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Review

Macroalgal assemblages as indicators of the ecological status of marine coastal systems: A review

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ABSTRACT

Macroalgae have been utilized as biological indicators of ecosystem health in many monitoring programs worldwide. These programs have utilized various methods to quantify macroalgal community structures. The aim of this study was to provide an overview of current progress by reviewing techniques and methods in both monitoring programs and impact evaluation studies that use macroalgal assemblage data. A total of 215 papers were selected and divided into four categories: macroalgal assemblage monitoring, macroalgal mapping, developing and employing ecological indices based on macroalgae, and developing and employing generic ecological indices including macroalgae. The number and goals of macroalgal monitoring programs are very different among geographical areas. In Europe, the recent European Union Directives led to the development of indices as tool to monitor the ecological quality of coastal systems. In other geographic regions, most studies focused on mapping the distribution of kelps or Fucales. This demonstrates the necessity to harmonize marine macroalgal monitoring, identifying common metrics and approaches in sampling design, field measurements, taxonomic resolution and data management, in order to develop standardized procedures which may allow data obtained to be compared.

1. Introduction

Marine coastal areas are particularly exposed to human pressures and are among the most exploited by human activities, thus a range of recent legislative approaches (e.g. Oceans Act in USA, Australia or Canada; Water Framework Directive and Marine Strategy Framework Directive in Europe, and National Water Act in South Africa) have been developed to address the challenges of maintaining sustainable marine coastal waters, habitats and resources (Borja et al., 2010)

The management of coastal systems requires the assessment of their ecological quality and capability, ideally identifying early warnings of change. In this contest, monitoring is recognized as a key tool for evaluating change in biodiversity and structure of ecosystems (Birk et al., 2012). Monitoring is undertaken to identify components of current systems (at species, community, habitat or ecosystem levels), with respect to various attributes (such as trends in specific populations, measures of diversity, community structure, ecosystem functions and services, ecosystem "intactness"), in order to recognize change in response to human-induced pressures or to management initiatives, at a range of scales in space and time (Birk et al., 2012). Over the past two decades, increased international attention has focused on coastal environments and the ways in which human-induced pressures are resulting in changes to ecosystems. This has led to demand for (i) robust methods to separate natural variation in ecosystems from change caused by human activities that can be potentially managed, and (ii) reliable and cost-effective indicators of such change (Martinez-Crego et al., 2010).

In this context, macroalgae have a long history of use in ecological assessments (Stevenson, 2014) because of their ecological importance (Steneck et al., 2002) and sensitivity to stress (Thibaut et al., 2015; Piazzi and Ceccherelli, 2020). Thus, macroalgal assemblages are widely considered as good ecological indicators for monitoring surveys and impact evaluation studies (Pinedo et al., 2007; Juanes et al., 2008; Guinda et al., 2008; Díez et al., 2012; Neto et al., 2012). Monitoring programs based on macroalgae have utilized various methods of mapping and sampling, and different ecological descriptors have been employed (Krumhansl et al., 2016; Duffy et al., 2019), and many ecological indices utilizing macroalgal assemblages have been proposed recently (Orfanidis et al., 2003; Ballesteros et al., 2007; Cecchi et al., 2014).

The aim of this study is to review techniques and methods that use

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macroalgal assemblages in both monitoring programs and impact evaluation studies in order to provide an overview of current progress. In addition, we evaluate the different approaches and highlight gaps in current knowledge.

2. Methods

A detailed literature search was carried out to locate literature on the use of macroalgae as biological indicators, and the monitoring and mapping of macroalgae beds. The databases searched included standard marine bibliographic sources (Science Direct, Web of Science, Wiley, Google scholar), and also web sites of marine research organisations. The literature search was made using a combination of key words including: kelp, macroalgae, macroalgae bed, Laminariales, Fucales, canopy-forming algae, large brown, turfing algae, coralline algae, monitoring, mapping, modelling, predict modelling, climate change, sedimentation, ocean acidification, productivity, temperature, nutrient, satellite, drone/AUV, single beam, multibeam, side scan sonar, Lidar, videography, ecological indices. The literature search ended on 2020.

Among all the papers retrieved regarding macroalgal field investigations, those reporting methods to assess the distribution and the ecological quality of macroalgal assemblages were selected and divided in relation to subject and geographic area of application. The nomenclature of algal taxa followed Guiry and Guiry (2021).

3. Results

A total of 215 papers were selected (Table S1) and divided into four categories: macroalgal assemblage monitoring (62), macroalgal mapping (90), developing and employing ecological indices based on macroalgae (51), and developing and employing generic ecological indices including macroalgae (12) (Fig. 1). Most papers concerned the European Union area (85) and North America (27); wide bibliography also referred to other European states (18), Australia (21) and New Zealand (22) (Fig. 2). Most of the 13 papers of Asian studies concerned Japan. The other papers were related to South/Central America (15), Africa (6), Antarctica (1) and Polynesia (1). Six papers were reviews or concerned general issues. The number of papers per year increased in the last two decades, shifting from a mean of 2.8 in the period 2001–2005 to a mean of 16.4 in the last five years (Fig. 3).

3.1. *State of the art of macroalgal assemblages monitoring*

3.1.1. *European Union*

The relationship between environmental quality and macroalgal assemblages has been widely investigated along the European coasts (Benedetti-Cecchi et al., 2001; Soltan et al., 2001; Arévalo et al., 2007; Pinedo et al., 2007; Falace et al., 2010; Piazzi et al., 2011). Monitoring tools have been developed in Europe to meet the requirements of the Water Framework Directive (WFD) and then the Marine Strategy Framework Directive (MSFD), both aimed at maintaining and improving the ecological status of marine coastal waters, habitats and resources, delivering an integrated ecosystem-based approach (Borja et al., 2010).

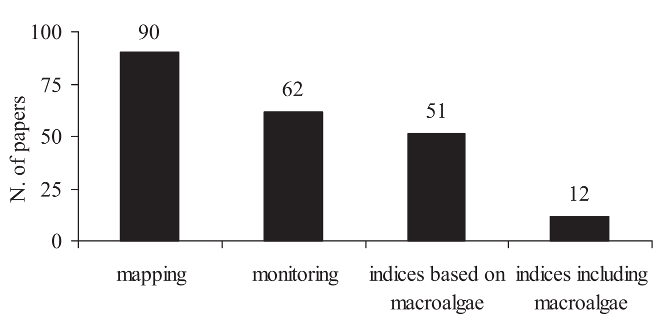

Fig. 1. Subject partitioning of papers considered in the review.

The WFD splits the coastal marine ecosystem into several biological quality elements (BQE), then compares the structure of these (such as species complement) individually before combining them and attempting to determine the overall condition; in contrast the MSFD concentrates on the set of 11 descriptors (biodiversity, alien species, fish stocks, food-webs, eutrophication, sea-bed integrity, hydro-morphology, contaminants in the sea, contaminants in seafood, litter and introduction of energy/noise) which together summarize the way in which the whole system functions (Borja et al., 2010). The MSFD requires all European marine waters to be in 'Good Environmental Status' (GES) which is reached when the 11 descriptors do not deviate significantly from the undisturbed state (Zampoukas et al., 2013). Both of these Directives need to evaluate the ecological status of coastal areas through the definition of 5 status classes (high, good, moderate, poor, bad). Their relative boundaries are obtained as the ratio between the values of the BQE observed and the values for these elements in a site where there is no or very minor disturbance from human activities (reference conditions) (Ballesteros et al., 2007).

The WFD states that macroalgae are a BQE for coastal rocky bottom habitats, to be used in defining the ecological status of a transitional (intertidal) or coastal water body (subtidal) (Guinda et al., 2008; Kelly, 2013). Specifically, the WFD outlines the criteria that need to be met for specific reference conditions for macroalgae: 1) taxonomic composition corresponds totally or nearly totally to undisturbed conditions, 2) there are no detectable changes in macroalgal abundance due to anthropogenic activities (Wells et al., 2007). In this context, several indices based on macroalgae or including macroalgae have been developed to assess the ecological quality of coastal marine ecosystems (Ballesteros et al., 2007; Wells et al., 2007; Juanes et al., 2008; Neto et al., 2012; Cecchi et al., 2014). Fucales are the most utilized biological quality elements and several population parameters (i.e. frond density, frond length frond-length/total frond-length ratio and taxonomic richness of epibionts) are considered valuable indicators of the ecological status of coastal waters (Wallenstein et al., 2013; de Casamajor et al., 2019; Mancuso et al., 2018).

3.1.2. *Other European areas*

The ALGAMONY project was developed to harmonize monitoring methodologies applied in Finland, Norway, Sweden and Denmark and to use common approaches (Moy et al., 2010). Two metrics were chosen for further common work: total cover of erect macroalgal species and the lower depth distribution limit of selected macroalgae (Moy et al., 2010).

The Norwegian Program for mapping and monitoring of marine biodiversity began in 2003 and integrates data on habitat and species distributions in coastal areas (Bekkby et al., 2013; Bartsch et al., 2015). The Norwegian Climate and Pollution Agency in 2003 funded a mapping and monitoring program of *Saccharina latissima* (Linnaeus) C.E.Lane, C. Mayes, Druehl & G.W.Saunders (Bekkby et al., 2013). The Nature Index (NI) includes a total of 65 indicators, including algae, and it is proposed as permanent tool of nature management and political planning (Oug et al., 2013).

In the Baltic Sea, long term monitoring started in 1993/1994 and 50 sites were added in 2006/2007 (Vahteri and Vuorinen, 2016). Field work was carried out by SCUBA on 50 m fixed transects to reveal temporal changes in *Fucus* belts. The abundance and distribution of nine of the most dominant macroalgal species were considered. A model based on publicly available macroalgal monitoring data estimated macroalgal productivity (Öberg, 2006).

3.1.3. *Africa*

South African kelp forests have been a valuable source of scientific information particularly as a result of the Kelp Bed Ecology Program that took place in the 1970s and 1980s but there has been little long-term monitoring of kelp forests in the region (Blamey and Bolton, 2017). Surveys were carried out on two exploited species of kelp, *Ecklonia*

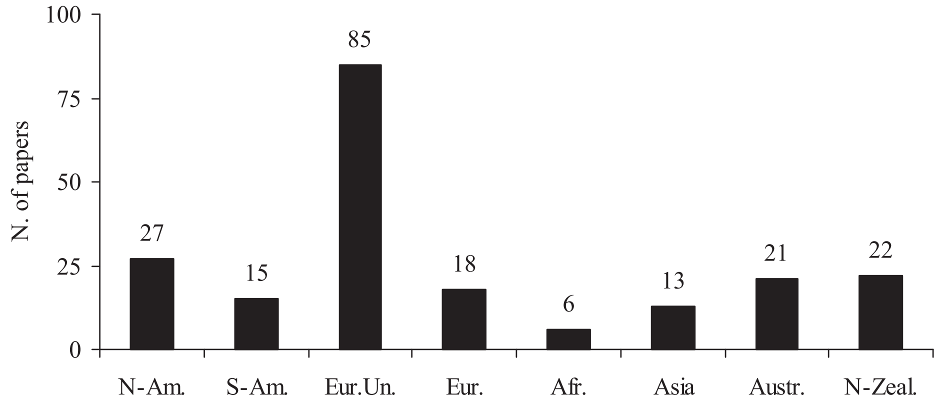

Fig. 2. Geographic partitioning of papers considered in the review.

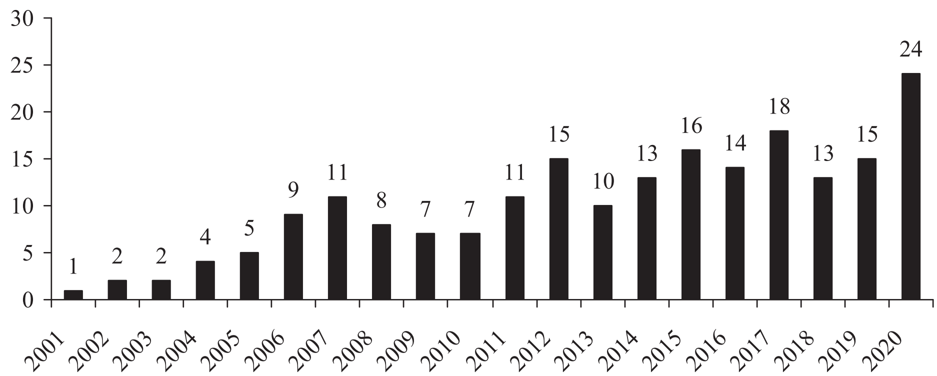

Fig. 3. Mean number per year of scientific papers on seaweed monitoring.

maxima (Osbeck) Papenfuss and *Laminaria pallida* Greville, that reach the surface at low spring tides (Anderson et al., 2007; Bolton et al., 2012). Monitoring was carried out on 900 km of coast using several methods: infrared aerial photography, multispectral aerial imagery, Landsat satellite imagery and physical mapping with hand-held GPS. Repeat photography of historical images gathered from different sources allowed the detection of visual changes in coastal habitats including the progressive easterly spread of *E. maxima* (Reimers et al., 2014). New mapping was obtained by multi-resolution satellite imagery (Dunga, 2020).

On the Mediterranean coasts of Africa, monitoring programs based on macroalgae have recently started through methods already applied in Europe (Chabane et al., 2018; Bahbah et al., 2020).

The suitability of macroalgae as a biological quality indicator for La Reunion reef flats was assessed linking the abundance and composition of macroalgae to water physicochemistry, in the context of the application of the WFD in the outermost French regions (Zubia et al., 2018).

3.1.4. North America

There is an extensive literature on the giant kelp *Macrocystis* forests and ecosystems in North America (Schiel et al., 2015). Californian kelps have been mapped and monitored since 1967 (Bell et al., 2015) through different projects such as the Long Term Ecological Research (LTER) program, the Santa Barbara Coastal Project (SBC) (Castorani et al., 2018; Bell et al., 2020), and the Channel Islands National Park (California) "vital signs" program (Davis, 2005).

In Washington State, *Macrocystis pyrifera* (Linnaeus) C.Agardh and *Nereocystis leutkeana* (K.Mertens) Postels & Ruprecht beds have been monitored through aerial surveys since 1988 (Van Wagenen, 2015). Oregon kelp forests have been long-term monitored through satellite imagery to evaluate the effects of climate changes (Hamilton et al., 2020). In Alaska, there is a monitoring program for *Nereocystis* undertaken by the Kachemak Bay National Estuarine Research Reserve (Mayne Island Conservancy Society 2010). In 2014 the Northwest Straits Commission launched a regional survey of *N. leutkeana* beds using a kayak-based survey (Bishop, 2016).

Long-term monitoring by a consortium of organizations called MARINe (Multi-Agency Rocky Intertidal Network) and led by the University of California Santa Cruz has carried out regular intertidal surveys at fixed monitoring sites along the entire Pacific Coast of North America and several East Coast sites. The monitoring focus is on key species within fixed plots, allowing the dynamics of rocky intertidal species to be monitored (https://marine.ucsc.edu).

PISCO (Partnership for Interdisciplinary studies of Coastal Oceans) kelp forest monitoring has been running continuously since 1999 in shallow (5–20 m depth) nearshore sites located on rocky bottom habitats through standardized visual census sampling protocols (http://www.piscoweb.org).

Reef Check has built a global network of volunteers SCUBA divers that monitor reefs by visual census worldwide through three programs: the Tropical Reefs Program, the Baja California-Mexico Program, and the California Program (Freiwald and Wisniewski, 2015).

In Canada, long-term data on kelp biomass and extent, spanning 30 to 65 years, has been used to monitor kelp beds (Filbee-Dexter et al., 2016). The Coastal Environmental Baseline Program, launched in 2016, extends over 5 years and aims to collect comprehensive baseline data on the state of 6 marine ecosystems (https://www.dfo-mpo.gc.ca/science/environmental-environnement/cebp-pdecr). In British Columbia, an estimate of the total standing crop, biomass, and kelp beds of *N. luetkeana* and *M. pyrifera* was performed for the first time in 2007 using colored infrared photography (Sutherland et al., 2008). A collaborative kelp forest ecosystem monitoring program was carried out from 2009 to 2013 in a shallow rocky reef ecosystem (Trebilco et al., 2014).

3.1.5. South and Central America

Surveys of *Undaria pinnatifida* (Harvey) Suringar were assessed in Argentina (Casas and Piriz, 1996), while in Chile the government has implemented a co-management and conservation plan, 'Management and exploitation areas for benthic resources' (MEABR), where several macroalgal species under commercial exploitation have to be monitored (Almanza and Buschmann, 2013). The populations of *Lessonia nigrescens* Bory in open access areas, exploited areas and Marine Protected Areas were monitored and compared along the northern Chilean coast (Vega et al., 2014). High-resolution satellite imagery was used to describe temporal and spatial distribution patterns of kelp beds in the Beagle Channel using Spectral Mixture Analysis (SMA) (Huovinen et al., 2020).

Various surveys of macroalgal assemblage structure, and its variation in time and space, have been developed in the tropical American zone (Carballo et al., 2002; Lopez et al., 2017) and macroalgal cover has been used as a key indicator of reef state (Bruno et al., 2009, 2014). In 2015, following repeated blooms of floating *Sargassum* spp., the French Interdepartmental Crisis Management Operational Centre produced maps over four areas of interest in the French Antilles islands to determine the extent of the algae bloom (https://emergency.copernicus.eu).

In Brazil, macroalgal assemblage have been monitored in impact assessments (de Szechy et al., 2017; De Paula et al., 2020). Recently, an ecological index based on macroalgae developed in Europe has been employed (Caldeira and Reis, 2019).

3.1.6. Australia

In southern Australia, there are both laminarian (*Macrocystis*, *Lessonia*, *Ecklonia* and the invasive Japanese kelp *Undaria*) and fucoid genera (mostly *Sargassum* and *Cystophora)* (Womersley, 1992). Despite the numerous scientific papers published about kelp and macroalgal assemblages in Western Australia, there are few studies based on long-term monitoring data (Wernberg et al., 2009; Edgar and Barrett, 1999; Hart et al., 2004). An Intertidal Reef Monitoring Program (IRMP) was established in 2003 at many marine reserves using standardised visual census methods to assess the abundance and diversity of algae and invertebrates (Hart and Edmunds, 2005). Investigations into the health of reefs on the Adelaide metropolitan coast has occurred since the late 1990s with surveys within the Reef Health research program, led by the South Australian Research and Development Institute (SARDI) Aquatic Sciences (Turner et al., 2007; Collings et al., 2008; Westphalen, 2008). In 2000, the Department of Primary Industries, Water and Environment and the University of Tasmania, started a project including mapping and survey of *Macrocystis* and *Undaria pinnatifida* beds (Edyvane, 2003). In Western Australia, algal functional groups were used as indicators of change to the benthic communities, through the analysis of digital imagery within a monitoring program established in marine parks (Bellchambers et al., 2009).

In the Perth region there are active monitoring programs which evaluate macroalgal assemblages to assess the quality of benthic habitat or water quality supported by the Department of Environment and Conservation, the Department of the Environment and Heritage and the Environmental Protection Authority (Smale et al., 2011).

The cover of macroalgae is included as descriptor in an index to assess the ecological status of coral reefs (Thompson et al., 2020).

3.1.7. New Zealand

In New Zealand macroalgae have been recommended as being suitable for use in monitoring programs based on a range of different research studies (Shears and Babcock, 2007; Wing and Jack, 2007; Shears, 2010; Shears, 2010, 2017; Hewitt, 2014), but the implementation of monitoring programs has not been consistent at local, regional or national scales. Hewitt (2014) reviewed marine environmental monitoring in New Zealand to assess whether comprehensive programs could be developed based on existing sampling programs and noted that a significant issue was the absence of standardised methodology, a poor current understanding of natural variability, and limited national coverage. There are some examples of long-term datasets gathered by researchers (e.g. Schiel, 2011) and through programs implemented by the Department of Conservation (e.g. Shears and Babcock, 2007a, 2007b; Shears, 2010) and Regional Councils (e.g. Shears, 2017). Hurd et al. (2004) summarised the studies that had reported quantitative macroalgal abundance at various locations in New Zealand.

Monitoring in New Zealand marine reserves has been focused especially on fish and invertebrates with the inclusion of few macroalgae species (Pande and Gardner, 2009, 2012; Battershill et al., 1993; Wing, 2006; Wing and Jack, 2007; Zintzen, 2014). Customary Fishery Protection Areas have established within the Ngāi Tahu rohe (boundaries) which are providing some baseline data (e.g., East Otago Taiāpure, D'Archino et al., 2019). Recently, the distribution and the conservation status of New Zealand macroalgae has been assessed and their potential as ecological indicators has been evaluated (D'Archino et al., 2019; Nelson et al., 2019).

3.1.8. Asia

Japan has the highest diversity globally of kelp species, with 38 species recorded from the region, and a long history of harvest for utilization as a food material (Fujita, 2011). Data about 86 species of habitat-forming seaweeds (fucoids and temperate kelps) have been collected from 1887 to 2014 at 7673 sites on the Japanese coast from warm to cold temperate zones, representing a tool to evaluate long-term changes (Kumagai et al., 2016; Arita et al., 2020). In 2003 Japan's Ministry of the Environment established a monitoring program which also included marine ecosystems and, since 2008, seaweed communities had been monitored at six sites, featuring the kelp and Fucales (Watanabe et al., 2012). A combination of destructive and nondestructive quadrat sampling methods, has been used to determine species composition, coverage, biomass and vertical distribution of seaweeds (Terada et al., 2019).

The ecological status and trophic level of rocky bottoms in an Iranian bay was assessed through the Ecological Evaluation Index developed in Europe (Alavian et al., 2018).

3.2. Mapping

The distribution and extent of macroalgal beds are key to planning monitoring programs and following the temporal evolution of the beds. A wide range of approaches have been employed to measure and map the extent of macroalgal beds. These approaches may be divided in indirect (optical sensors on satellites, aircraft and drones or acoustic devices) and direct methods (video and direct observations).

3.2.1. Indirect methods

3.2.1.1. Optical remote sensors on satellites. Remote sensors have been widely used to obtain data about the distribution of kelp beds (Deysher, 1993; Mora-Soto et al., 2020; Schroeder et al., 2019). Satellite imagery was employed to map *Macrocystis pyrifera* beds in the USA (Jensen et al., 1980, 1981, 1987; Cavanaugh et al., 2010, 2011; Augenstein et al., 1991; Finger et al., 2021), *Saccharina longicruris* (Bachelot Pylaie) Kuntze (formerly *Laminaria longicruris* Bachelot Pylaie) and other kelp beds in Canada (Simms and Dubois, 2001; Nijland et al., 2019), *Nereocystis luetkeana* and *Eualaria fistulosa* (Postels & Ruprecht) M.J.Wynne beds in Alaska (Stekoll et al., 2006), *Macrocystis* beds in New Zealand (Meng et al., 2015), kelp forests in Brittany (Belsher and Mouchot, 1992), Galicia (Casal et al., 2011a, 2011b) and the Baltic Sea (Lõugas et al., 2020), *Sargassum* and/or *Turbinaria ornata* (Turner) J.Agardh beds in Western Australia (Hoang et al., 2016), Thailand (Frouin et al., 2012, Noiraksar et al., 2014), Polynesia (Andréfouët et al., 2004; Betzabeth and de los Angeles, 2017), subtidal macroalgal beds in Turks and Caicos Islands (Mumby and Edwards, 2002), Brazil (da Silva et al., 2017) and Japan (Frouin et al., 2012) (Table S2). Remote sensors have been also tested to map floating macroalgal assemblages such as floating *Sargassum* canopies in the Caribbean Sea (Wang and Hu, 2016) and the tropical North Atlantic (Ody et al., 2019) and *Ulva prolifera* O.F.Müller (Hu et al., 2017; Qiu et al., 2018; Zheng et al., 2020) (Table S2).

The main problems related to the use of remote sensors are the shallow penetration in the water column, restricting mapping of the lower distribution limit of macroalgal beds, and the difficulty in distinguishing between seagrass and seaweeds and among different macroalgal taxa. Several studies have focused on exploring if macroalgae or seagrass species can be discerned from each other based on their optical signatures. In New Zealand, SPOT imagery allowed seagrass and red macroalgae to be distinguished (Israel and Fyfe, 1996). The MERIS configuration of spectral bands allowed the recognition of three indicator species for the Baltic Sea, *Cladophora glomerata* (Linnaeus) Kützing (green macroalga), *Furcellaria lumbricalis* (Hudson) J.V.Lamouroux (red macroalga), and *Fucus vesiculosus* Linnaeus (brown macroalga) (Kutser et al., 2006). MERIS imagery was also used to determine irradiance levels corresponding to the lower limit of kelps in the Azores (Amorim et al., 2015). The use of hyperspectral remote sensing in the Baltic Sea indicated that the depths where benthic macroalgae could be separated from each other did not differ significantly in clear or turbid coastal waters (Vahtmäe et al., 2006).

3.2.1.2. Aircraft. Aerial photographs are available dating back to the 1930s (Klemas, 2011) and were one of the first methods used to map kelp forests. Fyfe et al. (1999) used aerial photography (colour negative film and a UV filter) to map *Macrocystis* beds in Otago (New Zealand). A protocol to estimate the biomass and extent of *Nereocystis* and *Macrocystis* beds has been developed in British Columbia (Canada) using aerial colored infrared photography and digital mapping of kelp polygons directly from georeferenced digital images (Sutherland et al., 2008).

Airborne hyperspectral sensors have been widely used to monitor macroalgal beds in Artic fjords (Volent et al., 2007), in the Baltic Sea (Vahtmäe et al., 2012), in Spain (Casal et al., 2012, 2013) in Germany in the North Sea (Oppelt et al., 2012; Uhl et al., 2016), in Portugal (Gameiro et al., 2014), in the Indian peninsula (Ratheesh et al., 2019) (Table S3). LIF sensors could detect and characterize different macroalgae (red, green and brown) based on their specific pigments (Utkin et al., 2014).

3.2.1.3. Drones/UAVs. The use of drones, or unmanned aerial vehicles (UAVs), is an emerging technique in remote sensing. Drones are becoming more affordable, smaller and easier to maneuver, with significant potential for environmental studies. Miniaturized sensors are being developed/adapted for UAV, including hyperspectral imagers, LIDAR, synthetic aperture radar, and thermal infrared sensors (Klemas, 2015). UAVs can access remote areas and can provide higher resolution than satellites being closer to the targets and covering smaller areas, providing data for integration with satellite data and in field observations; UAV-mounted multispectral sensors proved accurate assessments of individual canopy-forming species (Rossiter et al., 2020). The use of a low-cost drone allowed coastal fish nursery areas to be identified and accurately mapped in Giglio Island (Italy) (Ventura et al., 2016). In Australia, UAV has been used to capture < 1 cm resolution data from intertidal reefs, providing reliable estimates of the dominant species *Hormosira banksii* (Turner) Decaisne (Murfitt et al., 2017). In the northern Gulf of Alaska, drones were used for intertidal monitoring of seagrass and macroalgae communities (Konar and Iken, 2017). UAVs have been used for monitoring intertidal and shallow subtidal macroalgal biodiversity through both RGB and multispectral imaging sensors in New Zealand (Tait et al., 2019).

3.2.1.4. Acoustic devices. Acoustic devices include Single Beam Echo Sounders (SBES) and Multi Beam Echo Sounders (MBES) as well as Side Scan Sonar (SSS). SBES and MBES provide accurate information about bottom hardness and roughness and have been used for bathymetric and habitat mapping especially in deep waters. SBES produces an echogram of the sea floor at a point directly below the transducer, while the MBES emits sound waves in a fan shape beneath the vessel and provides a fuller coverage. As echo sounders are not affected by water clarity, they have been successfully used in turbid and exposed waters and can be mounted on large vessels or small boats, and allow for surveys in shallow waters, although some limitations exist. Acoustic devices have restricted use in depths of 2–5 m, mostly due to sidelobe effect and bottom reverberation or multiple reflections (Madricardo et al., 2017). However, recently MBES have been used to map extremely shallow water in the Venice lagoon (Montereale Gavazzi et al., 2016) suggesting improved technology.

SBES and/or MBES have been used to map macrophytes in the artic fiords of the Svalbard Archipelago (Kruss et al., 2008, 2012, 2017), in shallow reefs in Victoria (Australia) (Che Hasan et al., 2014; Holmes et al., 2008; Schimel et al., 2020), in the Gulf of Maine (McGonigle et al., 2011), in the North Sea (Mielck et al., 2014), in Venice's lagoon (Madricardo et al., 2017), in Japan (Minami et al., 2010) and New Zealand (Pallentin et al., 2016).

SSS is an acoustic imaging device used to provide wide-area, high-resolution pictures of the seabed ideal for object detection and habitat mapping, and it has been also employed to obtain density and distribution of kelps (Zabloudil et al., 1991).

Acoustic investigations have been employed to obtain information about the primary production of an Australian temperate macroalgal system (Randall et al. 2020).

3.2.2. Direct methods

The most utilized direct methods to map sea bottom include observations (Blanfune et al., 2016b; Casas-Valdez et al., 2016) and video (Guinda et al., 2012) along transects perpendicular or parallel to the coast. These techniques are not suitable to map wide areas but they are useful for obtaining accurate maps of relatively small surfaces and are mostly used together with indirect methods to confirm information obtained through optical or acoustic devices.

Underwater hyperspectral imaging (UHI) has been deployed on underwater vehicles to map different targets on the seafloor including kelp (Johnsen et al., 2016) and coralline algae (Mogstad and Johnsen, 2017) outside the limits of passive remote sensing techniques.

Recently, several citizen science programs have been established to support kelp bed monitoring through the participation of trained volunteers in Australia (Westphalen, 2008), California (Freiwald and Wisniewski, 2015) and Ireland (Schoenrock et al., 2020).

3.2.3. Combined approaches

In most cases, mapping needs combined approaches in order to cover large areas and to obtain accurate identification of benthic habitats (Scanlan et al., 2007). Moreover, different approaches may be used at different depth ranges. Mapping was carried out using a combination of techniques e.g., aerial photography, side scan sonar, SBES and MBES, ROV, SCUBA diving and snorkeling in New Zealand (Kerr and Grace, 2005, 2006a, 2006b, 2013, 2015; Funnell et al., 2005; Leleu et al., 2012; Byfield, 2013).

A combination of data acquired with MBES and video has been used to map macroalgal beds in Victoria (Australia) (Ierodiaconou et al., 2007, 2011), in the Kent Island Group in south-eastern Australia (Jordan et al., 2005), and in East Antarctic (Bajjouk et al., 2015).

In the Mediterranean Sea, rhodolith beds between 50 and 100 m depth were mapped through the combination of MBES, SSS and underwater video (Barbera et al., 2012; Basso et al., 2016).

Percentage cover, biomass, distribution, and potential habitat mapping of macroalgae were assessed through a combination of high-resolution satellite data and field visual sampling techniques in Indonesia (Setyawidati et al., 2018), in the Antarctic Peninsula (Kotta et al., 2018) and in Canada (Fretwell and Boyer, 2010).

Remote monitoring of seaweed habitats has been achieved coupling high-resolution aerial and satellite imageries (Brodie et al., 2018; St-Pierre and Gagnon, 2020).

3.3. Sampling

Macroalgal assemblages can be studied by destructive methods through the total scraping of substrate (Piazzi and Ceccherelli, 2020), photographic methods associated with determination of main taxa/morphological groups (Piazzi et al., 2019b; Bellchambers et al., 2009) and visual census techniques (Díez et al., 2012).

Destructive methods are widely recognized as a suitable approach to describe the structure of assemblages and are necessary for studies on biodiversity and biomass (Arévalo et al., 2007; Piazzi et al., 2010). Destructive sampling allows the identification of cryptic species and collection of representative voucher material suitable for checking identifications in the light of changing/improved taxonomy in the future. However, these methods are not suitable to be used in sensitive habitats. The correct identification of organisms needs great expertise and time-consuming analysis of samples; thus, although the method is appropriate for studies with specific objectives or to identify cryptic alien species (Piazzi et al., 2018), it is difficult to process the high number of replicates required for ecological studies and monitoring surveys (Balata et al., 2011).

The use of photographic techniques together with determination of main taxa/morphological groups has been considered a suitable and cost-effective method to study macroalgal assemblages, particularly in the context of monitoring programmes and environmental impact assessment (Bellchambers et al., 2009; Cecchi et al., 2014). Moreover, photographic sampling involves less time spent underwater and the collection of a large number of samples such as required in ecological studies in habitats with high spatial variability (Benedetti-Cecchi et al., 2001).

The use of *in situ* visual sampling within frames enclosing a standard area of the substrate has been shown effective, but requires longer working time in the field than photographic methods. Thus, it is widely used in intertidal habitats (Vinagre et al., 2016) but its use is limited in subtidal habitats. Rapid Visual Assessment (RVA) methods have been proposed for a seascape approach (Gatti et al., 2015). RVA methods can capture additional information compared to photographic techniques, such as the size of thalli or the stratification of assemblages (Gatti et al., 2015). The swath/band transect method was used to monitor kelp beds in San Nicolas Island (California): permanent 10-m swaths, which run perpendicular to the main 100-m transect, were sampled by divers to determine densities of kelps (Kenner and Tomoleoni, 2020).

In Portugal, an ocean modular submersible platform was developed to *in situ* monitor seaweed assemblages (Santana et al., 2020).

3.4. Ecological descriptors

3.4.1. Species or morphological groups

Species-level identification has been the first approach to study macroalgal assemblages. This approach can enable a complete list of organisms to be assembled, the evaluation of the abundance of cryptic species, early identification of new introductions of alien species, and obtaining values of diversity (Konar and Iken, 2009). Thus species-level identifications are irreplaceable when addressing particular objectives. However, marine benthic algae are a very diverse organisms, for which identification at species level is a demanding task, and, in the case of small-sized species, often impossible in the field. Furthermore, identification of some species is based on subtle characters related to the particular conditions of the specimens examined (e.g. presence and morphology of reproductive structures); if these characters are not observable, a species-level identification is impossible even for a skilled taxonomist.

Groupings at marine benthos taxonomic levels higher than species (mainly family or order) is considered suitable for zoobenthos, but its use for macroalgal communities is not obvious (Díez et al., 2010). In fact, species belonging to the same supra-specific taxon (genus or family) may show different ecological characteristics as thalli with the same architecture but with a different spatial arrangement may respond differently to stress (Balata et al., 2011).

A widely used approach to describe macroalgal assemblages is grouping species into empirical morphological or ecological categories, reducing costs and processing time and enabling a larger number of replicates to be processed (Benedetti-Cecchi et al., 2001). In this context, functional group classifications have been widely used in oceanic habitats to describe assemblages (Vadas and Steneck, 1988; Lirman and Biber, 2000). A link between morphological habit and ecological function has been suggested, arguing that predictable patterns of growth forms emerge under given levels of environmental stress or disturbance; hence, morphological groups may allow prediction of stress/disturbance levels in given environments (Littler and Littler, 1980; Steneck and Dethier, 1994). These descriptors are particularly useful when community patterns are compared among different geographical regions or different types of habitats which host different sets of species (Phillips et al., 1997). The need to consider the spatial extent and design of any biodiversity monitoring programme when choosing cost-effective alternatives to species-level data collection has been also highlighted (Smale, 2010). In the Mediterranean Sea, an expanded concept has been proposed, in which the traditional morphological groups proposed by Littler and Littler (1980) are further subdivided based on thallus structure, growth form, branching pattern and taxonomic affinities (Balata et al., 2011). All these aspects contribute to determine responses to stress/disturbance. The 35 newly defined groups do not require professional taxonomic expertise necessary for species identification, and at the same time represent more natural groups that have been observed to show more uniform responses than the traditional functional groups (Balata et al., 2011).

3.4.2. Alpha diversity

Diversity of assemblages is a “taxonomy-based metric” (richness metric) very common in many ecological studies and widely shared among existing assessment methods (Birk et al., 2012). The alpha diversity considers the number of species in a community and/or the number of species and the relative abundance of individuals. Biodiversity can vary at different spatial and temporal scales (Gray, 2000) and the correspondence between values of diversity at small and large spatial scales is not self-evident but depends on patterns of spatial variability of each system (Gray, 2000). Thus, high values of diversity at small spatial scales may correspond to low values of diversity at larger spatial scale and vice versa, making patterns dependent on the spatial scale examined. Thus, alpha diversity could be considered below different spatial scales: point diversity (a single sample), alpha diversity (samples within a habitat), gamma diversity (the diversity of a larger unit, such as an island or landscape) and finally epsilon or regional diversity (the total diversity of a group of areas of gamma diversity) (Gray, 2000).

A decrease in species richness in stressed conditions has been widely described for macroalgal assemblages (Soltan et al., 2001; Arévalo et al., 2007); in fact, under stress or disturbance, sensitive species reduce or disappear leading to a reduction of alpha diversity (Piazzi et al., 2012; Piazzi and Ceccherelli, 2020). Also, the macroalgal functional diversity has been observed to be sensitive to environmental alterations (Balata et al., 2011). Thus, the number of taxa/morphological groups per sample has been considered an effective indicator of ecological quality (Cecchi et al., 2014).

3.4.3. Beta diversity

Beta diversity represents a further aspect of diversity, which may be evaluated at two different levels: between habitats, normally referred to turnover diversity, or within each habitat as the measure of the heterogeneity of assemblages (Gray, 2000).

Under stressed conditions, the importance of biotic factors in regulating macroalgal distribution decreases, and species occurrence and abundance mostly follow the gradient of stress intensity (Piazzi and Ceccherelli, 2020). The loss of structuring perennial species and the proliferation of ephemeral algae lead to a widespread biotic homogenization, and to a consequent reduction of beta diversity (Piazzi and Balata, 2009; Piazzi et al., 2011). Thus, the beta diversity of assemblages may be considered a valuable indicator of human pressure (Cecchi et al., 2014; Piazzi and Ceccherelli, 2020).

Beta diversity, in general, can be calculated through different methods (Gray, 2000). An effective method to be employed in monitoring survey and impact evaluation studies is the evaluation of the variability of species composition among sampling units (heterogeneity of assemblages). This variability may be measured in terms of multivariate dispersion calculated on the basis of distance from centroids through permutational dispersion multivariate analysis (PERMDISP; Anderson et al., 2006). Thus, any changes in compositional variability displayed by PERMDISP may be directly interpretable as changes in beta diversity (Anderson et al., 2006).

3.4.4. Sensitivity levels

Presence/absence and abundance of sensitive taxa is an "autoecology-based metric" in which taxa are categorized in relation to their sensitivity to stress or disturbance. Correlative and experimental studies highlighted major shifts in the structure of macroalgal assemblages subjected to several kinds of stressors (Gorgula and Connell, 2004; Rodriguez Prieto and Polo, 1999). Thus, the presence and abundance of some taxa/morphological groups may be considered as a main indicator of the ecological status of macroalgal assemblages (Pinedo et al., 2007). Recently, a method has been proposed to distinguish and measure the sensitivity to disturbance and the sensitivity to stress, the former causing mortality or physical damage and the latter physiological alteration of the sessile organisms (Montefalcone et al., 2017). Following this approach, a Sensitivity Level (SL) value has been assigned to each taxon/morphological group on the basis of its abundance in areas subjected to different levels of anthropogenic stress (Ballesteros et al., 2007; Cecchi et al., 2014).

3.5. Ecological indices

In the context of the international conservation of aquatic systems, the strategies currently adopted require the identification of biotic indices suitable for assessing the ecological quality of coastal marine ecosystems (Borja et al., 2010; Martinez-Crego et al., 2010). Over recent years, many indices have been developed following different approaches (Birk et al., 2012). The earlier indices considered sensitivity/tolerance of indicator species or ecological groups to stress induced by water quality alteration (Orfanidis et al., 2003; Rosenberg et al., 2004; Ballesteros et al., 2007), while more recent indices combine a variety of ecological descriptors and/or follow an ecosystem-based approach (Personnic et al., 2014; Rastorgueff et al., 2015; Thibaut et al., 2017).

3.5.1. Indices based on macroalgae

BENTHOS is an index developed on the Catalan coast (Spain, Northwestern Mediterranean) for rocky shore communities situated in the upper sublittoral. BENTHOS uses ordination tools (DCA analysis) and correlational evidence to classify samples and species along an environmental gradient, (Pinedo et al., 2007). The index showed a gradient from Fucales-dominated to *Ulva*-dominated communities, with other intermediate stations dominated by corallines.

CARLIT (CARtography of LITtoral and upper-sublittoral rocky-shore communities) is an index based on the degree of development of Fucales on rocky shorelines (Ballesteros et al., 2007). The sampling survey consists of a visual assessment from a small boat, driven as close as possible to the shoreline, to detect the dominant macroalgal community along the upper infralittoral rocky shore. The result is a partition of the rocky shoreline in several sectors, each one characterized by a community category (corresponding to a single community or combination of communities) (Ballesteros et al., 2007). The CARLIT index is officially recognized as an institutional monitoring tool in Spain since 1999 (Ballesteros et al., 2007), in Italy since 2004 (Mangialajo et al., 2007; Asnaghi et al., 2009; De La Fuente et al., 2018) and in France since 2006 (Blanfune et al., 2011, 2017). Several studies have assessed and implemented CARLIT (Blanfune et al., 2011, 2017; Cavallo et al., 2016; De la Fuente, 2015; Jona Lasinio et al., 2017) and this method has been applied to various regions in the Mediterranean Sea, for example in Albania (Blanfune et al., 2016a), Algeria (Bahbah et al., 2020), along the Lebanese coastline (Badreddine et al. 2018), in four Tyrrhenian Islands (Jona Lasinio et al., 2017), in the Adriatic Sea (Nikolić et al., 2011, 2013; Sfriso and Facca, 2011), and in the Alboran Sea (European Coast) (Bermejo et al., 2013). The distribution of *Fucus virsoides* J.Agardh in the Gulf of Trieste was assessed with the CARLIT method (Orlando-Bonaca et al., 2008). A simplified CARLIT method has been recently suggested (Blanfune et al., 2017).

EEI (Ecological Evaluation Index) was designed to estimate the ecological status of transitional and coastal waters (Orfanidis et al., 2011). The EEI index quantifies shifts in the marine ecosystem structure and function, evaluated by classifying marine benthic macrophytes in two groups (ESGs I, II) representing pristine and degraded ecological states. ESG I includes seaweed species with thick or calcareous thalli, low growth rates, long life cycles, and seagrass, whereas the ESG II includes sheet-like and filamentous seaweed species with high growth rates, short life cycles (opportunistic) and Cyanophyceae. Spatial and temporal changes in benthic macrophytic communities are identified by seasonal sampling of ecologically uniform non-overlapping permanent-polygons recommended for well-defined ecosystems, e.g., lagoons, shallow closed bays or permanent lines on open coasts (Orfanidis et al., 2003). Preliminary assessment of the ecological status of Slovenia coast with the EEI concluded that benthic macrophytes and EEI could be valuable tools for the implementation of the WFD within the Mediterranean eco-region (Panayotidis et al., 2004; Orlando-Bonaca et al., 2008), and it was also tested on the Albanian rocky shore (Gogo, 2015), Istrian coast (Iveša et al., 2009), in Iran (Alavian et al., 2018) and Brazil (Caldeira and Reis, 2019).

ESCA (Ecological Status of Coralligenous Assemblages) is based on analyses photoquadrats of coralligenous macroalgal assemblages in the Mediterranean Sea. Assemblage descriptors selected as metrics of the ESCA index are: presence/absence and abundance of sensitive taxa/groups (expressed as sensitivity level of assemblages), diversity of assemblages (expressed as α-diversity) and heterogeneity of assemblages (expressed as β-diversity). The three metrics were combined to give a final value for the multimetric ecological index, Ecological Quality Ratio, calculated as the ratio between the measured values and the value obtained in the reference condition. The ESCA index is the first ecological index using rocky, deep macroalgal assemblages to classify coastal waters (Cecchi et al., 2014). ESCA was tested in responses a gradient of human pressures e.g., sites with low human-induced pressure characterized by a stratified structure with a dominance of erect species, while turf algae were dominant in highly impacted sites (Piazzi et al., 2015b). Recently ESCA index was improved with sessile macroinvertebrates (Piazzi et al., 2017b) and used in impact assessment studies (Penna et al., 2018; Piazzi et al., 2019a, 2021b).

ICS (Index of Community Structure) has been developed as a single numeric descriptor to assess the structural state of macroalgal communities and to evaluate their relative development on rocky shores. Coverage of seaweed species was sampled in the field and treated by taxonomic groups, size classes and structural and functional groups. The ICS combines three sub-indices It (macroalgal cover), Is (taxonomic stratification) and Io (functional group). Six macroalgal communities corresponding to canopies (belts) distributed vertically on the shore have been investigated for several years in 14 sites in Brittany (France)

(Ar Gall and Le Duff, 2014).

CCO (Cover, Characteristic species, opportunistic species) was developed for the implementation of the European Water Framework Directory (WFD) in coastal waters, using intertidal macroalgal communities as bio-indicators (Biological Quality Element). CCO is based on the calculation of three metrics corresponding to the global cover of macroalgal communities, the number of characteristic species per topographic level/seaweed community, and the cover of opportunistic species. The final rating is obtained by pooling the scores of the three metrics (Ar Gall et al., 2016).

CFR (Calidad de Fondos Rocosos 'Quality of Rocky Bottoms') index is based on the analysis of seaweed communities throughout the depth gradient, and combines the richness of characteristic macroalgal populations, their total cover, the presence of opportunistic species and the physiological condition of the whole macroalgal community (Juanes et al., 2008; Guinda et al., 2014). This index uses an easy to apply methodology that does not require very precise taxonomic identifications because it is based on the assessment of general coverage of large characteristic macroalgae and opportunistic species. The index is considered to be very practical for extensive monitoring work or for its application to subtidal areas (Guinda et al., 2014).

The RSL (Reduced Species List) index was developed for intertidal seaweeds of the British Isles, based on species richness. RSL includes approximately 70 algal species and regional lists have been created for the different geographic areas in the British Isles. The index also includes a score that takes into consideration the physical nature of the habitat and the community structure. The changes in proportions of Rhodophyta and Chlorophyta species are considered to be indicative of anthropogenic influences and shifts in quality status. The Rhodophyta increase in species numbers with increased environmental quality, while the Chlorophyta adapt more readily to changes and increase with decreasing quality status. Several Ochrophyta (Phaeophyceae) species, that are large, cartilaginous and relatively hardy, are more likely to stay constant (Wells et al., 2007). The RSL index was reassessed for the rocky shores of the Atlantic coast of Andalusia (south-western Spain): based on anthropogenic pressures (water turbidity, nutrients, metal concentration and the distance to sources of stress), 19 sites along the coast were classified in quality states and then compared with water quality (Bermejo et al., 2012).

MaQI (Macrophyte Quality Index) is based on macrophytes and consists of two versions developed for experts, E-MaQI (Expert-Macrophytes Quality Index), and for rapid assessment, (R-MaQI). These have been recommended to the ARPAs (Regional Agencies for the Environment Safeguard) for monitoring surveys of Italian transitional waters. The index considers the ecological values of all the macroalgal taxa and seagrasses. In the expert version all taxa are identified to species level, including small epiphytes. The rapid version R-MaQI considers the Rhodophyta/Chlorophyta ratio, the general environmental conditions and presence/absence, biomass and species assemblages of some macroalgae and seagrasses, and the variability of some physico-chemical parameters (Sfriso and Facca, 2010; Sfriso et al., 2009).

MarMAT (Marine Macroalgae Assessment Tool) was developed in Portugal for intertidal rocky reefs. The index includes seven different metrics: species richness, proportion of Chlorophyta, number of Rhodophyta, number of opportunists/ESG I (ratio), proportion of opportunists, shore description, and coverage of opportunists. The index is based on a Reduced Species List for the Portuguese coasts (Neto et al., 2012).

RICQI (Rocky Intertidal Community Quality Index) is a quality index based on indicator species abundance, cover of morphologically complex algae, species richness, and faunal cover (herbivore and filter-feeder cover, proportion of fauna with respect to the whole assemblage). A conceptual model was proposed which describes successional stages of assemblages along a gradient of increasing environmental disturbance and associated values of the metrics included in the index (Díez et al., 2012).

QISubMac (Quality Index of Subtidal Macroalgae) was designed for the evaluation of the quality status of the water bodies along the French Channel and Atlantic coast, and is based on 14 metrics that consider the depth penetration, composition (sensitive, characteristic and opportunistic) and biodiversity of macroalgal assemblages and complies with WFD requirements (Le Gal and Derrien-Courtel, 2015).

ALEX (ALien Biotic IndEX) has been developed to evaluate biological invasions in soft-bottom macro-invertebrate assemblages (Cinar and Bakir, 2014). A modified version of ALEX has been recently proposed to evaluate biological invasions in macroalgal assemblages (Piazzi et al., 2015a). ALEX was applied in a Marine Protected Area where a recreational-fishing port is present, supporting the suitability of the index to detect spatial and habitat differences within a MPA where some non-indigenous macroalgae are at early stages of spread (Piazzi et al., 2018; 2021a). Moreover, ALEX was also employed in impact evaluation studies (Piazzi et al., 2020).

The Ecological Status has been assessed in Catalonia (northwestern Mediterranean, Spain) considering four different ecological strategies (competitor, indifferent, stress-tolerant, opportunist) of macroalgae, emphasizing the importance in the distinction between competitor and stress-tolerant species (Pinedo and Ballesteros, 2019).

PAN-EQ-MAT (PAN for general use, EQ for ecological quality and MAT for Macroalgae Assessment Tool) has been suggested as a tool that combines several features adapted from the RSL, CFR and MarMAT. Developed for the Azorean coastline, it considers intertidal rocky shore seaweed community features for the assessment of ecological quality of coastal water (Wallenstein et al., 2013).

3.5.2. Indices based on ecosystem properties which incorporate macroalgae

In addition to indices focused primarily on macroalgae, there are also indices based on ecosystem properties which incorporate macroalgae (Table S4).

EBQI (Ecosystem Based Quality Index) considers the whole structure and functioning of the ecosystems and it has been applied to different habitats (Personnic et al., 2014; Rastorgueff et al., 2015).

CAI (coralligenous assemblages index, Deter et al., 2012) has been applied in the monitoring program RECOR (Agence de l'eau RMC/ Andromède Océanologie) to describe the ecological quality of coralligenous habitats at 120 stations (distributed between 17 and 120 m) along the French Mediterranean coast since 2010 through photographic sampling (Holon et al., 2013). COARSE (COralligenous Assessment by Reef Scape Estimation) index uses SCUBA diving observations and measurements to gather data useful to evaluate the state of coralligenous reefs as an indicator of sea-floor integrity rather than coastal water quality (Gatti et al., 2015).

INDEX-COR is based on an integrated approach to assess the ecological quality of coralligenous reefs (Sartoretto et al., 2017).

MAES (Mesophotic Assemblages Ecological Status index, Cánovas-Molina et al., 2016), CBQI (Coralligenous Bioconstruction Quality Index, Ferrigno et al., 2017) and MACS (Mesophotic Assemblages Conservation Status, Enrichetti et al., 2019) have been developed on the basis of ROV (Remotely Operated Vehicle) photography and video footage in order to assess the status of mesophotic megabenthic assemblages from hard bottom substrates.

ISLA (Integrated Sensitivity Level of coralligenous Assemblages) was proposed as a method to differentiate between disturbance and stress to assess the ecological status of the coralligenous assemblages (Montefalcone et al., 2017).

Reef-EBQI (Ecosystem-Based Quality Index) is an integrative index recently developed for Mediterranean shallow, algae-dominated rocky reefs, between 1 and 10 m (Thibaut et al., 2017).

3.5.3. Comparison between indices

The proliferation of these indices has generated comparative assessment studies of implementation of some indices e.g., RLS vs CARLIT (Bermejo et al., 2014), EEI, E-MaQI and BENTHOS (García-

Sánchez et al., 2012), CARLIT vs EEI (Nikolić et al., 2011), COARSE vs ESCA (Piazzi et al., 2017a), COARSE vs ESCA vs ISLA (Piazzi et al., 2019a).

4. Discussion

Monitoring macroalgae appears to have greatest utility for identifying changes associated with specific stressors (such as sewage effluent, eutrophication, sedimentation, fishing practices, climate change) and measuring the effectiveness of management regimes, expressed in terms of recovery of populations or ecosystem function.

By the present study, numerous macroalgal monitoring programs have been developed, but there is a large variability among geographical areas. In fact, many regions have not been surveyed because they do not have the needed infrastructure or funding and/or occur in cold, turbid, deep, or wave-exposed environments far from road access (Krumhansl et al., 2016).

In contrast, in Europe, the recent E.U. Directives led to the development of indices as tool for monitoring the ecological quality of coastal systems. The application of these indices e.g., CARLIT, EEI, in the European Union has become part of regular monitoring, and the indices have been used to categorize areas affected by different degrees of pollution or anthropogenic impacts (Orfanidis et al., 2003; Mangialajo et al., 2007; Cecchi et al., 2014). The disappearance of Fucales/Laminariales and the predominance of turf algae are frequently used to characterize areas affected by significant environmental impacts (Ballesteros et al., 2007).

In other geographic regions, most studies focused on mapping the distribution of kelps or Fucales. Mapping macroalgae beds can be carried out in a number of ways depending on the scale of interest, by direct observations (e.g., videography, drop camera), or by indirect methods using remote sensing techniques (optical and acoustic). A critical review of 195 studies of the optical and acoustic remote-sensing techniques used to map seagrass beds showed that a multi-approach is needed, as there is no single technique that can acquire all the required data to map seagrass distribution (Hossain et al., 2015). It is evident from our review of mapping techniques that a multi-approach is needed to map macroalgae beds as the methods applied to date have advantages and limitations. The mapping methods need to be tailored to different habitats and environments, considering practicability, efficiency and, not least, cost. The size of the area and the depth range to be mapped also need to be considered. Intertidal macroalgae, subtidal beds, and species with surface floating canopies such as *Macrocystis*, require different approaches.

Optical remote sensing has developed hugely in the last decades and different sensors are currently available. However, penetration of the water column is still very shallow, especially in turbid water. Increasing sedimentation with the resultant turbid water has been recognized as a major threat to the coastal environment. With less light penetrating the water column, the maximum depth of macroalgae is expected to decrease (e.g., Desmond, 2016), thus monitoring the lower depth of macroalgae in coastal areas will give a measurement of change. At present, relatively few studies have tested mapping subtidal macroalgae forests in turbid water (e.g., Kutser et al., 2006; Vahtmäe et al., 2006; Casal et al., 2011b) and these usually required a multi-approach including ground-truthing with biological samples or images.

Satellite and airborne imagery have been successfully used in temperate water to map kelp beds, particularly species with floating canopies (e.g., *Macrocystis* and *Nereocystis*), and in clear water to map coral reefs, seagrass beds, and intertidal seaweed beds. The advantage of aerial surveys, in contrast to satellite imagery, is the ability to plan the flight time, as well as the capacity to choose the type of optical sensors to use. However, it is difficult to distinguish which seaweed species are in the water when examining an aerial photo. Dark rocks or rocks covered in short seaweed species are difficult to determine from aerial/satellite imagery. Satellite-born sensors and aerial photography have been effective, but these distant sensors cannot operate effectively in turbid temperate waters, and many image surveys do not account for changing tides (Bennion et al., 2019).

Mapping intertidal to shallow subtidal macroalgae and *Macrocystis* beds could be achieved with drones at an affordable cost compared to aerial surveys and commercial satellite imagery. With drone derived imagery, there will also be challenges in identifying macroalgae from the images, dealing with light reflection, attenuation through the water column, and changing water clarity over time.

Acoustic techniques such as side scan sonar, single beam, and multibeam echo sounders, have become more sophisticated and can provide a three-dimensional representation of the seafloor, and an indication of the associated habitat (Madricardo et al., 2017). Underwater videography is an effective method for mapping and ground truthing data acquired by remote sensing, usually at a large scale. Towed underwater video systems provide direct observations of species in their natural habitat, are cost-effective, simple to operate, and provide a valuable, non-destructive method to enable habitats to be monitored. As a mapping tool the main limitation is the spatial interpolation between the *in situ* data points or transects (Dekker et al., 2005).

A combined approach is considered effective in most cases (Byfield, 2013; Basso et al., 2016; Kotta et al., 2018) and it may represent a base for developing a standardized mapping protocol for seaweeds that can aid management and conservation efforts (Bennion et al., 2019).

5. Conclusions

The assessment of status and trends in macroalgal cover and quality is an emerging priority for ocean and coastal management. In fact, the number of scientific papers on this topic greatly increased in recent years, mostly in the European Union as consequence of the Water and Maine Strategy directives. However, the results obtained suffer as a consequence of limited coordination across the numerous programs that have been developed, which vary widely in goals, methodologies, scales and governance approaches. Also, it is now necessary to compare the large number of different indices available to see if there can be calibration between approaches. The comparisons conducted to date between ecological indices have mostly led to the same conclusion, namely that a multi-approach can be effective in providing more complete information, ranging from the community to the seascape level, about the alteration of ecological quality (Piazzi et al., 2017a). Thus, it is necessary to harmonize marine macroalgal observations, identifying common metrics and approaches in sampling design, field measurements, taxonomy resolution and data management, in order to develop standardized procedures which may allow to compare the data obtained within each coastal system (Duffy et al., 2019). Finally, suitable networks should be developed to ensure that information, from field surveys to data management, may be archived and shared among stakeholders to facilitate the development of monitoring plans and conservation measures.

CRediT authorship contribution statement

R. D'archino: Conceptualization, Writing - original draft. **L. Piazzi:** Conceptualization, Writing - original draft.

Declaration of Competing Interest

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

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Appendix A. Supplementary data

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