2019).

Natural capital assessment: NC value has been assessed as described in Vassallo *et al.* (2017) and Paoli *et al.* (2018) through emergy analysis. To create and maintain natural good and services, the environment must perform a work that requires an energy or material sources and a number of transformations. Emergy is the total amount of energy used directly or indirectly to generate a product. Since the fundamental energy that powers the biosphere is solar energy, the work done by the environment through all the transformations to obtain a product can be calculated as the total amount of equivalent solar energy. In such a way, emergy is able to attribute a value to environmental goods and services in terms of production cost necessary to obtain them (Odum 1996). Emergy analysis pertains to the so-called biophysical methods for NC evaluation. These methods allow to ascribe a value to a good or service on the basis of its intrinsic characteristics, regardless of market laws. Biophysical methods are then particularly suitable to be used in association with economic methods to obtain a full assessment (de Groot *et al.* 2010).

Table II. – UEVs employed for the MPA's emergy calculation, 15.20E+24 sej emergy baseline was used (Brown and Ulgiati 2010).

ITEM	UEV (sej/unit)	Reference
C	1.02E+08	Campbell *et al.* 2014
N	7.40E+09	Odum 1996
P	2.86E+10	Odum 1996
Sun	1.00E+00	Odum 1996
Rain	2.93E+04	Odum 1996
Wind	2.41E+03	Odum 1996
Currents	3.80E+04	Odum 1996
Geothermal heat	2.00E+04	Brown & Ulgiati 2010
Tides	7.20E+04	Brown & Ulgiati 2010
Runoff	6.61E+04	Odum 1996

NC is a stock of resources that nature stored in space and time and with a certain effort.

To evaluate MPA's NC, an inventory of all biocenosis, their surfaces and their biomass was realized. At this purpose previous studies about benthos and demersal fishes have been used (Guidetti *et al.* 2011). All items required to generate the biomass stocked in each biocenosis and in the entire MPA were assessed and then converted in emergy units. The required inputs are those allowing the photosynthetic process: through photosynthesis biomass is originated and stored in space and time. Considered items are then carbon, phosphorous, nitrogen, sun, wind, rain, tides, currents, runoff, all expressed in biophysical units (sej) using conversion factors (UEVs) in Table II. When emergy is calculated for each biocenosis, the sum of their values gives the MPA overall NC. In this work 15.20E+24 sej emergy baseline (Brown & Ulgiati 2010) was used for emergy calculation. The final unit of measure for NC evaluation is emergy-euro (em€), namely the emergy biophysical unit (sej) translated in money equivalent by means of appropriate conversion factors (Vassallo *et al.* 2017). The conversion of biophysical NC value in "virtual" money value or currency equivalent can be done by using the indicator named Emergy to Money Ratio (EMR) (Lou & Ulgiati 2013). This indicator is calculated as the ratio between the total emergy supporting a nation and its gross domestic product in the same year (Brown & Ulgiati 2004). EMR represents the average amount of emergy needed to generate one unit of money in the national economy (Odum 1996). The monetary value of NC for each habitat is then calculated by dividing the emergy value by the EMR. The monetary value of NC for the whole MPA is calculated as the sum of the monetary values of all the habitats (Vassallo *et al.* 2017). Even if the NC conversion in monetary units is not compulsory, most of all in a strong sustainability approach, nonetheless this makes results easily conveyable from scientists to managers and also from managers to general public or key stakeholders (*e.g.*, tourists or fishermen). Detailed results are presented in Paoli *et al.* (2018).

Table III. – List of selected and evaluated services framed in the context of CICES scheme.

Section	Division	Group	Class	Simple descriptor	Specific service evaluated
Provisioning (biotic)	Biomass	Wild animals (terrestrial and aquatic) for nutrition, materials or energy	Wild animals (terrestrial and aquatic) used for nutritional purpose	Food from wild animals	Wildlife exploitation for food purposes through professional artisanal fishing
Regulation & maintenance	Regulation of physical, chemical, biological conditions	Atmospheric composition and conditions	Regulation of chemical composition of atmosphere and oceans	Regulating our global climate	Climate regulation through Carbon storage by autotrophs
Cultural	Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting	Physical and experiential interactions with natural environment	Characteristics of living systems that enable activities promoting health, recuperation or enjoyment through passive or observational interactions	Watching plants and animals where they live; using nature to destress	Tourist use and economic impacts from: bathing tourism; pleasure boating; recreational diving; sport and recreational fishing

Table IV. – Items considered for emergy environmental costs of considered ES.

	ES					
	Bathing	**Diving**	**Recreational boating**	**Sport and recreational fishing**	**Professional and artisanal fishing**	**MPA institutional activity**
Fuel for users' journey	X	X	X	X		
Fuel for users' navigation		X	X	X	X	
Fuel for MPA vehicles						X
Electricity by activity	X	X				X
Natural gas by activity	X	X				X
Drinking water by activities	X	X				X
Means of transport			X	X	X	X
Consumer goods						X
Expenditures	X	X	X	X	X	
Human labor	X	X	X		X	X

ES assessment and budget implementation: The ES taken into account and evaluated were selected among CICES scheme (Haines-Young & Potschin 2011) (Table III) and in particular are: 1) wildlife exploitation for food purposes through professional artisanal fishing; 2) climatic regulation; 3) tourist use (*i.e.*, bathing tourism; pleasure boating; recreational diving; sport and recreational fishing); 4) economic impacts. Even if the list of ES provided by marine ecosystems is much greater (Liquete *et al.* 2013), this ES selection has been identified within the EAMPA project by MPA managers as a core set of services directly affected by the protection regime of the MPAs. This selection is also reported within the official National Account for the NC of the Italian ministry in the section dedicated to the MPAs (Comitato Capitale Naturale 2018).

Budget implementation was realized on an annual basis and according to Table IV from Marangon *et al.* (2008) and Visintin *et al.* (2014).

For each ES environmental annual costs and benefits are calculated through data collection and treatment from:

– questionnaires and interview campaigns for data gathering about users presences, habits and expenses, and users and economic operators resources' consumption;

– improvement of authorization system for ES fruition by users and economic operators in order to obtain data about their MPA attendance;

– inventory of resources consumed, detailed revenue and expenditure of the MPA itself;

– set up a specific data management system and a website for the rationalization of previous described data (biological data, questionnaires, authorizations) and project results.

In addition to costs and benefits related to ES, also those related to MPA institutional activities (*e.g.*, administrative and scientific activity) have been accounted.

Environmental costs and benefits are flows, respectively imposed to and generated from NC, as a consequence of ES fruition. Environmental costs are calculated here with a biophysical, ecological approach and then assessing the annual quantity of removed or damaged NC. This is an ecological donor side perspective since NC is the core of the cascade and the base from which ES arise. Environmental benefits are evaluated with both the economic, user side perspective, and the ecological, donor side perspective. When economic side is used, direct and indirect financial gains are estimated. When ecological side is embraced, the net ecological production of key components is assessed in biophysical units. Biophysical units are later translated in monetary equivalents as described for NC.

Economic benefits and costs are financial flows received and spent by the MPA.

Environmental costs: Environmental costs originate from the use of natural and man-made resources and the related impacts on natural environment due to the activities carried out by users during a year. They are therefore attributable to each ES. Environmental costs are divided into *direct environmental costs*, whose effects occur within the borders of the MPA (direct impacts), and *indirect environmental costs*, whose effects take place outside the MPA, sometimes even at very large distances (indirect impacts). Both costs categories are assessed with ecological donor side perspective using emergy analysis. The calculation of direct costs was not foreseen by the EAMPA framework and has been added as new budget module in the context of GIREPAM project.

Donor side direct environmental costs

Direct costs were assessed in the context of GIREPAM project for boaters, divers and fishermen (both recreational and professional artisanal). For boaters and divers, the cost is represented by the impacts made on the sensitive biocenosis (*i.e.*, *Posidonia oceanica* (Linnaeus) Delile and coralligenous) in terms of annual removal of NC. Specifically, boaters exert an impact due to the anchoring action on *P. oceanica* meadows. The mechanical damage due to the anchorage can be very significant especially on seagrass meadows, so much so that its survival is at risk. The exerted damage affects the entire habitat whose organ-

Table V. – Calculation formulas for environmental costs items.

Item	Data required	Data source
Fuel for users' journey	Average km travelled per user per presence	User questionnaires or authorizations
	Average consumption per km	Bibliographic or statistical sources
	Users attendance per year	Authorizations or monitoring
Fuel for users' navigation	Average expenditure per user presence for activities within MPA	Users questionnaires or authorizations
	Fuel price	Bibliographic or statistical sources
	Users attendance per year	Authorizations or monitoring
Fuel for MPA vehicles	Yearly consumption	Interviews to MPA staff
Electricity by activity	Annual consumption of operators and of the MPA institution	Interviews with commercial operators and MPA staff
Natural gas by activity	Annual consumption of operators and of the MPA institution	Interviews with commercial operators and MPA staff
Drinking water by activities	Annual consumption of operators and of the MPA institution	Interviews with commercial operators and MPA staff
Means of transport	Vehicle weight / life time	User questionnaires or authorizations
	Days of use in MPA for user presence	User questionnaires or authorizations
	Users attendance per year	Interviews with commercial operators and MPA staff
Consumer goods	Goods consumed in a year for carrying out the institutional activities of the MPA	Interviews to MPA staff
Expenditures	Average expenditure per user presence	User questionnaires or authorizations Commercial operators interview Bibliographic or statistical sources
	User attendance or total annual expenses for operators	Authorizations or monitoring and operator interviews
Human labor	Number of annual working hours dedicated to the ecosystem service	Operator interviews and authorizations Bibliographic or statistical sources

isms are impacted causing, in some cases, even changes in the trophic structure of the habitat (Francour *et al.* 1999, Backhurst & Cole 2000, Milazzo *et al.* 2002, 2004, Lloret *et al.* 2008). Divers instead impact through the physical contact with sea-bottom. Even if diving is traditionally considered an activity generating benefits without inflicting damage on the marine environment, its increase in the last decades has shown direct effects. In particular, the benthic calcareous organisms are affected due to the presence of species with high fragility and low growth rate (Milazzo *et al.* 2002, Ballesteros 2006, Lloret *et al.* 2006, Di Franco *et al.* 2009, Luna *et al.* 2009, Hammerton 2014, Betti *et al.* 2019). These negative effects are both assessed based on the presence of users yearly attending the MPA. For recreational and professional artisanal fishing, the impact is represented by the annual fish catches. Once the amount of biocenosis surface removed or damaged by boaters and divers was calculated, the value per unit of surface calculated by Paoli *et al.* (2018) has been applied to this area. The value associated to the withdrawn of fishes have been instead estimated as NC removal following Vassallo *et al.* (2017) and Paoli *et al.* (2018).

Donor side indirect environmental costs

Indirect environmental costs, on the other hand, concern the exploitation of the resources (*e.g.*, fuels, materials) necessary for ES fruition within MPA (bathing, boating, diving, fishing).

Table VI. – UEVs used for calculation of emergy associated to ES fruition.

Item	UEV		Reference
Diesel	1.81E+05	sej/J	Brown *et al.* 2011
Electricity	6.53E+04	sej/J	Brown & Ulgiati 2002
Water	7.61E+05	sej/J	Buenfil 2001
Methane	1.78E+05	sej/J	Brown *et al.* 2011
Wood	3.03E+04	sej/J	Buonocore *et al.* 2014
Fiberglass	3.77E+10	sej/g	Puca *et al.* 2017
Rubber	5.79E+09	sej/g	Puca *et al.* 2017
Aluminum	2.04E+10	sej/g	Buranakarn 1998
Steel	2.63E+13	sej/kg	Puca *et al.* 2017
Manpower	9.51E+12	sej/h	Pereira *et al.* 2013
Money	9.60E+11	sej/€	Pereira *et al.* 2013

Table V lists the items included in the calculation of the environmental costs associated to consider ES. For each item, the data necessary for the calculation of the quantity consumed are reported in Table IV.

Indirect environmental costs have been accounted as:

– impact on global warming by calculating the carbon footprint;

– consumption of resources through emergy analysis assessment.

The quantities of resources consumed and expressed in

Table VII. – Conversion factors for the application of the methodologies emergy analysis and carbon footprint.

Methodology	Conversion factor	Unit pf measure	Reference
Carbon footprint	36.92	€ 2015/tCO_2	Pereira *et al*. 2013
Emergy analysis	9.60E+11	sej/em€	EPA 2016

their specific unit of measurement (*e.g.*, grams, joules, kilowatt hours) are then transformed into the weight of CO_2 equivalents emitted (for calculation of carbon footprint) and into equivalent solar energy (sej, for the calculation emergy analysis).

The carbon footprint provides information related to the impact on the environment and humans as a result of greenhouse gas emissions. Emissions are quantified in tons of equivalent carbon dioxide produced by the use of energy and materials for carrying out anthropogenic activities. For the estimation of these emissions, the database published by the United Kingdom Government's Department for Business, Energy & Industrial Strategy was used (Hill *et al.* 2013). This database provides the conversion factors to transform the quantity of material or energy of each resource used into the corresponding equivalent CO_2. These conversion factors come from the Fourth Assessment Report of IPCC (Intergovernmental Panel on Climate Change; IPCC 2007) and refer to a 100-year Global Warming Potential (GWP 100). The conversion factors produced by IPCC take into account the greenhouse gas emissions generated during the entire life cycle of an item.

Emergy analysis, expresses resources required for ES fruition in the single unit of measurement of sej. At this purpose UEVs listed in Table VI were employed.

For both measures the sum of all the elements consumed provides the value of the total impact and the year was chosen as the temporal functional unit.

The values in emergy and carbon footprint units can be transformed into equivalent monetary units through the use of the conversion factors shown in Table VII (Pereira *et al.* 2013, EPA 2016). This conversion allows the two measures to be included in "cost-benefit" analyses. As a consequence, they represent tools to carry out an assessment of the damage generated and to implement mitigation measures.

Environmental benefits: Environmental benefits were accounted with both an economic and an ecological approach. In particular, when the economic approach is embraced, the environmental benefits are assessed as real or virtual yearly monetary benefit to humans associated with ES fruition. These benefits are called user side environmental benefits since they generate advantages only for humans, and not for nature, but they directly arise from the enjoyment of nature. On the contrary, when the ecological perspective is taken into account, the environmental benefits can be assessed as the monetary value associated to the biophysical production of the considered biocenosis and are then called donor side environmental benefits since they represent, first of all, a profit for nature.

User side environmental benefits

The environmental benefits associated with the following ES have been considered: 1) wildlife exploitation for food purposes; 2) climatic regulation; 3) tourist use; 4) economic impacts.

Wildlife exploitation for food purposes is the catches of biological resources due to fishing. The catches amounts are converted into monetary values at market prices.

Climate regulation is associated with the greenhouse gas cycles regulation performed by ecosystems. The sea, in particular, plays a fundamental role thanks to its ability to accumulate CO_2 that would otherwise be released into the atmosphere, increasing the greenhouse effect.

Tourist use refers more precisely to the physical and experiential use of plants, animals, marine and terrestrial landscapes and, therefore, to the environmental benefits which divers, sports fishermen, boaters and bathers appropriate. It is quantified through the number of users and its monetary value can be estimated through the contingent valuation method (CVM), and in particular by estimating the Willingness to Pay (WTP, Hawkins 2003). This represents the maximum amount that a user is willing to pay for a specific good or service, in this case for the preservation of the MPA, over the amount already paid to carry out the specific activity. CVM use can be controversial if applied in non-use value and it has been criticized for some weaknesses and biases that include, for instance, the failure of respondents to incorporate their personal budgets in valuation decisions, embedding effect, and overestimation of values (Diamond & Hausman 1994, Hausman 2012). Nonetheless, it is considered an effective method for estimating ES and it is a widely used for estimating non-marketed values given its wide applicability, flexibility and strong operability (Brander & Koetse 2011, Mutandwa *et al.* 2019, Bostan *et al.* 2020, Chu *et al.* 2020, Malinauskaite *et al.* 2020).

Economic impacts derive from the physical and experiential use of plants, animals and marine and terrestrial landscapes. In particular, it relates to direct, indirect and induced economic revenues associated with the activities of divers, sports fishermen, boaters and bathers exercised by tourists spending on the economic system. The direct effect is the impact of the expenses incurred by the user on the local economy, which would not have materialized in the absence of the aforementioned ES (*e.g.*, food, accommodation). The indirect effect is the impact on supply companies that respond to the greater local demand due to the direct effect (*e.g.*, food industry, maintenance). The induced effect is the impact deriving from the change in the level of income available to residents due to the greater demand for work.

Donor side environmental benefits

Donor side benefits, generated by the habitats or habitat components subjected to direct pressures and costs and described in the paragraph about donor side direct environmental costs, have been assessed. These benefits have been assessed only for the components subjected to direct cost in order to investigate if the impact exerted by humans is sustainable or not in a strong sustainability perspective. If costs are greater than benefits the NC is eroded as a consequence of ES fruition.

Table VIII. – Framework for budget calculation.

A	*BENEFITS PER YEAR*	B	*COSTS PER YEAR*
	ENVIRONMENTAL BENEFITS		ENVIRONMENTAL COSTS
a1	*ENVIRONMENTAL BENEFIT – economic user side*	b1	*INDIRECT ENVIRONMENTAL COSTS – ecological donor side*
	Wildlife for food		Pleasure boating
	Control of erosive phenomena		Recreational diving
	Nursery		Sport and recreational fishing
	Climate regulation		Professional artisanal fishing
	Tourist use		Bathing
	Economic impacts		MPA Institutional activity
	Scientific activity		
	Educational activity		
a2	*ENVIRONMENTAL BENEFITS – ecological donor side*	b2	*DIRECT ENVIRONMENTAL COSTS – ecological donor side*
	Coralligenous secondary production		Pleasure boating
	Fish secondary production		Recreational diving
			Sport and recreational fishing
			Professional artisanal fishing
a3	*ECONOMIC BENEFITS*	b3	*ECONOMIC COSTS*
	Current revenues		Current expenditures
	Capital revenues		Capital expenditures
	Reallocation of funds		Reallocation of funds
	TOTAL BENEFITS		TOTAL COSTS
	AGGREGATED NET BENEFIT (A – B)		
	NET USER SIDE ECONOMIC BENEFIT (a1 + a3 – b3)		
	NET DONOR SIDE ECOLOGICAL BENEFIT (a2 – b1 – b2)		

These components are: 1) coralligenous habitat (damaged by divers); 2) *P. oceanica* habitat (damaged by anchoring); 3) fish stocks (subjected to sport and recreational fishing as well as to artisanal professional fishing).

The benefits are then assessed as secondary production generated, taking into account the production/biomass ratio and the mortality rates of composing organisms according to the following equation:

$$\textit{Net secondary production} = \sum_{i=1}^{n} B_i \cdot \left(\frac{P_i}{B_i} - M_i\right)$$

where *Bi* is the biomass, *Pi/Bi* is the production/biomass ratio and *Mi* is the mortality rate of the considered organism *i*.

Since *P. oceanica* meadows in Liguria show a general condition of regression, the meadow was considered not able to compensate the damage (Montefalcone *et al.* 2007).

Economic benefits and costs: The economic benefits are the financial revenues of the MPA deriving, for instance, from national financing and self-financing.

The economic costs are the expenses associated with the maintenance of the MPA, including, for instance, the ordinary maintenance of buildings and structures, the scientific projects, the purchase of goods and equipment.

Budget accounting: All results obtained through the application of the described approach, namely cost and benefits associated to each service, allow obtaining all items to be included in the framework presented in Table VIII. From the budget three main results are obtained and reported as follows together with calculation formulas referring to Table VIII:

1) Aggregated net benefit: total benefits – total costs = A – B

2) Net economic benefit = user side environmental benefits economic side + economic benefits – economic costs = a1 + a3 – b3

3) Net ecological benefit = donor side environmental benefits – environmental costs = a2 – b1 – b2

Aggregated net benefit is a synthetic index of ecological and economic performances of the MPA, it is a weak sustainability indicator. Net economic benefit is a financial benefit taking into account also revenues directly related to ES fruition. Net ecological benefit is a strong sustainability indicator since it takes into account the benefits associated to protection regime intended as NC increase as well as costs imposed to the environment as NC decrease.

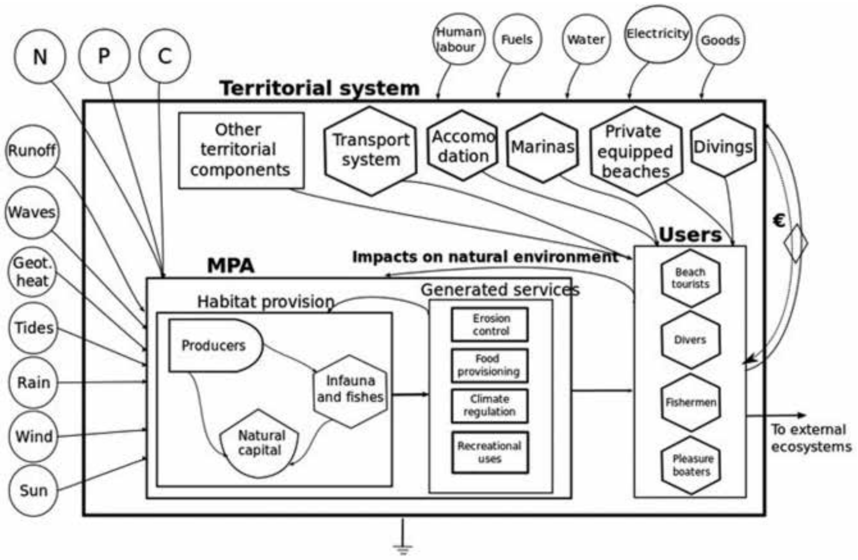

Fig. 3. – Emergy diagram of MPA and the surrounding territorial system.

RESULTS

Three main results have been obtained from the application of the framework: 1) the diagram of MPA functioning; 2) the value of NC in Portofino MPA; 3) the MPA budgets.

MPA functioning diagram

The diagram in Fig. 3 shows the analyzed system represented as a box. Inputs are depicted as circles around the main box while system components are represented within box boundaries as producers (bullets), consumers (hexagons), units with mixed or unclear behavior (little boxes) according to Odum's symobology (Odum 1967).

MPA is part of a wider territorial economic system to which is connected by means of users and their activities. Analogously MPA is connected to external ecosystems receiving and exporting materials and energy or playing important roles such as nursery or reserve area for fishes (Jennings 2009).

NC value in Portofino MPA

The NC value of the MPA and all the biocenosis included within its boundaries has been calculated in biophysical emergy units and later transformed in monetary equivalent (Table IX, Fig. 4). Biocenosis whose values are greater than 6 €/m², occupy the 19 % of MPA surface, but compose the 51 % of value. More in detail, habitats whose values are within the top range (9-18 €/m², namely coralligenous and caves) represent hot spots and are located in the MPA zones where the protection level is more severe. They occupy the 5 % of MPA surface representing the 20 % of NC value. Habitats in the 6-9 €/m² range (*P. oceanica* on soft bottom and on rocks) occupy the 14 % of surface mainly in the C shallow zone contributing to the overall NC value for the 31 %.

Table IX. – NC values for Portofino MPA.

	NC		Surface
	Sej	em€	m²
Photophilous algae	1.46E+18	1.52E+06	2.65E+05
Sciaphilous circalittoral algae	1.04E+16	1.09E+04	2.65E+03
Sciaphilous infralittoral algae	7.85E+17	8.18E+05	1.62E+05
Coralligenous	1.89E+18	1.96E+06	1.80E+05
Coastal detritic	7.98E+17	8.31E+05	5.68E+05
Muddy detritic	9.25E+17	9.63E+05	1.16E+06
Muds	1.05E+17	1.09E+05	3.11E+05
Caves	3.38E+16	3.52E+04	5.15E+03
P. oceanica dead matte	2.59E+17	2.70E+05	1.61E+05
P. oceanica and dead matte	3.53E+17	3.68E+05	1.02E+05
P. oceanica	2.22E+18	2.31E+06	3.64E+05
P. oceanica on rocks	8.69E+17	9.05E+05	1.34E+05
Sands	7.56E+16	7.88E+04	1.79E+05
Stones and pebbles	2.26E+16	2.36E+04	3.24E+04
Total	**9.80E+18**	**1.02E+07**	**3.63E+06**

The 66 % of the MPA surface hosts low value habitats (< 2 €/m²) that represent the 22 % of the Portofino MPA value. The overall value of Portofino MPA is 9.80E+18 sej equal to over 10 million of Euros.

Budgets results

The budget of the MPA takes into account five main components (Table VIII): 1) economic user side environmental benefits (a1): financial revenues obtained from the fruition of MPA ES; 2) ecological donor side environ-

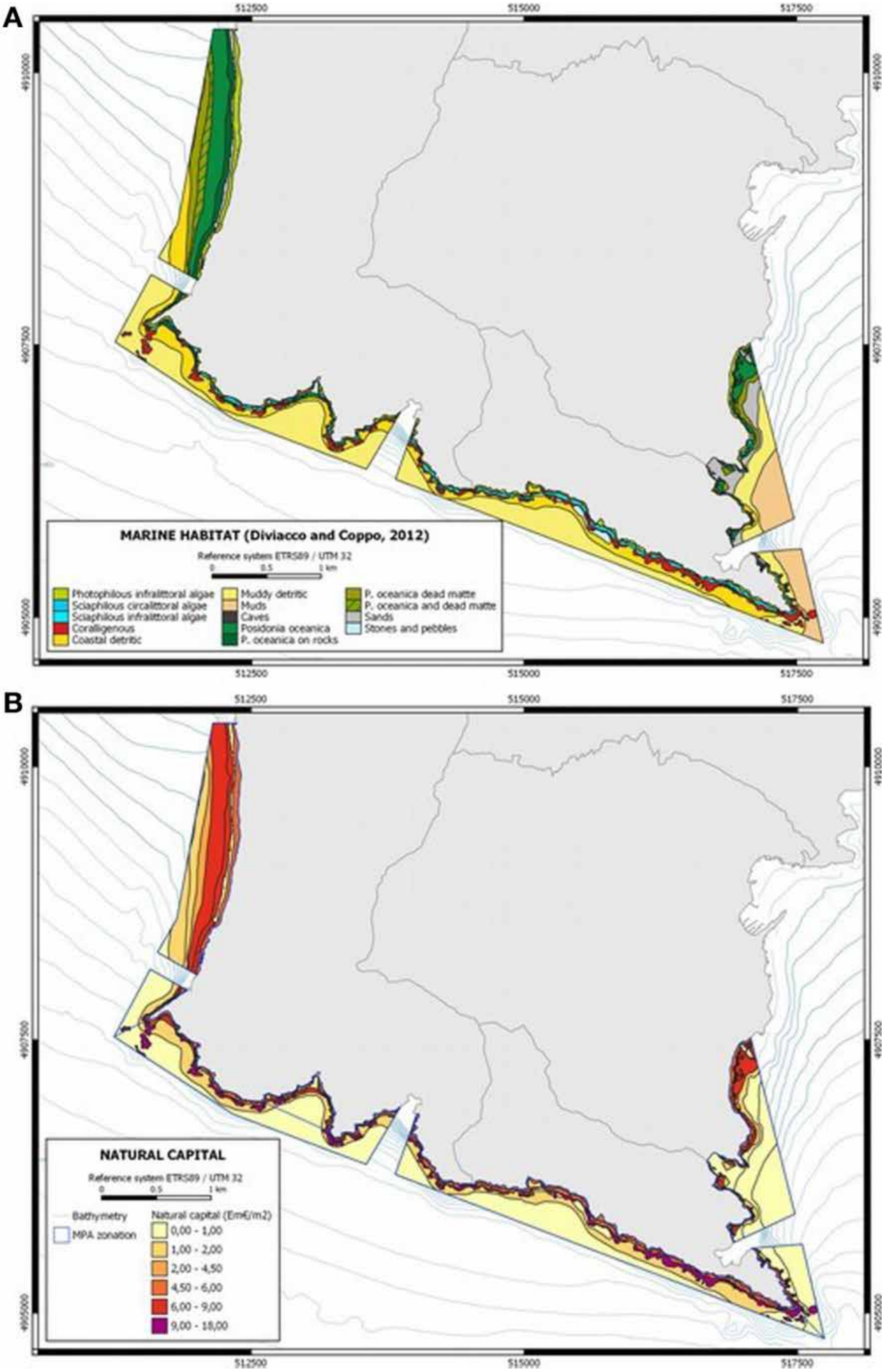

Fig. 4. – Maps of (**A**) habitats and (**B**) natural capital in Portofino MPA

Table X. – Modified budget for Portofino MPA, scientific and educational activity are not considered for the net benefit.

A	***BENEFITS PER YEAR***	€y^{-1}	B	***COSTS PER YEAR***	€y^{-1}
	ENVIRONMENTAL BENEFITS	27,639,387		ENVIRONMENTAL COSTS	13,265,532
a1	*ENVIRONMENTAL BENEFITS – economic user side*	*24,856,843*	b1	*INDIRECT ENVIRONMENTAL COSTS – ecological donor side*	*13,129,430*
	Wildlife for food	37,174		Pleasure boating	5,126,476
	Control of erosive phenomena	n.d.		Recreational diving	4,680,290
	Nursery	n.d.		Sport and recreational fishing	119,108
	Climate regulation	7,348		Professional artisanal fishing	79,976
	Tourist use	1,756,294		Bathing	3,032,989
	Economic impacts	23,056,027		MPA Institutional activity	90,591
a2	*ENVIRONMENTAL BENEFITS – ecological donor side*	*2,782,544*	b2	*DIRECT ENVIRONMENTAL COSTS – ecological donor side*	*39,337*
	Coralligenous sec. production	1,349,945		Pleasure boating	2,033
	Fish sec. production	1,432,599		Recreational diving	19,085
				Sport and recreational fishing	18,219
				Professional artisanal fishing	96,765
a3	ECONOMIC BENEFITS	843,156	b3	ECONOMIC COSTS	838,738
	Routine revenues	617,539		Routine expenditures	653,875
	Capital revenues	124,725		Capital expenditures	83,971
	Reallocation of funds	100,892		Reallocation of funds	100,892
	TOTAL BENEFITS	28,482,543		TOTAL COSTS	14,007,505
	AGGREGATED NET BENEFIT (A – B)	14,378,273			
	NET USER SIDE ECONOMIC BENEFIT (a1 + a3 – b3)	24,861,261			
	NET DONOR SIDE ECOLOGICAL BENEFIT (a2 – b1 – b2)	–10,482,988			

mental benefits (a2): gains in generated production for the environment; 3) economic benefits (a3): financial inputs from national and local administrations and from MPA activities (*e.g.*, licenses, sales, merchandising); 4) donor side ecological indirect costs (b1): environment impacts associated to users activities and MPA management calculated with carbon footprint and emergy; 5) donor side ecological direct costs (b2): damages to MPA habitats generated by users exploiting ES; 5) economic costs (b3): financial expenditures.

Portofino MPA returns to economy an aggregated net benefit (a1 + a2 + a3 – b1 – b2 – b3) of over 14 million of Euros per year, generating 78,469 €/ha/y of environmental benefits and 39,611 €/ha/y of net benefit. Benefits per year are 2 times greater than costs and are mainly due to environmental benefits (97 %) with a predominance of economic user side environmental benefits (87 %). The greatest benefit items are economic impacts (81 % of benefits) and tourist use (6 %). Analogously, environmental costs compose the 94 % of total costs. Donor side indirect costs represent almost the totality of environmental costs (93 %), with pleasure boating (36 %) and diving (33 %) being the main contributions. Donor side direct costs represent the 1 % of costs with professional artisanal fishing being the greatest item. The economic net benefit (a1 + a3 – b3) is positive and equal to almost 25 million of Euros per year, proving MPA economic sustainability while the ecological net benefit (a2 – b1 – b2) is negative (deficit of almost 10 million of Euros per year) (Table X).

DISCUSSION

This study is a first effort to synthesize biophysical and ecological information with economic measures within the context of ES theory and to provide a practical tool able to put into practice this integration as well as the ES cascade theory (Haines-Young & Potschin 2011). This effort is necessary if the scale and the intensity of growth of many economies are considered. In fact, these economies are completely dependent on natural resources exerting an ever-increasing pressure on ecosystems. This condition becomes critical since: 1) resources are finite and the excessive withdrawal leads to ecosystems degradation, also affecting ability to provide ES themselves; 2) all demands cannot always be fulfilled simultaneously generating trade-offs across ES, among beneficiaries and time periods. Concerns about the degradation of ES and the consequences on human well-being are more and more reflected in environmental policy (Bateman *et al.* 2013, Mace 2013). Over the past two decades, researches related to ES increased notably, also thanks to the development of several international initiatives that brought the ES theory to the attention of both scientists and policymakers (Nicholson *et al.* 2009, Seppelt *et al.* 2011). The Millennium Ecosystem Assessment, MEA (MEA 2005), the Convention on Biological Diversity Aichi Targets (CBD, UNEP 2010), The Economics of Ecosystems and Biodiversity, TEEB (Sukhdev *et al.* 2010) and The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, IPBES (Perrings *et al.* 2011), are examples of the initiatives developed in the last years. Against this background, the integration of the ES theory into real-world management decisions rose to prominence (Daily *et al.* 2009, de Groot *et al.* 2010, Goldstein *et al.* 2012, Maes *et al.* 2013, Martinez-Harms *et al.* 2015). As a consequence, the use of tools borrowed from financial analysis such as, for instance, cost benefit analysis, can be supportive, *e.g.*, as effective tool for choosing among alternative options (Pearce *et al.* 2006, Daily *et al.* 2009). In this context, these methods must be integrated in the ES cascade theory (Costanza *et al.* 1997, Turner *et al.* 2003, Norgaard 2010). The proposed approach suits these goals and it helps overcoming the gap between ecology and economy. The application to Portofino MPA allowed the calculation of NC value and a budget of MPA where ecological and economic magnitudes are assessed in the same unit of measure. In the budget, the costs imposed to the environment, as well as the benefits generated by ES (accounted not only as market prices but also as indirect benefits to users and territory), are included. In the specific case, the aggregated net benefit is positive since benefits per year (item A in Table X) are significantly greater than costs per year (Table X). This makes the MPA sustainable in a weak sustainability perspective.

Nonetheless, this result should be carefully interpreted. A vision taking into account only the aggregated net benefit could be ineffective: considering the budget result without splitting ecological and economic side can be short-sighted in assessing the environmental and economic consequences associated to a potentially unsustainable use of the ES. If some services are highly valued by market, an unsustainable use of NC might not appear in the aggregated calculation of net benefit. This is because the effects of ES fruition on NC status can be hidden by economic revenues: it happens when economic and ecological sides are uncoupled and when economy is not able to record negative externalities associated to NC depletion. This can lead to NC depletion and, consequently, in the long run, to the impossibility of using ES at the current level. In such a condition only weak sustainability can be achieved. To pursue strong sustainability NC must kept intact instead and this happens only if the ecological net benefit is zero or better in surplus.

As a consequence, in a strong sustainability approach two parallel budgets are required (Chiesura & de Groot 2003, Vitousek *et al.* 1997), an ecological one and an economic one. Considering ecological budget, NC is then a strong sustainability indicator if monitored through time: it must not diminish or even better it must increase.

In Portofino MPA, moreover, ES exploitation returns to the economy more than what the economy spends. In particular, if the sum of economic and ecological benefits (only economic side) is divided by economic costs, it can be seen that the exploitation of NC returns to economy 30 times the invested monetary resources. NC exploitation is then a very profitable activity.

The greatest part of the economic net benefit is due to the user-side component of environmental benefits and, in particular, to the economic impacts. Economic impacts include direct, indirect and induced economic revenues associated with the activities of MPA users: they also incorporate the part of income associated with sub-suppliers and that can be even generated in faraway lands. As a consequence, it can be said that NC exploitation exports wealth. Nonetheless, being the ecological budget negative, it cannot be assured that this wealth will be maintained in the long run. It could be appropriate to use part of this profit to restore the damaged NC.

Analyzing the environmental side of the budget, if only direct costs are compared with environmental benefits-ecological side, the result is positive. These items both are directly related to the MPA since the direct costs are those generated by users to MPA biocenosis while environmental benefits-ecological side are accounted as the value of net production generated by the same biocenosis. The MPA use of NC comprised within its border is then sustainable. But when the indirect costs are considered, the budget drop drastically down. Indirect costs are those associated to the impacts of all activities made by users and operators to get access to the ES located within the MPA (*e.g.*, fuels and material consumption to reach the MPA). Analogously to economic impacts, these costs can be generated in territories located far from the MPA borders.

Hence, the results show that the costs and benefits with the largest budget share generate effects external to the MPA borders that can hardly be handled by MPA managers.

It is then probably necessary to deal with the issue of sustainability considering a broader context able to take into account all consequences directly and indirectly generated. Even more so considering the possibility that some positive impacts, due to the establishment of the MPA conservation regime (e.g. nursery or protection from erosion effects) or some negative ones (e.g. the impact generated by lost fishing gears within and outside MPA borders) are not included in the budget.

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CAN ABANDONED SALINAS BE MANAGED AS COASTAL LAGOONS?

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ECOSYSTEM-BASED MANAGEMENT
COASTAL LAGOON
SALINA
HALOPHILE
HISTORIC MAPS

ABSTRACT. – Twenty-two of the Mediterranean coastal Salinas in S. France have been abandoned for salt extraction since 1950 and most are now available for nature conservation purposes. Their management is a challenge as the abandoned Salinas are human-modified systems, compartmentalized in salt ponds, with other important modifications in the landscape. These landscape modifications were studied in more detail in two examples by studying historic maps. Thus, for the Salin de Peyriac (Aude), an original endorheic lake (Etang du Doul) was converted into a pre-concentrating pond and nowadays persists as a hypersaline coastal lagoon. Salin des Pesquiers (Var) was created since 1848 within the Etang du Pesquier coastal lagoon. Therefore, the course of the Roubaud River, the tributary to this lagoon, was deviated to flow directly to the sea. Together with the creation of a circumferential canal this resulted in destruction of the original Roubaud delta and its fringing wetlands with ecotones in their freshwater – salt water transition zones. Both abandoned Salinas can in principle be managed as coastal lagoons, because the sediment surfaces of their ponds are located below mean sea level. Nevertheless, the Ecosystem-based management of these systems has to acknowledge a large degree of artificiality. Complete ecological restoration to historic conditions appears unrealistic and is often not desirable. Hence, abandoned Salinas offer many possibilities for creative and experimental management that should be followed by action monitoring and assessed by scientific studies.

INTRODUCTION

At the end of World War II, twenty-nine coastal Salinas along the Mediterranean coastlines in South France were exploited for salt extraction (Fig. 1). The exploited surfaces in the different Salinas ranged in 1950 from 17 ha to 3940 ha. In 2020, only five of those are still used for this purpose. While two smaller Salinas (Salin du Mas des Crottes and Salin de Esquineau) were incorporated into the enlarged Salin de Giraud, salt extraction has been completely abandoned in 16 of the smaller (*i.e.*, < 250 ha) and 6 of the medium-sized (*i.e.*, size in between 250 and 1000 ha) Salinas (De Wit *et al.* 2019), thus liberating 4019 ha for other land uses. This raises a question about spatial demands in society and spatial planning along the coastline.

The Salin de Garrouyas (137 ha) has been converted into industrial areas for the Port of Marseille-Fos; parts of Salin de Sainte-Lucie (70 ha of 441 ha) and Salin du Lion (30 ha of 61 ha) have been used for enlarging Port-la-Nouvelle and the airport of Marseille, respectively (De Wit *et al.* 2019). Projects for coastal developments, including marinas and holiday resorts, had been proposed for several abandoned Salina sites, *e.g.*, in the 1980's by the Société d'Aménagement du Port de Peyriac-de-Mer for Salin de Peyriac (Conservatoire du Littoral website). But, fortunately these projects were never executed and currently most abandoned Salinas are protected areas belonging to the Natura 2000 network (De Wit *et al.* 2019). How should these areas be managed as nature reserves? According to the Interpretation Manual of European Union Habitats – EUR28 (European Commission DG Environment 2013) "the salt basins and salt ponds may also be considered as lagoons, providing they had their origin on a transformed natural old lagoon or on a saltmarsh, and are characterized by a minor impact from exploitation."

The aim of the present study is to assess which environmental settings allow abandoned Salinas to be managed as coastal lagoons and what are alternative targets? Secondly, for the abandoned Salinas that will be managed as coastal lagoons, is it possible to adopt ecosystem-based management (EBS) and what does that imply?

LOCALIZATION, DISTRIBUTION AND ARCHITECTURE OF SMALL AND MEDIUM-SIZED SALINAS; ENVIRONMENTAL IMPLICATIONS OF ABANDONMENT OF SALT EXTRACTION

Figure 1 and Table I present the abandoned and still operating Salinas along the Mediterranean coastlines. These occur along the interior shorelines of coastal lagoons and on their coastal barriers (Languedoc region, Berre lagoon, Hyères). These Salinas have been constructed either directly within the lagoon or on their fringing salt marshes. Etang du Pesquier, the coastal lagoon on

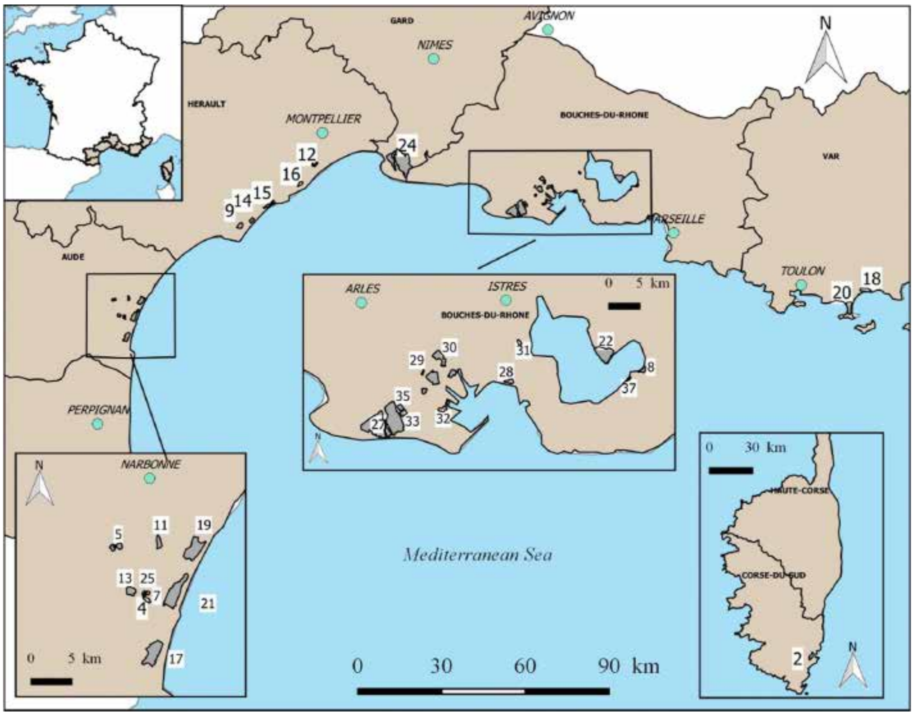

Fig. 1. – Distribution of Salinas exploited shortly after World War II in continental South France and Corsica. See Table I for names of the Salinas, indicating which Salinas have been abandoned and which are still being exploited. Creative Commons Licence, reproduced from De Wit *et al.*, 2019.

the Hyères peninsula, has completely been converted into a Salina. Other Salinas, (Sigean, Bagnas and Rassuen) occurred in depressions at a certain distance from the coastal lagoons, which were interconnected by man-built canals. Two large still operating Salinas and two abandoned medium-sized Salinas occur in the Rhône delta (Camargue) occupying former lagoonal and salt marsh environments. Some Salinas occur directly on the lowland coast, neither in a coastal lagoon nor in a delta (Fos-sur-Mer, Vieux salins d'Hyères). These latter lagoons have been constructed at the expense of salt marshes. Hence, most of the abandoned Salinas occur along the Gulf of Lion coastline, while smaller numbers of abandoned Salinas also exist in the Rhône delta, the Côte d'Azur and one in Corsica. Hence, most of the nowadays-abandoned Salinas were indeed created in coastal lagoons or on salt-marshes (locally designated as "sansouires") either bordering the lagoons or in the Rhône delta and are thus concerned by the citation above from Interpretation Manual of European Union Habitats – EUR28.

Salt extraction in Salinas is based on sun- and wind-driven evaporation and comprises two major steps, *i.e.*, i) concentrating the seawater to about one tenth of its original volume in pre-concentration ponds (Fr. partènements) and ii) using this concentrated brine with a salinity of about 290 g/L, to feed the crystallizer ponds (Fr. tables salantes). This allows to obtain a salt that is highly enriched in NaCl (table salt) in the crystallizer ponds. During the 19th century, experimental scientific studies of chemical composition of seawater and its changes during evaporation (Usiglio 1849) allowed to rationalize the process and improve the purity of the salt obtained. Accordingly, Salinas were designed in such a way that the concentrating seawater follows a flow trajectory across different pre-concentration ponds with increasing salinities. This allows for the sequential precipitation of calcium carbonate ($CaCO_3$) and gypsum ($CaSO_4$), before the brine enters the crystallizer ponds. Finally, from the crystallizer ponds, a highly concentrated brine rich in potassium (K+) and bromide (Br-) that remains after the precipitation of NaCl, is carefully collected and discharged thus preventing the impurities in the salt. This design implies that Salinas are compartmentalized environments comprising "salt ponds" or "salt-pans" of variable sizes, each surrounded by dikes with connections controlled by locks (Fr. martélières). Along the trajectory, the first pre-concentration ponds often have an irregular shape, while many of the next pre-concentration ponds and particularly the crystallizer ponds show very regular rectangular shapes. The latter feature has been used for the automatic assignment of functioning and abandoned Salinas in Corine land-cover (Bossard *et al.* 2000), assigned as Corine Land Cover (CLC) class 4.2.2 defined as salt-pans, active or in the process of abandonment. Nevertheless, this automatic assignment has resulted in a severe underestimation of Salina surfaces as the irregular pre-concentration ponds have been confounded with natural environments (De Wit *et al.* 2019). The dikes of the salt ponds can breach and erode away after maintenance by the salt extraction companies is abandoned, and this erosion particularly occurs in Salinas directly exposed to sea-surges and flash floods. Strong winds, which create wind erosion in exposed environments and strong water currents in

Table I. – List of Salinas abandoned since 1950 and still operating Salinas in South France. See the map in Fig. 1 for localization. Adapted from De Wit *et al.*, 2019.

Number (see Fig. 1)	Name of the salina	Creation date	Closure date	Owner[1]	Spatial planning designation[2]	Exploited surface 1950 (ha)
ABANDONED SMALL (< 250 ha) and MEDIUM-SIZED (> 250 ha and <1000 ha) SALINAS						
Narbonnais (Gulf of Lion)						
4	Salin de Tallavignes	1803	1962	Conservatoire	PA-CdL / N2000	37
13	Grand Salin de Sigean	1300	1968	Conservatoire	PA-CdL / N2000	96
25	Salin de Ferrand	1800	1962	Conservatoire	PA-CdL / N2000	17
7	Salin de Grimaud	1795	1962	Conservatoire	PA-CdL / N2000	19
21	Salin Sainte Lucie	1831	2005	Conservatoire	PA-CdL / Regional NR/ N2000-70 ha for Harbor	441
5	Salin de Peyriac	1300	1967	Conservatoire	PA-CdL / N2000	80
11	Salin de Campignol	1880	1963	DPM	N2000	75
Coastal lagoons of Thau and Palavas (Gulf of Lion)						
9	Salin de Bagnas	1789	1975	Conservatoire	PA-CdL / National NR / N2000	285
14	Salin de Castellas	1779	1967	Conservatoire	PA-CdL / N2000	173
15	Salin de Villeroy	1779	1968	Conservatoire	PA-CdL / N2000	186
16	Salin de Frontignan	1338	1968	Conservatoire	PA-CdL / N2000	218
12	Salin de Villeneuve	1100	1969	Conservatoire	PA-CdL / N2000	100
Camargue and Berre Lagoon						
29	Salin du Caban	1882	1970	Port of Marseille-Fos	N2000	567
30	Salin du Relai	unknown	1970	Port of Marseille-Fos	N2000	485
32	Salin de Garrouyas	1882	1950	Port of Marseille-Fos	Industry-Harbor	137
28	Salins Fos-sur-Mer / La Marronède	1833-1836	1985	Municipality of Fos-sur-Mer	N2000	205
31	Salin de Rassuen	1808	1953	Municipality of Istres	N2000- ENS	36
37	Salin de Jai	1923	1955	Communauté des communes	ENS	17
8	Salin du Lion	(822)-1802	1955	Communauté des communes	ENS, 30 ha Extension airport Marseille	61
Hyères (Côte d'Azur)						
20	Salin des Pesquiers	1848	1995	Conservatoire	PA-CdL / N2000	424
18	Vieux Salins de Hyeres	1200	1995	Conservatoire	PA-CdL / N2000	331
Corsica						
2	Salin de Porto Vecchio	1795	2001		Shortlisted for PA	29
SALINAS STILL EXPLOITED FOR SALT EXTRACTION IN 2020[3]						
Narbonnais (Gulf of Lion)						
17	Salin de La Palme	1884	2005	DPM	Salt extraction	412
19	Salin d'île Saint Martin	1910	2006	DPM, Municipality of Gruissan	Salt extraction	392
Camargue and Berre Lagoon						
24	Salin Aigues Mortes	13th century	–	Salins	Salt extraction	3940
27 (+ 33 + 35)	Salin de Giraud[4]	1856	–	Conservatoire, Salins	Salt extraction (5000 ha as PA-CdL / N2000)	2891 (+ 75 + 69)
22	Salin de Berre	1100	–	Salins	Salt extraction[5]	428

[1] Conservatoire: Conservatoire du Littoral (the French coastal protection agency), DPM: Public maritime domain (state-owned), Communauté des communes (a local public body of collaborating municipalities), Salins: Groupe Salins (multinational company dedicated to salt production).

[2] Abbreviations: PA-CdL: Protected Area owned by the Conservatoire du Littoral, N2000: Site belonging to Natura 2000 network, NR: Nature Reserve, ENS: "Espace Naturel Sensible" (a designation based on the Urbanistic code).

[3] Salt extraction was completely interrupted in the Narbonnais from 2006 to 2012.

[4] The Salin de Giraud (27) integrated the Salin de Esquineau (33) and Salin du Mas des Crottes (35), it was strongly enlarged between 1950 and 1975 to 12,000 ha. Since 2009 only about 7,000 ha is still exploited for salt extraction.

[5] Salin de Berre does not use seawater, but rather a brine from a salt mine.

coastal lagoons and salt ponds, can also contribute to the degradation of the dikes. Nevertheless, particularly at the more sheltered sites, more or less degraded dikes may persist as a vestige for many decades.

The above-mentioned rational well-organized management of the Salinas requires a fine-tuned hydraulic management. The small tidal amplitude in the NW Mediterranean is not sufficient to efficiently drive the hydraulic flow trajectories in the Salinas, as *e.g.*, used in the artisanal systems in Guérande on the Atlantic coast. Hence, pumping has been used in Salinas in the NW Mediterranean. The localization of the pumps depends on the topography and bathymetry of the Salina. Hence, Salinas with the soil surface above mean sea level have used pumps to take in the water from the sea and fill the first pre-concentration ponds. Gravitation force, sometimes combined with additional pumping has been used to drive the hydraulic flow trajectories. In contrast, Salinas created within coastal lagoons with the sediment surfaces located below mean sea level often functioned differently. In these cases, seawater intake into the pre-concentration ponds was often based on the principle of communicating vessels; the tendency of decreasing level in the successive pre-concentration ponds due to evaporation was compensated by inflow. Nevertheless, the flows driven by evaporation and gravitation were often not sufficient to drive the flow along the entire trajectory in the Salina and often pumping has been used to fill the crystallizer ponds. Pumping is expensive both in terms of energy costs (electricity) and in terms of maintenance (strong corrosion of metal parts in engines). As a result, abandonment of Salinas by the salt-extracting firms has resulted in all cases of immediate arrest of pumping.

To improve the yield and guarantee the stability of the salt-extraction process, the operating Salinas along the shorelines need to be protected from freshwater inflow from tributaries and runoff. Therefore, most of the Salinas have used a circumferential canal to intercept these freshwater flows from the catchments and deviate them directly into the lagoon or the adjacent sea. Such outlets were spatially separated from the intake of saline water into the Salina. The circumferential canal has often also been used for shipping the harvested salt (Fig. 2). In some cases, the course of tributaries was even deviated. These changes in the landscape destroyed the freshwater – saltwater gradients with their ecotones (De Wit 2011a). After abandonment of salt extraction, these gradients are not recovered spontaneously, because the natural course of the freshwater flow remains forced by the canal and is obtruded by dikes surrounding the ponds.

In conclusion, the abandoned Salinas that have become available for nature conservation management have inherited many features that imply a large degree of artificiality. At the same time, while pumping and salt-extraction have stopped, the artificial ecosystems have been subjected to a strong modification of their salinity regimes. Hence, particularly the hypersaline environments of highest salinities have often disappeared. Below, I discuss in more detail the major modifications in the landscape and ecosystems for two abandoned Salinas.

Fig. 2. – Boat used for transportation of salt harvest in a circumferential canal of a Salina in Languedoc. Note also the aerial electricity cables used for powering the pumps of the Salina. Reproduced from Leenhardt (1939, opposite page 66).

MAJOR LANDSCAPE AND ECOSYSTEM MODIFICATIONS REALIZED BY HUMANS FOR THE CREATION OF THE SALINA AND AFTER ITS ABANDONMENT; TWO EXAMPLES

Peyriac Salina and adjacent Etang du Doul

The Salin de Peyriac was created around 1300 (De Wit *et al.* 2019) in a semi-enclosed bay of the Bages-Sigean lagoon (Fig. 3A), to the north of the village of Peyriac-de-Mer. The Salina is clearly shown on the maps of Cassini (Fig. 3B) representing the situation about 1770 and the ordnance survey may (Fig. 4A) of the 19th century (Etat-major 1820-1866), which was based on field measurements realized in 1852 (Cavero 2010). Both maps show a valley adjacent to the Salina with a lake, the Etang du Doul. The valley of the Etang du Doul is an enclosed depression delimited by small hills (30-70 m) that was naturally endorheic (David & Carozza 2013). A threshold separated the valley from the semi-enclosed bay where the Salin de Peyriac was created. In the past, the volume and extension of this endorheic lake was thus the result from losses through evaporation and inputs from runoff and groundwater flows from its surrounding watershed, perhaps sometimes also by overtopping of water from the lagoon. The topographic map of 1950 (Fig. 4B), shows that at some time between 1852 and 1950, this area was modified to incorporate the Etang du Doul valley into the salt extraction enterprise at Peyriac-de-Mer. An inlet canal was created from the inlet in the lagoon and cut through the barrier into the valley (Fig. 4B, D). The Etang du Doul increased in volume. An outlet canal (Fig. 4B, D) was created to connect it with the pre-concentration ponds in the original Salina. Hence, since then Etang du Doul functioned as a pre-concentration basin in the extended Salina. Salt extraction was stopped in 1967. This did not impact the level of the Etang du Doul (see Fig. 4C, E, F) as it continued to exchange with the Salina and lagoon through the inlet and outlet canals. The salinity in the Etang du Doul has been maintained at around 1.5 to 2 times seawater salinity (Boutière 1974). However, occasionally the lake level decreased and its salinity increased when these exchanges were obtruded, and current management, therefore, aims to maintain these exchanges (Conservatoire du Littoral website).

Salin des Pesquiers (Hyères)

As shown by the Cassini map (Fig. 5A, B), in the 18th century, the Etang du Pesquier was a coastal lagoon located on the peninsula of Giens south of Hyères. This system represents a tombolo connecting the mainland to the island of Giens comprising two parallel coastal barriers with the coastal lagoon in between. The river Roubaud had its outflow located on the northern shore of the Etang du Pesquier (Aboucaya *et al.* 2011), where it created a small delta (see Fig. 5B). The Salina was created since 1848 in the northern part. Therefore, the Roubaud River was deviated and a canal was created (local name

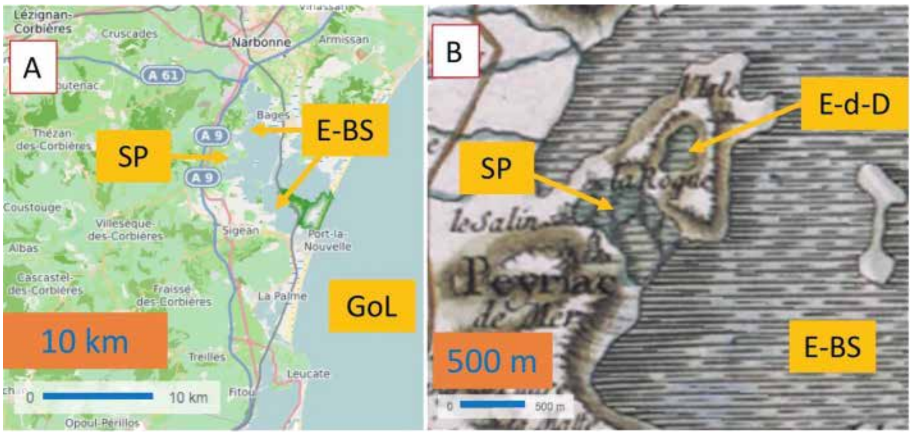

Fig. 3. – Salina in Peyriac-de-Mer (N° 5 in Fig. 1 and Table I). **A**: OpenStreetMap showing the location of Salin de Peyriac in a semi-enclosed bay of the Etang de Bages-Sigean; **B**: Carte de Cassini showing the Salin de Peyriac and the adjacent depression with the Etang du Doul endorheic lake in the 18th century. E-d-D: Etang du Doul, E-BS: Etang Bages Sigean, GoL: Gulf of Lion (W Mediterranean Sea), SP: Salin de Peyriac. Note difference in scale between panels. Source: IGN géoportail.

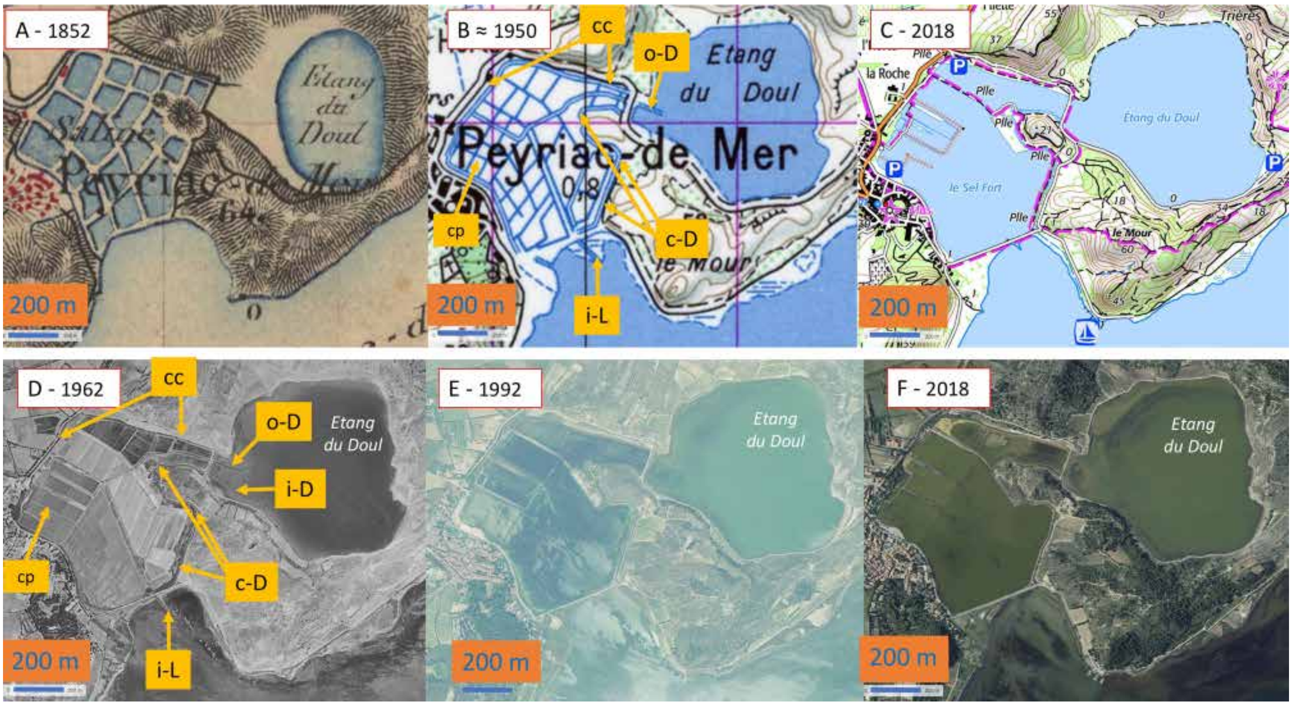

Fig. 4. – Salina in Peyriac-de-Mer. **A**: 19th century ordnance survey map (Etat-major 1820-1866) based on field measurements 1852 (Cavero 2010); **B**: 1950 IGN Topographic map; **C**: 2018 IGN Topographic map; **D**: Aerial photograph 1962; **E**: Aerial photograph 1992; **F**: Aerial photograph 2018. cp: crystallizer ponds, cc: circumferential canal, i-L: inlet from the lagoon into the canal connecting to Etang du Doul, c-D: canal connecting lagoon to Etang du Doul, i-D: inlet from canal into Etang du Doul, o-D: outlet from Etang du Doul into pre-concentration ponds in original Salina. Note same scale for all panels A to F. Source: IGN géoportail.

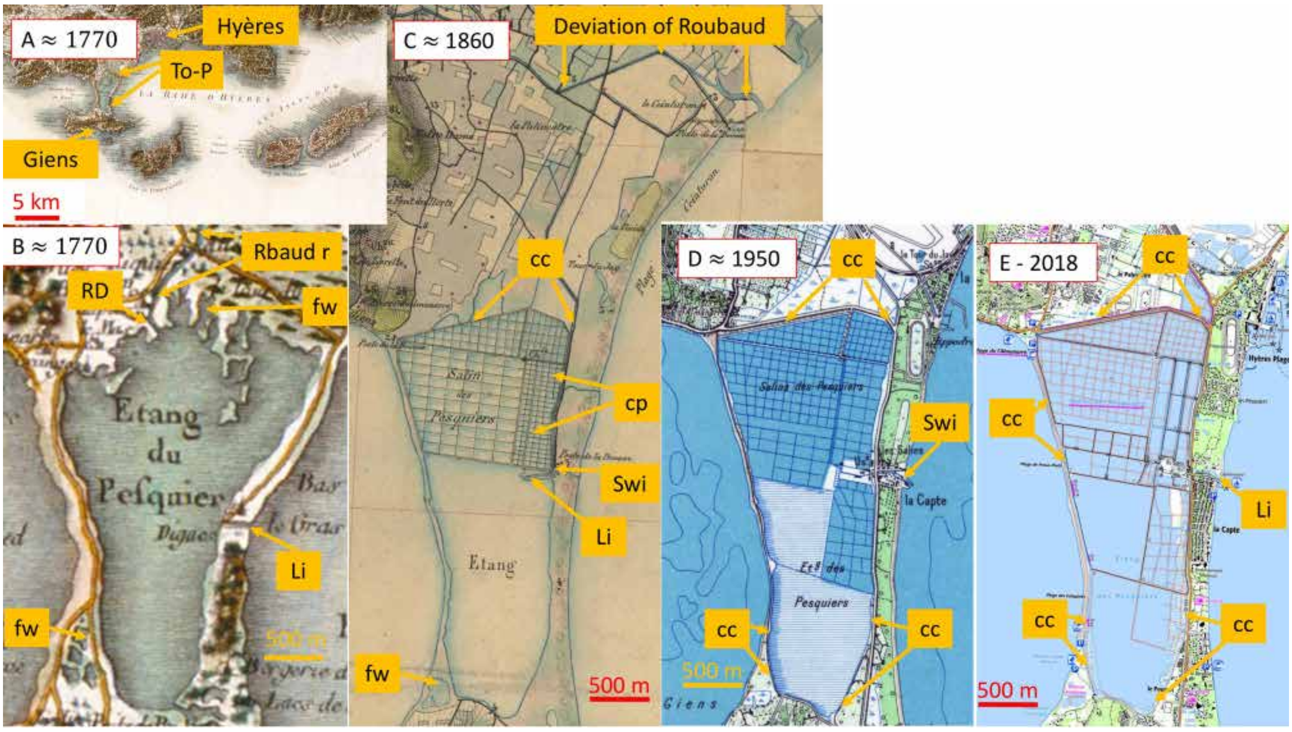

Fig. 5. – Etang du Pesquier and its progressive conversion in a Salina (Salin des Pesquiers). **A**: Carte de Cassini showing Rade de Hyères and Giens connected to mainland by a Tombolo with Etang du Pesquier; **B**: Carte de Cassini showing Etang du Pesquier; **C**: 19th century ordnance survey map (Etat-major 1820-1866), for which field measurements were performed between 1850-1860; **D**: 1950 Topographic map IGN; **E**: 2018 Topographic map IGN. cc: circumferential canal, cp: crystallizer ponds, fw: fringing wetland, Hyères: city of Hyères, Rbaud r: Roubaud river, RD: Roubaud delta in Etang du Pesquier, Li: lagoon inlet, Swi: seawater intake for salt extraction, To-P: Tombolo connecting Giens to mainland. Note same scale for all panels B to E, different scale for panel A. Source: IGN géoportail.

'canal de ceinture') as it is clearly visible on the Ordnance survey map (Fig. 5C). This destroyed the Roubaud delta and the large wetland complex including the former delta in the North became separated from the lagoon. Hence, the freshwater – saltwater gradients with their ecotones in the northern part were destroyed. While the southern part remained a lagoon with a modified inlet and fringing wetlands with freshwater-lagoon gradients still persisting in the SW corner. The Salina was progressively extended to occupy a large part of the southern lagoon as shown on the 1950 topographic map (Fig. 5D). This extension is already visible on the aerial photograph of 1920. During the same period the circumferential canal was also extended to the SW corner of the lagoon, but apparently not along the western barrier (as visible on aerial photograph of 1931). Nowadays, the circumferential canal also runs alongside that latter barrier (Fig. 5D). Salt extraction was finally abandoned in 1995.

Ecosystem-based management of abandoned Salinas

For implementing ecosystem-based management (EBM) in abandoned Salinas it is of paramount importance to document the changes in the landscape engineered by humans and understand how these have affected the ecosystem functioning. As shown before, these changes in the landscape go far beyond the compartmentalization of the Salina itself, but have most often included the fixation of the inlet and the destruction of the freshwater – seawater gradients with their ecotones. Hence, almost all Salinas have been surrounded by a circumferential canal and the Salin des Pesquiers even shows an example where the course of a river was deviated to elsewhere on the coast for restricting the freshwater input into the coastal lagoon. In Peyriac-de-Mer, the Etang du Doul valley, an enclosed depression in the landscape adjacent to the Salin de Peyriac has been completely modified by connecting it to the lagoon and the adjacent Salina. Hence the original natural endorheic lake, which is characterized by fluctuations of lake level and salinities, was converted to a permanent hypersaline lagoon.

A coherent and instrumental definition of EBM is provided by the US National Oceanic and Atmospheric Administration (NOAA). Accordingly, "EBM is an integrated management approach that recognizes the full array of interactions within an ecosystem, including humans, rather than considering single issues, species, or ecosystem services in isolation." (NOAA website). Indeed, as human-engineered ecosystems, the impact of humans cannot be neglected for Salinas. Table II lists the so-called core characteristics of EBM recognized by the NOAA and my recommendations on how this could be imple-

Table II. – Core characteristics of Ecosystem-based management (EBM) according the US National Oceanic and Atmospheric Administration (NOAA website), and the recommendations of the author on how these can be implemented for the management of abandoned Salinas.

Core Characteristics of EBM (NOAA)	Recommendations for management of abandoned Salinas
Adaptive and flexible, responsive	After deciding a target, Implementation of adaptive management supported by monitoring and research
Place-based with geographic areas defined by ecological criteria	Adopting a landscape/seascape approach, using ecohydrology, considering all the modifications engineered by humans that impact the ecosystem functioning (*i.e.*, transition zones, connectivity, etc.)
Cross-sectoral, considering interactions	Knowledge: Combine historic, geomorphological and ecological data and insights Law: Combine legislation on water, biodiversity, spatial planning and maritime issues
Proactive	Ecological restoration (prevent a dogmatic approach) and ecoengineering. Life projects as demonstration sites
Inclusive and collaborative – all levels of government, indigenous people, stakeholders	Discuss and co-construct the different options for targets with the stakeholders, including the local populations, and with local administrations

mented for the abandoned Salinas. The last characteristic in the table, *i.e.*, ‘Inclusive and collaborative concerning all levels of government, indigenous people, stakeholders’ appears as particularly pertinent for the coastal Salinas. Local populations often recognize the salt extraction activity as an important part of their local identity and are sometimes strongly in favor of keeping the “mémoire du sel” alive as a cultural heritage. Hence, for the abandoned Salina this may result in a demand for bringing back into exploitation and this has indeed been achieved by private initiative in the Salin de la Palme and Salin de l’île Saint Martin (see Table I). This option was also considered for the Salinas in Hyères, although it was finally not adopted. Bringing back into exploitation allows to conserve the hypersaline biological communities along the artificially maintained salinity gradients. The biodiversity of the pelagic and benthic biota in the ponds decreases with increasing salinity, and becomes progressively dominated by micro-organisms (reviewed in De Wit *et al.* 2019). The pigment-containing Alga *Dunaliella salina* (Dunal) Teodoresco, 1905 and extreme halophilic Archaea occur in high densities in the plankton of the most saline preconcentration and in the crystallizer ponds, successively, where they provide beautiful colorations. The hypersaline biodiversity is original as it comprises many strict halophiles restricted to these environments. Hence, the coexistence of small and medium-sized operating Salinas together with natural coastal ecosystems has resulted in increased biodiversity at the landscape level (De Wit *et al.* 2019).

Setting alternative targets for Salinas is a challenging task of primary importance, once it has been decided collectively with the stakeholders and the local public administrations to accept the permanent abandonment of salt extraction in the Salinas. Firstly, the topography and bathymetry of the ponds should be considered.

In the Mediterranean climate, the ponds with the soil above mean seawater level tend to become ephemeral salt ponds, which are submerged after autumn rainfall and dry out during spring or early summer. Their salinity results from the dissolution of the salt accumulated in the soils and increases strongly with evaporation. Such environments function as endorheic environments and provide interesting aquatic vegetation as the angiosperm *Althenia filiformis* Petit, 1829 and Charophytes including *Lamprothamnium papulosum* (K. Wallroth) Groves, 1916 and *Tolypella salina* Corillon, 1960 (Lambert *et al.* 2013; Mouronval *et al.* 2015). Nevertheless, these environments do not correspond to the generally accepted definition of coastal lagoons (Kjerfve 1994), but rather correspond to man-made surrogates of endorheic evaporitic ponds and lakes, which have decreased worldwide (Wurtsbaugh *et al.* 2017). When, these types of environments are subjected to strong wind erosion and breaching of dikes during sea surges and flash floods so-called ‘renaturalization’ can also be envisioned. Hence, the ponds will become progressively destroyed and finally it is possible to recover natural salt marshes and moving dune systems. This approach has been adopted for the northern part of the Salin de Sainte Lucie and for the 5000 ha in the Camargue that has been abandoned as Salina surface after a reduction of the exploitation in Salin de Giraud (De Wit *et al.* 2019). In conclusion, the abandoned Salinas, that comprise ponds with the soil above mean sea level, provide many interesting opportunities for nature development, although these are not suitable to be managed as coastal lagoons.

Abandoned Salinas that have the soil or surfaces of the sediments of their ponds located below mean sea level can in principle be managed as coastal lagoons. The two studied examples, *i.e.*, Salin de Peyriac and Salin des Pesquiers belong to this category. However, besides acknowledging the artificiality of the compartmentalization of the water body in Salinas, such a target should keep in mind that natural coastal lagoons are linked to a watershed and show a natural inlet that is not fixed by hard structures.

Can these features potentially be recovered together with a destruction of the compartmentalization, and is that desirable? Considering the example of the Salin des Pesquiers, which still was a natural lagoon in the 18th century (Fig. 5A, B). The theory of restoration ecology would consider this system a historical reference (Clewell & Aronson 2013). A dogmatic application of this theory would imply that the ecological restoration (see Table II) should i) destroy all the dikes and clear the lagoon, ii) destroy the circumferential canal, iii) restore the original course of the Roubaud river and restore the fringing wetlands, and iv) restore a natural inlet for which natural movements will be tolerated. This ambitious program appears of course unrealistic and probably it is even not desirable for the local populations, who may want to keep certain vestiges of the salt exploitation period. Moreover, particularly, along the coastal zone, it has to be considered that the historical reference state is probably not sustainable in face of global change (De Wit 2011b). For the Etang du Doul, ecological restoration would imply its conversion into an endorheic lake as still existing in the 19th century, but this solution has also been judged as non-desirable (Conservatoire du Littoral website). Hence, for EBM, the managers of abandoned Salinas have to accept a high degree of artificiality inherited from the exploitation period.

So, what can be done if it is not feasible to return to the historic conditions existing before the creation of the Salina? First of all, one can tolerate and even facilitate the erosion of the dikes in order to create larger ponds of more natural sizes. For the Salin de Peyriac it is shown that this erosion in the SE part of the Salina resulted in a larger water body, while vestiges of dikes remain along the W part, where the crystallizers occurred (Fig. 4E, F). On the other hand, the larger number of ponds also allows managers to experiment different environmental conditions in different ponds. Creative management may include the building of islets within the ponds to favor breeding bird colonies or developing a mosaic of ponds with different depths favoring different bird species. Reconnecting the abandoned Salina more efficiently with the sea allows to enhance the nursery functions for fishes and invertebrates. If oligotrophic conditions can be obtained and maintained in certain ponds, it appears interesting to achieve lower salinities (equivalent to seawater or below) to restore angiosperm meadows (*i.e.*, *Zostera noltei* Hornemann, *Ruppia cirrhosa* (Petange) Grande and *Cymodocea nodosa* (Ucria) Ascherson). Hence, it may be interesting to allow freshwater inflow in certain parts as these may contribute to achieve these lower salinities, while also contributing to locally creating freshwater – saltwater transition zones. This can be done by creating localized inlets from the circumferential canals regulated by locks. Nevertheless, this needs to be done with care, as it should be checked that the freshwater inputs do not carry high nutrient loadings, which will cause eutrophication of the lagoon, or chemical contaminants. The latter shows the importance of controlling water quality as requested by the EU Water Framework Directive (2000), meaning that the lagoon managers have to cope with different legislation, particularly also the EU Habitats (1992) and Birds directives (2009). Nevertheless, abandoned coastal Salinas present an interesting playground for experimental management, although it is most important that the management plan be discussed and concerted with the local stakeholders and public administrations and that the characteristics of EBM (Table II) are respected.

CONCLUSION

Abandoned Salinas that have the soil surface of their ponds below average sea level are good candidates for management as coastal lagoons. In contrast, the abandoned Salinas with soil surfaces above mean sea-level are more prone to be developed as series of endorheic temporal salt ponds or for promoting the recovery of a more natural system of salt marshes and dunes. The latter is possible in more dynamic environments, as *e.g.*, directly on the coast where wind and hydraulic processes are strong enough to reshape the landscape. In all cases, the EBM of the abandoned Salinas has to acknowledge a large degree of artificiality inherited from the exploitation period. The modifications in the landscape go beyond the compartmentalization of the Salina, and particularly have generally included the destruction of freshwater – saltwater transition zones and modified inlets. In addition, in Peyriac-de-Mer these modifications included converting an endorheic lake into a permanent hypersaline lagoon and in Salin des Pesquiers diverging the course of the Roubaud River. A complete ecological restoration to historic conditions before the creation of the Salina is often neither realistic nor desirable. While respecting the core characteristics of EBM, abandoned Salinas offer many possibilities for creative and experimental management that should be followed by action monitoring and assessed by scientific studies.

STUDY SITE AND METHODS

The study site comprised the littoral zone of the Mediterranean Sea in southern France (see Fig. 1) including the Gulf of Lion, the Côte d'Azur and the Tyrrhenian Sea in Corsica.

This paper is a mixture of an opinion paper enriched with a specific study on two abandoned Salinas, *i.e.*, Salin du Peyriac (Peyriac-de-Mer, Aude Départment, 43.088°N; 2.964°E) and Salin des Pesquiers (Hyères, Var Department, 43.066°N; 6.139°E). Figures 3, 4 and 5 have been compiled using websites Géoportail and its dedicated section "remonter le temps". The Casini map corresponds to the so-called "Marie-Antoinette-version". The following aerial photographs have been used.

Salin de Peyriac

IGNF_PVA_1-0__1962-06-30__C2446-0023_1962_F2446-2546_0181.jp2 (Fig. 4D)

IGNF_PVA_1-0__1992-08-06__C92SAA1532_1992_FD11_0097(1).jp2 (Fig. 4E)

Salin des Pesquiers

IGNF_PVA_1-0__1920-03-11__CCF00C-281_1920_CAF_C-28_0018.jp2 -

IGNF_PVA_1-0__1931-01-06__C3346-0561_1931_NP7_1205.jp2

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MAPPING OF *RUPPIA SPIRALIS* MEADOWS WITHIN THE SALTMARSHES OF HYÈRES (PROVENCE, FRANCE): A KEY SPECIES FOR AN ECOSYSTEM-BASED APPROACH

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RUPPIA SPIRALIS
SALTMARSHES
MAPPING
ECOSYSTEM-BASED APPROACH

ABSTRACT. – The saltmarshes of Hyères (Provence; France) are Mediterranean lagoons and wetlands deeply altered since Antiquity; they are an example of a socio-ecosystem characterized by industrial activities since 1848 for salt production and artisanal fishing. After the end of the industrial exploitation in 1995, saltmarshes of Hyères became property of the *Conservatoire du Littoral* (Coastal Protection Agency) in 2001 and managed by the *Toulon Provence Méditerranée* metropolitan area. Since then, access to the sites is still restricted in order to preserve the historical heritage of the site and biodiversity of birds. The present work aims to localize and map Magnoliophyta aquatic meadow habitats in order to: (i) better understand the water management of the saltmarshes, inducing the presence or absence of Magnoliophyta and (ii) assess the health status of macrophytes community. The main observed Magnoliophyta was *Ruppia spiralis*, Linnaeus ex Dumortier, its presence in such brackish waters can be considered as an indicator of good conservation status. Our results show a relatively high abundance of *R. spiralis*, strongly related with environmental conditions and water management. This study is the first step of an ecosystem-based approach; *R. spiralis*, as a primary producer, belongs to a major functional compartment of the ecosystem, which constitutes the saltmarshes.

INTRODUCTION

The saltmarshes of Hyères (Salin des Pesquiers and Vieux Salins, Provence, France; 550 ha and 330 ha, respectively) (Fig. 1) are two coastal lagoons that have been profoundly altered since the 16th century by human activities, *i.e.*, mainly artisanal fisheries and, since 1848, salt production. After the end of salt exploitation in 1995, the saltmarshes of Hyères became the property of the *Conservatoire du Littoral*, a French public agency that is government funded, in 2001. They are managed by *Toulon Provence Métropole* metropolitan area. Since then, access to the sites has been restricted in order to preserve the historical heritage and the waterfowl diversity. The Magnoliophyta *Ruppia spiralis*, Linnaeus ex Dumortier plays an important functional role as a primary producer and a habitat for juvenile fishes (Menéndez 2002, Casagranda & Boudouresque 2007, Lenfant *et al.* 2015). It can be defined as an ecosystem engineer (Verhoeven 1980). Within the two saltmarshes, a heterogeneous connectivity and a wide range of environmental conditions are induced by (i) a complex water circulation, (ii) a various depth of ponds and channels (from few centimeters to more than 1 m depth), (iii) the water level of numerous ponds which is sometimes more than 30 cm below the zero level (NGF: *Nivellement Général de la France* – General leveling of France).

The mapping and the assessment of *Ruppia spiralis* meadows and other macrophytes vitality have been held within the aquatic compartments of the two saltmarshes (*i.e.*, Salin des Pesquiers and Vieux Salins). Such survey can provide indices of the ecological status and the quality of the ecosystem functioning. The results presented in this study are intended to support the management team for improving the water management plan, while emphasizing the focus on the connectivity between the lagoon habitats and the open sea.

MATERIALS AND METHODS

The case study of the saltmarshes of Hyères: Coastal wetlands and lagoon along the Bay of Hyères (Hyères, Provence, Northwestern Mediterranean Sea, France) were deeply transformed by human activities since the Antiquity. The spread of the city of Hyères (named *Olbia* during the Antiquity) filled hundreds of hectares of wetlands, causing the fragmentation of the ecological continuity between the peninsula of Giens to the eastern part of the Bay. Since the Middle Age, two distinct areas were identified, Salin des Pesquiers and Vieux Salins. The first one was a coastal lagoon surrounded by wetland where an important local fishery was established providing considerable incomes (Réveillon 2018). The second, smaller, was exploited for salt production since Antiquity, but at an artisanal scale. From 1848 to 1995 these two areas were converted into an intensive salt

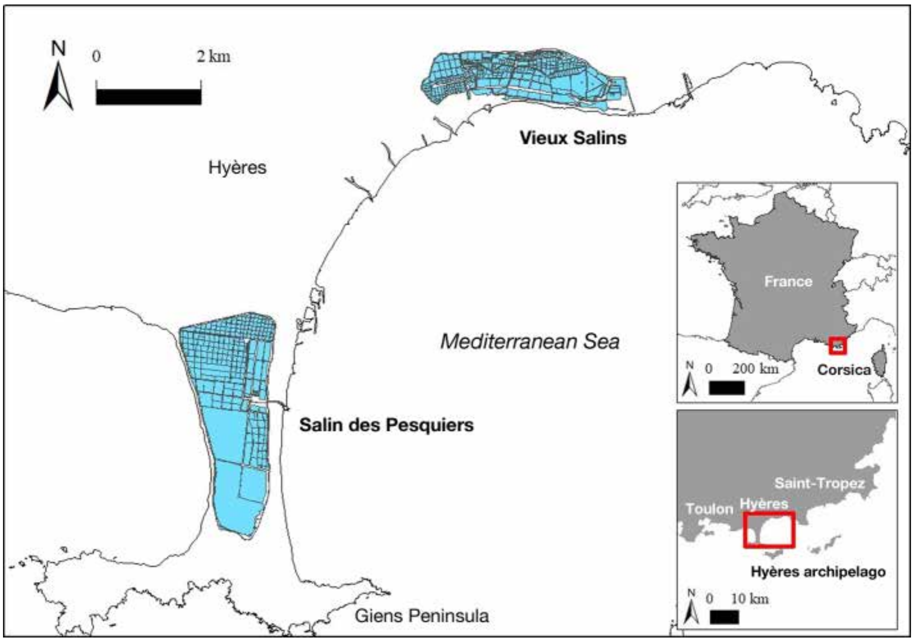


Fig. 1. – Saltmarshes of Hyères (Salin des Pesquiers and Vieux Salins) location.

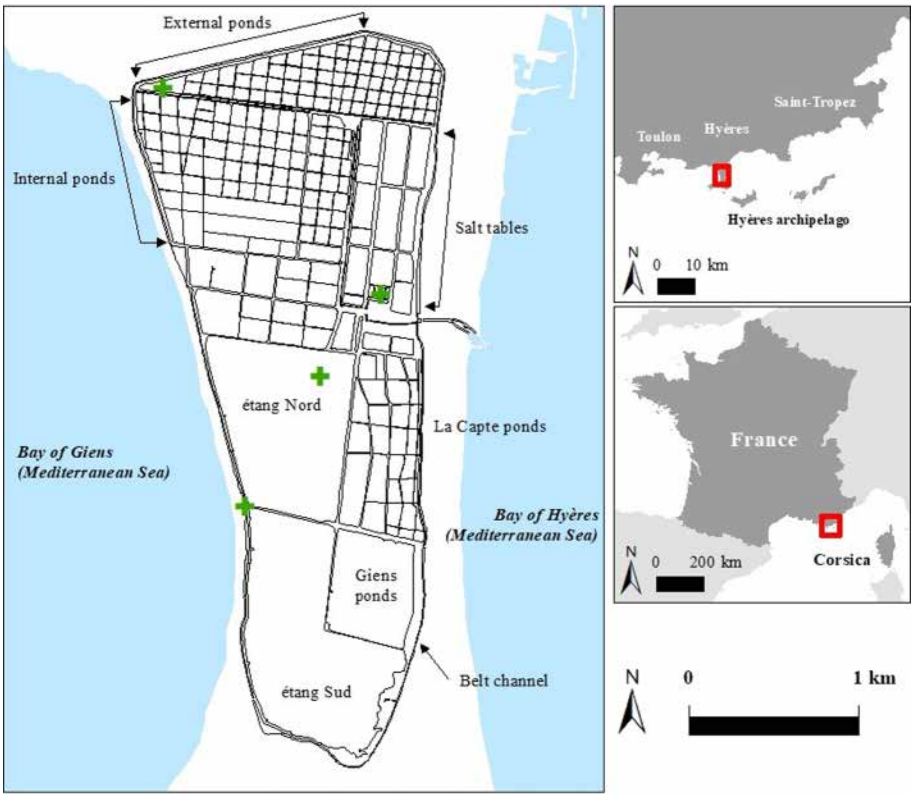


Fig. 2. – Toponymy within Salin des Pesquiers site. Green crosses correspond to sampling sites for biometry analyses.

production zone. In 2001, the whole site became the property of the *Conservatoire du Littoral* and managed by the *Toulon Provence Méditerranée* local authority. Since 2012, the sites are part of the Adhesion Area of Port-Cros National Park (Astruch *et al.* 2018). The first management goals and the water management plan were mainly focused on the historical heritage (*i.e.*, salt production, 'la *mémoire du sel*') and the conservation of waterfowl and wintering birds (Audevard 2017). A species-centered approach rather favored the so-called 'heritage taxa' (rare, threatened, charismatic). However, managers already identified the low connectivity with the open sea and related issues with fish assemblages (Conservatoire du Littoral, Toulon Provence Méditerranée & Parc national de Port-Cros 2011; CREOCEAN 2011). The water management is inherited from the previous salt exploitation (belt channel protecting the saltmarshes from fresh and seawater intrusions, low water level within the ponds, pumping, etc.), allowing the maintaining of the integrity of the sites but restricting its connectivity.

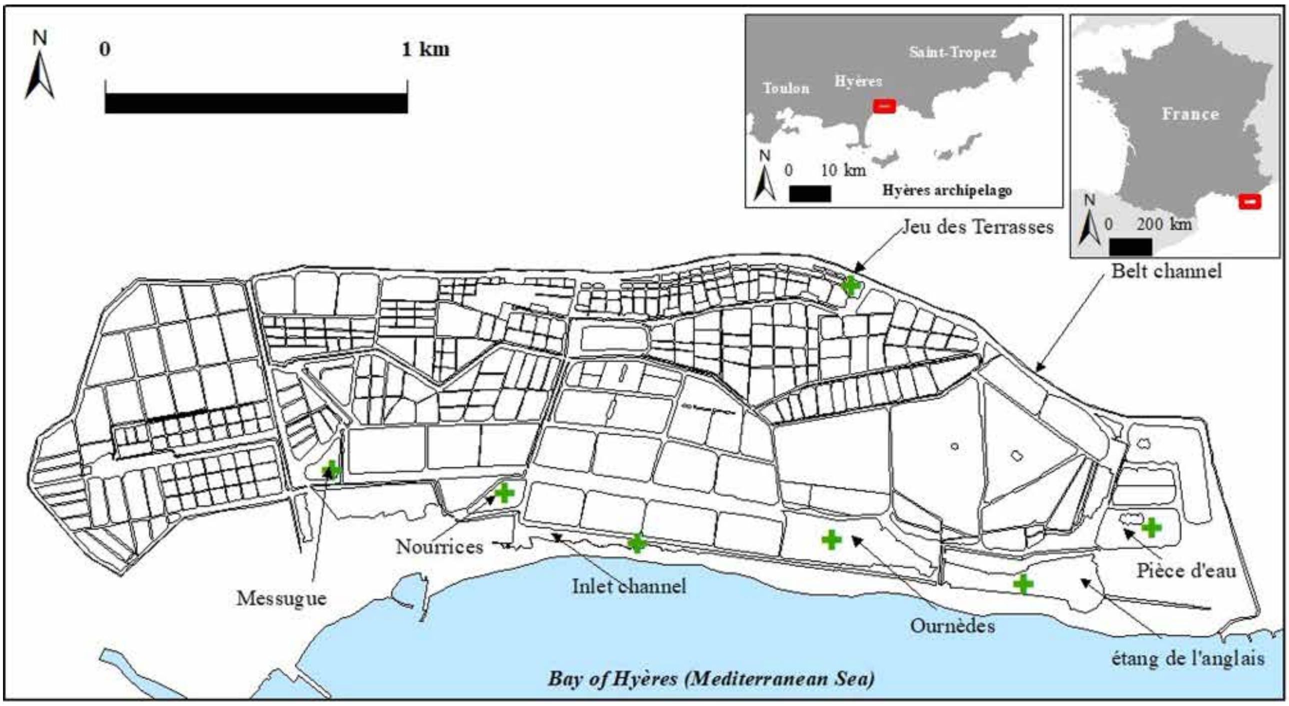


Fig. 3. – Toponymy within Vieux Salins site. Green crosses correspond to sampling sites for biometry analyses.

Table I. – Covering classes for the assessment of *Ruppia spiralis* cover.

Class	Cover (%)
0	0 %
1	0-1 %
2	1-5 %
3	5-25 %
4	25-50 %
5	50-75 %
6	75-100 %

The ecosystem engineer Ruppia spiralis: *Ruppia spiralis* Linnaeus ex Dumortier is characterized by a long, filiform and green stem. The ends of the leaves are regular and sharp with a diameter between 0.7 and 1.2 mm, the flower peduncle is very long (10-60 cm) and spiraled (Shili 2008). It is an annual/perennial species that grows in permanent and temporary lagoons and coastal brackish habitats. In France, *R. spiralis* spreads from the Mediterranean Sea to the Atlantic Ocean, the English Channel and the North Sea (Borel 2013). It is a euryhaline species able to withstand salinity ranging between 0 and 106 g/kg; its optimal growth occurs between 10 and 30 °C (Verhoeven 1980). This tolerance makes the species very competitive with other Magnoliophyta (Shili 2008). When drying occurs, the plant dies within a few days and only mature seeds survive. *Ruppia spiralis* vegetative phase starts in spring (April-May). After a few weeks, the flowers and fruits develop (May-July). Its regression starts from mid-summer (Shili 2008).

Data collecting and sampling: Data sampling has been held in May 2017, corresponding to both vegetative and flowering phase of *R. spiralis* meadow.

Mapping of *R. spiralis* meadow: covering (seven classes, Charpentier *et al*. 2003, Table I) has been visually assessed within all ponds and channels by foot or by kayak. The covering was then formatted in a Geographic Information System using ArcGIS® 10.5 software. The study site perimeter was previously digitized from the NGI's orthophotos (National Geographic Institute). Toponymy of main ponds and channel of both sites is presented in Figs 2 and 3.

Biometric analyses of *R. spiralis* were sampled at 11 sampling sites (4 at Salin des Pesquiers and 7 at Vieux Salins). On each sampling site, 3 replicates were sampled with a 20 cm × 20 cm frame (n = 33). The entire plant was sampled including roots and rhizomes until ~10 cm depth in the sediment. Analyses were held at the laboratory. After cleaning the samples from sediment and non-macrophyte organisms, flower and fruits were counted for each sample to estimate their density. Dry mass of epigenous (leaves, stem) endogenous (rhizomes, roots) of the plant and other macrophytes (*i.e.*, macroalgae) was measured after drying the samples 48 h at 70 °C. A Student's t-test was carried out to compare mean biomass of the two sites (Salin des Pesquiers and Vieux Salins).

RESULTS

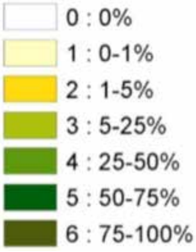
Mapping of Ruppia spiralis

Based on the surface area of each pond and the median of the cover class assessed during the sampling, we estimated 50 ha and 14 ha of *Ruppia spiralis* meadow respectively within the Salin des Pesquiers and Vieux Salins. Within Salin des Pesquiers, its covering is heterogeneous (Fig. 4, Table I bis). In the belt channel, which delimits the perimeter of the Salin des Pesquiers, we can observe a variable covering that ranges from class 0 to class 5. The Étang Nord shows a 5-class cover (50-75 %). Areas with an intermediate rate of class 2, 3 and 4 (1-5 %, 5-25 % and 25 – 50 %) are at the mouth of the belt channel with an upward gradient that ranges from class 2 (1-5 %) to class 4 (25-50 %). *Ruppia spiralis* meadow of Étang Nord lives in association with another macrophyte such as *Acetabularia acetabulum* (Linnaeus) P.C. Silva.

The covering of *R. spiralis* within Vieux Salins is also heterogeneous and limited to some ponds (Fig. 5, Table I bis). The ponds with the maximal covering of 6 and 5 classes are: Pièce d'eau (class 6, 75-100 %), a channel that extends between the Pièce d'eau and the Ilotes (class 6, 75-100 %); the channel between Jeu des Terrasses and Jeu du Petit Conseiller (class 6, 75-100 %); a part of the channel between the Étang de l'Anglais and the Jeu du Bassin 2 (class 5, 50-75 %); the channels located between the Jeu du Grand Conseiller and Farnosi and the Estagnet represent water outlets to the sewers (channels collecting the salt water after its journey in the connected ponds); here the covering rate is class 5 (50-75 %). The areas with

Fig. 4. – Covering classes of *Ruppia spiralis* at Salin des Pesquiers. The red circle corresponds to the location of *Lamprothamnium papulosum.*

Table I bis. – Colors of the covering classes of *Ruppia spiralis* used in Fig. 4 and 5.



⬜	0 : 0%
🟨	1 : 0-1%
🟨	2 : 1-5%
🟩	3 : 5-25%
🟩	4 : 25-50%
🟩	5 : 50-75%
🟫	6 : 75-100%

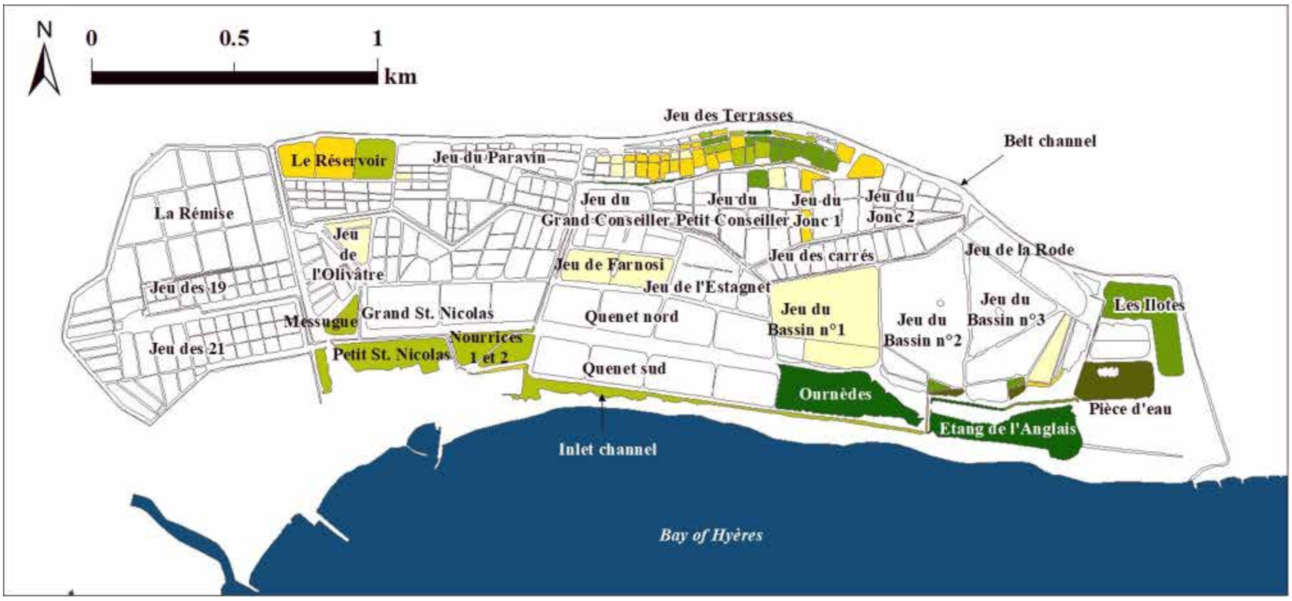


Fig 5. – Covering classes of *Ruppia spiralis* at Les Vieux Salins. The red circle corresponds to the location of *Zostera noltei.*

Table II. – Plant taxa recorded in saltmarshes of Hyères.

	Pesquiers				**Vieux Salins**			
Taxon	Étang Nord	Belt channel	External ponds	Salt tables	Nourrices	Étang de l'Anglais	Pièce d'eau	Belt channel
Acerabularia acetabulum	x							
Chaetomorpha linum	x	x		x	x			x
Cladophora sp.	x	x	x					x
Lophosiphonia obscura								x
						x		
Ruppia spiralis	x	x	x	x	x	x	x	x
Ulva rigida	x	x				x		x

Table III. – Biomass (gDW.m^{-2}) and flowers and fruits density (mean ± standard deviation) of *Ruppia spiralis* on the two sites (Salin des Pesquiers and Vieux Salins).

	Biomass	**Flowers.m^{-2}**	**Fruits.m^{-2}**
Salin des Pesquiers	65.6-267.2	1,017 ± 402	531 ± 630
Vieux Salins	97.2-557.5	1,613 ± 1,067	2,085 ± 1,639

an intermediate covering are the Ilotes (class 4, 25-50 %); Jeu de Terrasses (class 4, 25-50 %). A patch of *Zostera noltei*, Hornemann, was detected for the first time at the beginning of the inlet channel near the Nourrices 1 and 2, its covering is class 2 (red circle, Fig. 5).

Macrophytes community

The Macrophyte compartment is not very diversified; the most abundant taxa are *Chaetomorpha linum* (O.F. Müller) Kützing, *Cladophora* sp. and *Ulva rigida* C. Agardh (Table II). They are opportunistic species, which take advantage of high concentrations in nitrogen and phosphorus.

Of the 5 hydrophytic species identified by Borel (2013), only 2 were detected during the survey carried out in this work: *Ruppia spiralis* and *Lamprothamnium papulosum* (K. Wallroth) J. Groves. Several *Ruppia maritima* Linnaeus specimens were identified by Borel (2013), while no specimen was identified during this survey. In this regard, three assumptions can be made: (i) *R. maritima* meadow has regressed or disappeared due to too salinity and changes in water circulation, its optimum being 0.3-15 g/kg (Verhoeven 1980, Mannino *et al.* 2015); (ii) Borel (2013) allegedly confused *R. spiralis* with *R. maritima*; (iii) it may be that *R. maritima* was not found during our exploration. The criteria used for identification during our survey are those presented by Mannino *et al.* (2015): *R. spiralis* shows a regular and sharp apex, long leaves (15-17 cm), 4-6 carpels with a surface pollination while *R. maritima* has an irregular apex, short leaves (2-3 cm), 2 to 5 carpels and pollination occurs below the surface. These criteria were defined in 2015; they are more complete and they were not available at the time of Borel survey (2013).

A green algae species (Characeae) of heritage value, *Lamprothamium papulosum*, has been identified in a salt swamp surrounded by *Salicornia* bushes within the Salin des Pesquiers (red circle, Fig. 4). *Lamprothamium papulosum* is a species adapted to brackish waters, it develops in a few months from spring, preferably close to the edge of ponds and close to freshwater arrival. Among the Characeae species, *L. papulosum* is the most tolerant species at high salinity levels, it germinates only if the salinity is less than 20. Reproduction cannot occur when salinity is up to 40 g/kg (Réseau Suivi Lagunaire 2011).

Mean dry biomass of *Ruppia spiralis* (including epigenous and endogenous biomass) ranges from

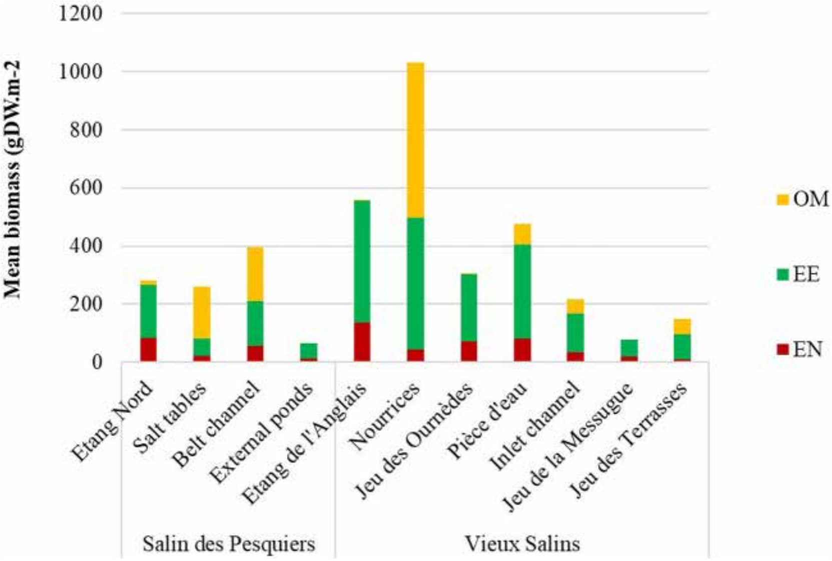


Fig. 6. – Biomass (gDW.m^{-2}) of *Ruppia spiralis* (EN = Endogenous; EE = Epigenous) and other Macrophytes (OM) measured in the different stations sampled in the saltmarshes of Hyères.

66 to 558 gDM.m^{-2} (Table III). The dry biomass is significantly higher in Vieux Salins (300 gDW.m^{-2}) than in Salin des Pesquiers (156 gDW.m^{-2}) (t test: p-value = 0.03). Endogenous and epigenous dry biomasses present different values according to sampling sites, respectively 8-136 gDW.m^{-2} and 54-454 gDW.m^{-2}. Other macrophytes biomass is also variable ranging from 0 to 535 gDW.m^{-2} (Fig. 6). The flowers and fruits density are also higher within Vieux Salins than Salin des Pesquiers (Table III).

DISCUSSION

Ruppia spiralis seagrass meadow within the saltmarshes of Hyères presents a good conservation status. Its relatively high mean biomass and overall covering correspond to the good health of this species, which find adapted conditions for its development. An increasing in the overall covering of *R. spiralis* meadow is also observed compared to Borel (2013) previous monitoring. However, *Ruppia spiralis* is a halophilous species, the only seagrass species that can thrive in such a range of salinity (*i.e.*, 0-106 g/kg). It seems important to consider that the highest abundance of a species does not indicate a climax configuration. A 100 % cover of *Ruppia spiralis* meadow could be linked to a lack of herbivorous at the scale of the saltmarsh ecosystem (*e.g.*, Anatidae). Moreover, herbivorous birds are known to contribute to the dissemination of submerged Magnoliophyta in other ponds (connected or not) thanks to the seeds contained in the feces (Clausen *et al.* 2002, Figuerola & Green 2002). In the case study of the saltmarshes of Hyères, the heterogeneous abundance of *R. spiralis* is linked to both artificialized water management and environmental condition; grazing by herbivorous birds has been observed on the field and Anatidae populations are known to be abundant in the area (Audevard 2017). The absence of other expected taxa such as *Zostera noltei* can be explained by both inadequate conditions (*e.g.*, high salinity, eutrophication) within the saltmarshes and the decreasing in the population at regional scale (Northwestern Mediterranean) (Pergent *et al*. 2014).

When comparing with other study cases of the Mediterranean Sea, mean biomass of *R. spiralis* within the saltmarshes of Hyères ranges among the highest value (Table IV). Consequently, we can consider that *R. spiralis* meadow within saltmarshes of Hyères is in good conservation status, although the associated plant communities are poorly diversified. Despite the good conditions, the ecosystem shows evidences of weakness, especially the fish compartment. The ponds that show potentially favorable features to increase the nursery role include: Étang Nord, belt channel (Salin des Pesquiers) and Nourrices, Petit Saint Nicolas, Jeu des Ournèdes, Étang de l'Anglais, Pièce d'eau, Ilotes (Vieux Salins) (Figs 2, 3), due to a widespread population of *R. spiralis*, their connection with open sea and the presence of several fish juveniles. However, the salinity and temperature measured in May 2017 show major values for Étang Nord (salinity greater than 60 g/kg). Such salinity range is not suitable for most fish species present in the saltmarshes of Hyères.

One of the objectives of the new management plan should be the maintaining of a salinity below 50 g/kg and a maximal temperature of 30 °C to limit the mortality of most juvenile species during the most critical period, between summer and fall. Even if *R. spiralis* can thrive in such euryhaline and eurytherm conditions (Verhoeven 1980), the enhancing of the nursery role and other lagoon functions of the saltmarshes of Hyères should be based on decreasing the maximum level of salinity. It is therefore important to ensure more regular water renewal for better oxygenation and to avoid too sudden variations in temperature and salinity, in order to promote the sediment mineralization and to limit opportunistic algal blooms such as *Chaetomorpha linum*, *Cladophora* sp. and *Ulva rigida*. In such ways, changes in macrophytes community could occur in a relative short period, conducing for example to the decreasing in halophilous species (*e.g.*, *Ruppia spiralis*) and the increasing in other macrophytes ecosystem engineers (*e.g.*, *Zostera noltei*, *Cymodocea nodosa* (Ucria) Ascherson, *Cystoseira barbata* (Stackhouse) C. Agardh).

CONCLUSION

The heterogeneous macrophyte communities within the saltmarshes of Hyères can be explained by the complex water circulation and the heterogeneous configura-

Table IV. – Dry mass (g.DW.m^{-2}) of *Ruppia spiralis* in other Mediterranean coastal lagoons and salt marshes.

Lagoons/saltmarshes	Biomass of *R. spiralis*	Sources
Camargue (France)	60-189	Verhoeven (1980)
Bahía del Fangar (Spain)	150-330	Calado and Duarte (2000)
Ichkeul Lagoon (Tunisia)	4-369	Casagranda and Boudouresque (2007), Shili (2008)
Smarlacca Valley (Italy)	52-411	Azzoni *et al.* (2001)
Hyères salt marshes	**66-558**	**Present work, Massinelli *et al.* (2017)**
Tancada, Ebro Delta (Spain)	61-656	Menéndez (2002)
Fra Ramon (Empordà, Spain)	95-802	Gesti *et al.* (2005)

tion of ponds and channels (depth, surface-area) of both sites (Salin des Pesquiers and Vieux Salins). *Ruppia spiralis* meadows present an overall good health status and dynamic, according to biometry descriptors and an increasing cover since the beginning of the environmental management in 2001.

These first data on the plant component of the ecosystem are the first step towards developing an ecosystem-based approach for saltmarshes of Hyères based upon a conceptual model of the socio-ecosystem (Massinelli *et al*. 2017, Astruch *et al*. 2019, 2020). This approach aims to improve the conservation and the management of the sites, taking into consideration the entire ecosystem rather than certain iconic taxa (Boudouresque *et al*. 2020), in the frame of the European Marine Strategy Framework Directive (MSFD) (Laffoley *et al*. 2004).

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FAVORING EXCHANGES BETWEEN THE SEA AND THE LAGOONS: A NECESSARY SUPPORT FOR THE RESTORATION OF THE FUNCTIONAL ROLE AS FISH NURSERY IN THE SALTMARSHES OF HYÈRES (PROVENCE, FRANCE)

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SALTMARSHES
LAGOONS
EXPERIMENTAL FISHING
NURSERY
FISH JUVENILES

ABSTRACT. – The salt exploitation of the saltmarshes of Hyères (*Salin des Pesquiers* and *Vieux Salins* sites) has led to the isolation of the lagoon from the sea to control evaporation. The question of whether or not the saltmarshes' ecological functioning is impaired with regard to the fish population is one aspect that has been studied at the ecosystem scale. During 2018-2019, 4 samplings with 4 mm-mesh size fyke nets (24 h fishing duration) confirmed that the two saltmarshes still have a relictual nursery role. If juvenile fishes were numerically dominant in the catches, adults were also present. A few euryhaline and permanent small-sized taxa (Atherinidae, Gobiidae, Syngnathidae) contributed to 90 % of the total abundance in the catches. The amphihaline migrator *Anguilla anguilla* (mainly adult stages) was the main contributor to the biomass of catches. Juveniles of commercial species such as *Dicentrarchus labrax*, *Sparus aurata*, Mugilidae and Soleidae were caught in autumn and spring when going back and forth between the lagoon and the sea. Their tolerance of a wide range of salinity enables them to benefit temporarily from good environmental conditions for their growth. However, considering the low connectivity and the high salinity of most of the ponds, most of those marine fish juveniles are probably trapped inside the lagoon. In both lagoons, the closer the sampling sites were to the open sea, the higher were the species richness and density of the juveniles, suggesting that other parts of the lagoons function as an ecological sink. The diversity of the fish assemblages of the saltmarshes of Hyères is directly dependent on the effectiveness of the connection with the open sea. Among the different actions emerging from this study, the introduction of a sluice gate between the northern pond (*Salin des Pesquiers*) and the input channel could significantly enhance exchanges and potentially provide support for fish resources at sea.

INTRODUCTION

The current management strategy for the Hyères saltmarshes (*Salin des Pesquiers* and *Vieux Salins* sites) is driven by the optimization and enhancement of the sustainability of biological functions at both sites. As a migratory stage for European birds, the Hyères saltmarshes are also an important site for their reproduction and feeding. The decades-old management of water circulation for the salt exploitation of the saltmarshes and the influence of bird protection NGOs has led to the isolation of both lagoons from the sea in order to control evaporation. Among numerous abiotic and biotic characteristics shared with other Mediterranean lagoons, the Hyères saltmarshes shelter a fish population, with numerous juveniles, and crustaceans (mainly crabs and shrimps). For the coastal marine species of the eastern part of the French Mediterranean coast, the saltmarshes of Hyères are the only lagoon sites in the Var, with the Villepey ponds near Fréjus (Provence, France) that they can frequent.

To determine, whether or not, their ecological functioning was altered, with regard to the fish population, was one of the aspects studied at the ecosystem scale. Some marine species need to spend a certain duration of their lifetime in estuarian or lagoon brackish waters, especially as juveniles, in order to grow, then, go back to the sea for adult life and reproduction. The nursery role of the Hyères saltmarshes has already been recognized by Poizat *et al.* (2004), Rosecchi *et al.* (2004), CREOCEAN (2011), but has not been studied. Only particular abiotic conditions of the water (temperature, salinity) and suitable habitats may sustain this functionality. The significant cover of a diverse aquatic vegetation, including the occurrence) of *Ruppia spiralis* Linnaeus ex Dumortier meadows (Astruch *et al.* 2019, 2020, Massinelli *et al.*

2020), has also been considered as an indicator of suitable habitats for the juvenile fish fauna in the present work.

In 2018-2019, four surveys were conducted to sample the juvenile population of teleost fishes in the saltmarshes. The periods when juveniles were present in the ponds and those when they migrated between the lagoons and the sea were identified: in May (spring), when eggs, larvae and juveniles of marine species enter (also adults), and in October (autumn), when fish juveniles and adults leave and go back to the open sea. During summer, they had enjoyed favorable conditions (warm, calm waters, light, fewer predators, see Beck *et al.*, 2001) for growing within the marshes.

In the present paper, we examine which species were present in the saltmarshes, the taxonomic richness and abundance of the teleost fishes sampled as juveniles or young adults according to their location within the saltmarshes. On this basis, we discuss how to facilitate exchanges with the sea in order to allow the life cycle of the concerned species to be achieved. The importance of the functional role of the saltmarshes as nursery for marine species in the central part of the Var is highlighted.

MATERIAL AND METHODS

The four scientific fishing campaigns were organized with the help of the manager of the Hyères saltmarshes and several local fishermen in May 2018, October 2018, March 2019 and May 2019. (Fishing has long been banned within the saltmarshes, but artisanal fishing is a traditional activity in the neighboring coastal zone). Qualitative sampling was conducted with a beach seine and quantitative sampling with fixed nets (both 4 mm mesh size), in channels and ponds where the depth allowed their deployment. At each site, sampling was carried out at 6 stations accessible from the paths (Figs 1, 2). At each station, 3 replicates of 24 h fishing duration were done on 3 consecutive days (4 replicates in March 2019). The temperature and the salinity of the water were measured at each station with a multi-parameter probe before hauling the nets. Other bi-monthly measurements taken by the managers of the sites at respectively 15 stations in the Salin des Pesquiers and 10 stations in the Vieux Salins, from 2013 to 2019, were used as reference. Species nomenclature is based on International Commission on Zoological Nomenclature and on International Code of Nomenclature for algae, fungi and plants.

Sampling strategy: The selection of the ponds to be sampled was made according to several criteria, such as: (i) the presence of *Ruppia spiralis* meadows and associated macroalgae community based on previous inventory and mapping (Massinelli *et al.* 2020) (50 ha meadows/550 ha in the Pesquiers lagoon; 14 ha meadows/330 ha in the Vieux Salins); (ii) the water circulation; (iii) the depth of channels and ponds; (iv) our ability to deploy fishing gear (depth, access, ground resistance); and (v) scientific knowledge (previous works on the Hyères saltmarshes by Poizat *et al.* (2004) and CREOCEAN (2011), mainly dedicated to adult fishes but also mentioning juveniles).

Fig. 1. – Localization of sampling stations in the Pesquiers lagoon according to the presence of *Ruppia spiralis* (% of covering by *Ruppia* is divided into 5 classes), depth and circulation of water (fixed nets sampling stations are in pink and beach seine sampling stations are in green).

The water circulation, in the Salin des Pesquiers, was mostly driven by gravity and led towards the irrigation of the northern pond, which was the deepest basin of the lagoon. The seawater entered at La Capte (station 1 in the middle of the western part, Fig. 1) and in the eastern part of the site by infiltration or driven by winter storms. For decades, the exchange of water between the sea and the lagoon had been made difficult by the requirements of the salt industry, by the constraints linked to the conservation of the historic salt works and to the conservation of nesting areas for birds. That was also the case in the Vieux Salins (Fig. 2).

Data analysis: The occurrence of species or families was calculated as catches per unit effort (CPUE) expressed in abundances per season per site and for each season at each site. Nonparametric comparison tests (U Test and Kruskal Wallis test) were used to compare the daily catches between sites (2), stations (6 at each site) and seasons (4 seasonal data collections). The variations were declared statistically significant when p was < 0.1). A principal components analysis (PCA) (Benzecri 1976a, b *in* Scherrer 2009) was performed on the basis of the daily log (CPUE+1) in abundance per taxa in order to evidence patterns of the juvenile assemblage according to sites, stations, seasons, and stations at each site.

RESULTS

Results of catches

In the catches, 39 taxa were recorded, among them 30 fish taxa and invertebrates: Carcinidae (crabs), Palaemonidae and Penaeidae (several shrimps), Sepiolidae (cephalopods), but fishes were dominant. The abundances in fish catches showed 98 % juveniles and adults of 'small size' species (*i.e.*, with a size of less than 15 cm) (Table I). The faunistic list encompasses typical taxa from the lagoon environment with numerous individuals, and other species such as *Z. ophiocephalus*, *Mullus*, *Engraulis,* with individuals occasionally present. The number of migratory species originating from peripheral environments (*i.e.*, brackish or marine waters) was higher than the number of sedentary species, underlining the importance of exchanges with the marine environment. The number of marine and freshwater species remained inferior to the number of species present in the nearby marine and freshwater environments.

Two species stand out for their abundance or their contribution to the caught biomass*: Atherina boyeri* (*Atherina* sp. *sensu stricto* because the number of *Atherina* was so high that each individual could not be identified to species level. But we are sure that *Atherina boyeri* was largely dominant in our catches) was dominant in abundance (CPUE catches per unit effort), its occurrence per site and per season was higher than 90 %. *Anguilla anguilla* (diadromous species protected by a national management plan) occurred at all stations, all seasons and dominated biomass in the catches. Caught individuals were mainly adults, up to 1 m in maximum total length (TL). Other species were caught as adult stage, in large numbers, or showed high occurrences in the catches. They belonged to euryhaline species such as *Sparus aurata*, several species of Mugilidae (*Chelon auratus, C. ramada and* Mugilidae) and Soleidae. However, the abundance of 98 % of the catches was composed of juveniles and adults of species smaller than 15 cm TL.

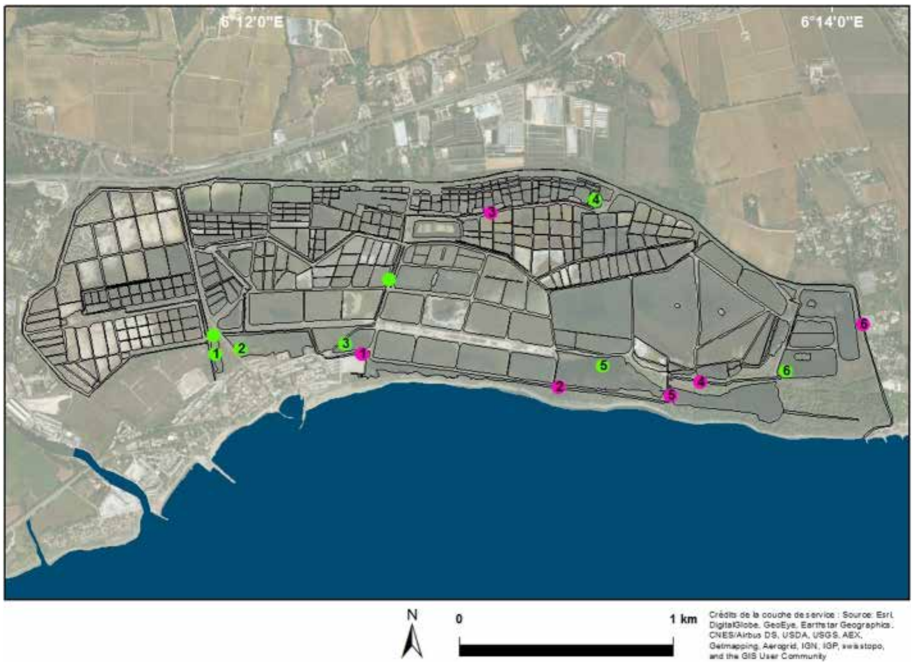

Fig. 2. – Sampling stations with fixed nets (pink) and with beach seine (green) in the Vieux Salins saltmarsh.

Table I. – Frequency of occurrence in the catches per 24 h sampling per taxon: species and families. Reproduction habitat of each taxon: M: marine water, F: fresh water, B: brackish water. Category of taxon according to its reproduction habitat. Kind of lagoon taxon. (Species nomenclature is based on International Commission on Zoological Nomenclature).

Species	Family	Reprod habitat	Category	Lagoon species	Occurrence per site		Occurrence per season				Occurrence per station											
					Pesquiers	Vieux Salins	May-18	Oct-18	Mar-19	May-19	Pesquiers						Vieux Salins					
											P1	P2	P3	P4	P5	P6	V1	V2	V3	V4	V5	V6
Anguilla anguilla	Anguillidae	M	Marine migrant	Characteristic	8%	20%	30%	3%	0%	14%	0%	0%	44%	0%	0%	10%	0%	11%	22%	44%	11%	33%
Atherina sp.	Atherinidae	F/B/M	Sedentary or marine migrant or freshwater migrant	Characteristic	90%	98%	94%	94%	100%	92%	85%	89%	67%	100%	100%	100%	100%	100%	100%	100%	100%	89%
Buglossidium luteum	Soleidé	M	Marine migrant	Complementary	3%	7%	0%	0%	0%	17%	0%	11%	0%	0%	0%	10%	22%	0%	0%	0%	0%	22%
Monochirus hispidus	Soleidé	M	Marine migrant	Complementary	13%	0%	0%	0%	67%	0%	31%	0%	0%	0%	31%	0%	0%	0%	0%	0%	0%	0%
Solea solea	Soleidé	M	Marine migrant	Characteristic	2%	2%	3%	0%	0%	3%	0%	11%	0%	0%	0%	0%	11%	0%	0%	0%	0%	0%
Chelon auratus	Mugilidae	M	Marine migrant	Characteristic	41%	15%	3%	33%	83%	31%	31%	56%	56%	33%	38%	40%	0%	22%	33%	33%	0%	0%
Chelon labrosus	Mugilidae	M	Marine migrant	Characteristic	8%	2%	6%	6%	8%	3%	8%	0%	11%	22%	8%	0%	11%	0%	0%	0%	0%	0%
Chelon ramada	Mugilidae	M	Marine migrant	Characteristic	22%	4%	6%	11%	58%	8%	8%	11%	56%	11%	31%	20%	0%	11%	0%	0%	0%	11%
Chelon saliens	Mugilidae	M	Marine migrant	Characteristic	8%	2%	0%	14%	0%	3%	0%	22%	33%	0%	0%	0%	0%	0%	0%	0%	0%	11%
Mugilidae indet.	Mugilidae	M	Marine migrant	Characteristic	32%	31%	64%	8%	33%	25%	46%	11%	33%	44%	38%	10%	44%	22%	11%	33%	33%	44%
Dicentrarchus labrax	Moronidae	M	Marine migrant	Characteristic	6%	11%	12%	0%	0%	17%	0%	11%	0%	33%	0%	0%	0%	22%	0%	0%	0%	44%
Dicentrarchus punctatus	Sparidae	M	Marine migrant	Rare	2%	0%	0%	3%	0%	0%	0%	0%	0%	0%	8%	0%	0%	0%	0%	0%	0%	0%
Diplodus puntazzo	Sparidae	M	Marine migrant	Characteristic	5%	2%	3%	0%	25%	0%	15%	0%	0%	0%	8%	0%	11%	0%	0%	0%	0%	0%
Diplodus sargus	Sparidae	M	Marine migrant	Characteristic	0%	4%	0%	0%	0%	6%	0%	0%	0%	0%	0%	0%	0%	11%	0%	0%	0%	11%
Diplodus vulgaris	Sparidae	M	Marine migrant	Characteristic	8%	15%	6%	3%	42%	14%	15%	0%	0%	0%	23%	0%	11%	11%	0%	11%	22%	33%
Sarpa salpa	Sparidae	M	Marine migrant	Characteristic	2%	9%	0%	0%	8%	14%	0%	0%	0%	0%	8%	0%	0%	22%	0%	0%	0%	33%
Sparus aurata	Sparidae	M	Marine migrant	Characteristic	32%	20%	36%	17%	25%	28%	8%	56%	33%	56%	31%	20%	11%	33%	33%	0%	0%	44%
Gambusia holbrooki	Poecillidae	F	Sedentary	Complementary	21%	2%	12%	17%	8%	8%	0%	0%	100%	22%	15%	0%	11%	0%	0%	0%	0%	0%
Lepomis gibbosus	Centrarchidae	F	Freshwater migrant	Complementary	0%	2%	3%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	11%
Gobiidae indet.	Gobiidae	B/M	Sedentary	Characteristic	21%	9%	30%	11%	17%	6%	8%	11%	33%	56%	15%	10%	0%	11%	0%	22%	11%	11%
Gobius cruentatus	Gobiidae	B/M	Sedentary	Characteristic	0%	15%	24%	0%	0%	0%	0%	0%	0%	0%	0%	0%	11%	33%	0%	22%	22%	0%
Gobius niger	Gobiidae	B/M	Sedentary	Characteristic	0%	2%	0%	0%	0%	3%	0%	0%	0%	0%	0%	0%	11%	0%	0%	0%	0%	0%
Gobius paganellus	Gobiidae	B/M	Sedentary	Characteristic	0%	2%	0%	3%	0%	0%	0%	0%	0%	0%	0%	0%	11%	0%	0%	0%	0%	0%
Pomatoschistus marmoratus	Gobiidae	B/M	Sedentary	Characteristic	6%	2%	0%	0%	33%	3%	0%	0%	0%	0%	31%	0%	11%	0%	0%	0%	0%	0%
Pomatoschistus microps	Gobiidae	B/M	Sedentary	Characteristic	11%	4%	0%	3%	25%	14%	8%	11%	11%	11%	15%	10%	0%	11%	0%	0%	11%	0%
Pomatoschistus sp.	Gobiidae	B/M	Sedentary	Characteristic	19%	50%	33%	3%	25%	67%	8%	22%	0%	11%	31%	40%	33%	44%	44%	56%	67%	56%
Zosterisessor ophiocephalus	Gobiidae	B/M	Sedentary	Complementary	5%	0%	0%	0%	25%	0%	8%	0%	0%	0%	15%	0%	0%	0%	0%	0%	0%	0%
Engraulis encrasicolus	Engraulidae	B/M	Marine migrant	Complementary	2%	0%	0%	3%	0%	0%	8%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Salaria pavo	Blenniidae	B/M	Sedentary	Characteristic	0%	13%	6%	0%	0%	14%	0%	0%	0%	0%	0%	0%	33%	11%	33%	0%	0%	0%
Syngnathus abaster	Syngnathidae	B/M	Sedentary	Characteristic	11%	39%	30%	6%	25%	36%	8%	22%	0%	11%	23%	0%	22%	44%	33%	56%	56%	22%

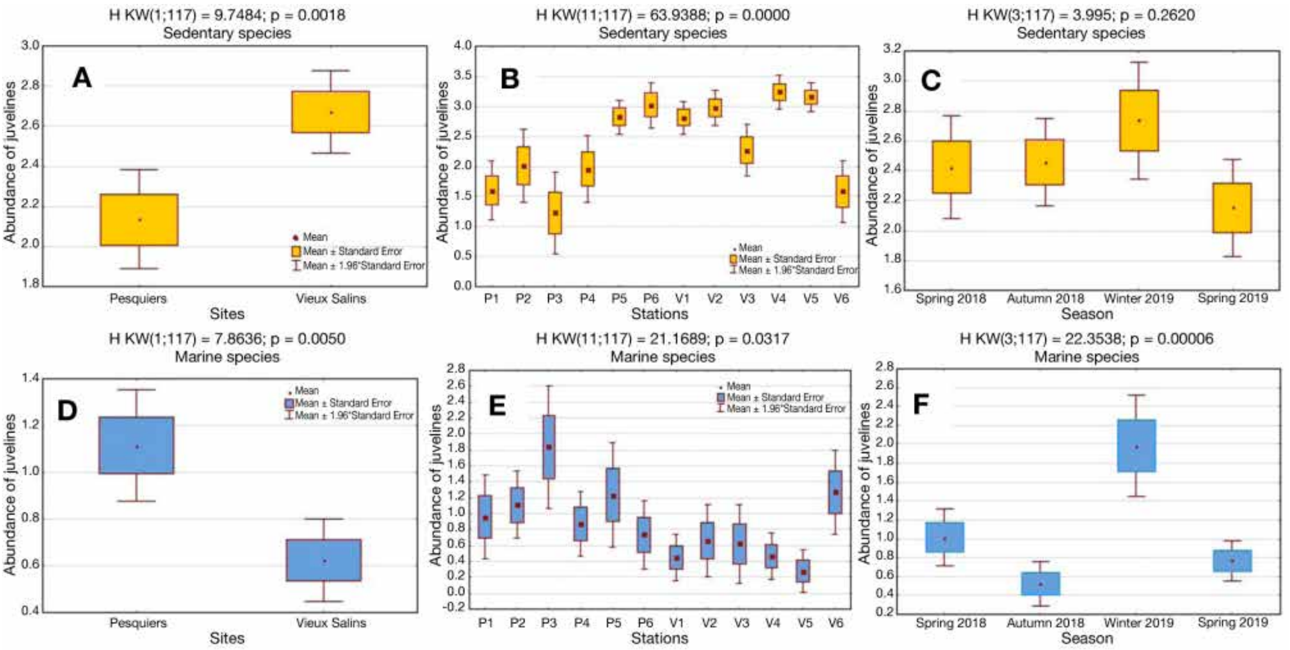

Fig. 3. – Abundance of fish juveniles in the catches according to sites, to stations and to seasons. Sedentary species are shown in the upper part, in yellow (**A**, **C**, **E**); marine migrants are shown in the lower part, in blue (**B**, **D**, **F**). Pesquiers stations: P1 to P5; Vieux Salins stations: V1 to V6.

The different taxa could be grouped in 3 categories according to a combination of their reproduction habitat, their behavior and environmental preferences: the sedentary permanent species, the marine migrants and the freshwater species.

Sedentary, permanent species: euryhaline small-sized taxa contributed to 90 % of the total abundance in the catches. *Atherina* sp., *Pomatoschistus* spp. and other Gobiidae, Syngnathus (*Syngnathus abaster*) as juveniles or adults were always present in the catches and at all sites. They completed their entire life cycle in the saltmarshes. The juveniles of sedentary species were more abundant in the Vieux Salins than in the Salin des Pesquiers (Fig. 3A).

Freshwater species: *Gambusia holbrooki*, *Lepomis gibbosus* were only present at stations receiving freshwater (station 3 in the Salin des Pesquiers and station 6 in the Vieux Salins lagoon).

Marine migrant species: mainly euryhaline species as *Dicentrarchus labrax*, Mugilidae, Sparidae as *Diplodus* spp., *Sparus aurata* and Soleidae juveniles or adults were especially abundant in May and October, when the majority of individuals came into or left the saltmarshes. Those individuals contributed mainly to the biomass in catches and they needed to return to sea for spawning. That is why the CPUE were higher at stations close to the open sea at both sites. The juveniles of the marine species were caught during March and May, when they achieved their trophic migration and settlement in the salt marshes. Several species of mullet (*Mugil* and *Chelon* spp.) were caught, in the spring and in fall, when they entered or left the lagoons. These species are considered to be tolerant to salinity variations (Kara & Quignard 2018) and they were particularly frequent in the catches at different sizes. The juveniles of marine species were more abundant in the Salin des Pesquiers than in the Vieux Salins (Fig. 3B).

The significant differences in log (CPUE+1) abundances among sites, stations and seasons are reported in Table II for the different taxa. The total abundance was significantly different among sites ($p = 0.013$), among stations ($p = 0.017$) and among seasons ($p = 0.08$). When all taxa were considered together, the seasonal variations of the abundance were less marked than the spatial variations.

The specific composition highlighted some differences in the fish catches among sites, in that 12 taxa showed significant differences. At the scale of the stations, differences were less obvious, with 8 taxa showing significant differences among stations. The inter-station spatial variations in the Salin des Pesquiers concerned 8 taxa, which is more than in the Vieux Salins (5 taxa). Many more taxa showed variations between seasons (16), demonstrating seasonal patterns in the coastal species assemblage using the lagoon. The marine migrant species taken together displayed a highly significant seasonal variation ($p < 0.000$) as well as *Anguilla anguilla* ($p = 0.004$). Sedentary species and freshwater species did not show this pattern ($p = 0.262$ and $p = 0.679$, respectively; Table III).

Finally, 18 taxa showed non-significant inter-site variations, 22 taxa non-significant inter-station variations, 14 taxa non-significant inter-season variations, but the

Table II. – Summary of the ANOVA comparisons (non-parametric U test and Kruskal Wallis test) of log(CPUE +1) abundance of juveniles between sites, stations (all), seasons (all) and stations at each site. Reproduction habitat of each taxon: M: marine water, F: fresh water, B: brackish water. (Species nomenclature is based on International Commission on Zoological Nomenclature).

Total abundance			**SITES**		**STATIONS**		**SEASONS**		**STATIONS**			
			2 sites		**All seasons, both sites, H(5, n = 117)**		**All stations, both sites, H(3, n = 117)**		**Pesquiers H(5, n = 63)**		**Vieux Salins H(5, n = 54)**	
Species	**Family**	**Reprod habitat**	**p**		**p**		**p**		**p**		**p**	
			0.013	**	**0.017**	**	**0.080**	*	**0.002**	***	**0.000**	***
Anguilla anguilla	Anguillidae	M	0.049	**	0.014	**	0.004	***	0.001	***	0.214	
Atherina sp.	Atherinidae	F/B/M	0.002	***	0.000	***	0.209		0.000	***	0.000	***
Buglossidium luteum	Soleidé	M	0.306		0.144		0.002	***	0.450		0.131	
Monochirus hispidus	Soleidé	M	0.007	***	0.014	**	0.000	***	0.024	**	1.000	
Solea solea	Soleidé	M	0.922		0.557		0.703		0.306		0.415	
Chelon auratus	Mugilidae	M	0.001	***	0.507		0.000	***	0.735		0.074	*
Chelon labrosus	Mugilidae	M	0.143		0.532		0.863		0.474		0.415	
Chelon ramada	Mugilidae	M	0.003	***	0.319		0.000	***	0.153		0.538	
Chelon saliens	Mugilidae	M	0.141		0.087	*	0.035	**	0.014	**	0.415	
Mugilidae	Mugilidae	M	0.788		0.372		0.000	***	0.225		0.475	
Dicentrarchus labrax	Moronidae	M	0.324		0.025		0.049	**	0.012	**	0.006	***
Dicentrarchus punctatus	Sparidae	M	0.364		0.504		0.522		0.571		1.000	
Diplodus puntazzo	Sparidae	M	0.400		0.084	*	0.000	***	0.374		0.415	
Diplodus sargus	Sparidae	M	0.128		0.497		0.208		1.000		0.538	
Diplodus vulgaris	Sparidae	M	0.284		0.191		0.001	***	0.139		0.435	
Sarpa salpa	Sparidae	M	0.060	*	0.106		0.022	**	0.571		0.038	**
Sparus aurata	Sparidae	M	0.273		0.065	*	0.272		0.082	*	0.090	*
Gambusia holbrooki	Poecillidae	F	0.002	***	0.000	***	0.688		0.000	***	0.415	
Lepomis gibbosus	Centrarchidae	F	0.288		0.396		0.467		1.000		0.415	
Gobiidae	Gobiidae	B/M	0.071	*	0.350		0.023	**	0.030	**	0.582	
Gobius cruentatus	Gobiidae	B/M	0.002	***	0.255		0.000	***	1.000		0.230	
Gobius niger	Gobiidae	B/M	0.288		0.504		0.522		1.000		0.415	
Gobius paganellus	Gobiidae	B/M	0.288		0.504		0.522		1.000		0.415	
Pomatoschistus marmoratus	Gobiidae	B/M	0.224		0.017	**	0.000	***	0.006	***	0.415	
Pomatoschistus microps	Gobiidae	B/M	0.141		0.801		0.011	**	0.986		0.538	
Pomatoschistus sp.	Gobiidae	B/M	0.000	***	0.250		0.000	***	0.221		0.217	
Zosterissessor ophiocephalus	Gobiidae	B/M	0.108		0.306		0.000	***	0.380		1.000	
Engraulis encrasicolus	Engraulidae	B/M	0.364		0.504		0.522		0.571		1.000	
Salaria pavo	Blenniidae	B/M	0.003	***	0.082	*	0.067	*	1.000		0.058	
Syngnathus abaster	Syngnathidae	B/M	0.000	***	0.132		0.013		0.343		0.263	

majority of those taxa were rare or presented a low occurrence per station.

The water temperature was recorded at the sampling sites at between 11.4 and 17.2 °C in March, 19.6 and 27.3 °C in May and 19.3 and 24.9 °C in October. Those relatively high temperatures are suited to juveniles' growth. The salinity in spring was between 26.7 and 28.7 g.kg^{-1}, and in autumn it was between 21.1 and 27.4 g.kg^{-1}, which are mean salinities lower than that of the Mediterranean Sea (about 38 g.kg^{-1}). The salinity measured in various other ponds could reach very high levels. The mean salinity of water calculated on a data set of 2 measurements per month in, respectively, 15 and 10 stations, from 2013 to 2019, was 74.8 g.l^{-1} in the Salin des Pesquiers and 55.5 g.l^{-1} in the Vieux Salins. The survival conditions for juveniles in shallow and over-salty ponds are undoubtedly limited for variable durations despite the high tolerance of the lagoon species for elevated salinities.

We performed a PCA analysis on the abundance of juvenile fishes in the catches. The descriptors of the sampling stations were salinity and temperature of the water at each sampling operation. The first two axes explained 23.21 % and 19.09 % of the variability respectively. Maps based on the first two axes (Fig. 4) showed differences between stations. The species mainly explaining variabil-

Table III. – Summary of the ANOVA comparisons (non-parametric U test and Kruskal Wallis test) of log(CPUE +1) abundance of juveniles between sites, stations (all), seasons (all) and stations at each site for families and categories according to the life traits. Reproduction habitat of each taxon: M: marine water, F: fresh water, B: brackish water. (Species nomenclature is based on International Commission on Zoological Nomenclature).

Taxon or category	**Reproduction habitat**	**SITES**		**STATIONS**		**SEASONS**		**STATIONS**			
		All seasons		**All seasons, both sites, h(5, n = 117)**		**All stations, both sites, h(3, n = 117)**		**Pesquiers H(5, n = 63)**		**Vieux Salins H(5, n = 54)**	
Family		**p**		**p**		**p**		**p**		**p**	
Anguillidae	M	0.049	**	0.014	**	0.004	***	0.001	***	0.214	
Atherinidae	F/B/M	0.002	***	0.000	***	0.209		0.000	***	0.000	***
Blenniidae	B/M	0.003	***	0.082	*	0.067	*	1.000		0.058	*
Centrarchidae	F	0.288		0.396		0.467		1.000		0.415	
Engraulidae	B/M	0.364		0.504		0.522		0.571		1.000	
Gobiidae	B/M	0.057	*	0.000	***	0.000	***	0.005	***	0.053	*
Moronidae	M	0.496		0.064	*	0.130		0.046	**	0.006	***
Mugilidae	M	0.001	***	0.727		0.000	***	0.128		0.653	
Poeciliidae	F	0.002	***	0.000	***	0.688		0.000	***	0.415	
Soleidae	M	0.152		0.031	**	0.000	***	0.153		0.042	**
Sparidae	M	0.846		0.138		0.057	*	0.470		0.033	**
Syngnathidae	B/M	0.000	***	0.132		0.013	**	0.343		0.263	
Marine migrants		0.005	**	0.032	**	0.000	***				
Anguilla anguilla		0.049	**	0.006	**	0.004	***				
Sedentary species		0.002	***	0.000	***	0.262	ns				
Freshwater species		0.007	**	0.000	***	0.679	ns				

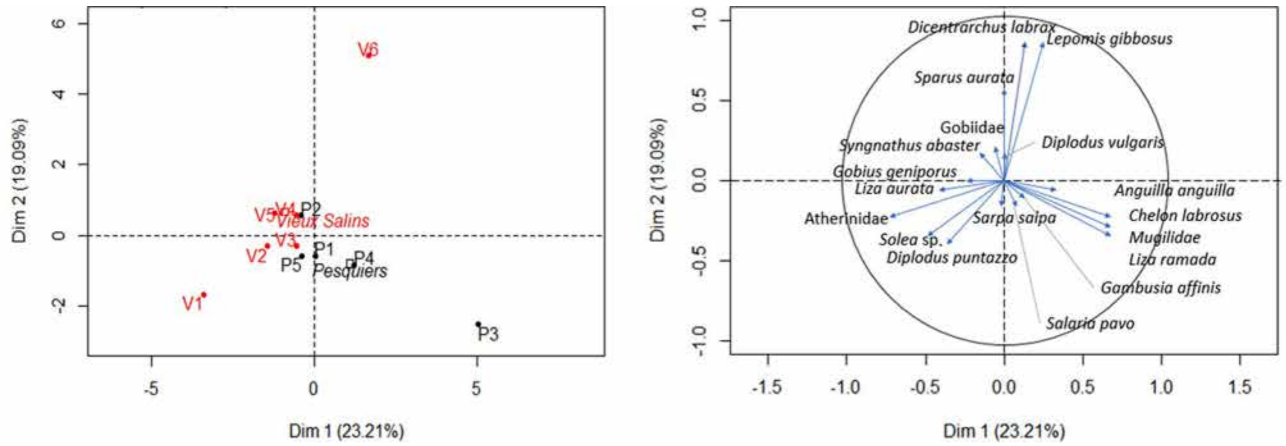

Fig. 4. – Principal Components Analysis of mean CPUE abundance per station (May, 2018). Pesquiers stations: P1 to P5; Vieux Salins stations: V1 to V6. The stations and taxa as supplementary vectors maps are defined by the first two factorial axes.

ity among stations were the seasonal marine migrants: the different species of Mugilidae, *S. aurata* and *S. solea* at both sites.

Stations P3 and V6 differed from the others because they received some freshwater provided by runoff, and this explained the presence of *Lepomis gibbosus* and the relative abundance of *Gambusia holbrooki*, which are freshwater species. Station V1 was directly in connection with the sea and marine species were dominant and abundant there. *Anguilla anguilla* and the different taxa of Mugilidae were abundant in the Salin des Pesquiers. *Solea solea*, *Diplodus puntazzo* and *Chelon auratus* abundance characterized the Vieux Salins catches. The taxa belonging to the sedentary taxa such as Gobiidae, *Syngnathus abaster*, and also *A. anguilla* were positioned near the first two axes of the CPA because they were largely spread among the monitoring stations showing low spatial and low temporal variations.

During autumn, the Mugilidae (*Chelon auratus*, *C. labrosus* and *C. saliens*) were particularly abundant at the P2, V3 and V2, V4, V6 stations. In spring, the arrival of *Sparus aurata* and *Solea solea* structured the juvenile assemblage at P2, V2 and P6. *Chelon ramada* is a marine migrant species, which contributed to the P3, P4, and P5 assemblage in spring. All other species have shorter vectors that stay closer to the centre of the axis, such as Gobiidae, showing they are more regularly present in the saltmarshes and less discriminating.

DISCUSSION

In the saltmarshes of Hyères, the broad scale distribution pattern of fish juveniles was influenced by the circulation of water between ponds and the seawater inlet. The composition of the fish juveniles' assemblages at each station was mainly related to the water characteristics. The management team at both sites controlled the opening and closing of the traps according to the basin water level and to the inputs and outputs according to basin water level, rainfall, evaporation and barometric tides. The stations situated closed to the open sea or with effective connections presented higher abundances of juveniles and a higher taxonomic richness due to marine migrants. The water circulation was also observed to induce direct effects on the structure of both lagoon fish assemblages. The frequent changes in abiotic conditions induced high spatial and temporal variability in the survey variables (temperature and salinity of water and taxonomic richness, abundance of juveniles) at site and at station scales. Even behavior effects were observed such as fish swimming upstream when the seawater was entering the sites.

At the sampling stations in the Hyères saltmarshes, the abundance of marine migrants in the catches served as markers of seawater entry in spring. Their abundance in autumn in the stations closer to the sea showed their need to return to the sea and their departure when the water conditions in the saltmarsh were changing. The other sites, not sampled during this study, seemed to be unsuitable for survival during the warm season because of the high salinity level and eutrophication of the water.

According to our sampling, the fish populations of the two lagoons were not very different in composition. The Vieux Salins, localized along the seashore, had a lower salinity and a high proportion of marine species, but they were less abundant than in the Salin des Pesquiers. They presented all the features of other Mediterranean lagoons with sedentary species whose entire life cycle is completed within the lagoons, and species coming in from the sea, entering at different development stages (eggs, larvae, juveniles) (Bouchoucha 2010, Kara & Quignard 2018). The marine species (*Dicentrarchus labrax*; *Sparus aurata*, *Chelon* spp.) benefited from this favorable environment for their growth from late winter to the end of autumn. The presence of species that were rare in lagoons and abundant in the sea, of which only the year class 0+ (*i.e.*, individuals less than 1 year old) was present in the lagoons, showed the temporary carrying capacity of these two lagoon sites with respect to the peripheral environments. Adults were also present and they might use the saltmarshes for feeding and reproduction. Some freshwater species such as *Gambusia holbrooki* and *Lepomis gibbosus* contributed to the specific richness at stations where salinity conditions were particularly low (May).

The study has confirmed the nursery role of the Hyères saltmarshes, but the water salinity and the circulation conditions were shown to jeopardize fish survival in most of the saltmarsh ponds. Hydraulic management of the sites is complex and obviously governed by prior management constraints unsuited to the seasonal rhythms of the migrations of fishes between the lagoon and the sea. As compared to the seasonal functioning of other unconstrained estuarine systems (Le Diréach *et al.* 2010, 2013, Kara & Quignard 2018), these saltmarshes might produce disappointing results for species whose life scenario depends on those migrations.

The present work has provided an update on knowledge regarding the fish compartment with the sampling of juveniles and regarding the current functionality of the aquatic compartments of both lagoons. That could prove useful as a basis for a more functionally oriented management system with an ecosystem-based approach (Astruch *et al.* 2020). The work sessions organized with managers, fishermen and scientists have favored the sharing of knowledge and points of view. The increase in awareness of the interest of the aquatic compartment and its populations (fishes but also invertebrates: shrimps and crabs, which strongly contribute to the diet of waterfowl) was a goal of this work.

The ecosystem-based approach applied here has undoubtedly offered a clearer vision of the functioning of the sites both for managers and scientists, including the constraints of both sites and the vicinity of the city of Hyères (runoff, pollution from the watershed, management of the mosquito population, landscape conservation, mass tourism, education). The various possible solutions to facilitate the circulation of water and exchanges with the sea were discussed, among them: (i) the cleaning and dredging of the belt canal, (ii) permanent or temporary gravity-based circulation of the northern pond in the Salin des Pesquiers (*e.g.*, decrease in mean salinity and eutrophication, reduction of pumping costs) and (iii) the use of a portion of channel as a lock.

In addition to the management propositions, specific actions have resulted from the SALSA program. For the Salin des Pesquiers, those actions were: (i) optimizing the circulation between the northern pond and the belt channel; (ii) optimizing the gravity-based water circulation permanently or temporarily in the northern pond; (iii) building a sluice gate between the northern pond (Salin des Pesquiers) and the input channel; (iv) strengthening the seawalls and levees of various ponds (northern pond especially) to sustain a rise in their water level (about 30 cm rise). In the Vieux Salins, a closed channel was even linked to the Étang de l'Anglais in order to increase the water circulation. Those actions are included in the new management plan for the Hyères saltmarshes (BRLi 2018) but some of them still need funding and feasibility studies.

Before the study and in the previous management plan, the water circulation system was mostly inherited from mode of operation when the salt was still exploited at both sites (before 1995) and bird conservation a priority. The present study has provided new scientific knowledge for management purposes and greater attention will now be paid to the aquatic compartments in the new management plan. The main benefits of the program derive from the collaboration built up between the different stakeholders working on field data collection: the fishermen, the managers and the scientists. The time spent in discussions and in trying to better understand the different points of view in relation to the aquatic resources, the management priorities for the saltmarshes and the conservation of the different compartments was the price to pay for developing a new management approach.

The Salin des Pesquiers has suffered from breaches in their levees close to the sea and to seawater invasions over the centuries. Before the development of the salt industry, the site was a lagoon (named in French: *Étang des Pesquiers*) and an important fishing ground for local fishermen (Faget *et al.* 2021). At that time, fishes used the lagoon to accomplish their life cycle. Since the Pesquiers lagoon was transformed with the development of the salt industry, managers have devoted considerable effort to the reinforcement of the seawalls and levees and the control of the water level. The exchanges with the sea remain limited and the survival of fishes highly compromised by the high salinity conditions of the water. The restoration of the functionality of the Hyères saltmarshes will now depend on a compromise between the priorities of bird and heritage landscape conservation and a kind of natural restoration that will undoubtedly be given added urgency by Global change and its impact along the Mediterranean coast (*e.g.*, sea level rise, warming, invasive species, etc.).

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IMPROVING THE MANAGEMENT OF THE SALTMARSHES OF HYÈRES (PROVENCE, FRANCE) USING AN ECOSYSTEM-BASED APPROACH

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SALTMARSHES
ECOSYSTEM-BASED APPROACH
MANAGEMENT
MEDITERRANEAN

ABSTRACT. – The management of saltmarshes, the complex ecosystem constituted by coastal wetlands and lagoons (SME – Saltmarsh Ecosystem), is often centered on the so-called 'heritage species' (rare, threatened and charismatic species). This 'species-centered' or 'taxon-centered' approach, a legacy from the 20th century, is fully understandable in areas where definitely and critically endangered species occur. However, an ecosystem-based approach, of course including species and higher taxa, but based upon the whole functioning of the ecosystem, from primary producers to *e.g.*, detritus feeders and top predators, would present advantages of paramount importance. The ecosystem-based approach (EBA) involves the management of the interactions between functional compartments, and the search for equilibrium according to the supposed baseline, ecosystem services and management goals. This approach offers a basis for considering the current global change. A conceptual model of the whole saltmarsh ecosystem, including lagoon bottom, water body and terrestrial adjacent habitats, has been established. Taking into account the high diversity of saltmarsh environments physically and biologically as well, this model is a frame that should be adapted to each case study. Here, the authors focus on two case studies in Provence (Vieux Salins and Salin des Pesquiers), northwestern Mediterranean. The weaknesses of the traditional species-centered approach and the advantages of an ecosystem-based approach are highlighted through novel applications.

INTRODUCTION

Saltmarshes are a worldwide ecosystem, the origin and structure of which vary according to climate (boreal, temperate, sub-tropical, tropical) (Danin 1981, Ayyad & El-Ghareeb 1982, Peinado *et al.* 1995, Asri & Ghorbanli 1997, Costa *et al.* 2009, De Wit *et al.* 2019), precipitation rate and variability over years (Callaway & Sabraw 1994), and tide amplitude (from microtidal, *e.g.*, Mediterranean Sea, to megatidal, *e.g.*, Atlantic Ocean) (Long & Mason 1983, Adam 1990, Vernberg 1993, Mc Owen *et al.* 2017). A high degree of similarity can be observed worldwide between saltmarsh ecosystems (Costa *et al.* 2009, Peinado *et al.* 2009). In temperate microtidal areas, saltmarshes are a complex combination of both wetland and coastal lagoon or estuarian system with evapotranspiration deficit (Boutière 1974, Mesleard *et al.* 1995, Duarte *et al.* 2002); hereafter, we will name the complex ecosystem they constitute SME (saltmarsh ecosystem). They play an important role at the interface of continental-terrestrial ecosystems and coastal marine waters, and provide a variety of ecosystem services (Nordlund *et al.* 2016, Himes-Cornell *et al.* 2018, Newton *et al.* 2018, Sy *et al.* 2018, O'Higgins *et al.* 2019): (i) breeding and nesting area for a large number of migratory waterfowl and wintering bird species (Furness & Greenwood 1993, Del Hoyo *et al.* 1996, Birdlife International 2004, Isenmann 2004, Thorup 2006), (ii) heritage (historical, landscape) for exploited or non-exploited areas (Borel 1996, Hérault 2010), (iii) sediment retention and trapping (Li & Yang 2000, Wood & Hine 2007), (iv) protection against erosion and submersion (Sharma *et al.* 2016, Lo *et al.* 2017) of urbanized areas, receptacle of the watersheds (King & Lester 1995), (v) cleaning of continental pollutants and contaminants (Fisher & Acreman 2004, Bromberg-Gedan *et al.* 2009, Calvo-Cubero 2014, Calvo-Cubero *et al.* 2014), (vi) nursery habitat for marine fish species (Beck *et al.* 2001) of high economic and fishery interest (Minello *et al.* 2003) such as the seabass *Dicentrarchus labrax* (Linnaeus, 1758) and the gilthead bream *Sparus aurata*

Linnaeus, 1758, (vii) essential habitat for adult fish such as the European eel *Anguilla anguilla* (Linnaeus, 1758) (Kara & Quignard 2018a, 2019a), (viii) primary production by Magnoliophyta, macroalgae and phytoplankton (Quintana *et al*. 1998, Curcó *et al*. 2002, Menéndez 2002, Quintana & Moreno-Amich 2002), (ix) grazing areas for cattle (Duncan & D'Herbes 1982, Andresen *et al*. 1990), (x) sites for extensive and intensive aquaculture (Lumare 1983, Boudouresque *et al.* 2020a), (xi) and an important carbon sink (Sousa *et al*. 2017).

Saltmarshes are often ecosystems that have been more or less artificialized and impacted for centuries (Bertness *et al*. 2002, Bromberg-Gedan *et al*. 2009) and are exposed to numerous threats (Ganju *et al*. 2017): (i) Habitat destruction by urbanization, industry (including salt production) and agriculture (Andresen *et al*. 1990, Mesleard *et al.* 1995, Tourenq *et al*. 2001); (ii) eutrophication (Quintana & Moreno-Amich 2002, López-Flores *et al*. 2006, Moseman-Valtierra *et al*. 2016) including increased herbivory issues (Holdredge *et al.* 2008, Alberti *et al*. 2011); (iii) global change, including invasive species and community shift (Castillo *et al*. 2000, Occhipinti-Ambrogi 2000, Occhipinti-Ambrogi & Savini 2003, Boudouresque *et al*. 2005, Bianchi 2007, Occhipinti-Ambrogi 2007, Boudouresque *et al*. 2011, Boudouresque & Verlaque 2012, Bianchi *et al*. 2013, Lascève 2014, Al Hassan *et al*. 2016, Boudouresque *et al*. 2017, Curado *et al*. 2018), the rise of sea level (Laborel *et al*. 1994, Moseman-Valtierra *et al*. 2016, Valiela *et al*. 2018) with the decline of plant diversity induced by stronger competition (Noto & Shurin 2017), warming (Boyer *et al.* 2012, Monllor *et al*. 2018), inducing higher evaporation and consequently higher mean salinity (Mollema *et al*. 2013), (iv) use of insecticides against harmful mosquitoes with potential high impact on non-target fauna (Poulin 2012), (v) erosion (Lo *et al*. 2017), (vi) contamination from human activities in the watershed (Usero *et al.* 2002).

For those reasons, saltmarshes are ecosystems of high concern and high heritage value; many international agreements aim at their protection and sustainable management (*e.g.*, RAMSAR, Habitat Directive Natura 2000, 92/43/EEC) (Matthews 1993, Evans 2012). However, the management of saltmarsh ecosystems is heterogeneous and varies according to uses (De Wit *et al*. 2019): (i) still anthropized areas without conservation goals (Moomaw *et al*. 2018), (ii) management including conservation within an anthropized or industrialized site: *e.g.*, Camargue (Tamisier 1991), Tijuana Estuary (Southern California, Callaway & Zedler 1998, 2004), Ebro delta (Romagosa & Pons 2017), (iii) totally protected areas managed for their conservation (*e.g.*, saltmarshes of Hyères; Astruch *et al*. 2019, De Wit 2020). Furthermore, management plans with conservation priorities are based on a species-centered approach. Management goals rather target the maintenance or the enhancement of 'biodiversity' and 'high-level' species or taxon populations, namely those that are emblematic, rare, aesthetic, charismatic or threatened (Bougrain-Dubourg & Terrasse 2001, Bourgeois & Vidal 2005, Bourgeois *et al*. 2008, Boudouresque *et al.* 2020b), to the detriment of 'ordinary biodiversity' species and those considered harmful (De la Blanchère 1878, Boudouresque 2014). Biodiversity (Dasmann 1968, Soulé & Wilcox 1980, Wilson 1988, Boudouresque 2014) is often seen by managers and stakeholders as equivalent to the number of species, a high number of species being erroneously considered as a health index of the ecosystem. Disturbances and stress could in fact be conducive to an increase in the number of species (Hastwell & Huston 2001, Boudouresque 2014). Although this species-centered approach has allowed the protection and conservation of numerous heritage value and key species, it may be considered as inappropriate for an effective management, as it does not take into account the ecosystem functioning (Boudouresque *et al*. 2020b). Particularly for saltmarshes, an ecosystem-based approach is needed.

The aim of the present work is to apply the Ecosystem-based approach (EBA) to the saltmarsh ecosystem (SME), including wetlands and Mediterranean lagoons. We propose a conceptual model of the SME to understand its functioning and provide an efficient tool to improve its management. Here, we focus on the example of the Hyères saltmarshes. The development of the EBA and the conceptual model of the SME were based on analysis of the literature and the authors' expert judgment. Original data were collected within the saltmarshes of Hyères on macrophytes, fish assemblages and the plankton community to complete the available information.

MATERIAL AND METHODS

The case study of the saltmarshes of Hyères: The coastal wetlands and lagoons along the Bay of Hyères (Hyères, Provence, north-western Mediterranean Sea, France) have been deeply transformed by human activities since Antiquity. The spread of the city of Hyères (formerly named Olbia in Roman times) covered hundreds of hectares of wetlands, breaking up the ecological continuity from the peninsula of Giens to the eastern part of the bay. Since the Middle Ages, two distinct areas were delimited, Salin des Pesquiers and Vieux Salins (Fig. 1). The first was a coastal lagoon surrounded by wetland where an important local fishery was established, providing considerable revenue (Réveillon 2018). The second, smaller, had been exploited for salt production since Antiquity, but at an artisanal scale. From 1848 to 1995, these two areas were converted into an intensive salt production zone; they were transformed into a grid of several dozens of pans (ponds) used to evaporate brine, with a network of canals to exchange water between the pans, and to bring in or remove from water between the pans and the sea. The evaporation ponds and canals have remained in place, even after salt production was discontinued. In 2001, the whole site became the property of the *Conservatoire de l'Espace Littoral*

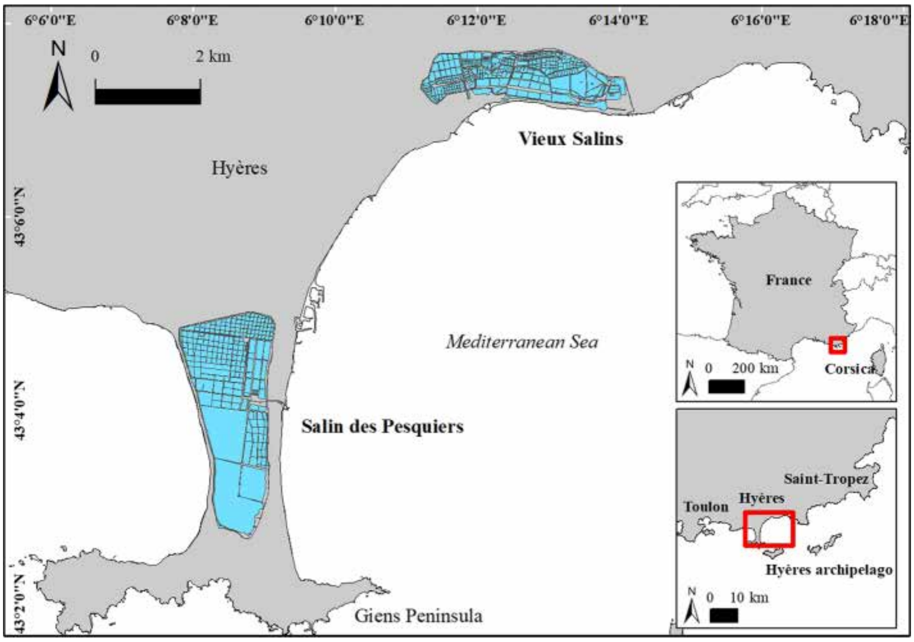

Fig. 1. – Location of the saltmarshes of Hyères (Salin des Pesquiers and Vieux Salins).

et des Rivages Lacustres (CERL) (coastal areas and lake shores conservation agency) and was managed by the *Toulon Provence Méditerranée* (TPM) local authority. During this period, the management goals and the water management planning were mainly focused on the historical heritage (salt production) and the conservation of waterfowl and wintering birds (Audevard 2017). A species-centered approach rather favored the so-called 'heritage taxa' (rare, threatened, charismatic). However, managers already identified the low connectivity between the saltmarshes (SME) and the open sea and related issues impacting fish assemblages (summer massive mortality, eutrophication) (Conservatoire du Littoral, Toulon Provence Méditerranée and Parc national de Port-Cros 2011, CREOCEAN 2011).

The concept of an ecosystem-based approach: The early concept of 'ecosystem' in ecology was proposed to understand and quantify flows (nutrients, carbon) in natural and anthropized ecosystems in the 1950s and earlier (Tansley 1935, Odum & Odum 1959, Odum 1998). Since then, saltmarshes, wetlands and lagoon ecosystems have been extensively studied (*e.g.*, Odum & Smalley 1959, Tamisier & Boudouresque 1994, Casagranda & Boudouresque 2007, De Wit *et al.* 2019), in part because of the aforementioned ecological services they fulfill. Since the seminal work of Teal (1962), several studies considering food webs and interactions between saltmarsh ecosystem functional compartments have been published (Nordström *et al*. 2015, Wang & Brose 2018). Today, the Ecosystem-based approach (EBA) is highlighted by the Marine Strategy Framework Directive (MSFD, 2008/56/EC) of the European Union (EU) (Laffoley *et al*. 2004, Halpern *et al*. 2010, Personnic *et al.* 2014) and is applied for fishery management worldwide (Turrell 2004, Rice 2005). It is regarded as an effective tool for the assessment and the management of an ecosystem corresponding to marine and coastal habitats of European interest (Habitat Directive Natura 2000, 92/43/EEC). In the framework of the MSFD, the EBA has been applied to four Mediterranean marine ecosystems to assess their quality: (i) *Posidonia oceanica* (Linnaeus) Delile seagrass meadow (Personnic *et al*. 2014), (ii) the coralligenous ecosystem (Ruitton *et al*. 2014); algae-dominated shallow rocky reefs (Thibaut *et al*. 2017) and underwater marine caves (Rastorgueff *et al*. 2015). An index, the Ecosystem-based Quality Index (EBQI), has been developed and tested to provide a standard tool for managers and stakeholders (Ruitton *et al.* 2017).

Hereafter, the conceptual model of the SME proposed is based on taxa from Mediterranean microtidal systems. The listed taxa are cited here as examples. The model itself is designed to be applied to non-Mediterranean SMEs encompassing other communities. The conceptual model was designed on the basis of expert judgment and the authors' knowledge, literature analysis and original data from the present study of the Hyères saltmarshes.

RESULTS

A conceptual model of the functioning of the saltmarsh ecosystem

Here, we propose a comprehensive conceptual model of the functioning of saltmarsh ecosystems (SME – wetland and lagoon). The conceptual model (Fig. 2) corresponds to an optimal functioning of the SME according to the management plan objectives (here, for the saltmarshes of Hyères). There is only one size of arrow. The goal of this scheme is not to assess the carbon flow between compartments (boxes) but to understand the fluxes between them. The description of the boxes in the conceptual

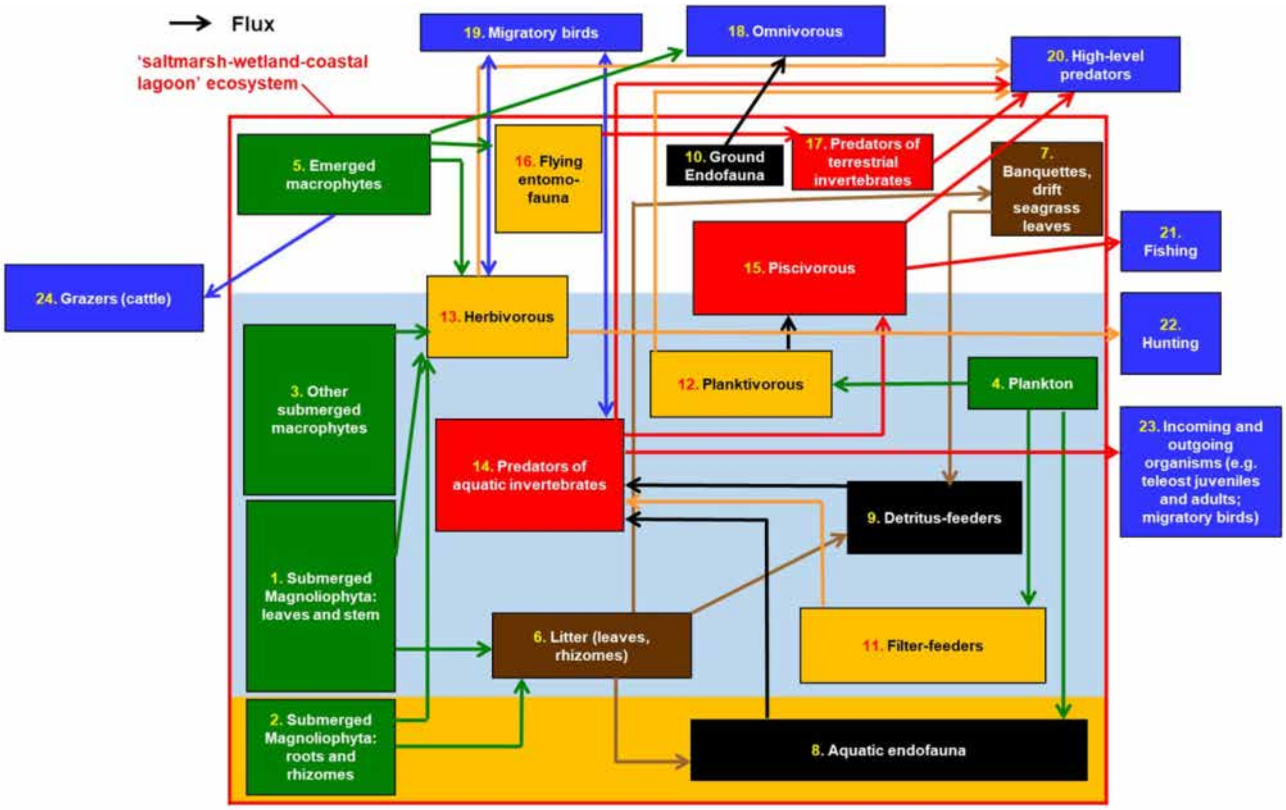

Fig. 2. – Conceptual model of the saltmarsh ecosystem (SME). Arrows correspond to the carbon flux between compartments (boxes). Box color: green: primary producers; yellow: primary consumers; red: predators; brown: detritus from primary producers; black: detritus-feeders and aquatic endofauna; blue: box out of the ecosystem but significantly interacting with it. The size of a box is not related to its 'weight' (importance).

model of SME is presented below. The cited taxa correspond to a northwestern Mediterranean context. The species or taxa described in the following boxes can belong to several of them according to trophic guilds.

(Box 1) Submerged Magnoliophyta

This box corresponds to the main aquatic primary producers of the ecosystem. These taxa are perennial, with maximum development between the end of spring and summer. Species occurrence in saltmarshes is strictly linked with the water salinity dependent on the connection with the open sea. *Ruppia spiralis* (Linnaeus) ex Dumortier (= *R. cirrhosa* (Petagna) Grande; Calado & Duarte 2000, Den Hartog & Kuo 2007, Ito *et al*. 2017) can tolerate euryhaline and hyperhaline conditions, from 0 to 106 g/kg (Verhoeven 1979, Mannino *et al*. 2015); *Stuckenia pectinata* (Linnaeus) Börner spreads in brackish or fresh water (Casagranda & Boudouresque 2007); *Zostera noltei* Hornemann can tolerate euryhaline conditions but less than *R. spiralis*; *Zostera marina* Linnaeus prefers marine-like conditions (Bernard *et al*. 2005); *Cymodocea nodosa* (Ucria) Asch. is an open sea species, widespread in the southern part of the Mediterranean, that can thrive in coastal lagoons (thermophilic, stenohaline) (Pergent *et al*. 2014). Two others marine Magnoliophyta are present in the Mediterranean Sea, *Posidonia oceanica* and *Halophila stipulacea* (Forssk.) Asch.; *P. oceanica* is strictly marine (Boudouresque *et al.* 2012) and *H. stipulacea* is an invasive Lessepsian species which can occur in brackish waters (Galil 2006). In this box, only leaves and stems are considered. Leaf and stem biomass can be driven by light availability, depth, nutrient availability and predation pressure. Low biomass or covering can be linked to inadequate environmental conditions while high biomass and maximum covering of the meadow could be explained by a relatively small population of herbivorous birds (*e.g.*, Anatidae). In both situations, it does not mean good ecosystem quality, the optimum corresponding to intermediate abundance (Gayet *et al*. 2012).

(Box 2) Buried parts of Magnoliophyta in the sediment

Roots and rhizomes are the buried part of the plant. Their biomass is related to the nutrient availability in the sediment. Even if the submerged part of the plant biomass is low, high endogenous biomass indicates a good state of health of the plant, which is adapted and able to accumulate reserves (Verhoeven 1980, Ferrat *et al*. 2003).

(Box 3) Other submerged macrophytes

Macroalgae can be abundant primary producers in SMEs. Most of them are Chlorobionta, mainly indicative of eutrophic conditions (*e.g.*, *Ulva* spp. Linnaeus, *Chaetomorpha linum* (O.F. Müller) Kützing, *Cladophora* spp. Kützing). The macroalgae assemblages are strictly linked with water conditions, light availability and eutrophication level. Compared to open sea macroalgae communities (*e.g.*, Infralittoral reefs with photophilous macroalgae), the species diversity is lower and is driven by annual changes of physical-chemical parameters (*e.g.*, salinity, dissolved oxygen, temperature) (Pérez-Ruzafa *et al*. 2008). In lagoons, which present a high connectivity with the open sea, Rhodobionta and Phaeophyceae can be more numerous, including perennial species such as *Cystoseira barbata* (Stackhouse) C. Agardh (Orfanidis *et al*. 2008, Réseau de Suivi Lagunaire 2011).

(Box 4) Plankton

Phytoplankton and zooplankton are key assemblages for the trophic network of the SME (Quintana *et al*. 1998). They constitute an important food supply for several boxes (planktivorous, detritus- and filter-feeders). Phyto- and zooplankton abundance are strongly correlated with a seasonal pattern (Riley & Bumpus 1946, Talling 2003). Species composition is linked to salinity and nutrient availability (Masmoudi *et al.* 2015). Status of plankton assemblages is a good indicator of the water column conditions. A high rate of Chlorophyll a could result from eutrophic conditions. The size of the organisms is also a good indicator that reflects the aging of the community. Extreme conditions can lead to the thriving of a single species (*e.g.*, *Artemia salina* Linnaeus, 1758 in hyperhaline conditions: Ollier 1964, Britton and Johnson 1987). Abundance of large individuals corresponds to an old stable population; abundance of small and young individuals indicates more unstable conditions. Early stages of Copepoda (nauplii) or Rotifera can be present in very high abundance, often corresponding to a rapid shift in water condition/circulation (Brucet *et al.* 2005, 2008).

(Box 5) Emerged macrophytes

e.g., *Salicornia* s.l. Linnaeus, *Spartina* Schreb (Molinier 1953, Molinier & Tallon 1970, Aboucaya *et al.* 2011). Biomass and specific richness depend on salinity (soil and water) and water control (flooding persistence) (García *et al*. 1993, Asri & Ghorbanli 1997, Curcó *et al*. 2002). Emerged vegetation (*e.g.*, *Spartina alterniflora* Loisel association) is considered as the main primary producer in the saltmarsh ecosystem (Teal 1962, Marañón 1998, Noble & Michaud 2016).

(Box 6) Litter

During the life span of aquatic Magnoliophyta, dead leaves usually shed from the summer. Rhizomes and roots can be uprooted by storms and herbivorous organisms. All these remnants constitute the litter. The abundance of the litter is correlated with the coverage and biomass of the living meadow. Litter is an important food supply for detritus-feeders, herbivores and part of the endofauna. It also can be exported to the *banquettes* (box 7). Litter increases when the salinity decreases (Curcó *et al*. 2002).

(Box 7) Banquettes

Composed of seagrass leaves and macroalgae detritus drifted onto the shore of the lagoon, *banquettes* provide habitat and trophic resources for detritus-feeders and part of the entomofauna. Thickness, taxonomic composition and abundance of the *banquette* are linked to seagrass meadow vitality and macroalgae abundance and can vary according to season (Virnstein *et al*. 1985).

(Box 8) Aquatic endofauna

Mainly represented by Annelida (*e.g.*, *Hedistes diversicolor* O.F. Müller, 1776) and Mollusca (*e.g.*, *Cerastoderma glaucum* Brugières, 1789, *Ruditapes* spp. Chiamenti, 1900), this box is composed of detritus- and filter-feeders. Abundance of individuals and species diversity are correlated with particle size of the sediment, dissolved oxygen rate, eutrophication and connection to the open sea (Ollier 1964, Lardicci *et al.* 1997, Breil 2014).

(Box 9) Detritus-feeders

The abundance of macro-invertebrates such as *Gammarus* spp. Fabricius, 1775, *Idotea chelipes* (Pallas, 1766), *Lekanesphaera hookeri* (Leach, 1814), *Carcinus maenas* (Linnaeus, 1758) and *C. aestuarii* Nardo, 1847 can be linked with seagrass biomass (Gravina *et al*. 1989, Casagranda *et al*. 2006, Özbek *et al*. 2012, Breil 2014). The mudsnail *Hydrobia* spp. Pennant, 1777 can be very abundant in saltmarshes and lagoons (Ollier 1964, Barnes 2005). This compartment, despite low biomass, has a key function, processing macrophyte matter to different trophic levels through fragmentation, thus accelerating the decomposition.

(Box 10) Ground endofauna

Mainly represented by Annelida, arachnids, and insects, it belongs to several trophic guilds (Chauliac 2005, Breil 2014).

(Box 11) Filter-feeders

This box is often represented by Mollusca such as *Mytilus galloprovincialis* Lamarck, 1819, *Ostrea edulis* Linnaeus, 1758, *Mya* sp. Linnaeus, 1758, *Ruditapes* spp., *Cerastoderma* spp., *Pinna nobilis* Linnaeus, 1758, Annelida such as *Ficopomatus enigmaticus* (Fauvel, 1923), *Sabella spallanzanii* (Gmelin, 1791), Ascidiascea such as *Phallusia mammillata* (Cuvier, 1815), *Botryllus* spp. Gaertner, 1774, *Microcosmus* sp. Heller, 1878, *Ascidiella aspersa* (Müller, 1776), etc. (*e.g.*, Perthuisot & Guélorget 1983, Gravina *et al*. 1989, Marchini *et al.* 2004, Soufi-Kechaou *et al.* 2019). Species diversity, abundance and size of individuals of the filter-feeder box depend on eutrophication level and food availability (see box 4). A high biomass could contribute to the eutrophication itself and leads to anoxic crises (Mayot *et al*. 2021). Oyster farming (mainly *Magallana gigas* (Thunberg, 1793) and *Mytilus galloprovincialis*) can contribute significantly to this box (Vaquer *et al*. 2000).

(Box 12) Planktivorous species

In coastal Mediterranean lagoon ecosystem and as well SME, the main taxa representing the Planktivorous box are *Atherina* spp. Linnaeus, 1758 (Le Diréach *et al*. 2013, 2021). This small sedentary teleost can be very abundant and can thrive in a wide range of abiotic conditions. *Atherina boyeri* Risso, 1810 spends its entire life cycle in SME (Maci & Bacet 2010, Kara & Quignard 2018b). This is a key taxon, which provides food for several higher trophic levels (*e.g.*, box 15: piscivorous teleosts and birds) (Kara & Quignard 2018a). Depending upon the level of connectivity with the open sea, non-sedentary planktivorous species can also occur, such as the European anchovy *Engraulis encrasicolus* (Linnaeus, 1758) (Manzo *et al.* 2013). The pink flamingo *Phoenicopterus roseus* Pallas, 1811 can belong both to this compartment and to box 14.

(Box 13) Herbivorous species

This box is composed of Anatidae (*e.g.*, *Tadorna tadorna* (Linnaeus, 1758), *Anas Penelope* Linnaeus, 1758) and other herbivorous birds and the teleost *Salpa salpa* (Linnaeus, 1758). The latter species is the only indigenous herbivorous teleost in the Mediterranean Sea that can also be observed in brackish waters (Kara & Quignard 2019b). Some small crustaceans, terrestrial gastropods and insects are also to be included in this box (see Casagranda *et al*. 2006). Finally, the rare southwestern water vole *Arvicola sapidus* Miller, 1908, a rodent, and the European hare *Lepus europaeus* Pallas, 1778 can occur.

(Box 14) Predators of aquatic invertebrates

(from open water or sediment endofauna) Represented by: (i) Shoreline birds, most of them being migrators, which use the SME for breeding and nesting (*e.g.*, the pied avocet *Recurvirostra avosetta* Linnaeus, 1758, the wood sandpiper *Tringa glareola* (Linnaeus, 1758), the pink flamingo *Phoenicopterus roseus*); (ii) wintering birds such as the little stint *Calidris minuta* (Leisler, 1812) and the Dunlin *C. alpina* (Linnaeus, 1758) occur and can be abundant (Audevard 2019); (iii) teleost fishes (*e.g.*, *Anguilla anguilla*, *Sparus aurata*, Soleidae, Mugilidae, etc.) (Kara & Quignard 2019a, b); and (iv) aquatic turtles, namely the European pond turtle *Emys orbicularis* (Linnaeus, 1758), and the red-eared slider turtle *Trachemys scripta* (Thunberg *in* Schoepff, 1792), even if their trophic regimes can be more diverse (Lascève 2014, Perrot *et al.* 2016).

(Box 15) Piscivorous species

Within the SME, piscivorous species are mainly birds: the lesser great cormorant *Phalacrocorax carbo* (Linnaeus, 1758), Laridae such as the little tern *Sternula albifrons* (Pallas, 1764), and Ardeidae such as the little egret *Egretta garzetta* (Linnaeus, 1766) and the grey heron *Ardea cinerea* Linnaeus, 1758) (Audevard 2017). The teleost fish *Dicentrarchus labrax* is also a piscivorous species (Haffray *et al.* 2006, Kara & Quignard 2019a).

(Box 16) Flying entomofauna

This box is highly diversified. Chironomidae and the mosquitoes *Aedes* sp. (Meigen, 1818) and *Culex* sp. (Linnaeus, 1758) (Poulin 2012, Roiz *et al*. 2015) are among the most abundant. Species diversity and abundance of individuals of the flying entomofauna are strongly linked with landscape and vegetation structure within the saltmarsh (Chauliac 2005, Roiz *et al*. 2015; Orthoptera: Lemonnier-Darcemont 2004; Coleoptera: Ponel 2005; Lepidoptera: Varenne 2015).

(Box 17) Predators of terrestrial invertebrates

This box is composed of a variety of taxonomic groups: (i) Some species of Chiroptera use wetlands as shelters and food resource (Cosson 2005, Flaquer *et al*. 2009, Barataud 2012, Naturalia 2015); (ii) Reptilia such as lizards, the threatened Spanish psammodromus *Psammodromus hispanicus* Fitzinger, 1826, the common wall lizard *Podarcis muralis* (Laurenti, 1768), the western green lizard *Lacerta bilineata* Daudin, 1802 (Joyeux 2005, 2011) and a snake, the Iberian three-toed skink *Chalcides striatus* (Cuvier, 1829) (SOPTOM 2017); (iii) frogs, *e.g.*, *Pelophylax* sp. Fitzinger, 1843 and the Mediterranean tree frog *Hyla meridionalis* Böttger, 1874 (Joyeux 2005); (iv) arachnids (Lycosidae Sundevall, 1833; and (v) Coleoptera Carabidae Latreille, 1802, *e.g.*, *Scarites planus* Bonelli, 1813 and *Carabus morbillosus* Fabricius, 1792.

(Box 18) Omnivorous species

The wild boar *Sus scrofa* Linnaeus, 1758 is an opportunist species; it is often considered a pest by farmers, because of the damage it can cause to crops (Sáez-Royuela & Telleria 1986, Schley & Roper 2003, Corbobesse 2008); however, it plays an important role in the functioning of a natural ecosystem (see *e.g.*, Barrios-García & Ballari 2012, Boudouresque *et al*. 2020b). The red fox *Vulpes vulpes* (Linnaeus, 1758) can be included in this box. The yellow-legged gull *Larus michahellis* Naumann, 1840, as an opportunistic species, is considered as omnivorous. The coleopter *Pimelia muricata* Olivier, 1795 is observed in sandy areas within the saltmarshes. Some fish species occurring in abundance within the lagoon habitats can belong to this box, such as *Chelon* spp. Artedi, 1793 and the flathead grey mullet *Mugil cephalus* Linnaeus, 1758 (Cardona 2006, Kara & Quignard 2019a).

(Box 19) Migratory birds

This box corresponds to migratory birds including waterfowl that can be listed in other boxes of the ecosystem (*e.g.*, Herbivorous species, Predators of invertebrates, etc.). It highlights the inputs and outputs from and to the SME by avifauna. The maintaining of this compartment is one of the main goals of the managers of the saltmarshes of Hyères (Birdlife International 2004, Audevard 2017, 2019).

(Box 20) High-level predators

This box can be placed both inside and outside the SME. It includes species that mainly live outside but close to the SME: (i) mostly raptors such as the western marsh harrier *Circus aeruginosus* (Linnaeus, 1758) (Audevard 2017, 2019); and (ii) mammals such as the red fox *Vulpes vulpes*, traditionally considered as harmful (see the human-centered approach; Boudouresque *et al*. 2020b), and the domestic cat *Felis silvestris catus* (Linnaeus, 1758) (Tranchant & Vidal 2003). Some Reptilia belong to this compartment, such as the Montpellier snake *Malpolon monspessulanus* (Hermann, 1804) (SOPTOM 2017).

(Boxes 21 and 22) Fishing and Hunting

Human activities that exploit the SME must be considered for their impact on waterbird and fish populations (Mathevet & Tamisier 2002, Tamisier *et al.* 2003), even when those activities are prohibited (due to illegal fishing and hunting).

(Box 23) Incoming and outgoing organisms

The SME is an essential habitat for fish juveniles (settlement of post-larvae coming from open sea spawning areas), and for adults looking for trophic resources, shelter and suitable environmental conditions (Kara & Quignard 2019a, b). Similar ecological functions are provided for bird populations, mainly waterfowl, which find breeding and nesting areas and trophic resources mainly from spring to the end of summer. This box highlights some of the main ecological functions and services of the SME (Himes-Cornell *et al*. 2018, O'Higgins *et al.* 2019).

(Box 24) Grazers

Cattle (cows, sheep and horses) in grazed pastoral areas can find an important food supply in wetlands (Duncan & D'Herbes 1982), interacting with other herbivorous species belonging to the SME (box 13) and structuring emerged Magnoliophyta communities (box 5). This compartment is external to the ecosystem (SME).

Diagnosis on the conservation status of the saltmarshes of Hyères

The *Ruppia spiralis* seagrass meadow within the saltmarshes of Hyères presents a good conservation status (Massinelli *et al.* 2017, 2021). Its high mean biomass reflects the good health of this species, which finds favorable conditions for its development. However, *Ruppia spiralis* is a halophilous species, the only seagrass species that can thrive in such a range of salinity (*i.e.*, 0-106 g/kg). The absence of other expected taxa such as *Zostera noltei* can be explained by both unsuitable conditions (*e.g.*, high salinity, eutrophication) within the saltmarshes and the decline of its populations at regional scale (northwestern Mediterranean; Pergent *et al*. 2014). A number of aquatic compartments (boxes) of the saltmarshes of Hyères are not well connected with the open sea. Fish assemblages are dominated by sedentary taxa adapted to euryhaline or halophile conditions (Le Diréach *et al*. 2021): (i) *Atherina* spp., (ii) *Pomatoschistus* spp. Gill, 1863, (iii) *Syngnathus abaster* Risso, 1827 (Kara & Quignard 2018b). When the connection with the open sea is better established, *e.g.*, within the belt channel of Salin des Pesquiers, a higher species diversity is observed, represented by: Mugilidae such as *Chelon* spp., *Mugil cephalus*, and migratory fish such as Sparidae (*Sparus aurata, Diplodus vulgaris* (Geoffroy Saint-Hilaire, 1817), *Sarpa salpa*), *Dicentrarchus labrax*, *Anguilla anguilla* (Kara & Quignard 2019a, b). The occurrence of juvenile individuals from these taxa can be considered as indicative of that connectivity. It is not only the lack of connectivity that can impact the functionality of the lagoon parts of the saltmarsh, but also inadequate salinity and DO (dissolved oxygen). Salinity is the main driver of fish assemblage composition (Malo-

Michèle 1979, Alliot *et al.* 1983, Bodinier *et al.* 2010), associated with oxygen (Person-Le Ruyet 1986) and habitat characteristics (Astruch *et al*. 2020, Massinelli *et al*. 2021). Salinity is also the main driver of plankton communities. The plankton community can react very fast after shifts in water condition. Sites well connected with the open sea show a low abundance of both phyto- and zooplankton and larger individuals and greater stability (*i.e.*, older populations). In less well-connected and stable areas, abundance is higher, particularly for phytoplankton, mainly represented by small individuals corresponding to a young population. As shown by Quintana *et al*. (1998), flooding episodes induce a dilution of the zooplankton and increased resources due to phytoplankton bloom. The saltmarshes of Hyères host a very high a species diversity of birds (the cumulative number, over several decades, is about 300 species), including waterfowl wintering species and shorebird species, most of them being of high heritage value (Audevard 2017, 2019). This is one of the main successes of the first management plan implemented in early 2000s.

The management policy has enabled the inclusion of the saltmarshes of Hyères within the Natura 2000 network (‘Rade d’Hyères’ FR9301613) as well as the Camargue and coastal lagoons of the Gulf of Lion. The monitoring of bird populations shows connectivity between the saltmarshes of Hyères, the Camargue and Gulf of Lion lagoons, all considered as nesting and wintering areas and privileged sites for migrations (Audevard 2019). Within the saltmarshes of Hyères, one habitat is considered a priority habitat of European interest, in the sense of the EU Habitat Directive: 1150-2 – Mediterranean lagoon. In addition, about thirty species of plant are of heritage value, with national or local protection status, highlighting the maintenance of a remarkable species diversity, despite the considerable transformations to which man has previously subjected these saltmarshes (Noble & Michaud 2016).

DISCUSSION AND CONCLUSION

The present work enables us to better understand the overall functioning of the SME (saltmarsh ecosystem – saltmarsh, including wetland and coastal lagoon) such as the saltmarshes of Hyeres and their management through an ecosystem-based approach (EBA). The next step should be to implement an Ecosystem-based Quality Index (EBQI) for the SME. The index should be tested with data provided from a wide range of Mediterranean SMEs: (i) size (surface area, mean and maximum depth), (ii) water conditions (salinity, temperature), (ii) connectivity to the open sea, (iii) eutrophication, (iv) gradient of human pressure other than eutrophication (fishing activities, non-indigenous species abundance and diversity, habitat destruction or withdrawal). This approach would provide the basis for an effective tool for assessment of the conservation status of Mediterranean SMEs. Obviously, assessment of the health status of coastal lagoon and wetland ecosystems is complex and several indices and methods already exist (Lepareur *et al*. 2013); most of these existing indices take into account only part of the ecosystem (plankton community, endofauna, macrophytes, etc.) and were developed to assess the quality of the water body within the framework of the Water Framework Directive. Further studies are needed to define a typology of saltmarshes following an EBA, *i.e.*, wetlands and coastal lagoons, in an Euro-Mediterranean context (Dyer 1998, Ibañez *et al.* 2002, Almeida 2016), according to: (i) the size of the system (including the watershed basin), (ii) the ratio between wetland and lagoon surface areas, (iii) the effectiveness of the connectivity with the open sea. The conceptual model should be adapted for each category by adjusting the weight of each box without changing its overall structure (*e.g.*, box 2 ‘submerged Magnoliophyta’ will get a higher weighting for a category including large Mediterranean lagoons, *e.g.*, Thau lagoon, southern France). It is a challenging perspective for better monitoring and management of such habitats of interest.

Despite the lack of innovative indices and typology, the ecosystem-based approach enables us to highlight ways to improve the functioning of saltmarshes considering consequences of the management at the ecosystem scale. Relative to the initial objectives of the first management plan for the saltmarshes of Hyères established in 2004, centered on waterfowl and saltmarsh landscape conservation, the results have met the expectations of the managers. Since 2019, the new management plan will not exclude these previous aims but will also consider the improvement of the whole ecosystem functioning (based on water circulation and connection with the open sea) together with better access to the sites for the public and the conservation of heritage landscape. This is a direct consequence of the application of the ecosystem-based approach. Our analysis underlines the need to improve (i) the water circulation in selected ponds where high salinity and low connectivity prevents their use by teleost fish, of which the juveniles could thrive there; inappropriate management of water circulation can lead to the trapping of juvenile fish that cannot escape to join the open sea adult population (Bruslé & Carbony 1992); (ii) the water quality along the belt channel is impacted by contaminated freshwater inputs from adjacent urbanized areas. In addition, the erosion of the shore and frequent submersion events have alerted the management team to the risk of the future ‘marinization’ of the area, in relation with the current increased rate of sea level rise (Blanfuné *et al.* 2016, Dieng *et al.* 2017). Major transformations are therefore expected and are of long-term concern in the future management plan. The former role of fishery, before the wetland/coastal lagoon system was converted into salt evaporation ponds, is known in the area (Chauvet 1986, Réveillon 2018). Historical data regarding the use of the Salin des Pesquiers lagoon as an active fishery proves that the functioning of the site was completely different from that of today. The functioning of the Salin des Pesquiers lagoon was close to that of a Mediterranean lagoon, including permanent connectivity with the open sea, sometime on both sides of the tombolo (Réveillon 2018). Taking into account global change, sea level rise and the increase in submersion events, a 'marinization' of the saltmarshes of Hyères is to be expected. Taking this 'marinization' into account constitutes an important challenge for the new management plan.

The upcoming decisions to be taken by managers will take into account all the following priorities: (i) Waterfowl conservation is of major importance not only at local scale but at regional scale (Audevard 2017); managers will therefore continue to favor the nesting of birds; but they will consider water circulation, flooding risk in surrounding urbanized areas and the future submersion of the north-western ponds of the Salin des Pesquiers (*e.g.*, with the destruction of a nesting site of the Kentish plover *Charadrius alexandrinus* Linnaeus, 1758). (ii) The awareness of the need to improve the water circulation, and the connectivity to the open sea, is a key factor for appropriate day-by-day management. (iii) It appears appropriate to use water management to conserve both the technical integrity of the site (belt channel, canals, ponds, locks, etc.) and the ecological functioning (*i.e.*, maintaining water level for bird nests, water circulation to allow fish to move in and out of the sites). (iv) Within Vieux Salins, a nozzle has been dug to improve water circulation without disturbing vehicle access for mosquito control. (v) A sluice gate will be installed between the poorly connected North Lagoon (Etang Nord, Salin des Pesquiers) to the inlet channel, creating a floodgate-like device for trapping the fish and allowing them to reach the open sea from the North Lagoon. (vi) Managers aim to improve access for the public, enhancing the awareness-raising role of the saltmarshes for the local population and tourists. (vii) The establishment of a small-scale salt production operation is planned in the Salin des Pesquiers; the aim is to make the public aware of the heritage value of the saltmarshes landscape as exploited salinas. (viii) The dredging of the belt channel will improve water circulation and quality and reduce eutrophication, nitrophilous macrophytes and *Ficopomatus enigmaticus* spread. (ix) A '*bourdigue*', *i.e.*, traditional Mediterranean fixed fishing gear trapping fish in shallow lagoons and channel systems, should be installed in the entrance channel of the Salin des Pesquiers; this device should not be used to catch fish from the saltmarsh but could serve as a tool to raise public awareness regarding a previous use of the site before industrial salt production (Réveillon 2018). This new management plan can obviously be described as 'interventionist'. Some authors favor a more passive recovery of the saltmarshes to improve ecosystem services (Almeida *et al*. 2017, De Wit *et al.* 2019, De Wit 2020). However, the configuration of the sites, particularly the Salin des Pesquiers, does not allow such passive restoration. Most of the ponds and channels are down to the mean sea level. A passive approach from the management team in this context would soon lead to the flooding of the whole site and the general transformation of the old evaporation ponds into a coastal lagoon. The choice of the managers is to control the water circulation and the infrastructure of the sites to accompany the inevitable changes and to preserve the SME as far as possible.

The Ecosystem-based Approach applied to SME management presented here must be considered as a logical development from current management activities within the scope of current guidelines (*e.g.*, Marine Strategy Framework Directive for European Union). Worldwide, the species-centered approach applied during the 20th and early 21st centuries has allowed the conservation of heritage, threatened and rare taxa (Boudouresque *et al*. 2020b) and the maintaining of extensive surface areas of well-functioning SMEs (De Wit *et al.* 2019, De Wit 2020). The present approach also takes into account those high-value taxa, but associated with the 'ordinary biodiversity' for enhanced consideration of the functioning of the ecosystem. Of course, the EBA does not rule out the specific protection and management of some threatened and rare taxa according to the management goals. The consideration of human activities within the SME is more consistent with management objectives taking into account both conservation of fauna and flora and the preservation of sustainable economic and social activities such as (i) pastoral activities (Duncan & D'Herbes 1982), agriculture (*e.g.*, ricefield: Tourenq *et al*. 2001, Lloret *et al*. 2005), (ii) fishing and hunting (although some authors consider hunting activities as disruptive, highlighting the need for hunting reserves) (Mathevet & Tamisier 2002, Tamisier *et al*., 2003), (iii) salt production (De Wit *et al*. 2019), and (iv) public access. This approach matches the socio-ecosystem concept, where the Human is considered as part of the system and contributes to its functioning, the impact being considered as not solely negative (Turner *et al*. 2008, Boudouresque *et al*. 2020b). Ecosystem services provided by SME (Himes-Cornell *et al*. 2018) are taken into account to highlight the benefits allowed by a well-functioning ecosystem. Issues regarding global change and its impact on ecosystem functioning are of concern worldwide and especially for the SME (Lloret *et al*. 2008, De Wit 2011, Green *et al*. 2017). The EBA takes into account these issues to understand the impact of warming, the rise of the sea level, submersion and the arrival of non-indigenous species in both terrestrial and aquatic compartments of the ecosystem. Within the Mediterranean, current monitoring of community shifts in marine ecosystems has evidenced a decline in the previously so-called 'high-value taxa', threaten by warming and the spread of invasive species (Francour *et al*. 1994, Lejeusne *et al*. 2010). A species-centered approach, based on an unachievable reference state of the ecosystem (*e.g.*, before the industrial revolution), cannot fit with appropriate management within the context of global change. On the other hand, the EBA can assess and monitor the functioning of the SME taking into account the expected changes. The rise in sea level (RSL) will contribute to an increase in marine conditions in the SME, aquatic compartments will shift to assemblages more connected with the open sea with, in some particular cases, a positive impact for teleost fish species of fishery interest, of which the juveniles settle in brackish water (*e.g.*, *Sparus aurata*, *Dicentrarchus labrax*). The RSL will also increase erosion of the seashore and submersion, resulting in the withdrawal of halophytic vegetation and the destruction of the nesting and breeding habitat essential for waterfowl. More than ever, the management of the SME, and particularly the saltmarshes of Hyères, calls for a comprehensive approach to anticipate and accompany the inevitable changes.

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DISTRIBUTION OF MANILA CLAM, *RUDITAPES PHILIPPINARUM*, INTO BERRE LAGOON ACCORDING TO THE ENVIRONMENTAL CONDITION

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RUDITAPES PHILIPPINARUM
BERRE LAGOON
FISHERY
STOCK
TEXTURE
HYDROLOGICAL DATA

ABSTRACT. – The Manila clam *Ruditapes philippinarum* (Adams & Reeve, 1850) is one of the most commercially exploited bivalve in the world (Dang *et al.* 2010). Since 2013, it has expanded into Berre lagoon. In 2017, the density was between 41 and 1,264 clams.m^{-2}, higher than that of other French sites (Mahé 2017, Sanchez *et al.* 2014). In February 2018, the professional clam fishery opened in the Berre lagoon. Starting in July 2018, an ecological crisis occurred and the clam stock was decimated (Mayot *et al.* 2020). Consequently, the fishery was closed. Following this event, the impact assessment was initiated with a clam stock study in spring 2019. During this campaign, 238 stations were sampled where the number and the size of *R. philippinarum* were assessed. To explain the clam distribution into Berre lagoon in relation to environmental conditions, hydrological condition and grain size were analyzed. These parameters are well known to control clam expansion (Le Treut 1986) and are believed to lead the clam distribution. Preliminary results show a total abundance of 552.07 million clams. No clear correlation was found between the hydrological condition, sedimentary data and the clams' distribution at a large scale into Berre lagoon.

INTRODUCTION

The Manila clam (or Japanese carpet shell) *Ruditapes philippinarum* (Adams & Reeve, 1850) is a bivalve mollusk native from the Indo-Pacific province. Nowadays, this is one of the most commercially exploited bivalves mollusks in the world (Dang *et al.* 2010). Manila clam is also 23rd on "the list of the most important 27 aquatic alien species voluntarily introduced in European freshwater and marine coastal waters" (Savini *et al.* 2010). The introduction of species into a new environment is one of the main challenges of global change in marine ecosystems (Cordero *et al.* 2017). In general, these species exhibit invasive behavior that can affect the ecology of ecosystems.

It seems to be the *R. philippinarum* behavior in the Berre lagoon. This area is located South of France, near Marseille urban city (Fig. 1). It covers 155 km² and its maximal depth is found to be at 9.5 m. In the fifties, at the beginning of the industrialization of the Berre coastal zone, two clams species were present in the lagoon: *Polititapes aureus* (Gmelin, 1791) and *Ruditapes decussatus* (Linnaeus, 1758) (Febvre 1968, Mars 1949). As a result of chemical pollution, which has increased sharply due to increasing industrialization and urbanization, fishing has been forbidden in 1957. In 1966, the hydroelectric power plant installation in the North of the lagoon brought huge freshwater and nutrient inputs into the lagoon (Mayot *et al.* 2020, Roux *et al.* 1985). This perturbation caused the extinction of many species, like clams (Le Corre & Garcia 1989). Since the 1990s, new laws have regulated the freshwater input volumes. These reductions have led to a significant improvement in the biodiversity of the Berre lagoon ecosystem (Mayot *et al.* 2020). In recent years, the shore benthic macrofauna biodiversity increased and was mainly constituted of Manila clams (Audry 2015; Mayot *et al.* 2020). New studies showed increasing densities in 2017, with an amount varying between 41 and 1,264 clams/m^{-2} (Mahé 2017). Note that this result exceeds the densities reported from other French sites but that these sites are exploited by fishing (Sanchez *et al.* 2014). As a result, the clam fishery was opened in February 2018.

In 2018, the occurrence of a massive input of freshwater during spring time (natural and by the hydroelectric power plant via EDF channel), the following high water temperature in summer time (30 °C) and the lack of wind has caused a major ecological crisis into the lagoon which began at the end of July 2018 (Mayot *et al.* 2020). The water column was stratified and O_2 could not reach the bottom layer. Consequently, clams missed O_2 and died. This resulted in a depletion of the clam stock, which ended the fishing activity in September 2018.

With this awareness, it appeared crucial to provide a recovery estimation of the *R. philippinarum* density and abundance after the ecological crisis. This study provides a clear overview of the crisis impact on the shore benthic macrofauna, especially *R. philippinarum*. Environmental

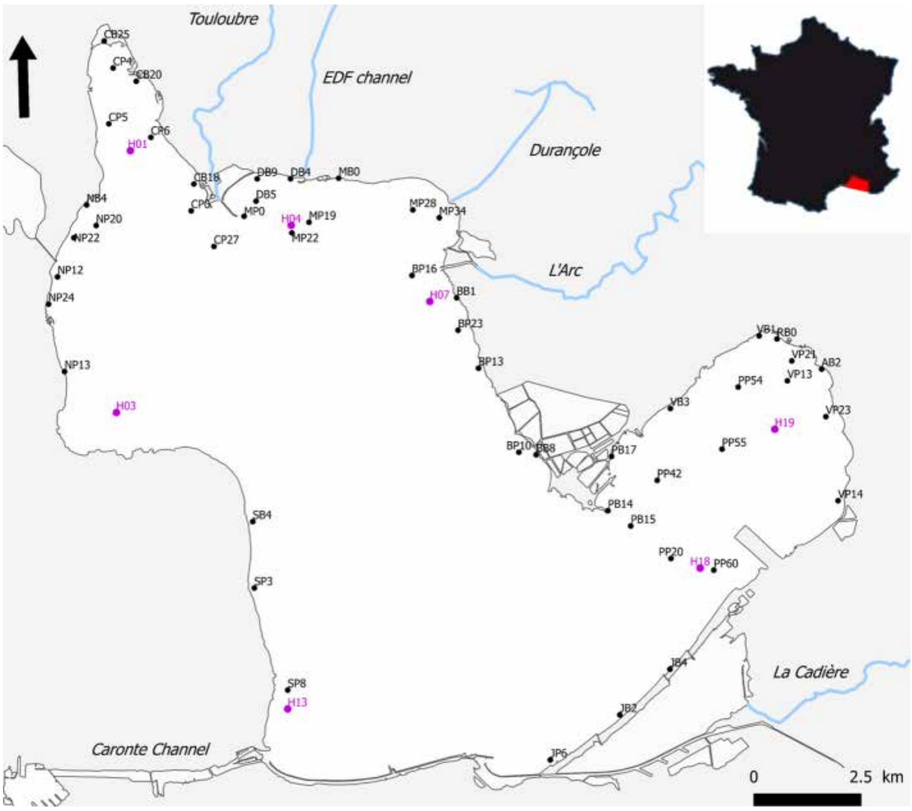

Fig. 1. – Situation of Berre lagoon in the south of France (red) with the localization of sediment station (black) and hydrological station (purple).

and economic officials expect such an ecological inventory from *R. philippinarum* for future decisions. Indeed, the objective of this study is to provide an overview of the distribution of clams in the Berre lagoon according to environmental conditions.

MATERIALS AND METHODS

Sampling campaign: To evaluate the clam distribution and density into Berre lagoon, a field investigation has been performed between the 27th of March and the 27th of July 2019. The lagoon has been divided into 19 layers according to the depth, the grain size *a priori* and the hydrologic conditions (following Berthou *et al.* 1997). Only the first 5 meters of depth were sampled. Indeed, no clams were expected below this depth due to the ecological crisis of 2018 (Mayot *et al.* 2020).

According to Bertignac *et al.* (2001), the sampling rate needs to be at least 10 stations per km² with 2 replicates to have an adequate precision (*i.e.*, 20 stations per km²). This strategy was created for the Arcachon Bay, where the tide plays an important role in clam biology and repartition. Into the Berre lagoon, it is not the case. In this campaign, the sampling rate was different depending on the layer specificity (depth and area) to better adjust the number of stations to the depth clam repartition. For the depth layer (between 2 and 5 meters), the sampling rate was 7 stations per km² with 3 replicates (*i.e.*, 21 stations per km²). For the shore layer (between 0 and 2 meters), two strategies were chosen (1) for the big shore layer, with an area superior to 1 km², the sampling rate was 10 stations per km² with 3 replicates (*i.e.*, 30 stations per km²) (2) for the shore layer with an area inferior to 1 km², the sampling rate was 20 stations per km² with 3 replicates (*i.e.*, 60 stations per km²). Using three replicates per station allows the data to be considered as normally distributed for statistical analysis.

Stations were randomly distributed inside each layer, with a distance minimal of 200 meters between them. GPS was used for reaching each station. A total of 238 stations were sampled. At each station, 3 replicates were sampled for repeatability estimations.

Biological data: Sampling was done by scuba diving using a 0.25 m² quadrat (0.50 m × 0.50 m). For each replicate, everything inside the quadrat was sampled. On the boat, bivalves were counted and *R. philippinarum* shell length was measured to the nearest 0.01 mm using a caliper. At each station, macrophytes distribution was visually evaluated.

Total abundance has been estimated using the protocol of Berthou *et al.* (1997) for the whole stock, for juvenile and for exploitable clams. According to Caill-Milly *et al.* (2003), clams with a shell of less than 17 mm can be considered as recruitment from the previous year. Into Berre lagoon, the minimum legal size for *R. philippinarum* is the same that for Mediterranean capture: the exploitable stock is all clams with a shell length larger than 30 mm (The Decree of 29 January 2013).

To have an idea of the clam biomass, a relation between shell length and weight, made on 1,347 *R. philippinarum* from Berre lagoon in 2017, was used:

Weight = 0.0002 × *Length*$^{3.0579}$ ($R^2 = 0.9627$, Mahé 2017)

Each observed clam was measured, and weight was calculated using this formula. Then, total biomass has been estimated using the same protocol as for total abundance (Berthou *et al.* 1997).

Hydrological data: For the hydrological parameters (temperature, salinity and dissolved oxygen), a probe (Hydrolab DS5) was used in 7 stations in Berre lagoon every month since 1994 (Fig. 1). Chlorophyll *a* and suspended matter were analyzed on sample taken at the surface and bottom on each station using a Niskin bottle.

Sediment data: During the campaign, sediment was sampled at each station (238 samples) and the visual aspect of the granulometry was estimated (clay, silt, sand, and presence of shell pieces). Analyses were done on a sub-sampling of 50 stations (Fig. 1). These stations were randomly selected in proportion to the number of stations per layer and on the visual aspect of granulometry observed during the fieldwork. These 50 samples were treated using two different protocols: (a) the Loss On Ignition procedure (LOI) was used to estimate the organic content (%MO) and the proportion of carbonates (%CaCO3) (b) the laser diffraction grain size analysis gives sediment size distribution of each sample. We chose to illustrate the texture using the scale proposed by Blott & Pye (2012).

Loss On Ignition (LOI)

The LOI procedure is a modification of the procedure described by Dean (1974). Each sample was subsampled and weighed to obtain a minimum of 10 g of sediment. The %MO was obtained after heating the sample to 550 °C for a minimum of 6 h. Then, the sediment sample was heated at 925 °C for 12 h to approximate the %CO2. Regular tests on replicates and carbonates standard made in CEREGE Sedim Laboratory show that there is a mean error of 10 % on carbonates estimations (D. Delanghe, pers comm). To calculate the %CaCO3 into the sediment, the %CO2 (with a 10 % error) was multiplied by the molecular conversion factor (2.27).

Laser diffraction grain size analysis

All of the 50 samples granulometry was performed using the laser diffraction grain size Beckman Coulter LS 13,320 laser granulometer (range of 0.04-2,000 microns in 132 fractions). The analytical parameters, procedures and accuracies are detailed in Lepage *et al.* (2019) and Psomiadis *et al.* (2014). All samples were mixed with a dispersing agent (0.3 % sodium hexametaphosphate) to disperse the clay particles. Each sample was subsampled to obtain an obscuration window of the laser between 8 and 16 % and the light polarization between 50 and 70 %. The calculation model (software version 5.01) uses Fraunhöfer and Mie theory. Each sample was analyzed 5 times (90 seconds each) and the result was an average of the 4 last passages because some small bubbles can perturb the integration phase after the rinsing phase.

Berre sediment displays different size distributions with multimodal occurrences from the clays to sand. In some samples, sands were present in weak quantities and clays were numerous. In this case, for statistical reasons (number of particle occurrences) it is difficult to stay within the obscuration intervals and catch the larger particles (sand) signal. Therefore, the sample was separated by sieving at 63 µm and analyzed in two times (one sample clay and silts, one sample sands) following Lepage *et al.* (2019). In this paper accuracies and reproducibility of the Coulter are detailed for standards, natural samples, known mixtures.

RESULTS

Results analysis was performed using Spyder (Python 3.7) and QGIS 2.18.21.

Biological data

During this sampling campaign, the mean *R. philippinarum* density was 33.59 clams.m^{-2} (Standard Deviation, SD 73.14) between 0 and 5 meters in depth. This density was very variable depending on the location. Few hotspots can be seen with mean density superior of 250 clams.m^{-2}: in the west, in the north and in the east (Fig. 2). Otherwise, clam density was low (around 15-30 clams.m^{-2}) or null. In more than 57 % of the stations, clam density was null.

The total abundance of whole stock was estimated at around 552.07 million (SD 75.02) equivalent to a total biomass of 2,903.62 tons (SD 432.68).

The abundance of juveniles has been evaluated to 106.78 million (SD 10.52) corresponding to 19 % of total *R. philippinarum* abundance. Total juvenile biomass has been estimated to 38.13 tons (SD 2.97). Only one hotspot was observed, in the west (the same as for the whole stock) where the mean density is superior to 250 juvenile clams.m^{-2}. Except for this spot, the juvenile density is quite poor: in 73 % of the stations, the mean density of juvenile was null.

Exploitable clam was found in the same zone as for the whole stock: in the west and in the east (mean density superior to 250 clams.m^{-2}). In the rest of Berre lagoon, mean density was very poor: in 79 % of the stations, zero exploitable clams were observed. The total abundance of exploitable *R. philippinarum* was estimated at around 193.05 million (SD 29.63), which corresponds to 35 % of global stock. Total exploitable biomass was evaluated to 1 837.69 tons (SD 270.42) between 0 and 5 meters. Accessible stock for fisherman was between 0 and 2 meters (fishing clams only by walking into the Berre lagoon), this stock was estimated to 582.26 tons (SD 78.21).

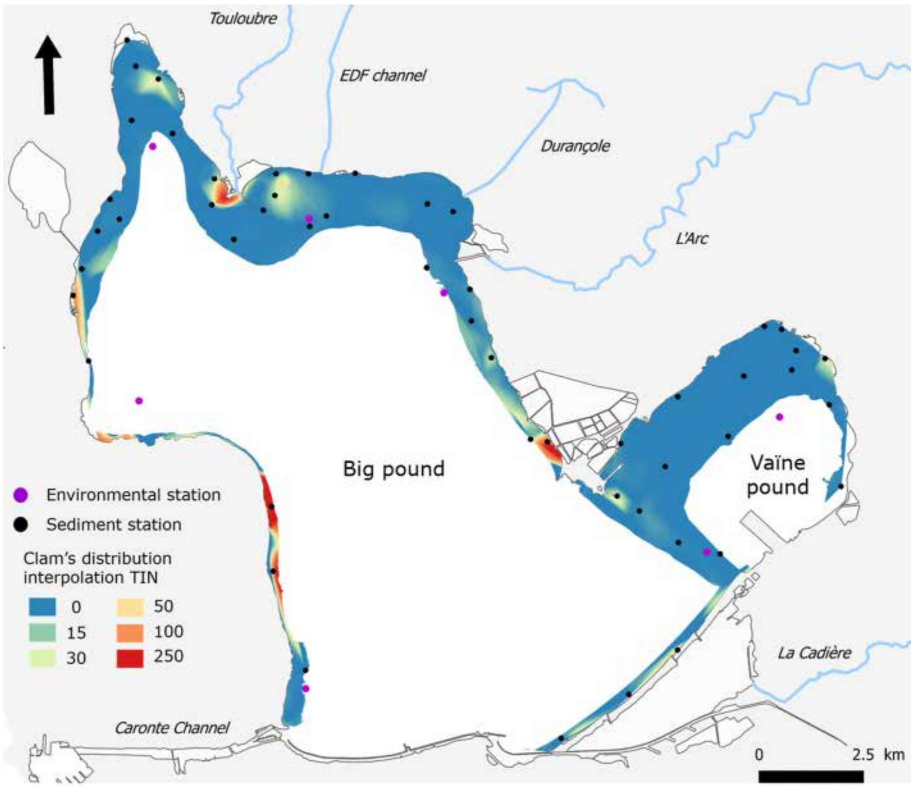

Fig. 2. – Result of clam's distribution by the interpolation from 238 stations (Triangulated Irregular Network with 2,000 columns and cells).

Hydrological data

In this study, 75 % of shell clam measured less than 32.19 mm. According to the von Bertalanffy curve, *R. philippinarum* needs around 3 years to reach 32.19 mm in Berre lagoon (data to be published). This means that these clams became established three years ago, in 2016. To link the hydrological condition to the clam distribution: temperature, salinity, chlorophyll *a*, dissolved oxygen and suspended matter will be studied starting from May 2016 and the average on each parameter on depth was calculated (Table I).

To compare the distribution of hydrological data, stations were studied separately (Fig. 2). Data for each variable at each station do not follow a normal distribution (Shapiro test, p-value < 0.05). Kruskall-Wallis test shows no differences between station for the water temperature, salinity and chlorophyll *a* (p-value > 0.09).

Table I. – Mean, standard deviation (SD), minimum (min) and maximum (max) of each sediment and hydrological parameters since May 2016.

	Mean	SD	Min	Max
MO (%)	6.29	3.71	0.89	16.33
$CaCO_3$ (%)	52.49	10.92	31.07	74.27
Temperature (°C)	16.77	7.16	2.59	30.07
Salinity	23.50	4.73	0	38.08
Dissolved oxygen ($mg.L^{-1}$)	8.03	2.42	0	16.42
Suspended matter ($mg.L^{-1}$)	5.24	8.29	0	81.50
Chlorophyll *a* ($\mu g.L^{-1}$)	6.40	8.76	0.24	106.77

Conover's test was used to make multiple comparisons of mean rank sums between each station for dissolved oxygen and suspended matter (p-value < 0.05) but no clear relation with the clam spatial distribution was found.

Sediment data

It should be noted that sediment data from SB4 station was taken off. The SB4 station had a very large number of clams (bigger mean density of 736 clams.m^{-2}, SD 398.14). The feeling was that the clams were packed like sardines. So far, it has been extremely difficult to sample the sediment compared to the predominance of clams. This point is therefore considered out of the general dynamics of the lagoon and has been removed from our results.

Loss on ignition (LOI)

Into Berre lagoon, the proportion of organic matter (% MO) is found to be quite low according to the vicinity activities, between 0.89 and 16.34 %. The proportion of the $CaCO_3$ (%$CaCO_3$) varies between 31.07 % and 74.21 % (± 10 %) (Table I). The dispersion of these values shows a large variation depending on the location in the lagoon, but without showing any particular pattern with the clams abundance.

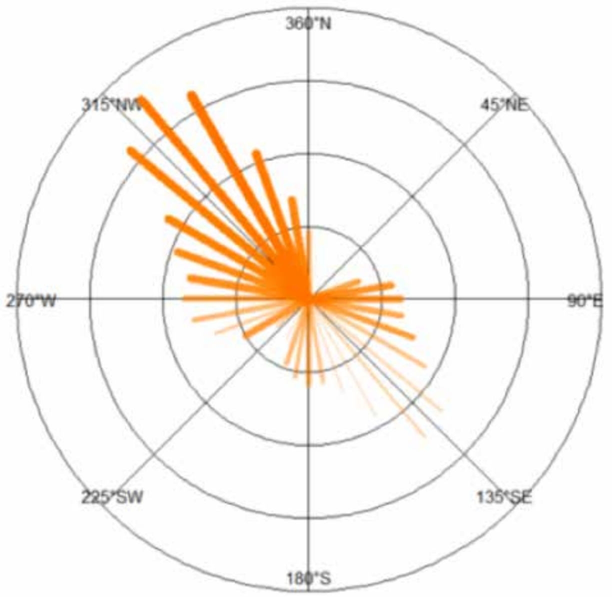

Fig. 3. – Wind direction rose from 2005 to 2019 by frequency (step of 2 %). The wind of 320° is the most frequent in 7.2 % of cases. Color and thickness represent the intensity of the wind. Except for winds from NW sectors, the wind intensity has little correlation with its most frequent direction. Winds from E-NE blow stronger but less frequently than SE winds. The SW wind of fairly strong intensity is infrequent (2 %) and corresponds to summer thermal breezes (from D. Nérini, data to be published).

Laser diffraction grain size analysis

The particle size varies greatly all around the lagoon, from very fine clay to very coarse sand. Well-sorted sands are found along the northwest exposed coastal areas while very fine clays and silts sediments are lying at low depths along the South East orientations coastal zones. The distribution is related to the main wind of the region, the Mistral, from the North-West sector (Fig. 3, Nérini 2000). Another identified pattern is linked with the industrialized and freshwater input zones directly connected to the lagoon or in its vicinity where size fractions display large heterogeneities with multi-modal distributions.

It seems that grain size distributions are influenced by wind directions but with no clear relation with the *R. philippinarum* density (Fig. 4). There is no particular pattern shown in Fig. 4 between the proportion of each texture and the increasing clam density at the stations. However, poor sorting distributions are highlighting anthropogenic areas.

To refine this observation, 4 groups of stations were defined depending on their clam density. The first group merges stations with a poor clams density (0.53 clams.m^{-2}, SD 0.65): SP8, JP6, JB2, PP60, PP20 (see station coding in Fig. 1 and station density in Fig. 2). The second group includes stations with 5.87 clams.m^{-2} (SD 4.43): PB15, BB1, BP16, MP19, CB18. The third group associates AB2, DB5, CB20, which have a mean *R. philippinarum* density of 28.44 (SD 3.33). And then, the last group regroups together stations with the big-

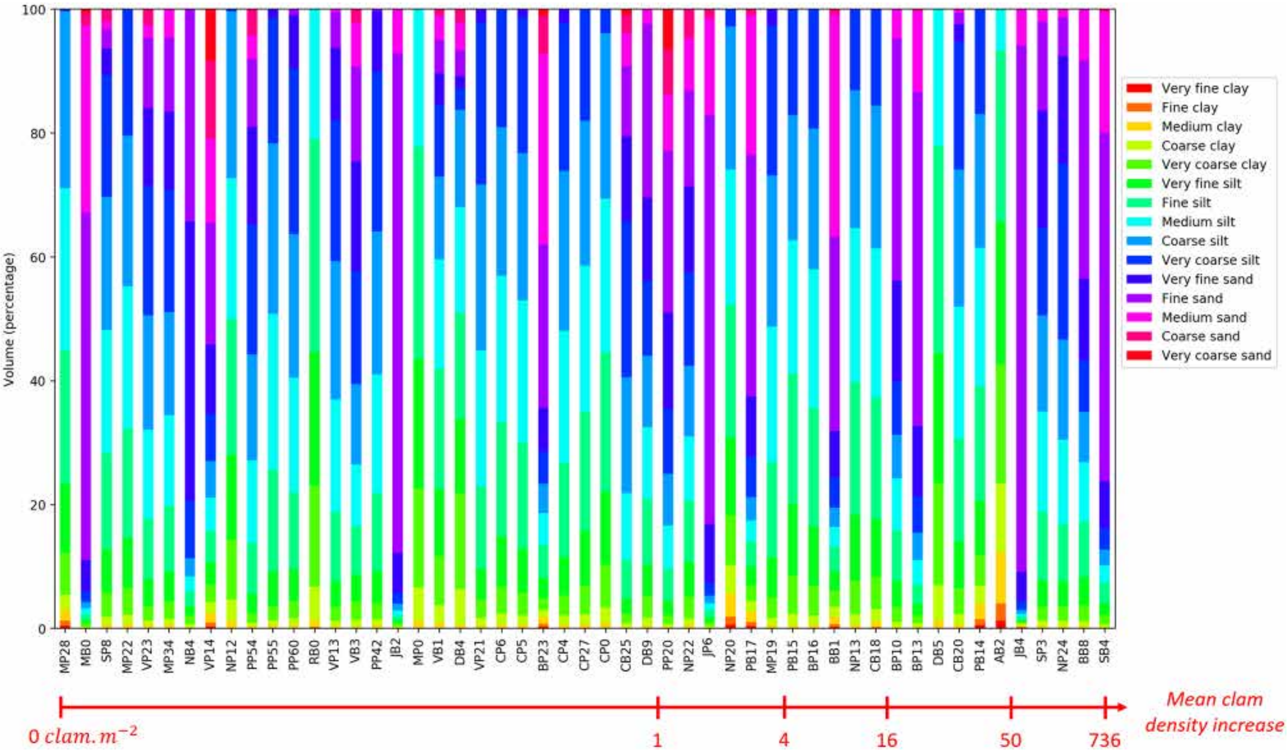

Fig. 4. – Stacked bar plot of sediment texture (scale of Blott & Pye 2012) depending on clam density (without H_2O_2 treatment).

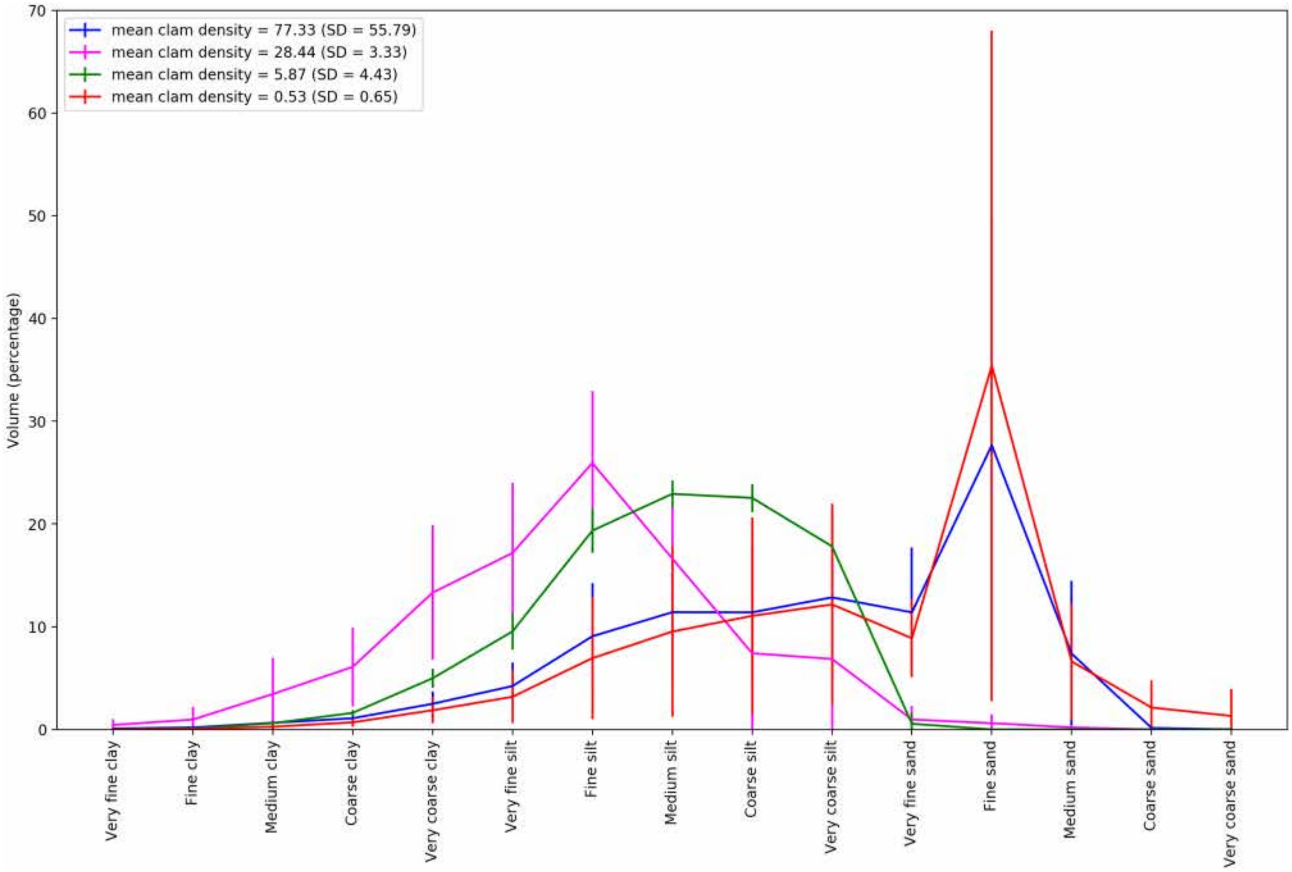

Fig. 5. – Texture of 4 groups of stations, according to the clam density. Number of stations by groups: blue = 6, red = 5, pink = 3 and green = 5.

ger clam density: 77.33 clams.m^{-2} (SD 55.79): NP24, SP3, PB14, BB8, BP10, BP13. Fig. 5 shows the texture of each group. Groups with a poor clams density, *i.e.*, group 1, and bigger clams density, *i.e.*, group 4, seem to display almost the same texture pattern. This observation is confirmed by statistical test (Mann-Whitney-Wilcoxon, p-value > 0.15).

No statistical difference between the depth of the group 1 and 4 using the Mann-Whitney-Wilcoxon method with a p-value > 0.39. Also, depth and textures from group 1 and group 4 do not show any particular correlation following this method.

Besides, sediment texture between the big pound and Vaïne pound was analyzed: no pattern between these two parts of the lagoon has been found.

Concerning the lack of consistency between the texture and abundance of clams, we supposed that the behavior of the particles in natural systems could influence the size distributions especially when organic matter and carbonates are present in the system (Slomberg *et al.* 2016). Another series of granulometry was carried out with removal of organic matter (with hydrogen peroxide, following Mikutta *et al.* 2005), but no pattern could be found.

DISCUSSION

An ecological inventory of a commercially exploited species such as *R. philippinarum* is important in a sustainable stock management principle for human exploitation (Caill-Milly 2012). Preliminary results show a total abundance of 552.07 million clams (SD 75.02) between 0 and 5 meters into Berre lagoon in 2019. Regarding the comparable results of Sanchez *et al.* (2014), since they have estimated their abundances using the same protocol during the year 2014 (Berthou *et al.* 1997). The Berre lagoon clam's stock is found to represent the second French stock, behind the Arcachon Bay (2,264 million), and in front of the Gulf of Morbihan (61 million).

The first objective of this study was to provide a snapshot of the density and abundance of clams after the ecological crisis of 2018 (Mayot *et al.* 2020). Historically, only data of accessible stock for fishermen has been estimated. The accessible stock was on clam with a shell length superior to 30 mm into the coastal zone (between 0 and 2 meters, regarding the Decree of 29 January 2013). According to the GIPREB data, in 2017 (before the opening fishing of *R. philippinarum* into Berre lagoon), the accessible stock was estimated to 2,200 tons. In this study, a sampling campaign was made in 2019, accessible stock was evaluated to 582.26 tons (SD 78.21). This reduction of more than 73 % of the accessible stock is found to be related to the anoxic crisis that occurred in the summer 2018 (Mayot *et al.* 2020).

This study establishes a "zero state" of the *R. philippinarum* stock after this ecological crisis. In the perspective of the reopening of the fishery, it would be interesting to follow a few layers of this campaign every six months to estimate the recolonization of the stock. This survey would allow monitoring the dynamics of the *R. philippinarum* population in a context of recolonization.

The second objective of this study was to understand the distribution of clams in the Berre lagoon as a function of the environmental conditions. Like all species, *R. philippinarum* requires optimal hydrological conditions to live. Water temperature needs to stands between 12 and 18 °C for the reproduction period and between 18 and 25 °C for optimal living conditions (Kang *et al.* 2016, Le Treut 1986). Salinity is another important environmental factor that controls the distribution of marine species, in particular through sudden changes in salinity that can have sub-lethal effects on the organisms (Nie *et al.* 2017). Salinity into Berre lagoon ranges between 12 and 32 – which allows optimal living conditions for *R. philippinarum* (Le Treut 1986). Oxygen is in normoxic condition, so clam can breathe normally. The suspended matter must remain as low as possible otherwise, the clam will stop filtering and be affected for reproduction or growth (Jones *et al.* 1993). According to Vincenzi *et al.* (2006), chlorophyll *a* can be used as a proxy of clam food and its optimal concentration for *R. philippinarum* biology is between 2 and 12 µg.L^{-1}. As shown in Table I, each hydrological parameter followed into Berre lagoon seems to be in the optimal range for clam biology all stations combined. This study does not show a link between abiotic conditions and clam distribution. This was quite surprising because according to Caill-Milly (2012), in the Arcachon bay, there is a strong link between the concentration of chlorophyll *a* (*i.e.*, food availability) and the stock state. Temperature also seems to play a role in the distribution of the stock into Arcachon Bay. These two parameters do not seem to be sufficient to explain the distribution of clams in the Berre pond.

With the preliminary results found in this study on a 50 sub-samples of the 238 stations, no clear relation is found between sediment data (texture from Blott & Pye 2012) and clams density. This result is unexpected compared to previous studies that showed a preference for *R. philippinarum* in sandy rather than silty environments (Le Treut 1986, Vincenzi *et al.* 2011). These studies seemed to highlight that with sand sediment, clams have a greater growth rate, higher maximum shell length and more successful juvenile settlement. On the contrary, into Berre lagoon, *R. philippinarum* densities do not seem related to the proportion of clay neither of sands (Fig. 4). For example, in AB2 station, a relatively high mean clam density (32 clams.m^{-2}) was observed with a high proportion of clay and silt (42.70 % of clay and 57.30 % of silt). Besides, the textures of stations with high clam densities show the same pattern as stations with low clam densities (Fig. 5). There is no statistical difference between these groups in texture and in depth. Our study may indicate that other environmental conditions have a predominant influence on clam growth, as shown by Sakamoto et Hirai (1984 *in* Artigaud *et al.* 2014), where no influence of sediment was observed with a salinity of 30 and a water temperature above 20 °C. In this lagoon, sediment composition and/or physicochemical conditions may have a greater influence on clam growth than grain size distribution. Overall, in the Berre lagoon, this study suggests that sediment texture does not appear to be a key factor in the distribution of *R. philippinarum*.

It should be stressed out that with the laser diffraction grain size analysis, the specter of grain size between 0.04 and 2,000 microns in analyses in made in one time. Often, coarse sediment (coarse sand and very coarse sand) are underestimated with this method (Lepage *et al.* 2019). According to Le Treut (1986), *R. philippinarum* can live in a habitat with very coarse sediment with rocks and shell pieces. Again, into Berre lagoon, it seems that with coarse sediment, no clam was found (Figs 4, 5) but this hypothesis needs to be confirmed by a mechanical sieving study. The question of the grain sizes and clam abundance possible relation remains therefore almost entire or local-dependent with winds directions and currents directions into the lagoons for nutrient-availability.

It's well known that *R. philippinarum* has an aggregated spatial distribution with a fine-scale (< 130 m, Beninger & Boldina 2014) but this analysis was made to try to understand the global repartition of clams at a large scale into Berre lagoon depending on hydrological and sediment data. Preliminary results submitted in this study do not show any correlation between environmental data and *R. philippinarum* distribution. The analysis is still in progress and another environmental factor can explain clam distribution into Berre lagoon.

The hydrodynamic regime can play a key role on the *R. philippinarum* biology, in the regulation of growth by the resuspension of food and on the reproduction by the transport of eggs and larvae (Abe *et al.* 2015, Kuwahara *et al.* 2016, Melià *et al.* 2004). However, the power of water current needs to be not too high because it can avoid the larvae settlement and cause their death (Le Treut 1986). The wind drives the resuspension of suspended matter than can feeds *R. philippinarum* according to the wind power, depth and also water current (Goulletquer 1989). Abe *et al.* (2015) show a significant relationship between current speed and growth rate of clam: with a water current of 0.15 m.s^{-1} the growth is higher than with a water current of 0.043 m.s^{-1}. They explain this difference by the role of current speed into the resuspension of suspended matter (such as microphytobenthos) as food sources for *R. philippinarum*.

Due to its shallow depth, the hydrodynamics into Berre lagoon is only caused by the wind that affects the entire water columns and causes currents and wind waves (Alekseenko *et al.* 2013, Paquier 2014). The two principal winds that affect the Berre lagoon are strong northwest wind, called Mistral, and southeast winds (Paquier *et al.* 2014). This lagoon is almost always under wind stress, with a wind speed more than 2 m.s^{-1} (Alekseenko *et al.* 2013). Strong wind, with a speed higher than 10 m.s^{-1}, is more common in winter and spring but still present in the other season (27.6 % from January to March, 26.6 % from April to June, 22.7 % from July to September and 23.1 % from October to December) (Paquier *et al.* 2014). Alekseenko *et al.* (2013) have shown the presence of a strong coastal jet into the Berre lagoon, which develops along the shore and under the wind in the big pound. This coastal jet creates a random bottom velocity near the shore. Depending on the wind, and the currents resulting from it, the clams are probably transported into the Berre lagoon in refuge areas, sheltered from hydrodynamics. These hypotheses must be confirmed by future studies.

In this study, only abiotic parameters were apprehended, but the distribution of clams may also depend on the biotic environment. A study is underway to look for traces of main clam diseases: the Brown Muscle Disease (BMD), the Brown Ring Disease (BRD), and the parasites *Perkinsus* (Dang 2009, De Montaudouin *et al.* 2016). The results of these experiments may be able to explain the distribution of clams. As far as predators are concerned, the best known predators of *R. philippinarum* are birds (such as gulls, seagulls and oystercatchers), fish (such as plaice, sea bream and triggerfish), crabs (especially green crabs) and starfish (Le Treut 1986). In the Berre lagoon, the amplitude of the tide is low (5-25 cm maximum, Nérini (2000)), so birds do not have access to clams. The main species fished in the Berre lagoon are European eels, mullet, sea breams and sea bass (GIPREB 2019), but so far, no evidence of predation has been found on *R. philippinarum* (GIPREB pers comm).

Finally, it is important to take into account that this study follows a major ecological crisis (Mayot *et al.* 2020). The distribution of clams in 2019 is therefore the consequence of this crisis of 2018. As no pattern of distribution is clearly obtained according to sediment structure or hydrological parameters, the survival (or refuge) zones should probably be linked to very local hydrodynamic force that could have preserved these zones from anoxic conditions. The future recolonization of Berre lagoon will be based on these high-density zones and the global hydrodynamic to disperses the larva. The continuous observation of the lagoon recolonization until stabilization would therefore be of high interest for the hypotheses tested in our study.

This work is part of a global project on *Ruditapes philippinarum* into Berre lagoon (GEPEPA). Many experiments on clam's biology are in progress (growth, reproduction, mortality, fishery). The purpose is to create a management model to find a sustainable way of exploitation for clams using DEB theory.

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Laws

Par le ministère de l'Écologie, du Développement Durable et de l'Énergie. Arrêté du 29 janvier 2013 modifiant l'arrêté du 26 octobre 2012 déterminant la taille minimale ou le poids minimal de capture des poissons et autres organismes marins (pour une espèce donnée ou pour une zone géographique donnée) effectuée dans le cadre de la pêche maritime de loisir. NOR : TRAM1240353A. JORF n°0045 du 22 février 2013 page 3024 texte n° 28 [en ligne] Disponible sur : https://www.legifrance.gouv.fr/affichTexte.do?cidTexte=JORFTEXT000027093867&categoreLien=id (consulté le 24 janvier 2017).

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A CITIZEN SCIENCE APPROACH FOR THE GENETIC ANALYSIS OF A *POSIDONIA OCEANICA* MEADOW IN THE ITALIAN SOUTHERN ADRIATIC SEA

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POSIDONIA OCEANICA
SEAGRASS
UNDERWATER VISUAL CENSUS
EPIPHYTE
MICROSATELLITE
GENOTYPING
CITIZEN SCIENCE
GENETICS

ABSTRACT. – A citizens' observatory, the Osservatorio del Mare a Molfetta (OMM), monitors the health status of the marine/coastal environment of the city. *Posidonia oceanica* (Linnaeus) Delile meadows are considered as bioindicators revealing the health status of the related marine/coastal ecosystem. To gather information necessary for the preservation of this ecosystem, OMM investigated the health status of the local San Vito-Barletta *P. oceanica* meadow extending along the coast of the Apulia region in the Italian Southern Adriatic Sea, which has been lately graded as 'disturbed'. Volunteers conducted underwater visual census of the epiphytes on *P. oceanica* leaves. Two campaigns of investigations at three different stations, namely Molfetta, Giovinazzo and Mola di Bari, evidenced differences in the distribution of the epiphytic communities. Furthermore, a population genetics study performed together with a scientific research centre using highly variable microsatellite markers showed an excess of heterozygosity and demonstrated the existence of at least two subpopulations within the meadow. Since the sampling areas were situated along the coasts of different urban centers, this situation may reflect the diverse anthropogenic pressures differing from city to city. Therefore, this study highlights the need to promote good conservation practices to preserve the San Vito-Barletta meadow and the whole related coastal/marine ecosystem.

INTRODUCTION

Posidonia oceanica (Linnaeus) Delile meadows are good biological indicators of the health status of marine ecosystem. Anthropogenic and environmental factors are causing the alarming decline of this marine flowering plant (Boudouresque *et al.* 2006, Montefalcone *et al.* 2019). The withdrawal of *P. oceanica* correlates with decay in associated biodiversity and ecosystem services. Yet public awareness of the importance of preserving *P. oceanica* meadows is still very low. The benefits provided by the ecosystem located in the intertidal zones are poorly understood (Nordlund *et al.* 2018). The European Union's Habitat directive (92/43/CEE) considers *P. oceanica* beds among priority habitats, which are among the major targets for the protection of the Mediterranean Sea (Telesca *et al.* 2015). Unfortunately, seagrass meadows are often not taken into account for the purposes of coastal management (Grech *et al.* 2012).

The Osservatorio del Mare a Molfetta (OMM, sea observatory at Molfetta) is a non-profit organization of volunteers operating in the city of Molfetta in the Apulia region of southern Italy.

Very limited information is publicly available on the widespread San Vito-Barletta meadow, a Site of Community Importance (SCI) located along the coast of the Apulia region. In 1991, a first map assessed this meadow as "not very lush" (Ministero Marina Mercantile 1991). An update performed in 2004 revealed a severe decline in the spatial distribution of the *P. oceanica* meadow, in particular northwards of Bari (Costantino *et al.* 2010). More recently, a local incidence assessment study related to the extension of the harbor of the city of Molfetta performed in April 2010 and October 2011 showed that *P. oceanica* had survived only within the natural cove named Cala San Giacomo. The meadow was highly fragmented displaying numerous patches of dead matte with a mean density value assessed as 'low sub-normal density' (LSD), thereby classified as 'very disturbed' (Andreassi 2016). So far, no other studies have been carried out on the San Vito-Barletta meadow.

Herein we have explored the health status of the San Vito-Barletta meadow using leaf-epiphyte biomass as biological quality elements (BQE) that are efficient indicators of the health status of *P. oceanica* (Costa *et al.* 2015). Then, the spatial pattern and level of genetic variability of this meadow was determined using highly variable microsatellite molecular markers allowing exploration of the correlation between genetic/genotypic variability and longevity of the *P. oceanica* population.

MATERIALS AND METHODS

Community-based environmental monitoring: OMM volunteers are of all ages, origins, gender, educational level, religious or political beliefs. We are determined to preserve the marine and coastal ecosystem of the city of Molfetta. For this purpose, numerous actions such as photographic exhibitions, conferences, seashore and seafloor clean-ups are performed to raise the awareness of the general public. The authors of the present study are among the OMM members (including researchers from other research fields) who wanted to perform this study and publish its scientific data. Each of the authors freely contributed to this study according to their own wishes/ability/free time. MdV led and coordinated the teamwork.

This study has been carried out using volunteers' personal devices including, but not limited to, cars, cell phones, GoPro cameras, snorkeling equipment and personal computers. The expenses of this study have been covered by all the volunteers. However, to complete the genetic part of this study and access the required instrumentation, OMM was helped by a public sector scientific research centre.

Study sites: The San Vito-Barletta meadow extends over an area of 12,459 ha corresponding to a covered area of 11,213.1 ha for a 65 km stretch of coastline situated in the Maritime Domain of Bari and Molfetta (Apulia Region, Southern Italy) at a depth between 3 and 16 meters, 17°2'9.996"E; 41°4'8"N (Natura2000, Cartografia-IT9120009). This meadow is a SCI (cod. IT9120009) established according to the Habitat Directive 92/43/CEE of the European Union.

The Geographic Information System (SIG) using the open tool Google Earth (where *Posidonia* shoots are visible) was used to select the sites for investigation. For underwater visual census, three shallow water habitats hosting *P. oceanica* were chosen: Molfetta (MF) at Cala San Giacomo (41°12'51.1"N; 16°34'37.0"E), Giovinazzo (GV) at Cala Spiriticchio (41°10'55.8"N; 16°40'56.9"E), and Mola di Bari (ML) (41°03'10.4"N; 17°06'34.1"E) (Fig. 1).

For DNA extraction and microsatellite genotyping, only two of the above sampling stations were chosen: MF at Cala San Giacomo where the seagrass meadow is confined within a semi-enclosed natural cove, and ML where *Posidonia* shoots extend along a longitudinal coastline (Fig. 1). MF and ML are 48 km apart from each other.

Underwater visual census and sampling: In June 2017 and June 2018, MF, GV, and ML sites were monitored by scuba diving between 4 and 6 m depth (5 m depth medium, one dive of about 40 minutes for each year and site). Photographs and videos were recorded using a GoPro Hero3 Black TM.

For DNA extraction and microsatellite genotyping, sampling was performed by scuba diving by two trained volunteers. Importantly, sampling only concerns old single outer leaves from vertical shoots that were selected at a minimum reciprocal distance of 10 meters.

DNA extraction and microsatellite genotyping: Epiphytes were removed with a razorblade from the surface of each leaf. Then, leaves were washed in distilled water, cut in half and stored at –80 °C before DNA extraction. Each half of a single leaf was ground to a fine powder in liquid nitrogen in a precooled sterile mortar. 100 mg of powder were transferred to an Eppendorf tube and processed for genomic DNA extraction with Plant DNAzol (Invitrogen) reagent according to manufacturer's protocol.

DNA quality was assessed through 0.8 % (w/v) agarose/1X Tris-borate-EDTA buffer (TBE) gels containing 0.5 mg/ml ethidium bromide and visualized by UV light. DNA purity and concentration was measured using NanoDrop ND-Spectrophotometer (Thermo Fisher Scientific). Samples were genotyped at 14 loci previously developed in *P. oceanica* (Procaccini & Waycott 1998, Alberto *et al.* 2003). Microsatellites were amplified in multiplex PCR reactions. Samples showing missing data were not considered for all the following analysis. To evaluate evolutionary influences, we tested for Hardy-Weinberg genotype frequencies. Hardy-Weinberg equilibrium (HWE) *P*-value, the probability of a given Multi-Locus Genotype (MLG) occurring n times as a consequence of different recombination events was calculated with the Genclone software 1.0. Clonal diversity was

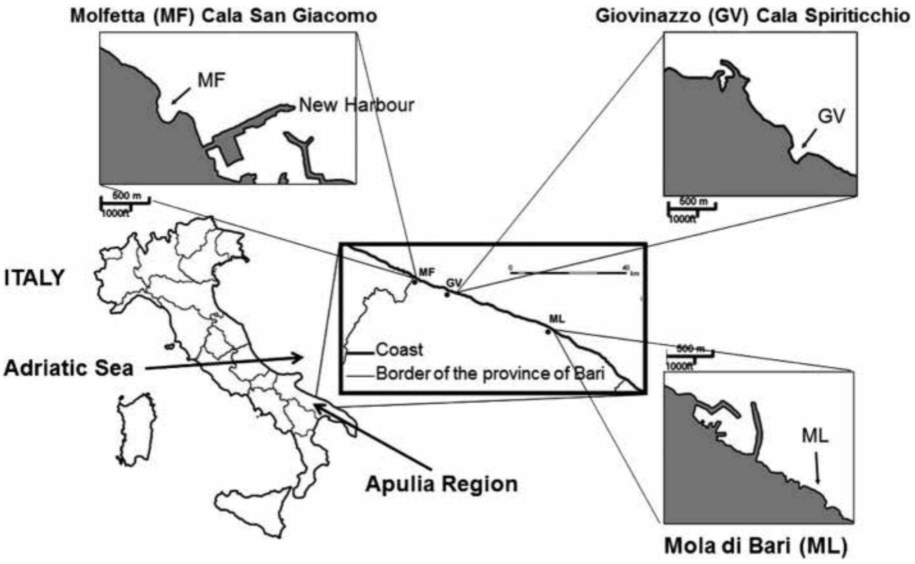

Fig. 1. – Location of the study stations in the San Vito-Barletta meadow. Molfetta (MF), Giovinazzo (GV), Mola di Bari (ML).

Table I. – Population genetic parameters for the two *Posidonia oceanica* populations analyzed. The following parameters are shown: number of samples genotyped (N), percent of polymorphic loci, number of multilocus genotypes (MLG), genotypic diversity (R_{MLG}), total number of alleles (Na), number of private alleles (Pa), observed (Ho), expected (He) and unbiased expected (uHe) heterozygosity and fixation index (Fis). All parameters are based on MLG values.

Station	N	Polym. Loci (%)	MLG	R_{MLG}	Na	Pa	Ho (SE)	He	uHe (SE)	Fis (SE)
Molfetta	20	71.43	13	0.632	27	5	0.362 (0.106)	0.301 (0.064)	0.314 (0.067)	–0.228 (0.155)
Mola di Bari	21	71.43	14	0.65	25	1	0.347 (0.105)	0.263 (0.062)	0.273 (0.064)	–0.101 (0.184)

estimated as $R = G – 1/N – 1$, with G representing the number of MLGs discriminated in the sample ramets after correcting for null alleles. The analysis on the HWE *P*-value as well as all the following analysis was performed on MLGs.

Genetic and genotypic diversity: The software GenAlEx 6.502 was used to estimate the number of alleles (Na), the effective number of alleles (Ne), the significance of Hardy-Weinberg (HW) deviations per each locus, the number of alleles per locus (Na/L), private alleles (Pa), observed (Ho), expected (He) and unbiased expected heterozygosity (uHe), Fixation index (Fis) per population (Table I).

Genetic differentiation: The inbreeding coefficient value (Fst) between the two sub-populations was calculated with GenAlEx 6.502. To visualize genetic relatedness, a Bayesian clustering analysis implemented in STRUCTURE 2.3.4 was utilized on the MLGs matrix. The number of clusters (K) was estimated using the online software STRUCTURE HARVESTER. To assess the component of genetic variance associated with different levels of analysis present in our dataset, an analysis of molecular variance (AMOVA) was performed using GenAlEx ver. 6.502. Components of genetic variance were computed at these hierarchical levels: between MF and ML, among and within sub-populations.

RESULTS

Underwater visual census of leaf epiphytes

At all stations, the epiphytic organisms on *Posidonia* leaves showed a clear apical-basal zonation mainly on the apical portion of the leaves. Differences were observed in both abundance and composition of leaf epiphytic communities (Fig. 2). In particular, biomass of filamentous epiphytes was more abundant on the leaves of station MF compared to GV and ML. These data imply that the health status of the San Vito-Barletta meadow is heterogeneous.

Genetic diversity

Genetic analysis was performed using highly variable microsatellites markers. The number of individuals sampled for each population was N = 20 at MF and N = 21 at ML. The proportion of polymorphic loci (71.43 %) was identical. The number of private alleles was significantly higher at MF (Pa = 5) than at ML (Pa = 1). 14 MLGs were found in ML and 13 in MF. Genotypic richness was lower than the average reported value in the literature (Jahnke *et al.* 2015) and slightly but not significantly higher at ML (R = 0,650) compared to MF (R = 0,632). Both MF and

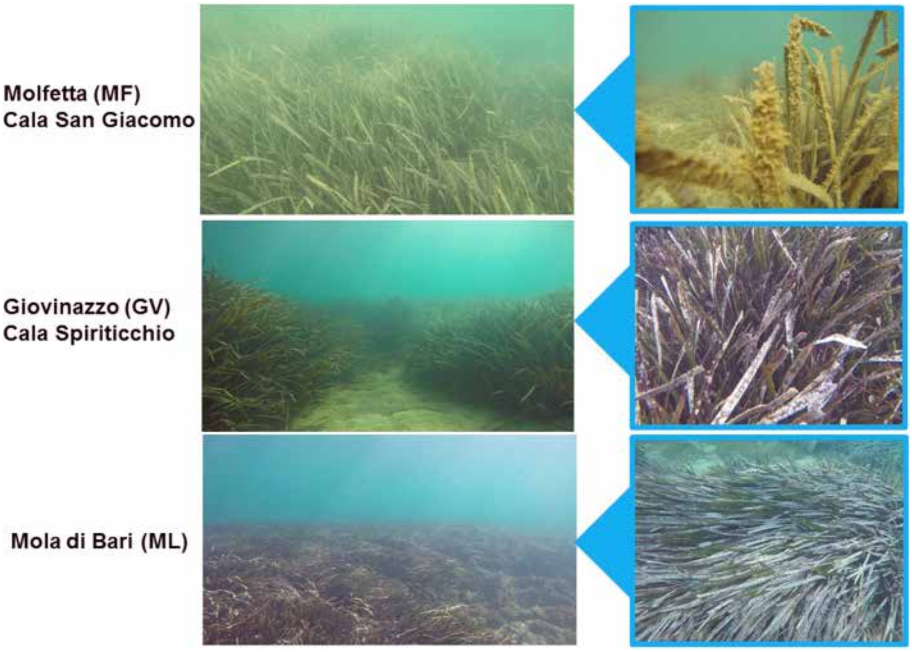

Fig. 2. – Distribution of *P. oceanica* leaf epiphytes. Molfetta (MF), Giovinazzo (GV) and Mola di Bari (ML).

ML populations exhibited an excess of heterozygosity values (Ho), which were higher than expected as shown by the uHe values. The fixation index (Fis) was less than zero in both populations indicating an excess of outbreeding (Table I).

Genetic differentiation

The genetic divergence between MF ad ML populations was assessed using the average differentiation value, Fst. Its value, Fst = 0.065 (p = 0.021) shows a slight but significant divergence among ML and MF populations to a level comparable to sub-population structure detected in other *Posidonia* meadows of the Mediterranean Sea (Micheli *et al.* 2015, Table II). Furthermore, the level of gene flow is sufficiently high to restrict the effect of genetic drift (Nm = 3.614) between ML and MF stations. Based on Evanno's delta K statistics, the Bayesian analysis of population structure led to the identification of 5 genetic groups (Fig. 3; mean LnP(K) = –365.26). Lastly, the percentage of variance among sub-populations (14 %) shows the existence of a sub-population structure within the San Vito-Barletta meadow (Table II).

DISCUSSION

Using a citizen's science approach and non-destructive methodologies, we have assessed the health status of the *P. oceanica* San Vito-Barletta meadow. Firstly, underwater visual census showed that the meadow is heterogeneous exhibiting obvious differences at the three sampling sites. Then, genetic investigations further illustrated this heterogeneity indicating the existence of at least two sub-populations with significant genetic differentiation.

Leaf-epiphyte biomass is one out of nine descriptors of *P. oceanica* judged as appropriate for the application of the European Water Framework Directive (Costa *et al.* 2015). We have observed substantial differences. Epiphyte biomass on the leaves of *P. oceanica* at station MF is more abundant than those at ML and GV. The difference between stations MF and ML may be due to the fact that these two stations are located 48 km apart (GV is in between stations MF and ML, 10 km away from MF). However the key features causing the above differences can be one or several environmental factors, either natural or anthropic, or concurrent. One of the main factors influencing leaf-epiphyte biomass is the enrichment in nutrients frequently resulting from human activities. Alteration in the flux of nutrients in waters alongside the coasts is crucial and can be initiated by physical disturbances such as piers, harbor extension and/or coastal erosion (Howarth & Marino 2006). Thus, it is possible that the enlargement of the harbor at Molfetta (which began in 2008) has modified the seawater currents, altering the flux of nutrients and explaining the increase in leaf-epiphyte biomass observed at station MF. In addition, suspended particles in the water column reduced water transparency at station MF. This may be another consequence of the construction of the new harbor at Molfetta, which through changes in coastal currents and swells may have impacted the dynamics of sedimentation. Light reduction mostly due to sediment resuspension is known to reduce growth rate, shoot biomass, storage of starch in the rhizomes, positive C-balance and diminution of the meadow (Ruiz & Romero 2001).

Genetic diversity improves resistance and resilience of seagrass populations to environmental changes (Hughes & Stachowicz 2004). The slow-growing and long-lived *P. oceanica* is a marine plant capable of enduring environmental change through its genetic reservoir (Procaccini *et al.* 2007). Genetic variability of the MF and ML populations was assessed using microsatellite markers. Stations MF and ML are located 48 km apart from each

Table II. – Hierarchical analysis of molecular variance (AMOVA). AMOVA is based on the 14 microsatellite loci utilized. *P*-values were based on 1000 permutation tests. The following parameters are shown: degrees of freedom (df), Mean Square (MS), Variance Components, Percentage Variation.

Source of Variation	df	MS	Variance components	Percentage variation	Fst
Among Subpopulations	1	11.170	0.573	14	0.065 (p = 0.021)
Within Subpopulations	25	3.439	3.439	86	
Total	73		4.013	100	

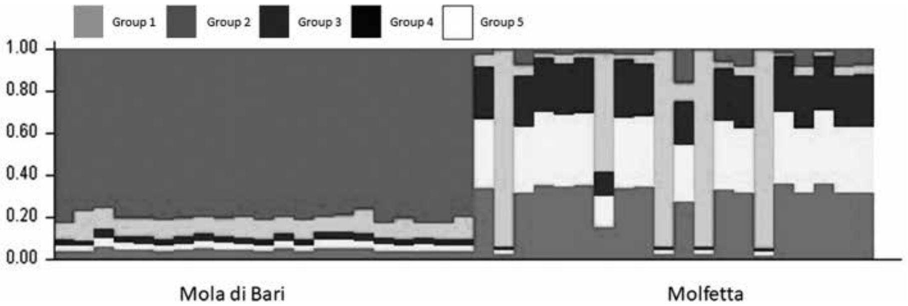

Fig. 3. – Bar plot resulting from the Bayesian analysis. The analysis was performed with the software STRUCTURE. For the two study stations, each multilocus genotype is represented as a vertical bar of different grey intensity in proportion to estimated membership coefficient (Q) to the K clusters.

other exposed to diverse urban environments. Our experimental data mirror this physical separation indicating the differentiation of stations MF and ML into two populations that are still closely related to each other. As shown by HWE *P*-values, not all of the loci were in HW equilibrium suggesting that these loci were under evolutionary influences generated by still unknown factors. Thus, stations MF and ML are most likely affected by diverse disturbance factors. However, both populations displayed an excess of heterozygosity that often occurs in *P. oceanica* (Micheli *et al.* 2015). Heterozygosity is related to fitness in various ways. The Fixation index suggests an excess of outbreeding in both populations, however *P. oceanica* is also a clonal species. Therefore, better-fitted heterozygous genotypes can become dominant via asexual propagation in a very confined environment with limited gene flow (Tomasello *et al.* 2009). This situation may pertain to the MF population, which is confined within a natural cove. Another possibility is that the excess of heterozygosity is due to the exposure to heterogeneous environmental conditions such as suspended particles in the water column observed at station MF altering flowering events. The existence of two populations at stations MF and ML that are genetically different is reinforced by the Fst value of 0.065 (p = 0.021). Moreover, the AMOVA analysis indicates genetic variations of 14 % between stations MF and ML and further supports the presence of two subpopulations.

In conclusion, our population genetic analysis was performed at two different sites of the San Vito-Barletta meadow using a minimum replication sample to decrease the impact of sampling on the meadow. Our data show: (i) the presence of two subpopulations, which have evolved under the pressure applied by diverse environmental and/or anthropogenic factors selecting more resistant clones; (ii) the moderate genetic and allelic richness displayed by both subpopulations is in line with the previously published means for *P. oceanica* meadows.

This study is the very first step in a wider ecosystem-based approach. It constitutes the first insight into the current health status of the ecosystem along the coasts of the Apulia region. *P. oceanica* meadows are biological indicators of the health status of the whole ecosystem because the ecosystem is dependent on *P. oceanica*, which provides oxygen, shelter, and stabilizes the coast, preserving it from erosion. However, to achieve a global view, and in phase with our ecosystem-based approach, we are also monitoring other parameters such as phytoplankton or *Ostreopsis ovata* Fukoyo, 1981 blooming (to be published elsewhere). Furthermore, our present data show that the two subpopulations of *P. oceanica* exhibit different health status; MF being more altered than ML. Therefore, it is more likely that the preservation of the whole ecosystem will require specific actions from site to site.

The previous genetic diversity of the San Vito-Barletta meadow is unknown. Thus, this study will be useful for a broader meta-analysis on *P. oceanica* populations with the goal of resolving the long-standing debate on the correlation between genotypic diversity and resistance/resilience. Our idea is to foster the use of genetic diversity as an essential tool in conservation management.

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TOXIC ALGAL BLOOMS ALONG THE URBAN COAST: A CITIZENS' POINT OF VIEW

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ALGAE
TOXIC BLOOMS
OSTREOPSIS OVATA
SOUTHERN ADRIATIC SEA
CITIZEN SCIENCE

ABSTRACT. – In this study, a Citizens' Observatory, the Osservatorio del Mare a Molfetta (OMM), has monitored the proliferation of the toxic microalgae *Ostreopsis ovata* along the coast of Molfetta, an Italian city on the Southern Adriatic Sea, where blooms have been reported since summer 2009. To establish an alternative low cost and effective monitoring protocol, planktonic cells were counted for four years in seawater sampled at two stations along the urban coast. We show that: (i) *O. ovata* blooms occur along the coast of the city of Molfetta; (ii) the presence of *O. ovata* was detected from May up until January of the following year; (iii) blooms of *O. ovata* are concomitant with those of non-toxic diatoms of the genus *Coscinodiscus*, which were predominant in the phytoplankton community; (iv) during the time frame of this study, *Ostreopsis* and *Coscinodiscus* proliferation start at the station closest to the urban centre. The traditional *O. ovata*-monitoring protocols, based on single sampling station/city every 15 days between June and September, need improvement. The weekly sampling of seawater along the urban coast from June to January is an effective and low-cost method to predict toxic algal blooms in our city. As a local citizen's observatory, we intend to support traditional monitoring programs by providing our data set to improve the surveillance.

INTRODUCTION

Benthic Harmful Algal Blooms (BHABs) are detrimental to the environment, marine organisms and humans (Faimali *et al.* 2012). The microalgae of the genus *Ostreopsis* live attached to benthic substrates such as brown seaweeds, rocks, shells of invertebrates and proliferate at high rates in shallow coastal areas close to the shoreline down to a few meters of depth. In the Mediterranean basin, *Ostreopsis ovata* Fukoyo, 1981, is one of the most common microalgae of tropical origin that produces palytoxin-like toxins (ova-toxins) (Fukuyo 1981, Penna *et al.* 2012). Ova-toxins represent a threat for human health via entry into the human food chain, inhalation or direct contact when cells are resuspended in surface seawater (Totti *et al.* 2010, Vila *et al.* 2012, Pelin *et al.* 2016). *Ostreopsis ovata* blooms in the Mediterranean Sea have become more frequent gaining the attention of scientific communities, managing institutions and public governance. Since the earliest 1990s, high concentrations of *O. ovata* have been recorded all around the Mediterranean Sea (Gallitelli *et al.* 2005, Kermarec *et al.* 2008, Blanfuné *et al.* 2015), along the Italian coasts, in the Ligurian, Sardinian, Tyrrenian, Ionian, and the northern and southern Adriatic Sea (Totti *et al.* 2010, Mangialajo *et al.* 2011). A notable bloom took place in 2005 in northern Italy along the coastline at Genoa where more than 200 cases of infection were recorded with clinical symptoms such as skin erythema, dispnoea, colds, fever and conjunctivitis (Ciminiello *et al.* 2006, Mangialajo *et al.* 2008, Tichadou *et al.* 2010). Most of the cases of infection were associated with inhalation rather than direct contact (Durando *et al.* 2007). Since summer 2009 in Molfetta (the city were the present study took place), regular monitoring of *O. ovata* blooms at coastal and recreational seawaters has been carried out by the regional agency for environmental protection every two weeks from June to September at a single sampling station (at Prima Cala in Molfetta), enabling the detection of blooms mostly in August and September (Ungaro *et al.* 2010, http://www.arpa.puglia.it).

Herein, trained volunteers from the Osservatorio del Mare a Molfetta (OMM) report the occurrence of *Ostreopsis* blooms at two public beaches on the waterfront of Molfetta. From July 2016 to December 2019, the concentration peaks of *O. ovata* were closely monitored on the basis of weekly sampling as well as its occurrence well beyond summer. Our data show that further efforts are required to understand the ecology of *O. ovata* and to improve the surveillance and alert systems in our city.

MATERIALS AND METHODS

Community-based environmental monitoring: OMM is a community of volunteers dedicated to promoting through the participative methodology of citizen science: i) the visual census and protection of coastal and marine ecosystems at urban level; ii) the public's right of access to environmental information in order to empower citizens to make key decisions relating to environmental issues. In addition, OMM provides incentives to scientists to engage with citizens in order to establish the absence or presence of a cause-and-effect association between marine biodiversity loss and habitual pollution/consumption practices.

Using their own devices such as light microscopes, computers, smartphones and cell counting chambers, OMM volunteers have carried out bibliographical research involving background reading of publicly available environmental data, sea water sampling, cell counts and data analysis.

Study sites and sampling: In the city of Molfetta, two public beaches which are crowded in summer 5 km apart from each other were chosen as sampling sites: Prima Cala, which is closer to the urban center (41°11’5N; 16°36’54”E), and Gavetone (41°11’37N; 16°38’11”E) (Fig. 1).

Benthos was collected weekly at 8:00 am from July 2nd 2016 to December 27th 2019. At each site, 40 ml of seawater samples were taken close to the shore from the bottom of four shallow tide pools (0.2-0.4 m depth) using a 10 ml syringe with the tip cut off and collected in a single tube.

Cell counting and identification of planktonic species: Within 2 hours after sampling, collection tubes were gently inverted 10 times. 1 mL of seawater was taken, dispensed into the Sedgewick-Rafter counting chamber, and planktonic cells were counted in duplicate under the light microscope. Photographs were taken using personal mobile phones by laying the camera on the eyepiece lens of the microscope. Microalgae identification was performed by the use of publicly available guides (Avancini *et al.* 2006; https://www.algaebase.org/)

Statistical analysis: Statistical analysis was performed with the GraphPad Prism software using an ANOVA model followed by Tukey’s post-test.

RESULTS

Ostreopsis ovata *abundance*

Fig. 1 shows the location of the two sampling stations, Prima Cala and Gavetone, located 5 km apart.

From July 2016 to December 2019, weekly sampling of seawater showed that maximum blooming of *O. ovata* occurred in summer 2016, 2017, 2018 and 2019 (Table I). The maximum peaks of proliferation were detected in August at Prima Cala and September at Gavetone with an average of 14.50 ± 13.27 days advance at Prima Cala relative to Gavetone. At the latter sampling station, one exception occurred in summer 2016 with a major peak in August. In general, *O. ovata* was more abundant at Prima Cala than at Gavetone with highest concentration peaks that were 2.77-, 6.18-, 1.6- and 1.2-fold higher during summer 2016, 2017, 2018 and 2019, respectively. The highest peak was measured during the first week of August 2018 at Prima Cala with a value of 13,242 ± 315.36 cells/ml (Table I).

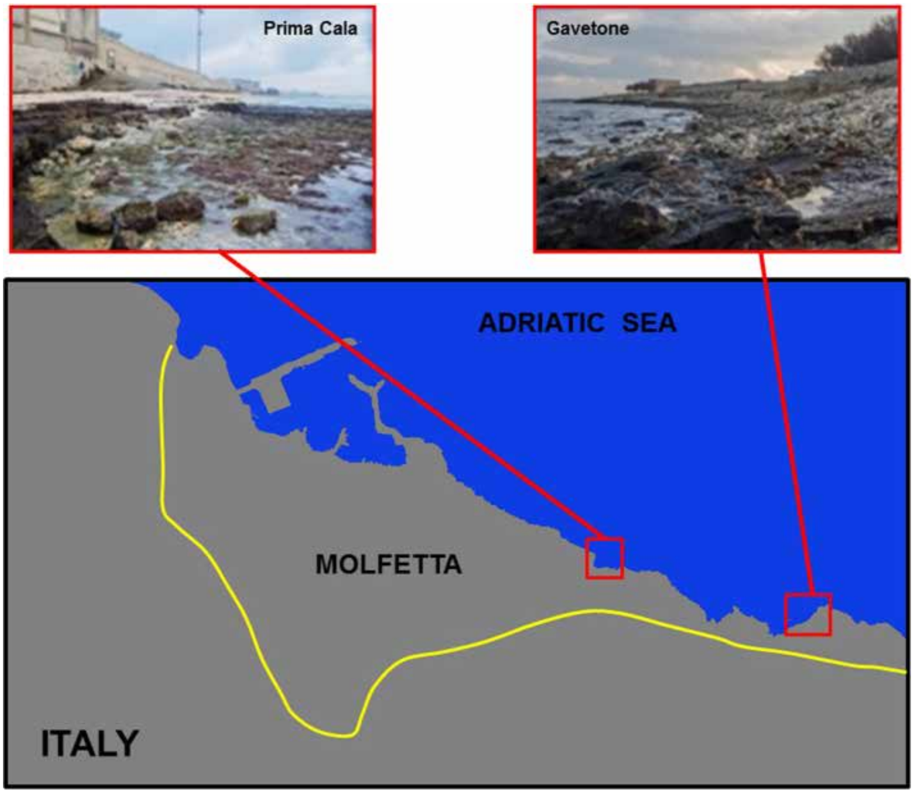

Fig. 1. – Study sites and sampling. In the city of Molfetta, two public beaches, crowded in summer, were chosen as sampling sites that are 5 km apart from each other. Prima Cala is closer to the urban centre (41°11’5N; 16°36’54”E) than Gavetone (41°11’37N; 16°38’11”E).

Table I. – Abundance of *Ostreopsis ovata* and *Coscinodiscus* sp. measured at the two sampling stations: Prima Cala and Gavetone. P < 0.05 is considered significant.

Species	**Sampling station**	**First detection**	**Last detection**	**Maximal peak occurrence**	**Cell number (cells/ml ± SD)**	**Significance**	**Difference between the two sampling stations**
Ostreopsis ovata	Prima Cala	Jul. 2016	Jan. 2017	Aug. 2016	6040 ± 14.85	P < 0.0001	P > 0.05
		Jun. 2017	Jan. 2018	Aug. 2017	6572 ± 197.98	P < 0.0001	
		May 2018	Jan. 2019	Aug. 2018	13242 ± 315.36	P < 0.0001	
		Jun. 2019	Jan. 2020	Aug. 2019	4791 ± 173.95	P < 0.0001	
Ostreopsis ovata	Gavetone	Jul. 2016	Jan. 2017	Aug. 2016	2173 ± 65.77	P < 0.0001	
		Jun. 2017	Jan. 2018	Sep. 2017	1063 ± 12.72	P < 0.0001	
		Jun. 2018	Jan. 2019	Sep. 2018	8064 ± 814.59	P < 0.0001	
		Aug. 2019	Jan. 2020	Sep. 2019	4140 ± 25.45	P < 0.0001	
Coscinodiscus sp.	Prima Cala	Jul. 2016	Feb. 2017	Aug. 2016	34553 ± 219.58	P < 0.0001	P < 0.01
		May 2017	Jan. 2018	Sep. 2017	27460 ± 2582.32	P < 0.0001	
		Apr. 2018	Feb. 2018	Sep. 2018	7983 ± 250.31	P < 0.0001	
		Mar. 2019	Jan. 2020	Aug. 2019	11232 ± 25.46	P < 0.0001	
Coscinodiscus sp.	Gavetone	Jul. 2016	Jan. 2017	Aug. 2016	3110 ± 57.98	P < 0.0001	
		Apr. 2017	Dec. 2017	Sep. 2017	7890 ± 33.94	P < 0.0001	
		Apr. 2018	Jan. 2019	Sep. 2018	4572 ± 280.01	P < 0.0001	
		Apr. 2019	Jan. 2020	Oct. 2019	9194 ± 463.15	P < 0.0001	

Ostreopsis ovata firstly appeared in July at Prima Cala and Gavetone in 2016 and in June at both sampling stations in 2017. In 2018, first detection occurred in May at Prima Cala and in June at Gavetone. Finally, in 2019, *O. ovata* was first detected in June at Prima Cala and only in August at Gavetone. Interestingly, the final detections of the dinoflagellata occurred in January of the following year. However, Tukey's post-test indicated that there is no significant difference (p > 0.05) in the distribution of *O. ovata* between the two sampling stations (Table I).

Abundance of associated diatoms

Other planktonic species have been observed together with *O. ovata*, in particular a variety of diatoms of the Bacillariophyceae class and the following genera: *Coscinodiscus*, *Licmophora*, *Gyrosigma*, *Cocconeis*, *Amphora*, *Campylodiscus*, *Achanantes*, *Bacteriastrum*, *Dactyliosolen*, *Tabellaria*, *Cyclotella*, *Navicula* and *Phaeodactylum*. *Coscinodiscus* and *Ostreopsis* were recorded as the predominant genera in summer. *Coscinodiscus* was more abundant than *Ostreopsis*, (with only one exception in summer 2018) and bloomed almost simultaneously in summer with *O. ovata* (Table I). *Coscinodiscus* reached peaks of proliferation higher than those of the dinoflagellate with concentrations that were 5.72- and 1.25- in 2016, 4.17- and 7.4- in 2017, and in 2019, 2.34- and 2.22-fold higher at Prima Cala and Gavetone, respectively. During summer 2018, *O. ovata* bloom started almost 4 weeks before than that of *Coscinodiscus spp.* at both sampling stations, and the dinoflagellate peaks were 1.62- and 1.01-fold higher than those of the diatom at Prima Cala and Gavetone, respectively. Tukey's post-test revealed that there is a significant difference (p < 0.01) in the distribution of *Coscinodiscus* between the two sampling stations (Table I).

DISCUSSION

In this study, volunteers at a citizens' observatory using their own devices have detected: i) the existence of *O. ovata* blooms along the coast at the city of Molfetta; ii) the presence *O. ovata* along the city waterfront from late spring up to winter; iii) the concomitance of *O. ovata* blooms with those of *Coscinodiscus*; iv) peaks of abundance of *Coscinodiscus* higher than those of *Ostreopsis;* v) both *Ostreopsis* and *Coscinodiscus* proliferations start at the station closest to the urban centre.

Recurrent health problems, *i.e.*, a respiratory syndrome caused by *O. ovata* blooming in the Mediterranean area, have necessitated the establishment of a surveillance and alert system in several countries including France, Italy, Monaco and Spain (Cohu *et al.* 2011a, b, 2013, Lemée *et al.* 2012, Vila *et al.* 2019). At Molfetta, at Prima Cala, regular public monitoring of the toxic microalga started in 2009 (http://www.arpa.puglia.it). The levels of *O. ovata* are measured during the summer season only. We have monitored the dinoflagellata over the whole year from July 2016 to December 2019 on a weekly basis at two sampling stations along the waterfront at Molfetta. *Ostreopsis* cells adhere to benthic substrates through filaments and mucilaginous substances forming mucilage and mats that can be easily detached from the benthos into the water column by high hydrodynamic conditions (*e.g.*, Vila *et al.* 2012, Mangialajo *et al.* 2008, Cohu *et al.* 2013). This method enabled us to demonstrate the presence of *O. ovata* over a long period of time that extends well beyond the summer season. *O. ovata* appears in spring and is present up to winter. *Ostreopsis* blooms have in fact been detected in October in the northern Adriatic Sea (Mangialajo *et al.* 2011). Moreover, our data show that peaks of proliferation occur at different time at distinct places. Pfannkuchen *et al.* (2012) demonstrated that *Ostreopsis* blooms might remain undetected with a high potential to affect human health at the coast. This may be particularly true with the present public surveillance system which triggers alert status when the cell density of *O. ovata* in the water column is above 30.000 cell/L, with a prolonged period of 1 to 7 days under conditions of low hydrodynamics and high temperatures (Funari *et al.* 2014). This idea is reinforced by our findings that *O. ovata* does not appear at the same time and is not present in seawater at the same concentrations at sites that are only 5 km apart.

Herein, we also report the co-occurrence of the proliferation of *Ostreopsis* and *Coscinodiscus*, which was observed during the four years of this study. One possible explanation is the formation of epiphytic dinoflagellate assemblage on macroalgae that have been previously described in the literature. Abundant components of these assemblages include *Ostreopsis* and the diatom of the genus *Coscinodiscus* (Vila *et al.* 2001). At the same time, a correlation between benthic diatoms species diversity and seawater quality has been recently described (Ryabushko *et al.* 2019).

In summary, this study shows that citizens using their own devices can provide useful data to monitor *O. ovata* blooms. In addition, since recurrent respiratory syndromes caused by inhalation of *O. ovata* have been recorded in the Mediterranean area, we recommend to policy makers and managers i) monitoring of the abundance of the toxic microalgae in the bioareosol, and ii) launching a territory-wide investigation on the algal respiratory syndrome in order to establish the appropriate threshold concentration to activate an effective alert system.

Further studies to elucidate the presence/absence of a relationship between *Ostreopsis/Coscinodiscus* proliferation and city-caused pollution such as agricultural and urban runoff are also necessary.

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THE UNIQUE FEATURE OF A *POSIDONIA OCEANICA* REEF AND LAGOON ALONG THE FRENCH MEDITERRANEAN COAST

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BARRIER REEF
LAGOON
POSIDONIA OCEANICA
SEAGRASS REEF
ANTHROPIC PRESSURES

ABSTRACT. – *Posidonia oceanica* is a seagrass, which can build original reef structures. A unique emerging *P. oceanica* reef was encountered in the Le Brusc Bay on the French Mediterranean coast. Extending over 1,600 m in width, the reef appears to be one of the last remaining large *P. oceanica* reefs. This reef has two distinct profiles: a barrier reef on the eastern front and a fringing reef on the western front. Located on the outer section of the bay, the barrier reef limits water turnover in the bay, thus forming a lagoon. The presence of three small passes in the inner section of the bay has shaped an original current dynamic. The unique morphology of the Le Brusc lagoon makes it a site of high naturalistic value. To assess the health status of the Le Brusc reef, the present study precisely maps it for the first time using unmanned aerial vehicle imagery coupled with field observations. We also studied its evolution since 1950 using historic aerial photographs. Results indicate a regression of the reef, mainly due to the artificialization of the coastal area and the development of harbor facilities, causing the loss of 29.3 % of the reef's surface including the disappearance of more than two-thirds of the barrier reef's profile. In addition, 8,180 m of boat stranding marks were observed in the barrier reef.

INTRODUCTION

Posidonia oceanica (Linnaeus) Delile is a seagrass endemic to the Mediterranean Sea. Vertical growth of *P. oceanica* results in the progressive burial of roots, rhizome fragments and leaf sheaths in the sediment forming a rot-resistant powerful rhizome stratum called 'matte' (Boudouresque *et al.* 2012). This unique biostructure can grow up to several meters high (Molinier & Picard 1952). In conditions of calm and shallow water, *P. oceanica* meadows can near the sea surface and leaves may partly emerge (especially at low tide or during the spring and summer seasons when the leaves are the longest), resulting in the formation of a reef. Different profiles of *P. oceanica* reefs have been identified. Unlike fringing or plateau reefs, which respectively grow parallel and perpendicular to the coastline, barrier reefs have the distinctive feature of being located in inner bay areas (Bonhomme *et al.* 2015, Rouanet *et al.* 2019). The sheltered innermost waters bounded by the reef (*i.e.*, the back-reef area) form an area with poor hydrodynamic and heavy sedimentation conditions. These lagoon-like conditions threaten the survival of *P. oceanica* and cause the regression of the reef in the lagoon and its spread towards the open sea (Molinier & Picard 1952, Boussard *et al.* 2019).

The Natura 2000 *Lagune du Brusc* site, located in the South of France near the city of Six-Fours-les-Plages, is a unique environment where *P. oceanica* forms a singular barrier reef structure. Unlike most barrier reef structures, located in the inner part of closed bays, the Le Brusc barrier reef grows on the outer edge of the lagoon on the northern side (Francour & Sartoretto 1991, Rouanet *et al.* 2008). The lagoon has the peculiarity of being delimited on its southern side by a string of small islands (Le Petit Gaou, Le Grand Gaou and Les Embiez Islands) allowing water renewal inside the lagoon through three passes. Despite its recognition as an exceptional natural site by several European and national management classifications – Natura 2000, special protected area of Mediterranean importance (SPAMI), Natural zone of ecological, faunal and floral value (ZNIEFF) and prefectural and municipal decrees – the Le Brusc lagoon suffers from major anthropogenic pressures (Holon *et al.* 2015). Since the beginning of the 21st century, major disturbances have been observed, such as the loss of the two species of seagrass meadows in the lagoon, *Cymodocea nodosa* (Ucria) Asch and *Zostera marina* Linnaeus (Couvray *et al.* 2020), compromising its fish nursery function (Kirchhofer *et al.* 2016). As the *P. oceanica* barrier reef plays a key role in the existence of the lagoon and the resulting ecosystem, assessing its present health status is crucial for the environmental management of this site.

We used an unmanned aerial vehicle (UAV) combined with orthophotography, field observations by snorkeling and hydroacoustic surveying to map the *P. oceanica* reef and Le Brusc lagoon. This work enabled us to produce an accurate representation of the Le Brusc *P. oceanica* reef and assess the direct impacts of boat stranding marks and harbor facilities on the reef, and supports the hypothesis of the indirect impact of water flow dynamic on the barrier reef health.

MATERIALS AND METHODS

Area mapping: The study area is located on the Mediterranean coast, in the South of France near the city of Six-Fours-les-Plages. Images of the Le Brusc lagoon site were taken in August 2019 and January 2020 using a Phanton 4 unmanned aerial vehicle (UAV). Up to 737 pictures were used for each photogrammetry. Orthomosaics were prepared using the DroneDeploy software program. Historic aerial photographs (source: National Geographic Institute, IGN) and, in particular the IGN 1950 aerial photograph, were used along with French administrative documents relative to the area to assess the development of anthropic facilities. Georeferencing and cartography analysis were performed using a Geographic Information System (QGIS 3.8, projection Lambert 93). *In situ* observations enabled us to validate the interpretation of the aerial photographs.

Bathymetry mapping: A hydroacoustic survey of the barrier reef and of the southern passes was carried out in 2018 by the engineering service SEMANTIC TS using a multi-beam R2SONIC 2020 echosounder. In very shallow areas, a mono beam eBEEM echosounder was used instead. Data collected with both sounders was merged to obtain the bathymetric chart of the Le Brusc lagoon and barrier reef with 0.25-meter precision. A previous hydroacoustic survey from 2006 conducted all around Les Embiez Island with 1-meter precision was used to determine the bathymetry of the full *P. oceanica* reef (fringing and barrier reefs).

Posidonia reef mapping: The Le Brusc *P. oceanica* reef was mapped according to the method detailed above. Visual differentiation of dead matte and living *P. oceanica* beds was possible for the barrier reef (north), but not for the smaller *P. oceanica* reef (westernmost southern pass) growing on rocky substrate. To fix the external limit of the fringing reef (north-west), 3 criteria were used: 1) absence of *P. oceanica*, or 2) presence of a vertical structure (*i.e.*, reef drop-off), or 3) closest to the 1-m depth limit (determined by the bathymetry mapping) following the natural topography of the meadow. The same criteria were used to fix the external limit of the barrier reef (north-east), except that the 0.75 m-depth bathymetry contour was followed to be in accordance with the depth of the reef's inner profile. To assess the global morphology of the barrier reef, we virtually separated the reef into 4 longitudinal and 5 transversal evenly distributed sections.

RESULTS

Lagoon and reefs

A fringing reef extends on the west side of the great barrier reef along the Les Embiez Islands, forming together a 1,600 m long *P. oceanica* reef from Le Brusc harbor to Les Embiez harbor (Fig. 1). In 2020, 18.7 ha were covered by the reef versus 26.4 ha in 1950. The lagoon covers 44 ha and the 3 southern passes maximum depths measured in the narrowest point are respectively, from west to east, 0.7 m, 0.8 m and 0.5 m. A second smaller *P. oceanica* barrier reef grows in the westernmost southern pass. It covers a total surface of 6,270 m^2 for 2,800 m^2 of living *P. oceanica* beds. From 1950 to 2020, we observe

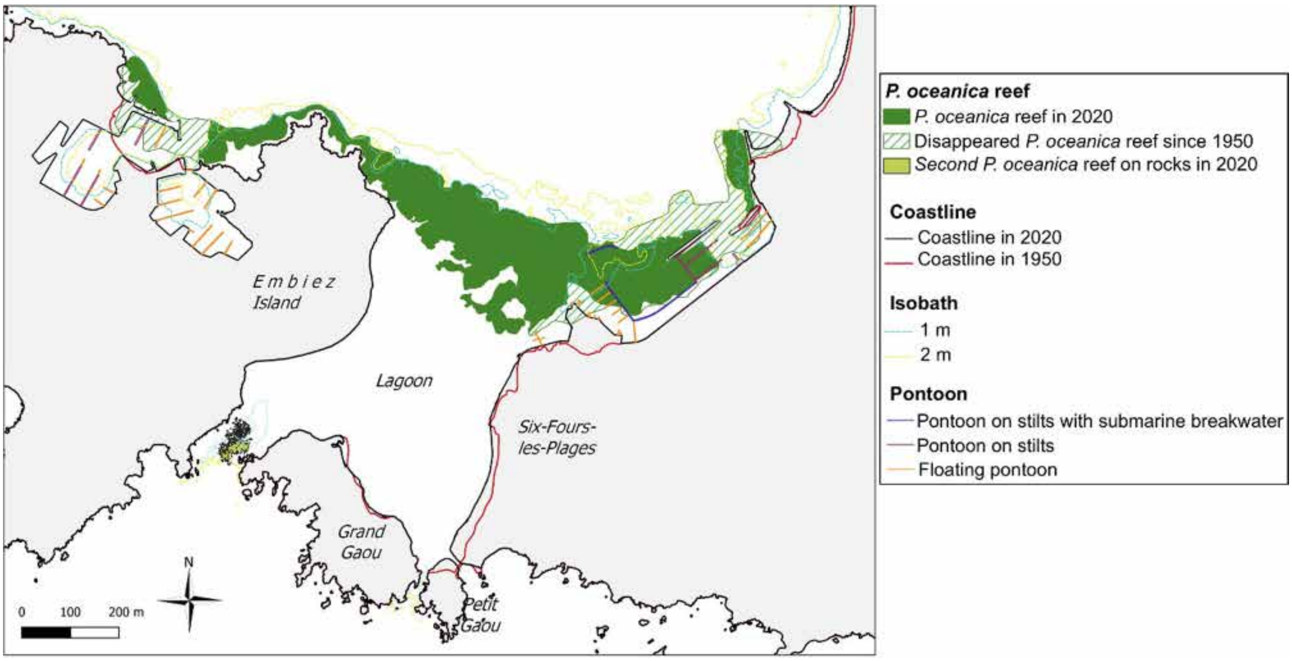

Fig. 1. – Le Brusc lagoon and *Posidonia oceanica* reefs, including harbor facilities and changes in the coastline between 1950 and 2020. The main reef in the north includes living *P. oceanica*, dead matte and natural passes. The second reef in the south includes only living *P. oceanica*.

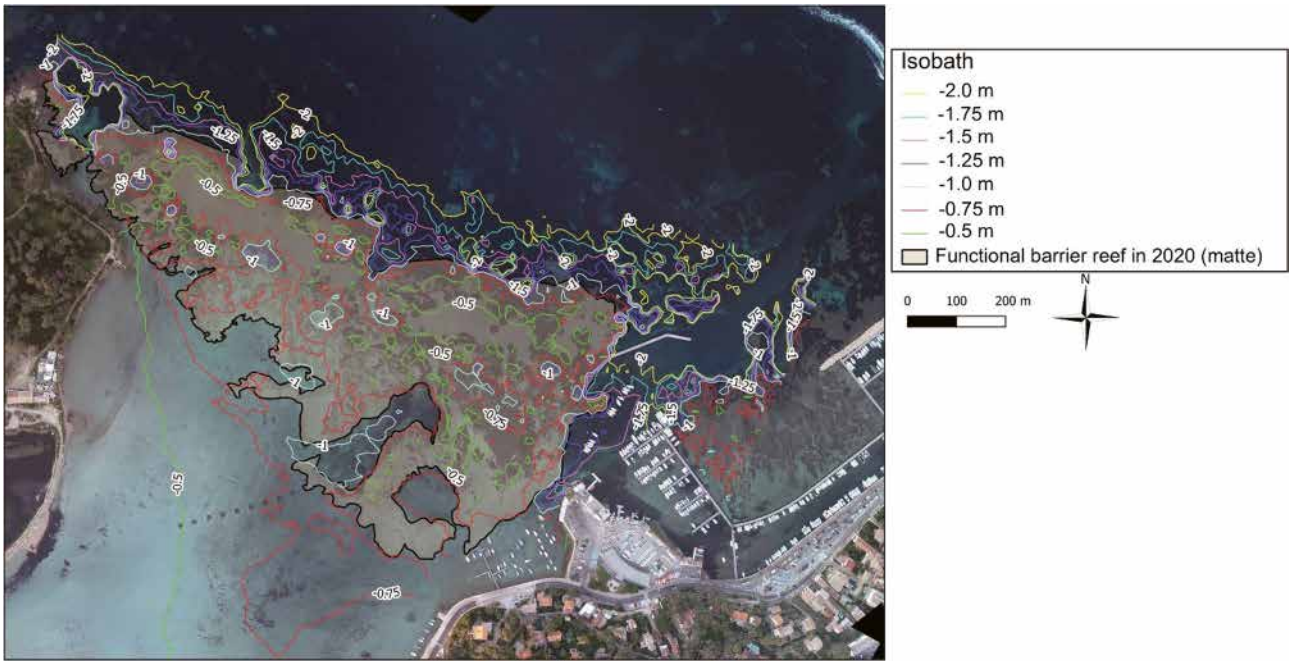

Fig. 2. – Functional barrier reef of the Le Brusc Lagoon and associated bathymetry. The functional barrier reef includes living *P. oceanica* and dead matte of the reef not directly impacted by the harbor facilities.

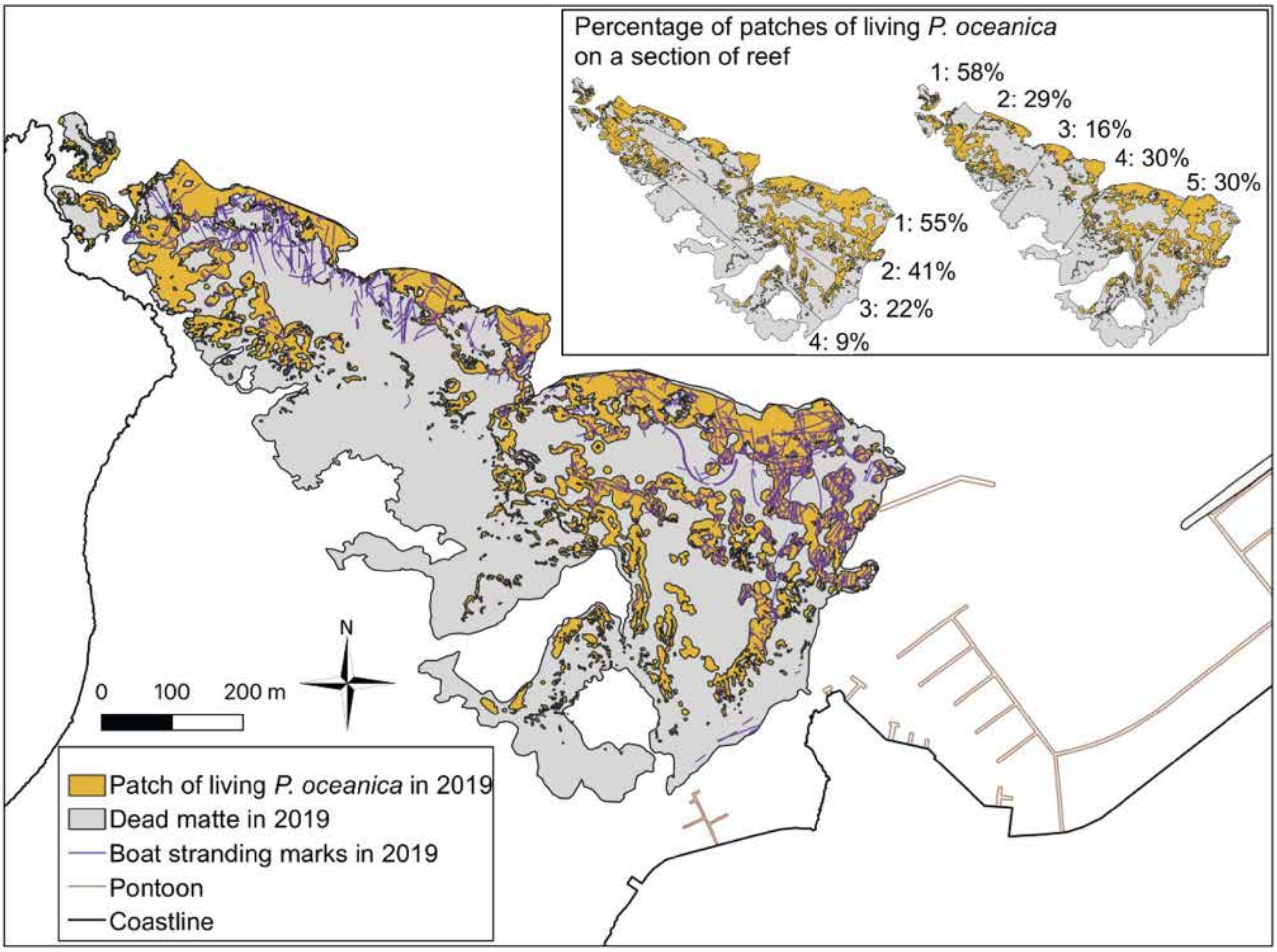

Fig. 3. – Patches of living *Posidonia oceanica* and dead matte in the functional barrier reef (*i.e.*, part of the reef not directly impacted by the harbor facilities) in 2019, including boat-stranding marks.

a regression of the lagoon surface and of the living *P. oceanica* mainly on the east side due to artificialization of the coastal area and the development of harbor facilities.

Great barrier reef

The cartographic analysis of the study site reveals the loss of the easternmost part of the barrier reef due the construction of the Le Brusc harbor facilities on it. In this area, only a few areas of silted dead matte and small patches of silted living *P. oceanica* remain. We therefore assume that the reef in this area no longer plays the role of a functional barrier reef for the lagoon. What remains of the barrier reef covers 10 ha and is delimited by two small passes. We consider this area as the functional barrier reef for the lagoon (Fig. 2). The morphology of the western pass resembles more that of a small water hole. The eastern pass is partly blocked by a boat launch ramp, pontoons on stilts and underwater breakwaters (Fig. 1). In 2019, our map revealed that patches of living *P. oceanica* only made up 29 % of the functional barrier reef. The patches were overall small and fragmented. We used the percentage of living *P. oceanica* on the reef surface as an indicator of the reef's vitality. We observed a difference in the proportion of the patches between the outer and inner areas of the barrier reef. The patches of living *P. oceanica* accounted for 55 % of the reef's total surface at the outer limit versus 9 % at the inner limit. We also observed a higher proportion of patches in the eastern and western peripheral sections of the reef in comparison with the centre (Fig. 3).

DISCUSSION

The Le Brusc reef, made up of both a fringing and a barrier reef, has been precisely mapped for the first time in this study. We have shown that this reef was continuous between the present Embiez harbor in the west and Brusc harbor in the east in 1950 (Fig. 1). Since then, Embiez harbor, built in 1962, and the construction of the main jetty and the digging of the channel, broke the continuity of the reef, isolating its westernmost section. On its eastern side, the successive developments until 2019 of the old Le Brusc traditional fishing harbor induced the destruction of part of the barrier reef. The reef suffered its greatest damage after the construction of the nautical base and the boat launch ramp in 1970. The artificialization of the beach just north of the Le Brusc harbor most likely also enhanced the degradation of the eastern part of barrier reef. In addition, successive coastline modifications to the east of the lagoon may also have had an indirect impact on the reef. As a consequence, the reef lost 29.3 % of its surface between 1950 and 2020.

Extreme temperature and salinity conditions (compared to those in open waters) arise in bodies of water enclosed by barrier reefs, generating stress for *P. oceanica* (Molinier & Picard 1952, Boussard *et al.* 2019). According to theoretical *P. oceanica* barrier reef dynamics (Molinier & Picard 1952), reefs tend to regress in the lagoon area and expand towards the open sea. Consequently, the presence of dead matte in a *P. oceanica* reef is common and not necessarily a sign of a declining health. At first glance, we can consider the decreasing outer to inner gradient of living *P. oceanica* observed in the Le Brusc barrier reef as not alarming (Fig. 3). However, the Le Brusc lagoon has the unique feature of receiving seawater through three inlets in the South. The reduced size and shallow depth of these passes, as well as the presence of a second *P. oceanica* reef in the main pass, reduces swell and wave action in the lagoon while allowing the water to circulate. The specificities of the Le Brusc barrier reef is its location on the outer part of the bay, which allows it to benefit from the rate of water-turnover made possible by the passes. Historically, the water flow used to enter the lagoon through the main southern pass, flowed along the eastern coastline of the lagoon and exit the lagoon through a natural pass east of the barrier reef (Blanc 1958). We can assume that the environmental conditions were suitable enough for *P. oceanica* due to the sufficient water-turnover. Nowadays, this large pass on the barrier reef is blocked by the nautical base of the Le Brusc harbor. Only two narrow inlets remain, one on the western end of the barrier reef and the other close to the former large pass in the east. The recent construction of new pontoons equipped with underwater breakwaters near this second inlet will most likely further reduce the water flow (Figs 1, 2). As described above, we observed an outer to inner gradient of the reef's vitality, which could be consistent with theoretical *P. oceanica* barrier reef dynamics (Molinier & Picard 1952). However, due to recent developments in the Le Brusc harbor, short-term changes in the vitality pattern should be assessed in the years to come. We also observed a higher vitality on both sides of the *P. oceanica* barrier reef, certainly linked to the presence of the passes, which favor water renewal. However, there were more patches of living *P. oceanica* on the western side of the reef than on the eastern side (respectively 58 and 30 %) (Fig. 3) even if the main flow was historically on the east side of the barrier reef (Blanc 1958). These results suggest the possible indirect effects of harbor facilities on the water flow and on the vitality of *P. oceanica* barrier reefs. Additional research on the hydrodynamics of the lagoon should be performed to complete our observations.

Francour & Sartoretto (1991) observed an abnormally high mortality rate of the Le Brusc barrier reef in the decades prior to their study. In line with their observations, our study showed that less than a third of the 1950 barrier reef surface area remained in 2019 (Fig. 3). We therefore hypothesize that a high level of human activity in this area (Holon *et al.* 2015) is a strong factor to explain the observed decline of living *P. oceanica* on the Le Brusc barrier reef. For instance, increasing boating activities increase the stress on the reef. In 2019, we recorded 8,180 meters of boat stranding marks on the barrier reef. Despite sailing and motorboats being prohibited in this area, several boat strandings are observed each year on the reef (personal observations). Such erosion of the barrier reef has also been observed at Port-Cros Island (Augier & Boudouresque 1970). Considering the very slow elongation rate of *P. oceanica* rhizomes, these marks accumulate year after year and fragilize the reef. Damaging or altering the vitality of such reefs is therefore irreversible on a human time scale.

The Le Brusc reef is one of the last remaining large *P. oceanica* reefs on the French Mediterranean coast. The goods and services provided by such ecosystems, such as climate change mitigation (Boudouresque *et al*. 2016), give grounds for their immediate protection. The barrier reef is vital to the Le Brusc lagoon ecosystem and is a part of our natural heritage. Although the lagoon is now closed to swimmers and sailing and motor boats to protect its nursing grounds, constant direct and indirect anthropogenic pressures continue to threaten the barrier reef and the lagoon on short time scales.

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STUDY OF THE *POSIDONIA OCEANICA* MEADOWS' UPPER LIMIT WITH GEOREFERENCED UNDERWATER PHOTOGRAMMETRY

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MAPPING
PHOTOGRAMMETRY
SEAGRASS
GEOREFERENCING
UNDERWATER

ABSTRACT. – Photogrammetry can be used to generate maps based on a large number of photographs and the tie points between them. Commonly used in terrestrial mapping with drones, georeferenced photogrammetry is currently seldom used in underwater studies. In order to obtain a centimetric positioning accuracy through direct georeferencing, we developed a floating platform encompassing two submerged cameras synchronized with a RTK GNSS at the surface. This device can be used from 1 m to 10 m depth in clear waters and the photographic views of the seafloor, with millimetric resolution, make it possible to map shallow habitats previously impossible to detect with classic methods such as acoustic sounding and aerial photographs. With this in mind, we used the platform to shoot 3 400 photos over a distance of about 500 m along the upper limit of the shallow *Posidonia oceanica* meadows of the Mugel, a creek in La Ciotat (France). The underwater photographs and the position data from the GNSS were synchronized to provide each picture with geographic coordinates. All the photos were then processed to build an orthomosaic of the meadow limit. Owing to its high resolution, the orthomosaic enabled the mapping of various features of the seascape such as the position of the limit with a centimetric accuracy, dead matte patches, litter detritus, artificial objects (moorings, wrecks) and marine organisms living on the seafloor.

INTRODUCTION

The meadows formed by the endemic seagrass *Posidonia oceanica* (Linnaeus) Delile constitute one of the most important marine habitats of the Mediterranean Sea due to the ecosystemic services (*e.g.*, shelter, nurseries, protection against coastal erosion, carbon sink, oxygen production) they provide (Boudouresque *et al.* 2012). Despite these key roles, they have been subject to anthropogenic impact resulting from human activities (*e.g.*, anchoring, trawling, coastal development, dredging) for decades. These activities lead to major changes and the fragmentation of the seascape formed by the meadows (Abadie *et al*. 2018a).

The shallowest extension of *P. oceanica* meadows, called the "upper limit", is the place where most interactions with human activities occur (*e.g.*, coastal development, tourism, leisure nautical activities). Due to the low depth of the upper limit – that may reach the surface in the case of barrier reef meadows – its mapping with conventional acoustic probes mounted on motorized boats is prohibited. Most of the time, the upper limit is mapped using aerial photographs with the disadvantage that *P. oceanica* meadows can be mistaken with rocks, dead matte and litter detritus.

Taking into account the limits of the classical mapping methods, the aim of this work was to develop a georeferenced photogrammetric technique to obtain underwater orthomosaics of *P. oceanica* meadows. Furthermore, this research involved the study of the information that can be extracted from photogrammetric products for ecosystem-based management purposes.

MATERIALS AND METHODS

The study took place at the Mugel Creek in the Bay of La Ciotat in the south of France within the Parc National des Calanques (Fig. 1A). Anchoring is forbidden in this small creek, which is characterized by shallow depths in its north-western part and an extensive *P. oceanica* meadow covering most part of the seafloor. Photogrammetric data (position and underwater photographs) were acquired on 27th August 2019 using the floating platform developed by Abadie *et al*. (2018b). Two hybrid cameras (Canon M50® and a Sony A6000®), equipped with 11-mm wide angle lenses in waterproof cases, were synchronized with a RTK GNSS (North RTKite®). The floating platform was operated at the surface above the western meadow limit over a distance of about 500 m by a snorkeller and 3 368 photos were taken in about 90 minutes.

After the image acquisition, the position data and the underwater pictures were synchronized by time, and XYZ attributes were written in the photographs' EXIF using ViewMap software developed by Seaviews. Photogrammetric processes, *i.e.*, the creation of a tie point cloud, the generation of a Digital Elevation Model (DEM) and finally the building of an orthomosaic were performed using the Metashape software from Agisoft. Finally, the orthomosaic was exported in ViewMap to generate

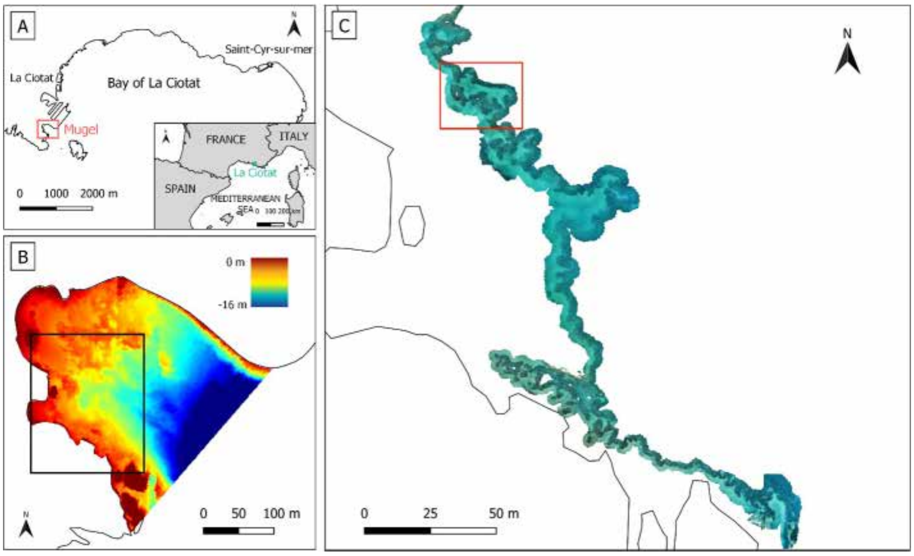

Fig. 1. – **A**: Study site (red frame) in the Bay of La Ciotat. **B**: Bathymetric map of Mugel Creek (data from the Litto3D program of the Hydrographic Service of the French Navy). The black frame represents the area where the meadow upper limit was mapped. **C**: Underwater orthomosaic of the meadow limit.

photographic tiles that allowed a better visualization to detect and map the various elements of the seafloor (*e.g.*, meadow, dead matte, moorings, rocks, wrecks, litter, sediments).

All the individuals of the fauna within the meadow and on the nearby various substrates were identified up to the species when possible. Their location was pinpointed and stored in a georeferenced computer file exploitable in a Geographic Information System (GIS). In order to investigate the potential of the biological data that can be extracted from the orthomosaic, all the substrates and habitats were manually mapped in ViewMap on a restricted portion of the whole orthomosaic, corresponding to a 350 m² surface area with depths ranging from 2 m to 7 m (Fig. 1B, C).

RESULTS

The orthomosaic clearly showed the *P. oceanica* meadow's limit that was very contrasted with the fine sediments at its edge (Fig. 1C). The pixel resolution obtained was 1 mm for a positioning precision of 7 cm over the whole area mapped. The high resolution allowed detection of dead matte and litter detritus areas as well as *P. oceanica* meadows, fine sediments and rocks (Fig. 2A). Artificial substrates were also identified such as old concrete blocks, tyre mooring systems (Fig. 2A), and small wrecks.

Five different sessile species were identified with various abundances: two snakelock anemones *Anemonia viridis* Forsskål, 1775, 285 sea-cucumbers *Holothuria* sp., two red starfishes *Echinaster sepositus* (Retzius, 1783), 57 brown sea-urchins *Paracentrotus lividus* (Lamarck, 1816) and eight fan mussels *Pinna nobilis* Linnaeus, 1758 (living and dead individuals). Most of the detections occurred on bare substrates, *i.e.*, rocks and sediments, and at the edge of the meadow. *Holothuria* sp. was mainly detected on the dead matte at the edge of the meadow. *P. lividus* individuals were observed in the cracks of shallow rocky bottoms. Dead *P. nobilis* were found lying on the dead matte while the erect ones were situated within the *Posidonia oceanica* meadow near the edge.

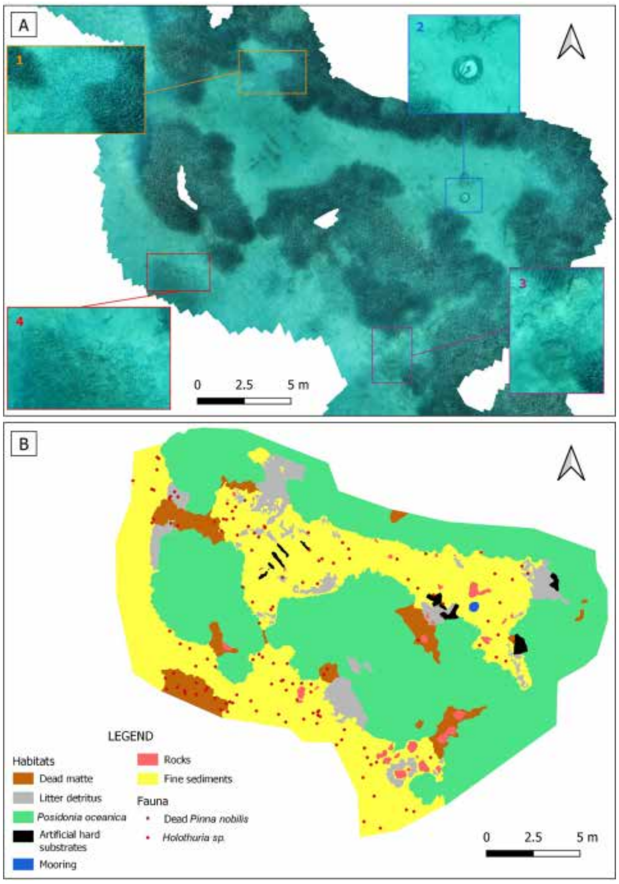

Fig. 2. – **A**: Orthomosaic of the focused area with zooms on special habitat patterns: 1: detritus litter at the edge of the *P. oceanica* meadow; 2: mooring system; 3: rocks and detritus litter; 4: dead matte with *Holothuria* sp. individuals. **B**: Map of the marine habitats and the identified fauna.

The map of the restricted area showed a distribution of the litter detritus and the dead matte at the edge of the *P. oceanica* meadow (Fig. 2B). Artificial concrete blocks were visible at the edge of the meadow, the latter covering them partially. A mooring system (an old tyre filled with concrete) and an assemblage of metallic structures were found on the main fine sediment patch (Fig. 2A). The spatial data showed that *P. oceanica* (living and dead parts) covered 63.7 % of the zone while fine sediments occupied 35.5 % and artificial substrates 0.8 % (Table I). When focusing on *P. oceanica* only, the dead parts of the plant (dead matte and litter detritus) represented 17.2 % of the total surface (Table I).

DISCUSSION

The aim of this research was to investigate the potential of orthomosaics generated by underwater georeferenced photogrammetry to study several functional compartments of the *P. oceanica* ecosystem in the framework of an ecosystem-based management strategy.

When compared with other mapping techniques, georeferenced underwater photogrammetry shows several key advantages. Acoustic sounders and airborne lidar are designed for bathymetric study and habitat mapping. They do not provide data allowing sessile fauna detection unlike underwater photogrammetry. Airborne photogrammetry is able to map near-surface marine habitats (up to 2 m depth) such as coral reefs and seagrass meadows (Casella *et al.* 2017) but requires still water. Moreover, their flying altitude – around 30 m – provides orthomosaics with a centimeter or decimeter resolution. The underwater photogrammetric technique deployed in the present study, with its millimeter resolution, allows the detection and identification of the sessile fauna of centimetric size. Its two main drawbacks are the range of the mapping efficiency, *i.e.*, the size of the mapped area, which is far lower than that of acoustic and lidar techniques; and its range, which depends on the water transparency since the photographs are shot from the surface. In the clearest Mediterranean water, its maximum range is around 10 m.

This work focused on the upper limit of the Mediterranean seagrass meadow, a place where many natural (*e.g.*, hydrodynamics, interaction with land ecosystems) and anthropogenic (*e.g.*, boating, fishing, tourism, coastal development) influences occur (Holon *et al.* 2015). Over the last decades the meadow's upper limit has been mainly mapped using aerial orthophotographs, with the drawback that dead matte, litter detritus and rocks can be mistaken for living *P. oceanica*. Underwater photogrammetry removes this uncertainty because of the photographic view that allows the mapping of the different habitats and substrates with virtually no erroneous identification (Rende *et al.* 2015). As highlighted in the results of this research, this technique also makes possible a spatial analysis of the different living and dead parts of *P. oceanica* meadows, the litter detritus and the dead matte representing about 20 % of the surface covered by the plant. Due to their important ecological role (*e.g.*, habitat, source of food, carbon sink), these areas should be taken into account in an ecosystem-based approach for the *P. oceanica* system (Boudouresque *et al.* 2015).

Underwater photogrammetry is also a powerful tool to detect and map artificial structures and more specifically illegal mooring systems composed of concrete blocks with a chain linked to a surface buoy. This system can be very harmful for *P. oceanica* meadows at shallow depth by pulling off the leaves, resulting in the generation of a dead matte area within the range of the length of the chain (Montefalcone *et al.* 2008). The capacity of photogrammetry to detect them paves the way for studying the impact of mooring systems on seagrass meadows with a spatial approach linking anthropogenic pressure and the ecological status of the seafloor surrounding them.

The identification and detection of sessile species is another advantage of underwater photogrammetry, thus allowing further investigations of the spatial distribution of marine organisms according the seascape characteristics (Abadie *et al.* 2018b). In the framework of this study, the key species *Paracentrotus lividus* of the *P. oceanica* ecosystem was detected, although not within the meadow but at its edge on a rocky substrate. The endemic – and currently under threat of extinction – species *Pinna nobilis*

Table I. – Area covered by marine habitats and their proportion in the focused area.

Habitat	Area (m^2)	Relative proportion (%)	Overall proportion (%)
Posidonia oceanica			
Living meadow	183.6	82.8	52.0
Dead matte	17.8	8.0	5.0
Litter detritus	20.3	9.2	5.7
Other habitats/substrates			
Fine sediments	125.3	–	35.5
Rocks	3.6	–	1.0
Artificial hard substrates	2.3	–	0.7
Mooring device	0.2	–	0.1

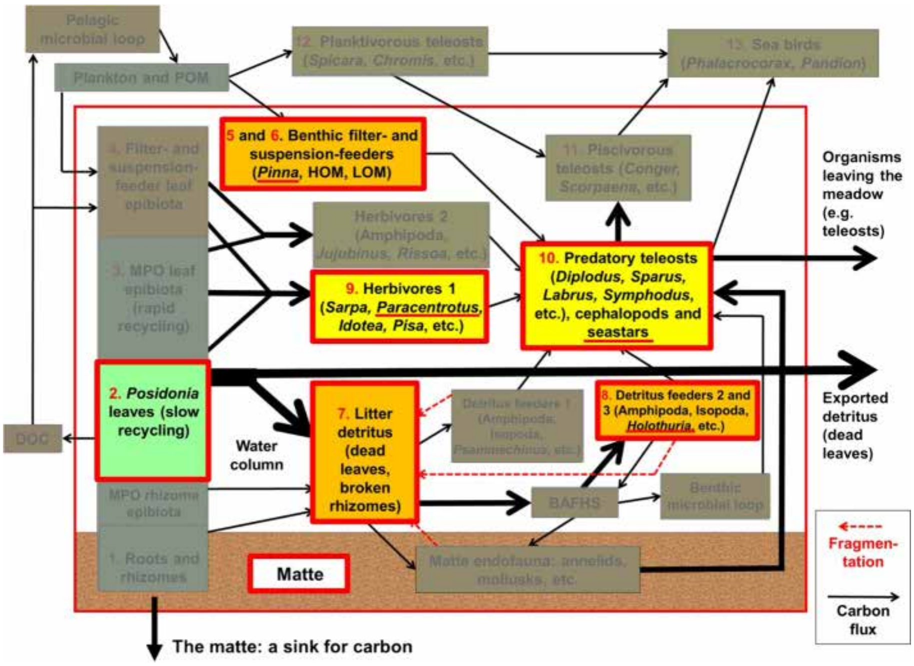

Fig. 3. – Conceptual representation of the *Posidonia oceanica* ecosystem composed of functional compartments and the interactions between them. Compartments framed in red are the ones investigated with photogrammetry in this study. Modified from Personnic *et al.* (2014).

was also detected. Unlike *P. lividus*, individuals of *P. nobilis* were found within the meadow near its edge. It was impossible to tell which ones were living and which ones were dead however, except for two individuals that were lying on their side on dead matte. These observations on *P. nobilis* distribution corresponds to that made by Coppa *et al.* (2010) who found that this bivalve settles mainly on dead matte and within the meadow close to the edge. The difficulty of detecting sessile species within the canopy is mainly due to the length of the leaves at the moment of the data acquisition (August). In order to increase the detection capacity of sessile species within the meadows, we recommend performing the data acquisition at the end of autumn or during winter when *P. oceanica* leaves are shorter and meadows are sparser.

It is clear that underwater photogrammetry is suited to the study of both spatial features of marine habitats and the distribution of several benthic species. In the framework of an ecosystem-based management strategy for the *P. oceanica* ecosystem, this method can be used to investigate several functional compartments (Fig. 3) of the conceptual model suggested by Personnic *et al.* (2014). It is obvious that photogrammetric mapping is not able to replace scuba diving data sampling. It is rather a low cost and effective tool to obtain additional spatial data on an area of interest.

Recent innovations in the acquisition of spatial data have the potential to provide new insights on the spatial heterogeneity of seagrass meadows and the organisms that live within or near them. The approach presented in this work thus appears relevant in the framework of an ecosystem-based management strategy for *P. oceanica* meadows. More work remains to be done on the analysis of the photogrammetric products to apprehend their true capacity in terms of the exploitable information. Furthermore, the spatial analysis of orthomosaics still requires automatization for more objective and effective study.

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TOWARDS THE DEVELOPMENT OF ECOSYSTEM-BASED INDICATORS OF MANGROVES FUNCTIONING STATE IN THE CONTEXT OF THE EU WATER FRAMEWORK DIRECTIVE

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MANGROVE
BIOINDICATION
WFD
ECOSYSTEM-BASED MANAGEMENT
BENTHIC COMMUNITY

ABSTRACT. – European Water Framework Directive is enforced in five tropical French Oversea Territories where mangroves are present. Developing bioindication tools to support the ecosystem-based management approach of the Directive is needed. A series of expert workshops was organized and led to the proposal of a strategy and of an applied research program to develop bioindication tools. The proceedings of the workshops are presented as a case study, as this is the first time such an integrative ecosystem-based approach is proposed in mangroves, combining structural and functional aspects, from forest structure to benthic community functioning.

INTRODUCTION

Among environmental legislations aiming at reducing human impact on ecosystems, the European Water Framework Directive (WFD, 2000/60/EC) set the objective to reach "good ecological status" of coastal and transitional waters. Bioindication, status assessment of biological quality elements (BQE) in WFD terminology, is one of the means proposed to assess this ecological status. Bioindication tools must be developed in French Oversea Territories (OTs) as they are for continental Europe waters (Birk *et al.* 2012). As Marine Strategy Framework Directive (MSFD, 2008/56/EC) is not enforced in French OTs, WFD and its ecosystem-based management approach is the central tool for coastal and transitional waters management. French Guiana on the equatorial Atlantic coast of South America, Martinique, Guadeloupe and Saint Martin Islands in the Caribbean, Mayotte Island in the West Indian Ocean, are the five French OTs where both this environmental policy is enforced and where mangroves are present.

Mangroves are potentially subject to different kind of anthropogenic pollutions mediated by water: as an interface between land and sea, land-originated pollutions transit through, and as sediment deposition area, they are a sink for inorganic and organic contaminants. They are also sensitive to hydrological changes due to human activities. For this reason, it has been proposed to include mangrove ecosystem in the assessment of the ecological status of transitional and coastal waters, even if not identified among the standard WFD BQEs that have been mainly designed for non-tropical areas. This ecosystem-based approach is also currently applied for WFD bioindication tools development in coral reefs and seagrasses ecosystems (Le Moal *et al.* 2016)

Unlike mangroves at global scale, which have lost a third of their surface area in twenty years (Splading *et al.* 2010, Hamilton & Casey 2016) and are still in decline, the surface area of mangroves in the French OTs has been relatively stable over the same period (Roussel *et al.* 2009, Fromard & Proisy 2010, Jeanson *et al.* 2014). Nevertheless, available data on the pollution levels of these mangroves or their ecological status are very limited. The impact of the pollutions on mangrove ecosystem needs to be investigated and potentially operational bioindicators need to be identified. Beyond the ability to reflect the level and impact of pollution, the technical and financial feasibility to deploy either long-term monitoring or single diagnosis, is a crucial aspect of the bioindication tools.

In the scientific literature, numerous studies are assessing anthropogenic impact on mangroves, through different perspectives of interest and identify potential bioindicators defined as "physiological and biochemical responses to anthropogenic perturbation with consequences at different biological complexity levels, from species to ecosystem" (Mc Carty & Munkittrick 1996). But most of these studies are punctual, limited in time and space, deal with a single type of pollution, compare extremely contrasted sites *i.e.*, pristine *vs* highly degraded, and focus on modifications observed in one or few compartments of the mangrove ecosystem through few parameters: soil organic matter composition (Aschenbroich *et al.* 2015), organic matter mineralization and primary production enhancement (Penha-Lopes *et al.* 2010, Molnar *et al.* 2014), soil heterotrophic community (Bouchez *et al.* 2013), crab population dynamics and feeding (Bartolini *et al.* 2009, 2011), RNA/DNA ratio in crabs (Amaral *et al.* 2009) or oxydative stress in oysters (Ramdine *et al.* 2012), shrimp population (Penha-Lopes *et al.* 2011), abundance of generalist *vs* specialist species of sponges (Díaz *et al.* 2004) or Bryozoa (Creary 2003), mudskipper population structure (Kruitwagen *et al.* 2006), mangrove tree leaves pigment concentration (MacFarlane & Burchett 2001; MacFarlane 2002) and respiration (Herteman *et al.* 2011), canopy and tree community structure (McDonald *et al.* 2003, Lovelock *et al.* 2009, Herteman *et al.* 2011), tree productivity (McDonald *et al.* 2003) or mortality (Duke *et al.* 2005, Schaffelke *et al.* 2005) for instance (see Dirberg 2015a for a review). Choosing among them the most relevant ones to be used and deployed in the WFD's integrative ecosystem approach to assess ecological status is not straightforward:

– There is a variety of situations both within and between our five OTs of interest: anthropogenic pressures, mangroves types, associated biota, and ecological conditions are diverse;

– The complexity of the ecological status apprehension in mangrove ecosystem requires an holistic transdisciplinary approach;

– The stakes associated with the cost and mandatory implementation of environmental policy are high.

For these reasons, a transdisciplinary panel of experts from diverse scientific and environmental management background, was gathered and asked to set up a strategy to develop bioindication tools for WFD water bodies ecological status assessment in French overseas mangroves. This is a joint initiative from the French Biodiversity Agency (OFB) and the National Museum for Natural History (MNHN), with experts from French National Center for Scientific Research (CNRS), French National Research Institute for Sustainable Development (IRD), Aix-Marseille University, Toulouse University, Conservatoire du Littoral, Nantes University, French geological survey (BRGM), and University of the French Antilles.

This paper summarizes, as a case study, the proceedings and the proposed strategy from the expert group

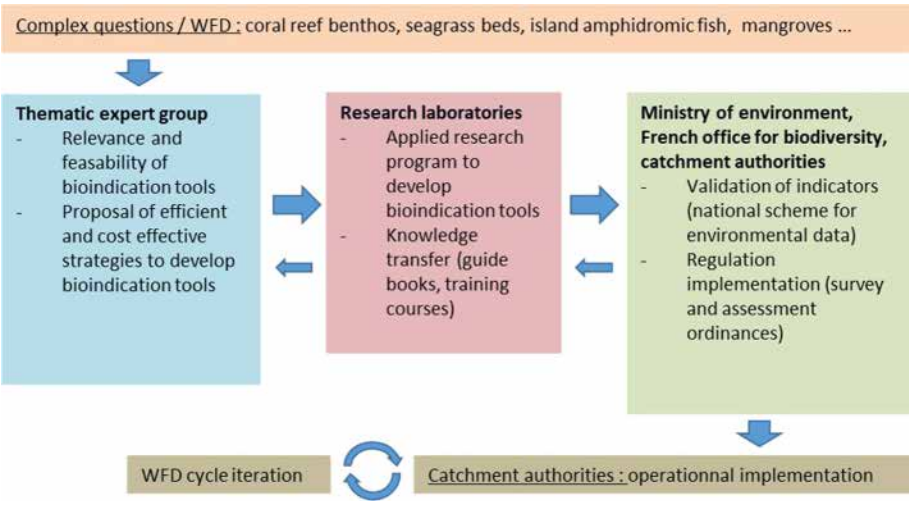

Fig. 1. – Scheme of the general organization of WFD bioindication tools development for French OTs.

workshops that led to the deployment of a 5-year transdisciplinary research program on bioindication in mangroves ("Thematic expert group" box and first part of "Research laboratories" box on Fig. 1). This developmental phase will allow field sampling and analysis. First bioindication tools are expected to be available at the end of this phase for routine deployment by either private engineering consulting, national or regional public environmental agencies or research laboratories.

Two 3-days workshops, gathering 14 and 16 people, were held in Paris in September 2015 and May 2016 at National Museum of Natural History.

EXPERT GROUP WORKSHOPS PROCEEDINGS

Building the context of the expertise

The main objective of the first workshop was to bring everybody to a common level of knowledge on WFD requirements and share views on this issue from the perspective of the different expert's background.

1) A bibliographic review on bioindication in mangroves (Dirberg 2015a) was prepared and sent to the participants to prepare the first workshop.

2) A presentation and discussion of the conceptual framework of the WFD were organized as it raises immediately many questions to any person who is not already acquainted with it and its vocabulary. Going through the process of questioning and sharing thoughts on the WFD concepts and objectives, even if it does not seem to deliver measurable results, can be seen as a very good mean to build a first common agreement. Hence for pragmatic reasons, and as first common decision, questions about the definition of "good" when talking of "good ecological status" (of which we cannot ignore the political and philosophical dimension), the possibility to find or define reference conditions, the relevance of the WFD water bodies delineation from mangroves perspective, the possibility to untangle the contribution of different anthropogenic pressures, or global change, when facing ecological impact, were discussed and acknowledged as problematic. But it was decided they should not hamper the main objective to develop tools for helping to maintain or improve ecological status of mangrove ecosystems. These questions will be kept in mind and reformulated as the project evolves.

Focussing on anthropogenic pressures allows sharing transdisciplinary knowledge and experience, and delivered first important results: (i) Identification and prioritization of known or potential anthropogenic pressures affecting mangroves in each OT (Dirberg 2015b); (ii) Description of how these pressures could affect any component of the mangrove ecosystem and identification of parameters that would vary with the impact in a predictable way; (iii) Identification of potential sampling sites in each OT, sites either known to be impacted, or as little disturbed as possible to be considered reference sites.

This process does not allow to limit significantly the number of parameters that are potentially relevant bioindicators, but it clarifies the needs, the constraints, the background and hence the possibilities in each of the OTs of interest.

Setting the practical objectives

The second workshop objective was to choose the parameters to be measured in the field and set up the strategy to develop the bioindication tools.

As the final objective was to assess the "ecological status of the waters through bioindication in the mangroves", we needed to define explicitly what objects we were actually looking at. Thus, the concept was broken down in a list of more practical and explicit descriptors, structural and functional, to characterize the ecological status of a mangrove itself:

– Forest structure dynamics;

– Mangroves tree growth;

– Regenerative capacity of the ecosystem;

– Species abundance and diversity (species with a life trait depending on mangroves);

– Functional characteristics of the sediment (in particular the organic matter degradation process);

– Eutrophication signs.

In the WFD perspective, to be qualified these descriptors will have to be compared to a reference, either historical or theoretical. Beyond these descriptors that allow assessment of the present mangrove status, two vulnerability descriptors are proposed to be added to take into account the context and its expected evolution:

– Vulnerability to sea level rise (*i.e.*, landward accessibility to mangrove migration);

– Vulnerability to foreseen urban / agricultural / industrial development.

For each descriptor, potential parameters among those sensitive to the anthropogenic pressure identified at the first workshop, and associated methods, could be proposed, but many other considerations on scientific, technical, organizational and financial issues were considered:

– As the response time to pressures is highly dependent on the level of biological complexity (from the biochemistry of a single organism, to community or ecosystem levels, from fast to slow) and as this biological levels are observed at very different spatial scales (see Martínez-Crego *et al.* 2010), the combination of methods proposed to give information on the descriptors should cover different spatio-temporal scales, from station to river basin, from season to decade.

– For mangrove forest, mangroves trees and macrofauna species descriptors, we can find robust methodologies in an extensive literature and no further development seems necessary. There is less literature on the functional characteristics of the mangrove sediment and organic matter degradation pathways and dynamics (Molnar *et al.* 2013, Luglia *et al.* 2014, Pascal *et al.* 2014, Aschenbroich *et al.* 2015, David *et al.* 2019 for instance) and none presents an integrative approach including benthic macro- and meiofauna, fungi and prokaryotes within the same study. This benthic community has a pivotal role in mangrove ecosystem functioning (Nagelkerken *et al.* 2008; Aschenbroich *et al.* 2016, 2017). Therefore, we must pay particular attention to this compartment and its functional aspect in the development phase of our bioindication tools. Finally we found only one publication (Carugati *et al.* 2018) that attempts to combine a set of parameters covering our different descriptors in an integrative ecosystem-based approach to assess biodiversity and ecosystem functioning related to mangrove degradation. This case study compares highly contrasted mangrove sites, one pristine and the other with a massive dieback, and does not link the observed impact on functioning and biodiversity to any specific pressure. We must be able to establish links between pressure and impact for management purposes and we must be able to assess status not only in extreme degradation situations (as this is not the usual situation) but also in situations of moderate pollution.

– For the developmental phase of the project, a common standard data set from the different OTs is needed hence deploying the same sampling strategy and methods in this different ecological contexts and species assemblages.

– More parameters than those that will be retained beyond the developmental phase of the project need to be investigated, in order to have an in-depth view and then be able to choose the most relevant ones. Pollution levels must be measured in situ, as water sampling sites monitored under the WFD for assessment of chemical status do not provide adequate information to allow linking impacts to pressures.

– Potential reference sites are difficult to find, and may not exist, as in Mayotte, Martinique and Guadeloupe, population is dense and human activities are everywhere. The least impacted sites will serve as reference sites for the development phase. Reference for the final WFD status assessment will have to be define. Sites on other islands of the region could be considered. A review of archeozoological records of species known to be linked to mangroves is also proposed to provide some historical context and tackle the shifting baseline syndrome.

– In French Guiana, human population density is much lower and access to mangroves is more difficult. Coastal mangroves are highly dynamical, depending on the Amazon River sediments loadings (Fromard *et al.* 2004, Anthony *et al.* 2010). On the contrary, estuarine mangroves inland along the polyhaline area show different vegetation structure and are more stable (Fromard *et al.* 2004) but are probably also more affected by local anthropogenic pressures. The mangrove along the coast is more directly affected by the Amazon River discharge. Hence the anthropogenic pressures they are facing are out of control of French authorities and are not considered within WFD perspectives. Therefore, only estuarine mangroves are here targeted and the mangrove sampling sites were chosen, in a first time, along the Cayenne estuary, moving away from Cayenne, the main city of French Guiana.

– We need to choose carefully the sampling stations to limit as much as possible the ecological conditions discrepancies between samples and maximize the signal that could be linked to the different levels of pollution. This means:

• Choosing sites within similar mangrove zonation: riverine *Rhizophora* zonation in French Guiana, either *Bruguiera* or *Rhizophora* dominated zonation in Mayotte, seaward *Rhizophora* zonation in Martinique and Guadeloupe;

• Measuring *in situ* the tidal level and immersion time to ensure the comparability;

• Core sampling in similar conditions: anticipate underground roots distribution and crab burrows to avoid them, take into account soil micro-topography to avoid local low points with potentially very different immersion time.

– Mangrove macrofauna: crustacean, molluscs, insects, birds but also sessile Bryozoa, sponges, ascidians, and other taxa could be potentially bioindicator, but we lack ecological knowledge on most of them, and the necessary work required to fill the gaps is not compatible with timeframe and budget allowed for this project. Benthic invertebrates (meio-, meso- and macrofauna) living within sediments and their bioturbation functional roles (Aschenbroich *et al.* 2017) were prioritized in this study since they are known as bioindicators for the WFD in temperate areas. As crabs are key engineer species in mangroves (Kristensen 2008), minimal information should be collected. As cryptic and burrowing animals, crabs can be difficult to monitor (Kent & McGuiness 2006). Thus, crabs burrow counting (Skov *et al.* 2002) with measurement of opening size of burrows (Micheli 1991) was proposed as a minimum proxy to crabs abundance.

– Among technical constraints, we need to be able to go to the field, with one small boat, one or two cars, hence not too many people, collect samples for different type of analysis, make *in situ* measurements, bring back the samples in good conditions, and allow time for the subsampling, measurements and sample preservation at laboratory.

– Temporal and spatial natural variation of the different parameters cannot be tackled at the same time we were sampling for testing the full set of parameters. This has to be done in a second phase with a dedicated sampling strategy. The first phase should allow reducing the num-

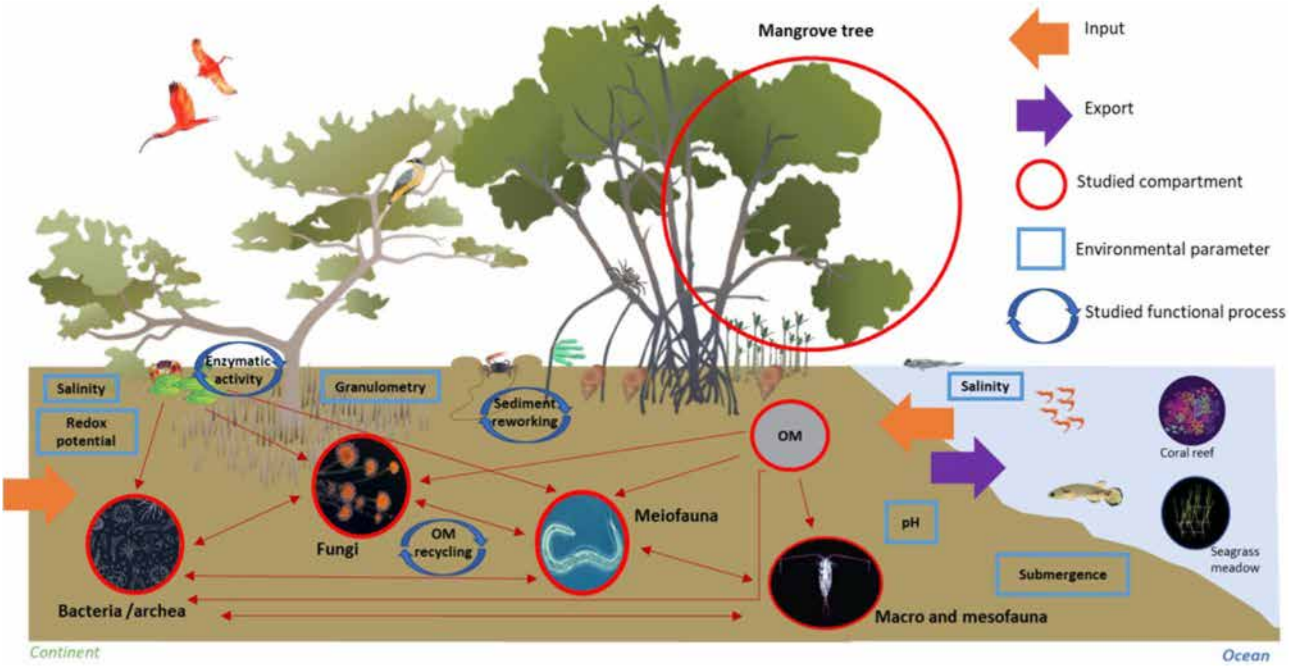

Fig. 2. – Simplified conceptual scheme of the studied compartments and functional processes of mangrove ecosystem toward the development of ecosystem-based indicators of mangroves functioning state (Figure credit: Maud Fiard).

ber of necessary measured parameters and come out with a lighter field protocol for this second phase.

– The lack of research facilities or equivalent accessible laboratory workspace in Saint Martin excluded it from the first round of sampling.

The resulting choices as a research program

The sums of the needs and constraints led to the proposal of a strategy (Dirberg 2017) that became an applied research program. We summarize the content of this program as a result of the presented expertise process (Fig. 2). Protocols will be detailed in dedicated publications.

Sampling sites

In each of the four sampling OTs, sampling sites were proposed by local experts. The sampling team visited the sites prior to sampling to confirm the selection and checked the accessibility in reasonable distance/time/conditions from the laboratory, and that ecological conditions were similar.

Hence, we have chosen 3 sites in French Guiana along the Cayenne river, 4 sites in Martinique, 5 sites in Guadeloupe, 2 *Rhizophora* dominated sites and 3 *Bruguiera* dominated sites in Mayotte, making 17 sites in total (Table I). These sites are either affected by different type of known pressures (agricultural, industrial, domestic waste) or potential local reference.

Studied parameters

The final choice of studied compartments and parameters is compatible with a dense one station-a-day organization with 5 peoples, one boat, two cars.

In situ measurement

Environmental parameters:

– Pore water salinity

– Water level recording (HOBO probe)

On sediment core samples

On each station, 3 cylindrical sediment cores were sampled (10 cm diameter, 18 cm in length), sliced at the lab for subsampling (10 slices: 0-1-2-4-6-8-10-12-14-16-18cm). On each slice, following parameters were measured:

Potential bioindication parameters:

– Sediment reworking measurement (one-week incubation after fluorescent tracers deposit) as a proxy of endofauna activity.

– Biomass, abundance, diversity of microbes and fungi (genetic tools).

– Diversity, density and biovolume of small macrofauna.

– Diversity, density, and biomass of meiofauna.

– Potential enzymatic activity measurement (Biolog Ecoplates®) tested as a potential integrative bioindication tool.

Table I. – GPS coordinates (WGS84) of the 17 sampling stations.

Site	Station	st_code	Latitude	Longitude	Station type / pressure
French Guiana	Crique Fouillée	S1	4.914780	–52.337759	Urban
French Guiana	Confluence	S2	4.897008	–52.374365	Low, domestic
French Guiana	Petit Cayenne	S3	4.858881	–52.399868	Reference station
Mayotte	Dembéni1	DS	–12.844892	45.194823	Urban
Mayotte	Dembéni2	DP	–12.837679	45.190321	Reference
Mayotte	Malamani1	MS	–12.921955	45.152809	Sewage water
Mayotte	Malamni2	MP	–12.923628	45.152893	Local reference
Mayotte	Zidakani	ZI	–12.785458	45.096780	Reference ?
Martinique	Baie du Trésor	S4	14.766701	–60.883034	Reference
Martinique	Pointe Marin	S5	14.447821	–60.878443	Sewage
Martinique	Pointe Merle	S6	14.561594	–61.010904	Agriculture
Martinique	Cohé du Lamentin	S7	14.602466	–61.021394	Urban/industrial
Guadeloupe	Intermédiaire	IN	16.2775	–61.5488	Urban
Guadeloupe	Décharge	DE	16.2594	–61.5469	Landfil site, Urban
Guadeloupe	Babin	BA	16.3388	–61.5294	Reference
Guadeloupe	Fajou	FA	16.3509	–61.5906	Reference
Guadeloupe	Goyave	GO	16.1379	–61.5743	Urban, Agriculture

– Biochemical tracers' concentration and/or ratio (fatty acids, pigments) as a proxy of organic matter degradation processes.

Environmental parameters:

– Physical: Redox potential, pH, sediment granulometry, pore-water salinity.

– Chemical: organic contaminants (PAHs, PCBs, pesticides, phthalates, PBDE, alkylphenols) and inorganic contaminants (heavy metals), C:N ratio**.**

On litter bags

On each station, litter bags filled with 10 *Rhizophora* leaves, deposited on site and then sampled at 0, 5, 10, 20, 30 days or 0, 7, 14 days.

Potential bioindication parameters:

– Biomass, abundance, diversity of microbes and fungi (genetic tools).

– Potential enzymatic activity measurement (Biolog Ecoplates®) tested as a potential integrative bioindication tool.

– Biochemical tracers' concentration and/or ratio (fatty acids, pigments) as a proxy of organic matter degradation processes.

Forestry quadrats

Potential bioindication parameters and environmental parameters:

– Tree species and density.

– Tree diameter measurement.

– Tree sanitary status.

– Saplings counting.

Crabs burrows quadrats

In three 1 m² quadrats, measurement of each burrow aperture width to the closest centimeter as proxy of crabs abundance.

Remote sensing

Characterization of soil occupation of the catchment upstream each mangrove site.

Delineation of mangroves for long term monitoring

CONCLUSION

WFD represents an important opportunity both for environmental management and for scientific research. As illustrated in this case study, ecosystem-based management sets a real challenge to science as understanding the complexity of mangrove ecosystem and taking into account functional parameters are necessary to develop the bioindication tools. This has led to the development of a transdisciplinary research project that might have not been possible without this impulse. Bringing together experts from very different cultures and professional goals, and getting them to build together a project that meets the standards of academic research and the needs of environmental managers, is also a challenge, but is necessary in the context of environmental policy implementation and ecosystem-based management. Finally, another important by-product of the process presented here, is the increased exchange between science and management, the mutual understanding of each other's needs, the increase in experience, knowledge and of concerns of everyone, that ecosystem-based management stimulates.

The first phase of the transdisciplinary applied research project described here should end in 2021. Sampling in French Guiana occurs in 2017, Martinique and Mayotte in 2018, Guadeloupe in 2019. First scientific results are under the process of publication. Environmental managers and scientific researchers will meet several times by 2021 to discuss the results and the opportunity to transfer them into WFD long term monitoring.

Ecosystem-based approach is a management standard within other EU environmental policy like Main Strategy Framework Directive (MFSD) and leads to the development of dedicated Ecosystem-Based Quality Index methods (Boudouresque *et al*. 2015, Thibaut *et al*. 2017). From a WFD perspective as developed in France, the integrated ecosystem approach developed in French OTs and illustrated here for mangroves, is new. It is justified by the complexity of tropical ecosystems and the relative lack of knowledge, compared to European lake and river ecosystems for instance. This study is providing some basic knowledge that was lacking on French OTs mangroves. And, from a global perspective, it is the first attempt to combine all these structural and functional parameters, in an integrative ecosystem-based approach for ecosystem-based management.

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INTERSPECIFIC COMPETITION AND INVASIVE CAPACITY OF *HOLOTHURIA (ROWEOTHURIA) ARGUINENSIS*: DOES ITS DIET REPRESENT A DANGER FOR NATIVE MEDITERRANEAN SPECIES?

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HOLOTHURIA (R.) ARGUINENSIS
INVASIVE SPECIES
DIET
COMPETITION
ALGERIAN WEST COAST
NATIVE MEDITERRANEAN SPECIES
ALGAE
FORAMINIFERA

ABSTRACT. – The aim of this study is to investigate the diet of the invasive species *Holothuria (Roweothuria) arguinensis* Koehler & Vaney, 1906 and its competitive potential in food intake, with native Mediterranean holothurian species in Salamandre province (Algeria). Crustaceans are widely consumed by all native species [21.33 %, 16 %, 10 % and 9.06 %, respectively in *Holothuria (Panningothuria) forskali* Delle Chiaje, 1823, *H. (Platyperona) sanctori* Delle Chiaje, 1823, *H. (Holothuria) tubulosa* Gmelin, 1791 and *H. (Roweothuria) poli* Delle Chiaje, 1824]. Foraminifera are also well appreciated by *H. (R.) poli* and *H. (H.) tubulosa* (respectively with 17.33 % and 29.33 %). *Holothuria (R.) arguinensis* stands out with high consumption of bivalve fragments (24.80 %), sponges fragments (11.6 %) and nematodes (6.80 %). However, cyanophyceae, crustaceans and foraminifera are not consumed by this invasive species. There was a very highly significant difference (Permanova, $p < 0.001$) between *H. (R.) arguinensis* and the four native Mediterranean holothurians. *Holothuria (R.) arguinensis* prefers food resources which are little or not consumed by the other native sea cucumbers; it could therefore be considered as a “specialist” species which shows no tendency to competition with the other holothurians species.

INTRODUCTION

Holothurians, commonly known as sea cucumbers, are marine organisms belonging to the phylum of Echinodermata, which comprises more than 1400 species (Conand 1994, Navarro 2012). Holothuriida are among the most common invertebrates in the benthic compartment of the Mediterranean *Posidonia oceanica* (L.) Delile (1813) ecosystem (Francour 1990, Mezali 2008) and play an important role in the organic matter recycling within the food web of this ecosystem (Zupo & Fresi 1984, MacTavish *et al*. 2012). During their foraging, these deposit-feeders collect selectively the richest organic matter particles (Mezali & Soualili 2013, Belbachir *et al*. 2014), which could favor ecological niche partitioning between species.

Holothuria (Roweothuria) arguinensis is a northeastern Atlantic species (Thandar 1988, Rodrigues 2012), which has recently been spreading into the Mediterranean Sea, where it has been recorded on the Algerian coast (Mezali & Thandar 2014). Due to the colonization of a different area of the Mediterranean Sea, *H. (R.) arguinensis* may compete with native holothurians species and consequently constitute a potential danger for them. Through this work, two questions were considered: (1) what are the food resources consumed by *H. (R.) arguinensis* and the Mediterranean native species *H. (P.) forskali, H. (P.) sanctori, H. (H.) tubulosa* and *H. (R.) poli*? (2) is there a competition for food resources between native sea cucumbers and *H. (R.) arguinensis*?

MATERIAL AND METHODS

Holothurians were sampled during spring 2018 at the Salamandre site (Mostaganem province, Algeria) (35°54’N; 0°03’E) (Fig. 1) at 3 m depth. This site shows a reduced biodiversity and a degraded *Posidonia oceanica* meadow (Belbachir 2018). Five Holothuriida species are found at the prospected area. *Holothuria (R.) poli* is the most abundant species, it can cover its body

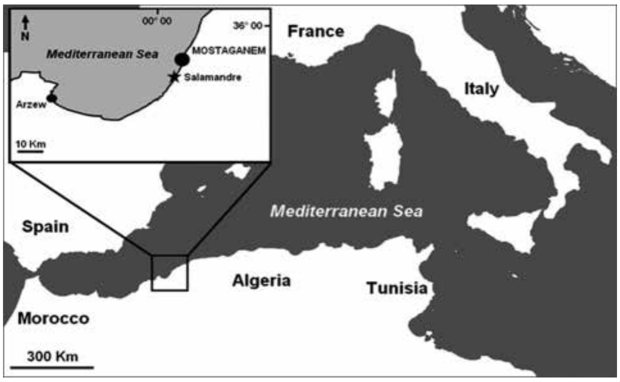

Fig. 1. – Geographical location of the Salamandre site (full star character) where samples of the five holothuroids were collected (from http://www.histgeo.ac-aix-marseille.fr, modified).

with a thin layer of sand (Mezali 2008). This species has a hard body and inhabits the detrital bottoms and the *intermatte* (spaces without living *P. oceanica* leaves within the meadow) (Mezali 2004). *Holothuria (H.) tubulosa* is found much closer to the hard bottom and within the *P. oceanica* meadow (Francour 1990). *Holothuria (P.) forskali* and *H. (P.) sanctori* have a soft body and constitute cryptic species that are found fixed on hard substratum, under rocks and in the eroded vertical edge (thickness of the *matte* that is observed at the level of the *intermattes*) of the *P. oceanica* meadow (Mezali 2008). *Holothuria (R.) arguinensis* is an invasive species recently recorded in Algerian shallow water areas (Mezali & Thandar 2014). At the prospected site, this species is usually found on rocky or sandy substrate.

Ten adult individuals (20 cm average contracted length) were collected for each of the five holothurians species found at the Salamandre site [*Holothuria (P.) forskali*, *H. (P.) sanctori*, *H. (H.) tubulosa*, *H. (R.) poli* and *H. (R.) arguinensis*]. Each individual was dissected and the contents of its digestive tract were carefully collected for microscopic observations. The contact method of Jones (1968), modified by Nedelec (1982), was used for the digestive content analysis (see Belbachir & Mezali 2018). Permutational multivariate analysis of variance (PERMANOVA) (Anderson 2001) was carried out using R v3.4.1 software (R Core Team 2017) to test the dissimilarity of diet items among holothurians digestive tract. Diet composition data were visualized in two-dimensional space through non-metric multidimensional scaling (NMDS) (Clark 1993). The use of the R v3.4.1 software enabled us to introduce confidence ellipses based on the variability existing between the replicates (the confidence level used is 68 %).

RESULTS

Diatoms and algae are relatively well represented in all the analyzed holothurian guts. The highest rate of diatoms is obtained for *Holothuria (P.) forskali* (28.66 %) and the highest percentage of algae is obtained for *H. (P.) sanctori* (31.33 %) (Fig. 2).

Bivalve fragments (24.80 %), sponges (11.6 %) and nematodes (6.80 %) are relatively widely consumed by *H. (R.) arguinensis*. Furthermore, these three food resources are little consumed by the native holothurians species (Fig. 2). *Holothuria (R.) arguinensis* do not consume foraminifera, crustacean and cyanophyceae, unlike most of the native species, which appreciate these food resources (Fig. 2). The leaves (dead and alive) of *P. oceanica* are consumed by all sea cucumbers, but in very small proportions. The highest proportion of dead *P. oceanica* leaves (2.67 %) is obtained in *H. (P.) forskali* gut and the highest percentage of alive *P. oceanica* leaves (3 %) is obtained in *H. (R.) arguinensis* gut (Fig. 2). The PERMANOVA analysis reveals a dietary difference among the five studied holothuroids ($p < 0.001$). According to the non-metric multidimensional scaling (NMDS)

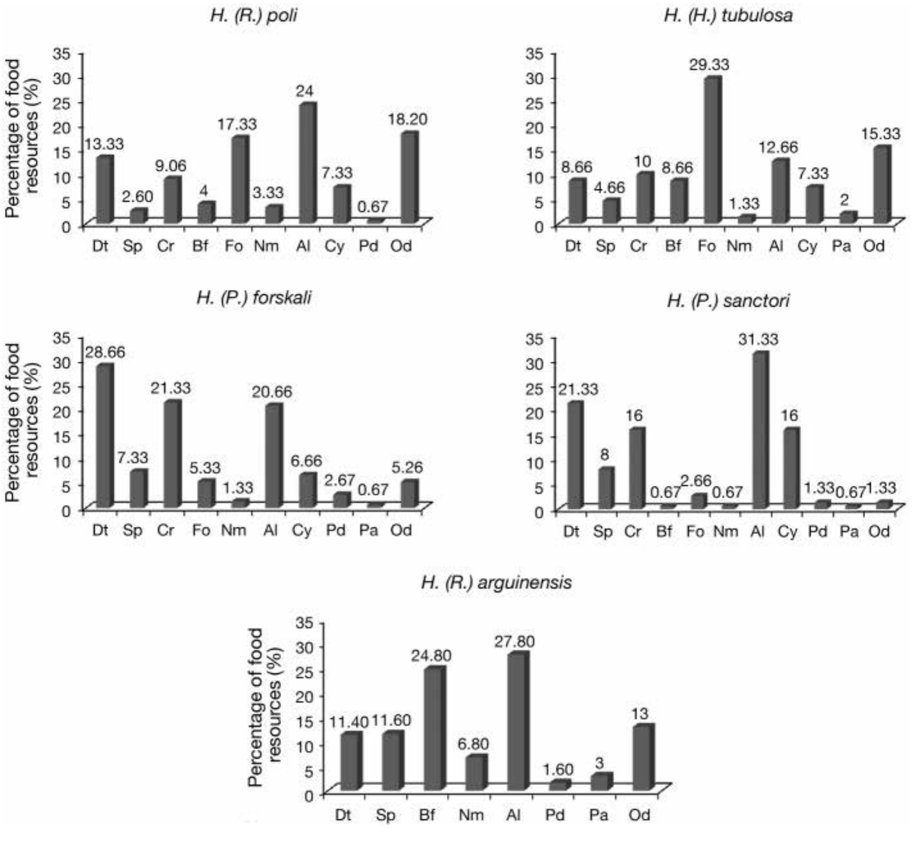

Fig. 2. – Percentage of food resources in the diet of the studied holothurians. Diatoms (Dt), Cyanophyceae (Cy), Macrophytes algae (Al), *Posidonia oceanica* alive leaves (Pa), *Posidonia oceanica* dead leaves (Pd), Foraminifera (Fo), Crustacean (Cr), Sponges (Sp), Nematodes (Nm), Bivalve fragments (Bf), Organic detritus (Od).

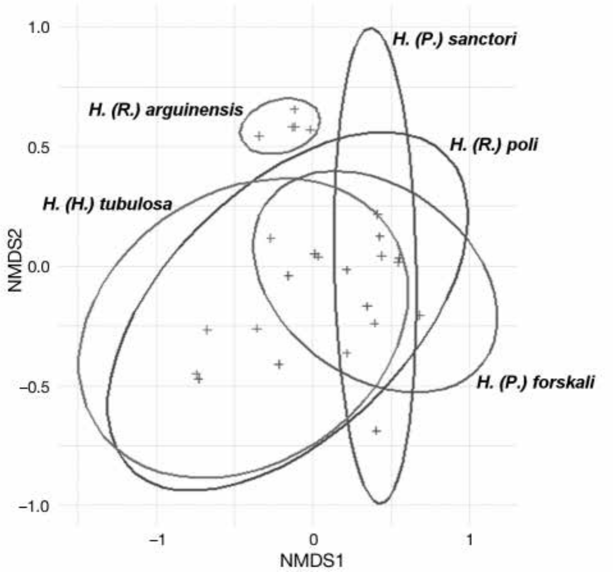

Fig. 3. – Non-metric multidimensional scaling (NMDS) analysis based on Bray-Curtis distances highlighting the relationship between the studied holothurian species at the Salamandre site, according to their diet.

presentation, *H. (R.) arguinensis* ellipse is clearly distinguished from those of the native species (Fig. 3).

DISCUSSION

The food resources of the five studied holothuroids are very diverse. Statistical analysis (PERMANOVA and NMDS) enabled us to distinguish the invasive *Holothuria (R.) arguinensis* from the native holothurian species group [*H. (P.) forskali*, *H. (P.) sanctori*, *H. (H.) tubulosa* and *H. (R.) poli*] in terms of diet. *Holothuria (R.) poli* is much closer to *H. (H.) tubulosa*, since foraminifera are the common food item shared between these two species. *Holothuria (P.) sanctori* and *H. (P.) forskali* are also close to each other in terms of diet; these two holothuroids greatly prefer the vegetal component and crustacean in their diets. Through non-metric multidimensional scaling (NMDS) presentation, we could claim that there is no diet overlap between *H. (R.) arguinensis* and the other native species. The invasive species ellipse is narrower than those of the other species, suggesting that it could be a specialist species. *Holothuria (R.) arguinensis* does not compete with the four studied indigenous Mediterranean sea cucumbers, as long as it consumes some food resources, which are infrequently used by the rest of the sea cucumbers. There are also some food resources consumed by native species, which are not consumed by this invasive species.

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