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RELEVANT PUBLICATIONS ON MARINE CITIZEN SCIENCES

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Presented by Simone Panigada, Chair of the ACCOBAMS Scientific Committee

Issue: presentation of relevant publications on marine Citizen sciences

1. Action requested

The Scientific Committee is invited to:

a) **note** the information provided in publications on marine Citizen sciences when developing recommendations on this issue.

2. Background

Resolution 7.6 “Work Programme and Budget for the triennium 2020-2022”, adopted by Parties at MOP7 in November 2019, requests the Scientific Committee to:

- review the current citizen sciences initiatives in the ACCOBAMS area;
- evaluate the relevance of “Citizen Science” input of cetaceans’ sightings in expert-supervised databases;
- produce basic guidelines on the use and how to gather information.

This document is a support document to the ACCOBAMS-SC14/2021/Doc37, “Review of the current citizen sciences initiatives in the ACCOBAMS Area”.

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Review

Tracking Marine Alien Macroalgae in the Mediterranean Sea: The Contribution of Citizen Science and Remote Sensing

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Abstract: The accelerating rate of the introduction of non-indigenous species (NIS) and the magnitude of shipping traffic make the Mediterranean Sea a hotspot of biological invasions. For the effective management of NIS, early detection and intensive monitoring over time and space are essential. Here, we present an overview of possible applications of citizen science and remote sensing in monitoring alien seaweeds in the Mediterranean Sea. Citizen science activities, involving the public (e.g., tourists, fishermen, divers) in the collection of data, have great potential for monitoring NIS. The innovative methodologies, based on remote sensing techniques coupled with in situ/laboratory advanced sampling/analysis methods for tracking such species, may be useful and effective tools for easily assessing NIS distribution patterns and monitoring the space/time changes in habitats in order to support the sustainable management of the ecosystems. The reported case studies highlight how these cost-effective systems can be useful complementary tools for monitoring NIS, especially in marine protected areas, which, despite their fundamental role in the conservation of marine biodiversity, are not immune to the introduction of NIS. To ensure effective and long-lasting management strategies, collaborations between researchers, policy makers and citizens are essential.

Keywords: non-indigenous species; Mediterranean Sea; monitoring; managing; citizen science; remote sensing; Landsat 8 OLI

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1. Introduction

Non-indigenous species (NIS), organisms introduced from beyond their natural (past or present) geographical region and outside of their natural dispersal potential, are a major threat to biodiversity and natural ecosystem functioning [1–4]. NIS that have large established and expanding populations may become invasive alien species (IAS), which could result in significant environmental, socioeconomic and human health impacts [2,5–8].

In the marine environment, IAS may have substantial negative impacts on native biota (e.g., substitution of native species and biodiversity loss due to habitat modifications, alterations in community structure and ecosystem service changes) [4,5,9]. For instance, they compete with native species, and may change native benthic communities, perhaps leading to an impoverishment of subtidal communities [10,11]. Moreover, they may cause degradation of seagrass meadows, having a severe negative impact on coastal protection [5].

The high number of NIS make the Mediterranean Sea a genuine hotspot for marine biological invasions, in terms of both the number of species and the rate of introduction [12–14]. The conspicuous increase in the rates of introduction and expansion of NIS can be correlated with the intensified research efforts involving marine NIS and the increase of stakeholder involvement and of citizen science initiatives [14,15].

Marine protected areas (MPAs), whose major aim is biodiversity conservation, may be highly affected by NIS invasions, and the impact of NIS on marine resources and habitats can be significant, even highly detrimental [16–20]. To date, the effect of MPAs on NIS is still not fully understood, and it is unclear whether or not MPAs favor NIS expansion via tourism activities, e.g., boat anchors and diving [21–23]. Since NIS represent serious threats at multiple levels, they have to be targeted by scientists, conservation managers, policy makers and citizens to increase the amount of information on their distribution and spread dynamics and impacts, with the main goal of taking prevention and mitigation measures. In the marine realm, the management of NIS is particularly difficult [24]. Prevention is certainly the cheapest and most cost-effective option for reducing the risk of future NIS introduction and the spread of ones that have already emerged [25]. In this respect, monitoring and surveillance plans, which greatly assist managers and policy makers in their decisions on the prevention or mitigation actions to be taken, are crucial. Regularly updated, space-temporal information on the distribution and abundance of NIS is fundamental for the assessment of effective management plans [26,27]. However, traditional monitoring and mapping methods (field survey and sampling campaigns) are time-consuming, costly and limited in space and time. In this respect, citizen science activities and remote sensing may be useful complementary tools for monitoring the distribution and spread of NIS.

Citizen science activities, involving the public (e.g., tourists, fishermen, divers) in the collection of data (regularly updated and validated by scientists), in addition to their important role in educating and improving public awareness, could be useful resources for collecting space-temporal information on NIS distribution, abundance and spread [19,28–30]. When properly designed, citizen science activities can provide scientifically reliable data on species' distribution and abundance [30,31]. Currently, the number of citizen science initiatives has increased enormously thanks to the new technologies and social media [15,19,32].

Remote sensing techniques (RS), using passive multispectral imaging sensors, operating in the visible (VIS), near infrared (NIR) and shortwave infrared (SWIR) on both satellite and airborne platforms, have been even more widely used for providing Earth observation (EO) data for the continuously growing applications in different sectors, including sea water monitoring at different scales [33–36]. In particular, these optical sensors, measuring the energy reflected and emitted from the Earth's surface, allow researchers to not only map the extent of shallow benthic or intertidal marine habitats, but also to identify key marine species [37]. Satellite sensors at intermediate ground resolution (0.3–1 km), such as the moderate resolution image spectrometer (MODIS), managed by NASA, or Sentinel-3, operated by ESA, systematically provide the so-called "Ocean Color" data. This information, linked to parameters like chlorophyll or pigment concentration, can even support the detection and monitoring of NIS in shallow water seabeds, floating on the sea surface or in the water column. In general, mapping NIS through RS in coastal and shallow water may require higher ground resolution that can be suitably supported by other remote sensing sensors, which are currently available and based on airborne (including UAV—unmanned aerial vehicle) or other high-resolution (HR) orbiting satellites and very high-resolution (VHR) optical sensors. The HR satellites can provide a repetitive coverage for monitoring evolving phenomena with ground resolution until 30 m (i.e., Landsat 8 Operational Land Imager (OLI) operated by NASA, and Sentinel-2 MultiSpectral Instrument (MSI) operated by ESA).

In addition, the same ground resolution is provided by the PRISMA (Hyperspectral Precursor of the Application Mission) hyperspectral sensor, made recently available by the Italian Space Agency (ASI), which, because of its hundreds of acquisition bands in the spectral ranges of interest, may constitute a very promising tool in this specifying sector. Moreover, the currently available sensors based on satellite (VHR), airborne or UAV technologies, may provide tailored solutions to specific seabed, water quality and NIS detection and monitoring needs [34]. Once suitably corrected for atmospheric noises (e.g., turbidity, clouds), the EO data provided by this new generation of polar satellite multi/hyperspectral sensors have proven to be effective and operative for environmental conservation and the monitoring of the coastal marine ecosystems and water quality, even in moderately turbid shallow waters [35,36].

These RS techniques are recognized as effective tools for determining species diversity and distribution, for quantifying biomass and primary production, as obtained from the photosynthetically available radiation (PAR) and leaf area index (LAI), and for monitoring their changes over space and time in shallow waters [36,38–43]. In any case, they must be combined with in situ measurements [44] of biophysical parameters of interest in order to support the proper calibration/validation of the EO data. Moreover, these data may be widely exploited for the multiscale/multitemporal systematic monitoring and mapping of the increasing presence of NIS in shallow water and marine ecosystems, which are typically affected by their worldwide spread, driven by rising effects of oil/gas shipping (especially in the Mediterranean Sea), eutrophication/pollution and climate change. When the high water turbidity (>100 NTU—Nephelometric Turbidity Units)) limits the exploitation of RS techniques, the vertical side-scan sonar can be a suitable acoustic integrated tool for mapping the vegetation, including NIS, on the seabed of coastal zones [45].

As biological invasions are highly impacting the Mediterranean Sea biodiversity, and many coastal zones and shallow waters are vulnerable to NIS invasion, the use of cost-effective systems, e.g., citizen science and remote sensing, may offer many important contributions to the knowledge and management of NIS [19,46–48].

In this paper we demonstrate that citizen science and the operative multispectral satellite sensors (remote sensing) can be used as effective complementary tools for scientific and extensive monitoring in the Mediterranean Sea, with potential as NIS early warning and monitoring systems, fundamental to support ecosystem-based sustainable management. Relevant literature, updated until November 2020, was searched for using standard scientific databases (e.g., Google Scholar, Web of Science, Scopus and ResearchGate) and analyzed. The search was performed using various combinations of the following list of keywords on the subject: "citizen science", "remote sensing", "marine", "alien", "non-indigenous", "invasive", "species", "macroalgae", "Mediterranean Sea".

2. Citizen Science

Out of 17 records, only 5 were related to alien seaweeds and citizen science activities. Indeed, the great majority of records concerned the monitoring of marine invasive fishes (e.g., the project "AlienFish" launched by Ente Fauna Marina Mediterranea and the project "Is it Alien to you? Share it!!!" launched by the iSea online platform) [15,49–51]. All of the five selected papers (hereafter referred to as the cases), were related to the monitoring of three invasive *Caulerpa* taxa, *Caulerpa cylindracea* Sonder (along the Sicilian and Ligurian coasts, in the western Mediterranean), *Caulerpa taxifolia* (M. Vahl) C. Agardh (Spanish, French, Italian and Croatian coasts) and *Caulerpa taxifolia* var. *distichophylla* (Sonder) Verlaque, Huisman and Procaccini (along the Maltese coasts, in the central Mediterranean) in the Mediterranean Sea [19,52–58] (Figure 1). All the *Caulerpa* taxa showed invasive behavior with significant impacts on the native communities.

According to the authors of [59], *C. cylindracea* was first recorded in the Mediterranean Sea off the coasts of Libya in 1990, whereas the authors of [60] date the first record back to 1985 in Tunisia. This alga is able to compete with native species and may change native benthic communities, leading to an impoverishment of subtidal communities [10,11,61]. In particular, it may enter into competition with native seagrasses, such as *Posidonia oceanica* (L.) Delile, mainly when their meadows are stressed and degraded [10,62,63]. It is also able to clog and break fishing nets by the mats it forms. *Caulerpa taxifolia* (M. Vahl) C. Agardh (invasive aquarium strain) was first recorded in the Mediterranean in 1984 [62]. It affects photophilic algal communities, causing a drastic reduction in diversity, and it is also able to compete with *P. oceanica* (for the interception of light or the utilization of nutrients) and interfere with it by the production of secondary metabolites (allelopathy) [64,65]. *Caulerpa taxifolia* var. *distichophylla* is the most recently introduced *Caulerpa* in the Mediterranean Sea. It was first recorded in Syria in 2003 ([66] as *C. mexicana*). It showed invasive behavior in Sicilian waters, with significant impacts on native ecosystems [67–69]. Similar to *C. cylindracea*, it is also able to clog and break fishing nets [67].

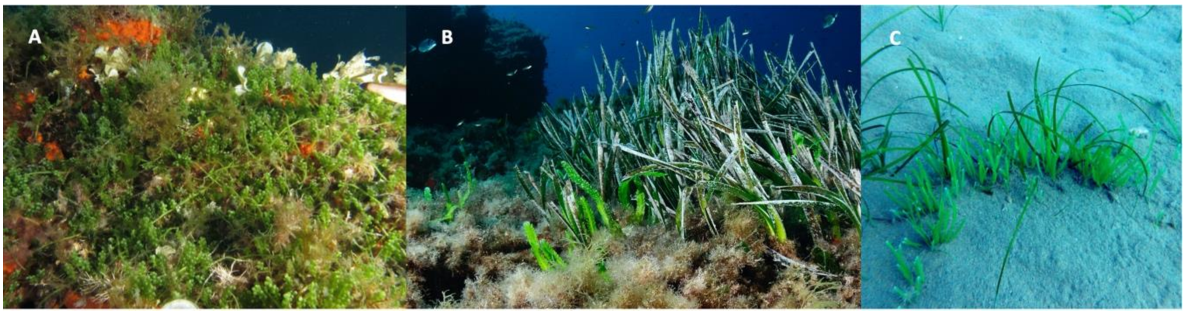

Figure 1. *Caulerpa cylindracea* with macroalgae and sponges (Secca del Toro—Favignana, 15 m depth; photo by Sergio Zanoni; from Reference [70]) (**A**). *Caulerpa taxifolia* in a *Posidonia oceanica* meadow (Strait of Messina, 12 m depth; photo by Alessandro Pagano; from Reference [70]) (**B**). Small patches of *Caulerpa taxifolia* var. *distichophylla* at Termini Imerese (photo by Marco Toccaceli; from Reference [56]) (**C**).

Cases 1 and 2, referring to the monitoring of *C. cylindracea* with the exclusive involvement of citizen scientists, also stressed the second mission of citizen science activities, i.e., to promote environmental and scientific education and awareness in engaged participants. Case 3 refers to a monitoring campaign (1991–1992) on *C. taxifolia* organized in the Mediterranean coast of France [52], involving sea users to whom brochures on the species were distributed. Case 4 was related to international campaigns for public awareness on *C. taxifolia*, supported by the European Commission (see [54]), organized by French, Italian, Spanish and Croatian scientists, who elaborated leaflets and posters for distribution to sea users. In Case 5, the involvement of citizen scientists (e.g., snorkelers and recreational divers) in the monitoring of *C. taxifolia* var. *distichophylla* was a complementary activity to the field research surveys.

2.1. Western Mediterranean

2.1.1. Case 1

Case 1 refers to the citizen science project "*Caulerpa cylindracea*–Egadi Islands" [19,55], addressed to different groups of volunteers (i.e., citizens, fishermen, snorkelers and divers) that aimed to collect data (place, date, depth and substrate coverage %) and photos on the distribution of *C. cylindracea* within the Egadi Islands MPA (Aegadian archipelago). Data were sent through a mail address, a Facebook page, the MPA website or with the filling in of a form available online or in the MPA's offices. This MPA (instituted in 1991), the largest Italian MPA, is located approximately 7–9 km from the western coast of Sicily (Italy, Tyrrhenian Sea). Only data validated by the scientific team of the project were gathered in the database.

The project registered approximately 160 sightings of *C. cylindracea,* mainly recorded at Favignana, the largest island. The alga was found in different habitats, between 0 and 40 m depth, on rock, rock with sediment and sand. It was also recorded in valuable habitats such as vermetid reefs, *Cystoseira* communities (upper infralittoral zone) and coralligenous formations (Figure 2). The alga was more frequent in *Cystoseira* communities and vermetid reefs than in coralligenous formations, showing coverage values mainly ranging from 20% to 50%, but also reaching values higher than 50% in the vermetid reefs. Anchoring activities, mainly carried out by pleasure boats, seem to have encouraged the spread of *C. cylindracea*. This was also highlighted by some biological traits of the alga, e.g., the ability of creating bridges with its stolons over native communities and forming compact multilayered mats that were able to trap the sediment.

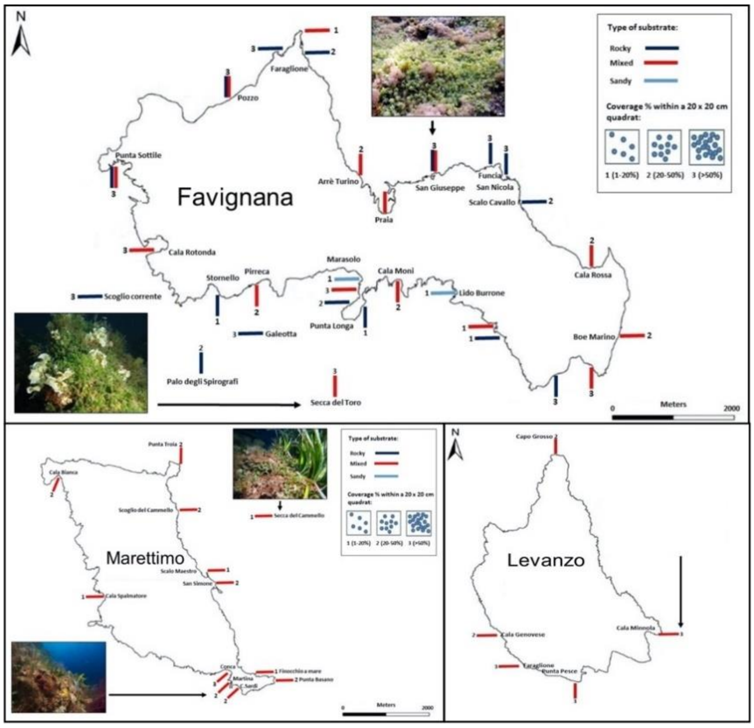

Figure 2. Occurrences of *Caulerpa cylindracea* within the Egadi Islands marine protected area (MPA) (modified from Reference [55]).

Records concerning other NIS and cryptogenic species (see Reference [71]: a species that cannot be included with confidence among native nor among introduced species) were also gathered, e.g., *Asparagopsis taxiformis* (Delile) Trevisan, *Aplysia dactylomela* (Rang, 1828), *Fistularia commersoni* (Rüppel, 1838), *Rhopilema nomadica* (Galil, Spannier and Ferguson, 1990) and the biofouler worm *Branchiomma bairdi* (McIntosh, 1885), and would have been favored by the increase of sedimentation among the stolons of *C. cylindracea*. In the wake of this project, the project "Aliens in the sea", which aimed to collect data on 19 NIS (including *C. cylindracea*) along the Sicilian coasts, was launched in 2017. The project is still active and allowed for the gathering of new records of *C. cylindracea* from the Egadi Islands MPA [32].

2.1.2. Case 2

Case 2 was related to the monitoring of *C. cylindracea* along the Ligurian coast, including the Portofino MPA [57]. The monitoring was carried out by Reef Check Italia Onlus (RCI), a non-profit organization involving volunteer divers in the Mediterranean Sea that developed protocols for coastal environment monitoring. Volunteer divers, after a one-day intensive training course, were able to make observations on the presence/absence and abundance of the target species.

The monitoring (2006–2014) provided useful information on the spread of this IAS, highlighting the expansion in distribution and abundance of this alga. Since 2006, the species has spread rapidly and new sites have been recorded. *Caulerpa cylindracea* was first recorded in Portofino MPA in 2007 at San Fruttuoso Bay, one of the three boat corridors within the MPA, and since then, it has rapidly spread throughout the MPA, which appears to be severely affected by this NIS. Along the Ligurian coast, the alga was mainly recorded in the coastal rocky bottoms (74% of alga occurrences) and between 5 and 10 m depths.

2.1.3. Case 3

Case 3 refers to a monitoring campaign (1991–1992) on *C. taxifolia* organized in the Mediterranean coast of France [52]. Brochures were distributed to sea users, who were requested to report sightings and information on this alga. Dives were also carried out in the newly colonized sites. Data from Spain and Italy were also gathered, and a rapid increase of the spread of the alga has been observed since 1990. Useful information on the biology, ecology and dynamics invasion of the alga was also collected. Five different stages in the invasion process were described and the colonization effects on *P. oceanica* meadows were also observed. The authors also highlight the role that maritime traffic and fishing may play in the spreading of the alga.

2.2. Mediterranean Countries—Case 4

Case 4 was related to some monitoring campaigns on the spread of *C. taxifolia* conducted in Mediterranean countries affected by the invasion of this species [53,54]. The first international campaign for public awareness on *C. taxifolia* was organized in 1993 and 1994 by the Laboratoire Environment Marin Littoral (LEML) and supported by the European Commission (see Reference [54]). French, Italian and Spanish scientists participated in the campaign organization and distribuited elaborated leaflets and posters to sea users. In 1997, the second international campaign for public awareness was organized to map the distribution of *C. taxifolia* in the Mediterranean Sea (see Reference [54]). This campaign, coordinated by the LEML and the Groupement d'intérêt scientifique (GIS) Posidonie, was always supported by the European Commission. The message was, "Wanted *Caulerpa taxifolia*. If you find this seaweed, do not help it to spread, and phone us". Leaflets, containing information on the biology, ecology, dissemination methods and spread of the species, were distributed to sea users in Spain, France, Italy and Croatia as well as to scientific institutions. Several institutions and associations (e.g., Lions Clubs, divers, Institut National des Sciences et Technologies de la Mer, etc.) contributed to this public awareness campaign. The majority of known locations were confirmed by sea users and new locations were gathered. All the information was rigorously verified, the sightings were mapped and the reports were regularly updated. This campaign was an effective tool for updating the distribution of *C. taxifolia* and also for helping to plan measures to slow down the spread of the alga. The authors of [53] described the status of *C. taxifolia* invasion at the end of 2000 in six Mediterranean countries (Spain, France, Monaco, Italy, Croatia and Tunisia). The data were also obtained with the support of public awareness campaigns (the distribution of pamphlets and posters). The authors reported that 80% of the area colonized was along 500 km of coastline between Toulon (France) and Genoa (Italy), supporting the hypothesis that the origin of the introduction was Monaco.

2.3. Central Mediterranean—Case 5

Case 5 refers to a monitoring activity (2016–2017) of *C. taxifolia* var. *distichophylla* in Maltese waters [58]. The citizen science monitoring activities, complementary to field surveys, lasted from July 2015 to December 2017, and in particular involved snorkelers and recreational divers. A poster with representative photographs of the alga and details on how to report the species was prepared and distributed to local diving schools and dive clubs, as well as being shared online through social media.

Records received through the citizen scientists, regularly validated, came from sites different from those surveyed by the researchers. Certainly, records from the citizen scientists provided useful information on the habitats preferred by the alga and also helped to document the significant change in the distribution of *C. taxifolia* var. *distichophylla*, confirming that the species is rapidly expanding in Maltese waters.

3. Remote Sensing

Out of six records only one record was related to alien seaweeds and remote sensing. The majority of the detected articles concerned alien plants. The paper refers to the indo-pacific NIS *Hypnea cornuta* (Kützing) J. Agardh in the Mar Piccolo of Taranto, where it was recorded for the first time in 2000 [72]. The first Mediterranean report of the species was from Rhodes Island, Greece (in Reference [73] as *H. valentiae* (Turner) Montagne).

In 2014, the distribution of *H. cornuta* in the Mar Piccolo was mapped using the Landsat 8 OLI multispectral optical sensor in combination with quantitative sampling. Four stations were considered within the two inlets of the Mar Piccolo of Taranto in order to carry out quantitative sampling. Thalli of *H. cornuta* were only found in two stations, Battendieri and Cimino, and the highest biomass values were registered at Battendieri (the station of the first finding). The map of spatial distribution was assessed by means of the EO data provided by Landsat 8 Operational Land Imager (OLI) sensor (Figure 3). The OLI multispectral data were previously corrected for atmospheric noise, scattering/attenuation from image-derived aerosol optical depth (AOD) and adjacency effects. Subsequently, they were classified using a supervised maximum likelihood (ML) parametric algorithm and trained using point sampling data. Although a thematic accuracy superior to 80% was achieved with respect to the available sea truth data, the model could be further improved by expanding the sampling schema.

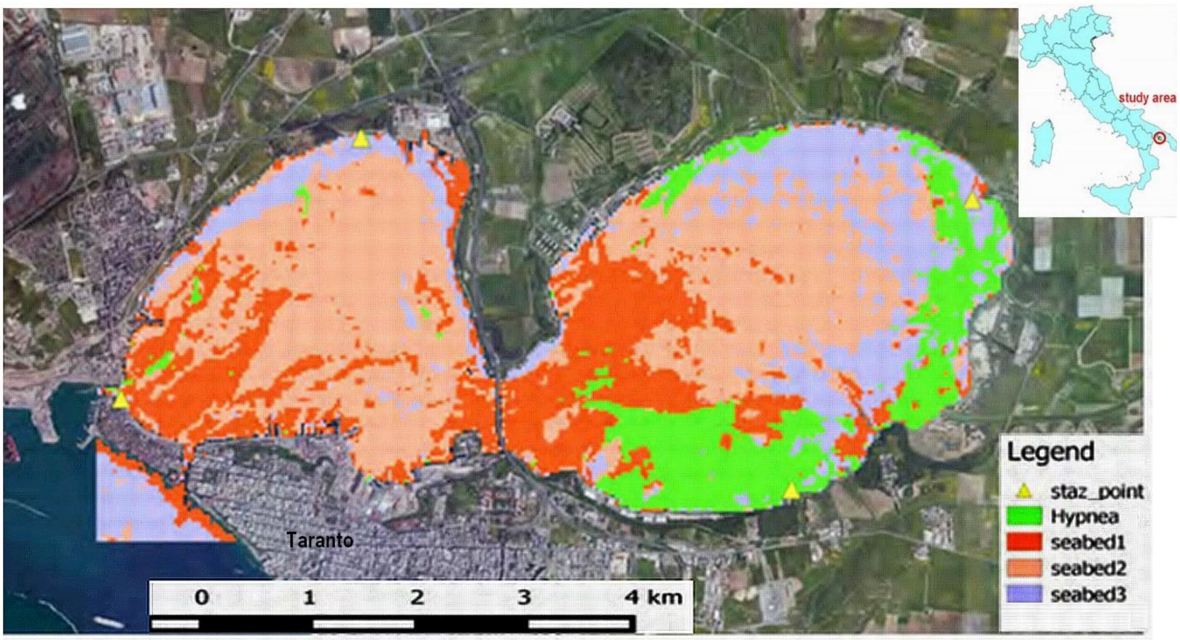

Figure 3. Thematic map of *Hypnea cornuta* from Landsat 8 Operational Land Imager (OLI) data acquired on August 2014 (from Reference [72]).

The distribution map, obtained from the remote sensing satellite techniques, was in agreement with the in situ collected data, seeing as *H. cornuta* was confined to the second inlet of the Mar Piccolo of Taranto. On the basis of the achieved and upgradable results, the authors highlight the promising integration between the remote sensing techniques and in situ/laboratory methods for mapping the distribution of aquatic alien species in shallow waters. Thus, in general, this technology could be a useful tool for suitably supporting the sustainable management of these threatened and fragile coastal environments.

4. Discussion

Our bibliographic search highlights that, despite the fact that citizen science and remote sensing have become increasingly important for NIS monitoring, the number of papers related to alien seaweed monitoring in the Mediterranean Sea using these two techniques is still very low. In the Mediterranean Sea, remote sensing techniques have been mainly used for the detection and prediction of invasive plants and the assessment of their impact. In extra Mediterranean areas; however, these techniques are widely used for mapping NIS or floating marine algae, such as the *Sargassum* species [74,75]. According to the authors of [74], the compact airborne spectrographic imager (CASI) is a suitable tool for mapping NIS, such as *Codium fragile* spp. *tomentosoides* (Van Goor) P.C. Silva in Mahone

Bay, Nova Scotia. Instead, several satellites were used to monitor large pelagic *Sargassum* in the tropical North Atlantic, in particular: the medium resolution imaging spectrometer (MERIS, on board the ENVISAT (ENVIronmental SATellite) with a spatial resolution of 300 m; the moderate resolution imaging spectroradiometers (MODIS, on board the AQUA and TERRA satellites); the visible infrared imaging radiometer suite (VIIRS, on board the SNPP (Suomi National Polar-orbiting Partnership) NASA satellite) with a coarser spatial resolution of 1 km and 750 m, respectively; the high-resolution sensors on board Landsat platforms with a 30 m resolution in coastal areas; the recently launched ESA higher resolution satellite sensors, namely, the ocean and land color instrument (OLCI, 300 m) on board the Sentinel-3; and the MultiSpectral Instrument (MSI, 60-20-10 m) on board the Sentinel-2 [75].

With respect to citizen science in the Mediterranean Sea, the great majority of papers concern the monitoring of marine invasive fishes. In extra Mediterranean areas; however, citizen scientists (fishers, bathers, sailors, volunteers from associations, etc.) have greatly supported the monitoring of NIS. For instance, in the north-eastern Atlantic Ocean, the North Sea and New Zealand, there has been monitoring of large NIS, such as *Sargassum muticum* (Yendo) Fensholt and *Undaria pinnatifida* (Harvey) Suringar [76], which are easy to map because they can also reach the sea surface; additionally, there has been good monitoring of NIS in the ports of Le Havre and Antifer (Normandy, France) [77].

Even if they are few, the reported case studies show that these two techniques may be useful tools to support the traditional methods of alien seaweed monitoring in the Mediterranean Sea. Indeed, they could be effective as early warning instruments of new introduction and as detectors of distribution changing over time.

Citizen science activities, besides improving public awareness, may be very useful in collecting data on introduction and spread of NIS over spatial and temporal scales that otherwise would remain hidden. The goal is achieved and is really effective when the activities are properly designed and communicated and volunteers are appropriately prepared and motivated. In this respect, social medias can have an important role in making the information on the temporal and spatial spread within a certain area easily available [78], and also involving as many categories of volunteers as possible (e.g., see References [49,79,80]). The higher the number of categories we involve, the higher the number of habitats which may be monitored. Since local volunteers know the environment where they live, they are the true early warners of new introductions in their area.

Recently, the interest of the scientific community in these complementary tools has increased (e.g., see References [19,79,81–83]) as proven by the citizen science initiatives on the monitoring of NIS launched in the Mediterranean Sea (e.g., see References [15,49,84]). Indeed, a lot of species that are easy to identify (see the reported case studies), could be included within citizen science initiatives with the dual purpose of raising awareness and early-warning detection [13,85]. Since MPAs are not immune from NIS invasions (as shown in the present study and References [18,19,32]), the involvement of local citizens is also essential for setting management actions to effectively prevent and control marine bioinvasions. Indeed, restrictions upon activities generating economic benefits (tourism, boat anchors, diving, etc.) could raise obstacles to their acceptance from the local population [86,87]. High-quality and continuously updated information on the distribution, spread dynamics, abundance, and pathways of their introduction might greatly assist managers and policy makers in prioritizing prevention or mitigation actions, and for conservation planning [2,88–91].

Advances in remote sensor/platform technologies and processing algorithms are enhancing marine habitat mapping, through an even smaller spatial resolution and better color discrimination. Even if some parameters, such as the submerged depth, may limit NIS detection and quantification from space, satellite imagery datasets with various spatial, spectral and temporal resolutions provided, for instance, by the MODIS, VIIRS, OLCI and MSI satellite sensors, are successful tools to map NIS distribution at a large scale and to detect NIS abundance consistent with in situ observations [75]. Furthermore, CASI is a promising tool for mapping and monitoring NIS in subtidal habitats [74].

5. Conclusions

Biological invasions are an ongoing phenomenon and many NIS are expanding their distribution in coastal waters with negative impacts on the environment, human health and the economy. The Mediterranean Sea is one of the most important biodiversity hotspots in the world [92], but it is also subjected to complex and severe anthropogenic pressures, e.g., biological invasions [2]. Therefore, monitoring the distribution of NIS, in order to have accurate and regularly updated information, is essential for the conservation and management of marine ecosystems and the implementation of policies [89].

To support the ecosystem-based sustainable management of affected coastal areas, rapid and accurate tools for assessing and mapping the abundance and distribution of NIS are required. For this purpose, besides the traditional methods, other powerful and cost-effective methods for monitoring and detecting NIS currently exist, including citizen science activities and EO-based innovative techniques using airborne [35] and satellite [72,74,75,93,94] platforms.

Currently, space technology is globally emerging by private spaceflight and the aerospace industry, with faster, cheaper and better access to space programs and EO data [95,96]. Citizen science is also increasingly developing thanks to technological developments, adequate training courses for citizen scientists and continuous validations of data quality [97,98]. Therefore, these space-based cost-effective methods, complementary with those for in situ sample gathering and analysis, can provide powerful tools for monitoring NIS introductions and their spread, especially in areas like MPAs. Exploiting these synergies is becoming increasingly necessary to ensure effective and long-lasting management strategies based on multidisciplinary collaborations between researchers, policy makers and citizens.

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An integrated approach for cetacean knowledge and conservation in the central Mediterranean Sea using research and social media data sources

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Abstract

1. Sources of data other than those derived from conventional research protocols may contribute valuable information to fill gaps in knowledge about cetacean occurrences and diversity in a given area and help address conservation issues.
2. The performance of a method to examine cetacean communities based on presence records systematically derived from shared photographs and videos posted by boaters and maritime operators on social media (e.g. YouTube and Facebook) combined with patchy distributed visual/acoustic data collected by researchers has been evaluated.
3. Records (*N* = 1,274) gathered over a 10-year period (2008–2017) have been used to obtain insights into species' presence and habitat selection in a scattered study area of the central Mediterranean Sea (Italy). The effectiveness of the method, practical and theoretical advantages, limitations, and challenges of using data originated from social media for research and conservation purposes are discussed.
4. Seven out of the eight cetacean species regularly residing in the Mediterranean have been reported in the area, with different relative densities. Maximum entropy modelling techniques have been applied to the datasets derived from (a) social media, (b) research surveys, and (c) the combination of the two, using six fixed variables as proxies for cetacean presence. Distance from the coast and depth emerged as the main variables predicting encounters, with specificities related to the ecology of the species.
5. The approach was reliable enough to obtain broad-scale, baseline information on cetacean communities in the region, on the basis of which initial conservation recommendations and future research programmes can be proposed.
6. With the increasing need for studying whale and dolphin population ecology coming from national/international directives, support from citizens to aid research may act as a practical, inexpensive solution to gathering extensive spatial–

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temporal data for regional-scale monitoring and for the development of management priorities.

KEYWORDS

cetacean, conservation, distribution, Maxent, Mediterranean Sea, social media

1 | INTRODUCTION

Cetaceans are an integral part of the pelagic and coastal fauna of the Mediterranean Sea. They are major consumers at most trophic levels, with key influences on marine community structure and services (Estes, Heithaus, McCauley, Rasher, & Worm, 2016). They are vulnerable to short-term natural and anthropogenic threats caused by activities on land and at sea, but also to the long-term chronic and cumulative effects of various stressors (Pace, Tizzi, & Mussi, 2015). Knowledge about their distribution and abundance in the Mediterranean basin is still limited with heterogeneous data (Mannocci et al., 2018), and many ecological and conservation questions remain unanswered. Studying cetaceans presents a number of challenges and is severely limited by both logistical and financial constraints. Data collection across different locations and habitats over a number of years, or even decades, requires a large workforce and substantial budget, consequently constraining opportunities for monitoring and research, and resulting in important data gaps (Braulik et al., 2018; Lodi & Tardin, 2018).

It has been recently suggested that other sources of data than those derived from conventional research protocols could help address major knowledge gaps and conservation challenges (Caitlin-Groves, 2012; Di Minin, Tenkanen, & Toivonen, 2015; Goffredo et al., 2010; Klemann-Junior, Villegas Vallejos, Scherer-Neto, & Vitule, 2017; McKinley et al., 2017). For example, incidental sightings by sea users (e.g. recreational sailors and fishermen, professional fishermen) may be a cost-effective method to obtain valuable scientific background information in uninvestigated regions or to acquire data over a wide geographic area (Embling, Walters, & Dolman, 2015; Robinson et al., 2013). With the advent of the Web 2.0 world, and thanks to advances in portable electronic devices (smartphones) and applications during the last decade, people are able to record data, images, and locations about species sightings in the wild at any time and share them on various social media platforms. This unwitting 'citizen sensor network' (Caitlin-Groves, 2012; Goodchild, 2007) allows non-scientists to considerably contribute to research, whether they intend to or not (Bonney et al., 2009).

The opportunity of exploiting social media posts as a way to scan and retrieve different information collected and transmitted by 'connected' citizens, and the use of such information (e.g. text, pictures, or videos) is developing fast (Di Minin et al., 2015; Dylewski, Mikula, Tryjanowski, Morelli, & Yosef, 2017; Mikula & Tryjanowski, 2016). Both conservation science (Barve, 2014; Daume, Albert, & Von Gadow, 2014; Di Minin et al., 2015; Papworth et al., 2015; Richards & Friess, 2015; Roberge, 2014; Saito et al., 2015) and scientific research (e.g. Daume, 2016; Dyderski et al., 2016; Giovos, Ganias, Garagouni, & Gonzalvo, 2016; Leighton, Hugo, Roulin, & Amar, 2016; Mikula & Tryjanowski, 2016; Mori et al., 2017) are increasingly using this novel approach to extract information from various types of social media platforms, such as Facebook (FB), YouTube (YT), Twitter, and Instagram, to provide new insights into the study of certain species or help identify knowledge gaps in their ecology and/or conservation (Dylewski et al., 2017).

In social media, each post contains information about when the content was created or shared; when geotagged, the videos, pictures, and text have geographic coordinates, or place name, showing the location where they were taken or posted from. Thus, social media content bears great potential for monitoring target species at greater spatial scales and temporal resolution than many other available data allow (Longley, Adnan, & Lansley, 2015; McKinley et al., 2017). Of the various types, online video-sharing applications have been shown to have high interactive/usage level in the social media scenario (Khan, 2017; Ricke, 2014). YT is the most well-known video-hosting service, with more than a billion users consisting of nearly 33% of the Internet populace (YT, 2016). Through greater use of smartphones with video recording capabilities, YT represents a tool that facilitates rapid accumulation of data from shared recordings, including those depicting animal sightings and behaviour (Dylewski et al., 2017; Yosef & McPherson, 2016). FB is another commonly used social platform to release texts, videos, and photographs that makes up about 18% of global social media activity (Chaffey, 2016). By tracking photographs and videos posted on platforms such as YT and FB from boaters, maritime tourists, divers, sea lovers, and professionals, potentially valuable information on cetacean species can be extracted (Giovos et al., 2016), providing much larger datasets than could be collected by traditional research alone. Although this approach can provide a substantial array of scientific and conservation benefits, limitations such as data fragmentation and over-reporting in high-use areas (Bird et al., 2014) should be properly evaluated and potential results interpreted with caution (Hann, Stelle, Szabo, & Torres, 2018).

In this paper, the performance of a method to detect and examine cetacean communities based on presence records systematically derived from shared photos and videos on YT and FB combined with patchy distributed visual/acoustic systematic and non-systematic data collected by researchers and stranding data, is introduced and evaluated. Records gathered over a 10-year period (2008–2017) off the

Lazio region (central Mediterranean Sea, Italy) as a paradigmatic site are used to assess the effectiveness of the method, and indicate practical and theoretical advantages, opportunities, limitations, and challenges of using data originated from social media for both research and conservation purposes.

2 | METHODS

2.1 | Study area

The study area is located in the Tyrrhenian Sea (Italy), a zone of the central Mediterranean featuring one of the most complex marine structures in the seas surrounding the Italian peninsula. The study area covers about 39,000 km^2 (Figure 1) and was specifically selected to include a variety of environmental features (e.g. bathymetries) and a range of different habitats (seagrass meadows; hard-bottom communities with coastal banks, cliffs, and caves; seamounts; sand and mud). In the northern section, it includes the islands of Giglio and Giannutri (Tuscany Archipelago), and the islands of Ponza, Palmarola, and Ventotene (Pontine Archipelago) are in the southern portion.

Two marine protected areas (MPAs) are found in the region: the MPA Islands of Ventotene and S. Stefano and the MPA Tor Paterno bank, the only Italian MPA completely submerged. Giglio and Giannutri islands are part of the Tuscany Archipelago National Park and are included in the Pelagos Sanctuary for Mediterranean Marine Mammals, an international area classified as a Specially Protected Area of Mediterranean Importance, subject to an agreement between Italy, Monaco, and France for the protection of marine mammals. In addition, a number of Sites of Community Importance and Special Protection Areas under the Natura 2000 European network are included in the study area (see the Italian Ministry of Environment list by region, http://www.minambiente.it/pagina/sic-zsc-e-zps-italia), and more recently, the marine region including the Pontine Archipelago was acknowledged by the International Union for Conservation of Nature (IUCN) Marine Mammal Protected Areas Task Force (MMPATF) as an Important Marine Mammal Area (IMMA; discrete portions of habitat, important to marine mammal species that have the potential to be delineated and managed for conservation) for the sperm whale (*Physeter macrocephalus*) in the Mediterranean (IUCN-MMPATF, 2017).

The study area includes seven river estuaries, from north to south: Albegna, Fiora, Marta, Mignone, Tiber, Garigliano, and Volturno. The Tiber is the major source of organic material in the Rome coastal area and nearby regions, and is also one of the main contributors of heavy metals in the Mediterranean Sea (Inghilesi et al., 2012; Montuori, Aurino, Garzonio, Nardone, & Triassi, 2016). At about 3 nmi off the two Tiber mouths, there is a terminal comprising two single-point moorings handling crude oil and petroleum products. Navigation, anchoring, diving, and fishing activities are banned within a radius of 0.75 nmi from each single-point mooring. These types of structures are reported to attract some dolphin species (Triossi, Willis, & Pace, 2013).

In the offshore zone of the study area, there are two seamounts, Tiberino and Albano; the former rises from the sea bed at 600 m to a depth of 250 m, the latter from 800 m depth to 300 m (Würtz & Rovere, 2015). It is known that seamounts attract pelagic top predators, particularly sea turtles and cetaceans (Fiori, Paoli, Alessi, Mandich, & Vassallo, 2016).

2.2 | Data sources

Data on the occurrence of cetaceans in the study area over a 10-year period (January 2008 to December 2017) were obtained from three different sources: (1) social media and other sighting records collected by citizens during recreation activities or work; (2) published and unpublished data collected by researchers during scientific surveys; and (3) stranding records from the official (governmental) national stranding network and from other data sources (social media, local newspaper

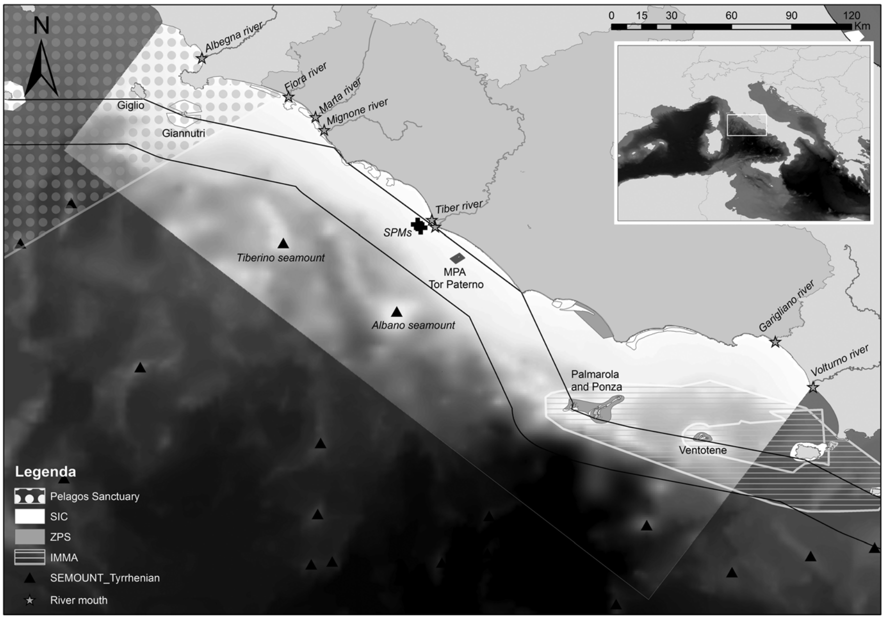

FIGURE 1 The study area is situated in the central Mediterranean Sea (Tyrrhenian Sea, Italy). The two black lines define the Italian coastal baseline and the limits of the Italian territorial sea

websites, further stranding database). All data were organized by season, defined as summer (July to September), autumn (October to December), winter (January to March), and spring (April to June).

For social media data, YT and FB were scanned for all available cetacean visual information (video footage and/or photographs). Both searches were restricted only to videos/photographs shared in personal accounts and not in official YT and/or FB channels from research projects, institutes, or non-governmental organizations, in order to ensure independency in the sample collections (as in Giovos et al., 2016). Three keyword categories were used for searches (for a full list of keywords, see Supporting Information, Table S1): locations (e.g. Ostia, Ponza island), cetaceans (dolphins, whales in Italian language, i.e. delfini, balene), and name of the species regularly present in the Mediterranean Sea (both common in Italian language and scientific). All keywords were extracted using a case-insensitive search technique, and variations of the same word (e.g. 'dolphin', 'dolphins') were compiled as the same keyword. This process, which was repeated iteratively until no new video/photograph was encountered, giving confidence that all available information was identified, resulted in 889 records.

To maximize the efficiency and quality of data retrieval, storage, and analysis, the way in which data were registered was standardized. To avoid biases, records were collected independently by two experienced scientists and the reliability coefficient (Cronbach's alpha; Cronbach, 1951) calculated. The coefficient was very high (0.96) for both YT and FB. The list of records was then filtered to guarantee reliability, reduce variability, and eliminate outliers or more trivial errors. As in Giovos et al. (2016), double entries, different fragments of the same footage, and different pictures of the same sighting were used only once in the final analysis in order to ensure that each dolphin encounter was represented exclusively by a unique video or set of photographs. Moreover, videos with more than one sighting were also excluded, resulting in 557 records being retained for the analysis (Table 1).

The minimum requirements for a species occurrence record (the type of data in this study) include taxonomic identification, data collection time (date or season), and geographical coordinates or location of the sighting. For species identification, only the photographs and video footage in which the diagnostic features of the taxon (i.e. body size and shape, coloration pattern in the dorso-lateral area of the animal's body) were clearly visible were selected. In most cases, this was a relatively easy and straightforward process, and the reliability coefficient between the two experienced researchers was 99%. On a few occasions, when the species could not be reliably determined or confirmed (e.g. animals far away, dorso-lateral area not clearly visible), videos/photographs were recorded as 'undetermined'. Then, geo-referenced/location data available from each video/photograph included in the analyses were identified to determine the locations in the study area (see Supporting Information, Table S1). When the geographical coordinates were unavailable, or locations were not assessable from clear landmarks or other precise descriptions, the originator of the data was contacted to obtain further details on the location of the sighting. For each of the 557 records, the following variables were identified or estimated and extracted by researchers: number of individuals observed, behaviour, presence of immature individuals, interaction with the observation boat, presence of other vessels in the vicinity, presence of professional fishing boats (i.e. trawlers or artisanal boats), and concurrent recreational fishing by observers. Any other information, such as unusual events (e.g. deformities or mutilations, nurturant behaviour) from social media videos was also collected and reported in this study (see Supporting Information). Finally, to ensure data quality, the accuracy of the data extracted from each social media video/photograph was checked by a third researcher after inclusion in the database to verify and validate the data. Along with data collected by citizens and posted on YT and FB, 40 verified and validated cetacean records directly reported to researchers by citizens were considered and included in the social media dataset (Table 1).

As for research data, cetacean sighting records in the study area were collected from different monitoring/research surveys covering various coastal and offshore areas, cetacean species, time frames, and methods (Table 1). The following data sources were used: (a) survey data (n = 326 records) collected from ferries by ISPRA within the Fixed Line Transect Mediterranean Monitoring Network project; (b) boat-based survey data (n = 100 records) collected from different research campaigns by Sapienza University of Rome; (c) boat-based and aerial survey data (n = 56 records) compiled in OBIS SEAMAP (http://seamap.env.duke.edu/; Halpin et al., 2009); (d) boat-based survey data (n = 9 records) collected from a research campaign by Tuscia University.

Finally, cetacean stranding records (n = 191) were extracted from the National Data Bank (n = 164 http://mammiferimarini.unipv.it), which is a real-time updated database by the Italian Stranding Network supported by the Italian Ministry of Environment and managed by the Natural History Museum of Milan and the University of Pavia (Pavan et al., 2013), and from other sources (n = 19 from local newspapers, social media, and personal information; n = 8 from GeoCetus http://geocetus.spaziogis.it/index.php).

The distribution of all records from social media, research surveys, and stranding datasets is reported in Figure 2. Social media accounts have a more coastal diffusion than the research sources, with an overlap between research effort and social media data in three areas within the study region: (1) Albegna river estuary and Giglio island; (2) Mignone/Marta/Fiora river estuaries; and (3) Tiber river estuary and Tor Paterno MPA.

2.3 | Data analysis: Group size

'Best estimate' group sizes were used for social media and research datasets. ANOVA was applied to test differences in species group size between the two datasets. Linear models were used to examine changes each season for species for each dataset. When the assumptions of linear models were violated, generalized linear models (GLMs) were fitted using a Poisson distribution if there was evidence of over-dispersion (as in Dwyer, Clement, Pawley, & Stockin, 2016). Analyses were carried out in Past version 3.20 (https://folk.uio.no/ohammer/past/; Hammer, Harper, & Ryan, 2001).

TABLE 1 Descriptive features of the three datasets used in this study

Data	Method	Provider	Source/project	Period	No. of records
Social media	—	Citizens	YouTube (video)	2008–2017	220
Social media	—	Citizens	Facebook (video)	2008–2017	190
Social media	—	Citizens	Facebook (photograph)	2008–2017	147
Social media/Direct reporting	—	Citizens	Call/mail to researchers	2008–2017	40
Research surveys (visual)	Ferry-based fixed line transects	ISPRA	Fixed line transects Mediterranean monitoring network project (Arcangeli, Marini, & Crosti, 2012)	2008–2017	326
Research surveys (visual and acoustic)	Systematic boat-based line transects and non-systematic ('haphazard'; sensu Corkeron et al., 2011) boat based in closing mode (cetaceans were approached after detection)	Sapienza University of Rome	Costa Concordia project	2012–2016	55
Research surveys (visual)	Non-systematic boat-based surveys	Sapienza University of Rome	Fishery projects	2008–2012	35
Research surveys (visual and acoustic)	Systematic boat-based line transects and non-systematic ('haphazard'; sensu Corkeron et al., 2011) boat based in closing mode (cetaceans were approached after detection)	Sapienza University of Rome	CETYS/TASM projects	2017	5
Research surveys (visual)	Systematic aerial line transects	Tethys Research Institute; Observatoire PELAGIS University La Rochelle—CNRS	Lanfredi & Notarbartolo di Sciara, 2011; Van Canneyt, 2016	2009–2012	37
Research surveys (visual)	Systematic boat-based line transects	IFAW, Song of the Whale Team; Tethys Research Institute; Italo-Tunisian Cetacean Research Project	Boisseau, 2014; Lanfredi, & Notarbartolo di Sciara, 2014; Tringali, 2015	2012–2014	19
Research surveys (visual)	Boat-based line transects	Tuscia University	Seabirds/dolphins project	2013–2014	9
Strandings	—	Italian stranding network	Stranding National Data Bank (Pavan, Bernuzzi, Cozzi, & Podestà, 2013). Data downloaded from http://mammiferimarini.unipv.it [March 26, 2018].	2008–2017	164
Strandings	—	Various	YouTube, Facebook, local newspaper	2008–2017	19

(Continues)

TABLE 1 (Continued)

Data	Method	Provider	Source/project	Period	No. of records
			websites, call/mail to researchers		
Strandings	–	Centro Studi Cetacei	GeoCetus Data downloaded from http://geocetus.spaziogis.it/index.php on March 26, 2018	2008–2017	8
Total					**1,274**

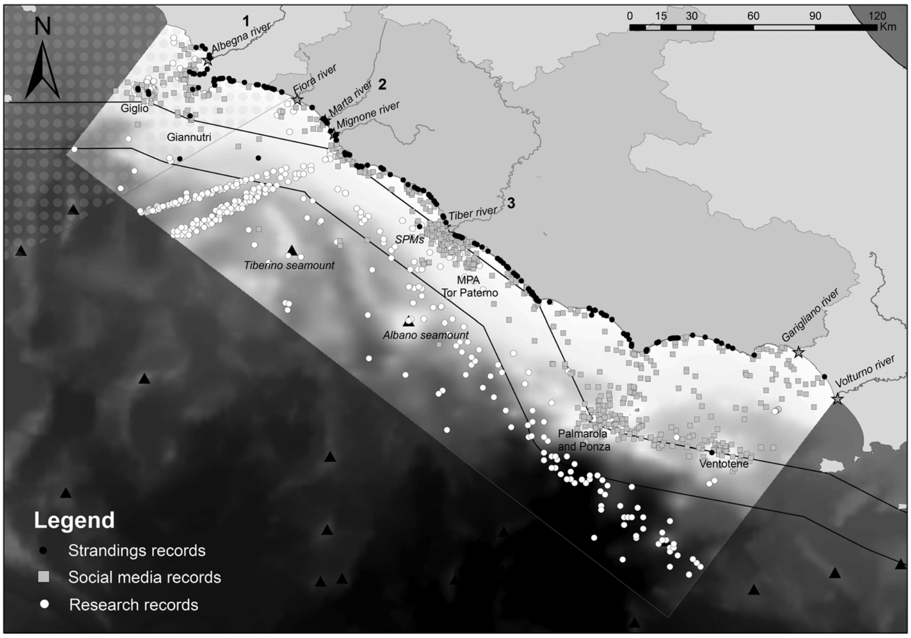

FIGURE 2 Distribution of all records analysed in this study (Tyrrhenian Sea, Italy) over the 2008–2017 period. Black lines mark the Italian coastal baseline and the limit of the Italian territorial seas. White dots indicate cetacean sightings collected by researchers, light grey squares represent social media accounts, and black dots are the locations of stranding events. An overlap between research efforts and social media records occurred in three areas: (1) Albegna river estuary and Giglio island, (2) Mignone/Marta/Fiora river estuaries, and (3) Tiber river estuary and Tor Paterno Marine Protected Area

2.4 | Data analysis: Spatial relative densities

Kernel density estimates (KDEs) were used as a measure of the species relative densities (i.e. 'use of space', as in Hauser, 2006). KDEs were calculated based on a combined dataset of research and social media records in ArcGIS version 10.2.2 (Environmental Systems Research Institute, Redlands, CA) setting cell size of 100 m square and a search radius of 20,000 m. The same scaling factor across the study area was used for each species, allowing comparison of relative density values. KDEs were calculated only for species with more than 15 records. The resulting KDE maps were visually assessed in order to recognize any high-density areas (i.e. hotspots; Clement, 2005).

2.5 | Data analysis: Habitat suitability modelling

A characteristic of social media data is the absence of a designed sampling scheme on a geographic scale, so the observation density over both space and time is more representative of observer concentration than of the targeted data points themselves (Garcia-Soto et al., 2017). The absence of correction for heterogeneous observations may lead to important and significant biases and hence to spurious conclusions on spatial patterns of species distribution (e.g. a seasonal increase in the number of observers may lead to the conclusion that there is a seasonality in the presence of species). It has been suggested that a convenient way to address such issues involves pooling opportunistic, heterogeneously distributed presence-only data with controlled data (e.g. scientific survey data based on a predefined protocol) when available (Pagel et al., 2014) and use the latter in a global model to calibrate the purely opportunistic data. The maximum entropy method (MaxEnt version 3.3.3, http://www.cs.princeton.edu/~schapire/maxent/), which generates pseudo-absences ('background points') to fill the gaps and hence enable the analysis of presence-only data (Kramer-Schadt et al., 2013; Renner et al., 2015), was applied to model the relationships between environmental predictors and the occurrence records of different cetacean species in the study area, using research and social media datasets separately and then as a combined dataset.

Six fixed variables were selected to generate models: depth, slope, Euclidean distance from the shoreline, Euclidean distance from estuaries, Euclidean distance from seamounts, and Euclidean distance from main harbours; these were then used as proxies of factors that could affect the cetacean presence and distribution (Bombosch et al., 2014; Breen, Brown, Reid, & Rogan, 2016; Correia, Tepsich, Rosso, Caldeira, & Sousa-Pinto, 2015; Gómez & Cassini, 2015; Pace et al., 2018). The environmental variables were obtained from geographic information system raster layers.

Since MaxEnt accounts for sampling biases via correction features that consider sampling effort (in which the spatial bias in the sightings data is transferred to the background data by approximating areas where the probability of detection is non-zero; Phillips et al., 2009), the bias file feature to input a layer representing the area of the sampling effort for each dataset (research and social media) was used, as recommended by Stolan and Nielsen (2015), validated by Syfert, Smith, and Coomes (2013), and applied by Pace et al. (2018). Sampling bias was regulated by adding a specific bias originated from occurrence data and applying it as a template for the extraction of background points in effort areas (Bombosch et al., 2014; Elith, Kearney, & Phillips, 2010; Fourcade, Engler, Rödder, & Secondi, 2014; Pace et al., 2018). The number of background points was set separately for each dataset to generate the same background density in all models (research, social media, and combined). Then, each background and presence location was associated with the series of environmental variables, and MaxEnt was set to eliminate duplicates at the same location to reduce pseudo-replications and spatial autocorrelation of samples (Hammond, O'Keefe, Aldrich, & Loeb, 2016). For each dataset (research, social media, and combined), distinct MaxEnt models were run by stratifying data per different species, using default regularization parameters with maximum iterations up to 500 to reach convergence. MaxEnt was controlled using only linear, quadratic, and product feature classes, restricting it to produce relatively simple models to minimize the likelihood of overfitting the data (Merckx, Steyaert, Vanreusel, Vincx, & Vanaverbeke, 2011; Merow, Smith, & Silander, 2013; Syfert et al., 2013). A minimum of 15 presence points per species was used (Pearson, Raxworthy, Nakamura, & Peterson, 2007), limiting MaxEnt predictions to four species for the social media dataset (fin and sperm whales, bottlenose and striped dolphins) and five species for the research dataset (the aforementioned four and Cuvier's beaked whale).

The descriptive power of each model was assessed by calculating the area under the receiver operating characteristic curve (AUC; Thorne et al., 2012). This metric determines model discriminatory power by comparing model sensitivity (i.e. true positives) against model specificity (i.e. false positives). The AUC values range from 0 to 1; when AUC value is 0.5, it means that model predictions are not better than random; values below 0.5 are worse than random, and higher values denote improving precision. The relative contribution of individual environmental variables to each of the resulting models was estimated by jackknife analysis, which then measured the percentage contribution and permutation importance for each variable (Baldwin, 2009). Jackknife analysis included creating a model excluding one variable, followed by the generation of a model using only the omitted individual variable (Moura, Sillero, & Rodrigues, 2012), thus providing an indication of how well the model performed when an environmental variable was omitted and additionally how each variable contributed to the model individually (Bombosh et al., 2014).

Using ArcGIS, spatial prediction maps of habitat suitability based on MaxEnt outputs were generated, which depict habitat suitability across the region investigated with values ranging from 0 (very unsuitable habitats) to 1 (very suitable habitats).

3 | RESULTS

3.1 | Social media (citizens') dataset

The searches in social media platforms produced 557 cetacean group encounters (annual mean = 60 ± *SD* 24 groups), with 410 unique videos (220 in YT and 190 in FB) and 147 records of FB photographs. The duration of all videos totalled 12 hr 38 min, with an average of 2 min 20 s and 1 min 16 s per footage in YT and FB respectively. The number of records found in YT and FB was different over the years, with a clear increasing trend in FB (see Supporting Information, Figure S1).

Species were positively identified in 92.5% (*n* = 515) of the social media records. It was not possible to recognize the species in 4% of YT videos (*n* = 9), in 8% of FB videos (*n* = 15), and in 12% of FB pictures (*n* = 18), but species were accurately determined in the 40 sightings collected by citizens and directly reported to researchers. The list of the observed species and their occurrence by year and season for all citizen accounts (*n* = 597; 557 found in social media platforms and 40 collected as direct reporting) is presented in Table 2.

Two species of dolphins were encountered regularly: 65.6% of sightings were of bottlenose dolphin *Tursiops truncatus* (*n* = 392) and 20.6% were striped dolphins *Stenella coeruleoalba* (*n* = 124). Other species encountered were 3% fin whales *Balaenoptera physalus* (*n* = 18), 2.3% sperm whales *Physeter macrocephalus* (*n* = 14), 0.8% common dolphins *Delphinus delphis* (*n* = 5), 0.5% Risso's dolphins *Grampus griseus* (*n* = 3), and 0.2% killer whale *Orcinus orca* (*n* = 1). In 7% of the observations (*n* = 42) it was not possible to determine the species. Citizens' sightings have been principally reported in the summer (47.7%), followed by spring (25.9%), autumn (16.4%), and winter (10.2%). Three species were sighted in all seasons (bottlenose dolphin, striped dolphin, and fin whale).

Bottlenose and striped dolphin groups were frequently observed with immature individuals (37.5% of sightings, *n* = 147 and 27% of sightings, *n* = 34, respectively) all year round (at least one group with immatures per month). Immatures were principally observed in groups of bottlenose dolphin in summer (40% of the seasonal sightings; *n* = 71) and autumn (50% of the seasonal sightings; *n* = 35), from August to October; however, they were also spotted in 33% of the winter encounters (*n* = 16) and 26% of the spring ones (*n* = 25).

TABLE 2 Cetacean species occurrence in the area investigated (Tyrrhenian Sea, Italy) over the study period (2008–2017) by year and season. Data are shown for each dataset (social media, research, and strandings)

		Social media/citizens' dataset											Research dataset											Strandings dataset										
		'08	'09	'10	'11	'12	'13	'14	'15	'16	'17	O	'08	'09	'10	'11	'12	'13	'14	'15	'16	'17	O	'08	'09	'10	'11	'12	'13	'14	'15	'16	'17	O
Fin whale	Sp	–	–	1	–	–	–	2	–	1	–	**4**	–	4	12	–	–	–	5	1	–	–	**22**	–	–	–	–	–	–	–	–	–	–	–
	Su	1	2	–	3	1	–	1	–	1	–	**9**	5	2	13	5	–	3	2	–	–	1	**31**	–	–	–	–	–	1	–	–	–	–	**1**
	A	–	–	–	–	–	1	–	–	–	3	**4**	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	W	–	–	–	1	–	–	–	–	–	–	**1**	–	–	–	–	–	–	2	–	–	–	**2**	1	–	–	–	–	–	–	–	–	–	**1**
	O	**1**	**2**	**1**	**4**	**1**	**1**	**3**	–	**2**	**3**	**18**	**5**	**6**	**25**	**5**	–	**3**	**9**	**1**	–	**1**	**55**	**1**	–	–	–	–	**1**	–	–	–	–	**2**
Sperm whale	Sp	–	–	1	–	–	–	–	2	–	–	**3**	–	1	–	–	–	1	–	–	–	–	**2**	1	–	–	–	–	–	–	–	–	–	**1**
	Su	–	–	–	–	1	1	–	1	1	6	**10**	–	–	4	1	–	–	1	–	–	–	**6**	–	–	–	–	–	–	–	–	2	–	**2**
	A	–	1	–	–	–	–	–	–	–	–	**1**	–	–	2	–	–	–	–	–	1	–	**3**	–	–	–	–	–	–	–	–	–	–	–
	W	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	–	–	**1**
	O	–	**1**	**1**	–	**1**	**1**	–	**3**	**1**	**6**	**14**	–	**1**	**6**	**1**	–	**1**	**1**	–	**1**	–	**11**	**1**	–	**1**	–	–	–	–	–	**2**	–	**4**
Cuvier's beaked whale	Sp	–	–	–	–	–	–	–	–	–	–	–	–	2	–	3	–	–	–	–	–	–	**5**	–	–	–	–	–	–	–	–	–	–	–
	Su	–	–	–	–	–	–	–	–	–	–	–	2	–	3	3	–	–	1	–	1	–	**10**	–	–	–	–	–	–	–	–	–	–	–
	A	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	W	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	O	–	–	–	–	–	–	–	–	–	–	–	**2**	**2**	**3**	**6**	–	–	**1**	–	**1**	–	**15**	–	–	–	–	–	–	–	–	–	–	–
Killer whale	Sp	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	Su	–	–	–	–	–	–	–	–	1	–	**1**	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	A	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	W	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	O	–	–	–	–	–	–	–	–	**1**	–	**1**	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Risso's dolphin	Sp	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	Su	1	–	1	–	–	1	–	–	–	–	**3**	–	–	1	–	–	–	–	–	–	–	**1**	–	–	–	–	–	–	–	–	–	–	–
	A	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	W	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	**1**
	O	**1**	–	**1**	–	–	**1**	–	–	–	–	**3**	–	–	**1**	–	–	–	–	–	–	–	**1**	–	–	–	–	–	**1**	–	–	–	–	**1**
Rough–toothed dolphin	Sp	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	Su	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	–	**1**	–	–	–	–	–	–	–	–	–	–	–
	A	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	W	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	O	–	–	–	–	–	–	–	–	–	–	–	–	–	–	**1**	–	–	–	–	–	–	**1**	–	–	–	–	–	–	–	–	–	–	–
Bottlenose dolphin	Sp	2	4	9	5	9	10	11	17	16	13	**96**	–	1	–	3	4	1	3	2	1	–	**15**	–	–	1	1	–	2	–	–	3	3	**10**
	Su	3	8	9	14	7	23	15	27	42	29	**177**	2	5	6	13	9	14	2	5	–	1	**57**	1	–	–	3	–	–	1	3	1	2	**11**
	A	3	2	8	7	3	11	6	10	4	16	**70**	–	–	–	1	6	4	3	3	–	2	**19**	–	–	–	–	1	–	–	–	3	–	**4**
	W	2	2	7	6	3	4	7	4	9	5	**49**	–	–	–	–	3	3	–	–	–	–	**6**	–	–	1	–	–	2	1	1	2	–	**7**
	O	**10**	**16**	**33**	**32**	**22**	**48**	**39**	**58**	**71**	**63**	**392**	**2**	**6**	**6**	**17**	**22**	**22**	**8**	**10**	**1**	**3**	**97**	**1**	–	**2**	**4**	**1**	**4**	**2**	**4**	**9**	**5**	**32**
Common dolphin	Sp	–	–	1	–	–	1	–	–	–	–	**2**	–	–	–	–	–	–	2	–	–	–	**2**	–	–	–	–	–	–	–	–	–	–	–
	Su	–	1	–	–	–	1	1	–	–	–	**3**	2	–	–	4	–	–	–	–	–	–	**6**	–	–	–	–	–	–	–	–	–	–	–
	A	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–

(Continues)

TABLE 2 (Continued)

		Social media/citizens' dataset											Research dataset											Strandings dataset										
		'08	'09	'10	'11	'12	'13	'14	'15	'16	'17	O	'08	'09	'10	'11	'12	'13	'14	'15	'16	'17	O	'08	'09	'10	'11	'12	'13	'14	'15	'16	'17	O
	W	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	O	–	**1**	**1**	–	–	**2**	**1**	–	–	–	**5**	**2**	–	–	**4**	–	–	**2**	–	–	–	**8**	–	–	–	–	–	–	–	–	–	–	–
Striped dolphin	Sp	2	3	4	1	11	4	–	2	5	5	**37**	–	–	10	2	1	5	5	3	–	–	**26**	2	1	2	1	3	1	3	1	–	1	**15**
	Su	2	5	9	9	7	8	3	4	6	10	**63**	7	5	38	25	–	10	6	7	–	–	**98**	–	1	2	1	–	1	–	–	1	4	**10**
	A	3	4	–	3	1	2	1	1	1	2	**18**	–	–	2	–	–	4	4	–	–	1	**11**	1	3	2	–	3	1	1	1	2	5	**19**
	W	–	1	–	1	2	–	–	1	–	1	**6**	–	–	–	–	–	1	7	2	–	–	**10**	5	3	4	1	6	36	1	1	9	8	**74**
	O	**7**	**13**	**13**	**14**	**21**	**14**	**4**	**8**	**12**	**18**	**124**	**7**	**5**	**50**	**27**	**1**	**20**	**22**	**12**	–	**1**	**145**	**8**	**8**	**10**	**3**	**12**	**39**	**5**	**3**	**12**	**18**	**118**
Undetermined dolphins	Sp	–	–	2	–	1	2	1	1	2	3	**12**	–	4	4	4	1	1	2	1	5	–	**22**	–	–	1	–	2	–	–	–	3	3	**9**
	Su	1	1	–	2	3	2	–	3	5	3	**20**	2	8	38	20	–	5	1	1	–	–	**75**	–	–	–	1	–	1	–	–	1	1	**4**
	A	–	1	–	2	–	2	–	–	–	–	**5**	–	–	4	–	21	1	–	1	–	1	**28**	1	1	1	–	2	1	–	1	–	3	**9**
	W	–	–	–	–	–	1	1	2	–	1	**5**	–	–	–	–	–	3	4	1	1	–	**9**	–	–	–	–	–	6	4	–	1	1	**12**
	O	**1**	**2**	**2**	**4**	**4**	**7**	**2**	**6**	**7**	**7**	**42**	**2**	**12**	**46**	**24**	**22**	**10**	**7**	**4**	**6**	**1**	**134**	**1**	**1**	**2**	**1**	**4**	**8**	**4**	–	**5**	**8**	**34**

A: autumn; O: overall; Sp: spring; Su: summer; W: winter.

Likewise, immature striped dolphins were spotted in groups during the summer (30% of the seasonal sightings; n = 19), but they were also observed in 33% (n = 2), 24% (n = 9), and 22% (n = 4) of the winter, spring, and autumn encounters respectively.

Records derived from social media also provided information on spinal deformities, mutilations, and nurturant behaviour in the bottlenose dolphin (see Supporting Information, Figure S2).

3.2 | Research dataset

Research surveys (vessel n = 122, ferry n = 326, and aerial n = 37 based) accounted for a mean of 49 ± *SD* 38 annual cetacean group encounters. Differences in the mean annual number of sightings between social media and research datasets were not statistically significant (two-way ANOVA: F = 0.48, df = 19, P > 0.5).

Nine species (Table 2) were recorded, of which 33.4% of sightings were of striped dolphins (n = 162), 20% bottlenose dolphin (n = 97), 11.3% fin whales (n = 55), 3.1% Cuvier's beaked whales *Ziphius cavirostris* (n = 15), 2.5% sperm whales (n = 12), 1.6% common dolphins (n = 8), 0.2% Risso's dolphins (n = 1), and 0.2% rough-toothed dolphin *Steno bredanensis* (n = 1); 27.6% records were undetermined species (n = 134). Four sightings were mixed groups of striped and common dolphins, one of bottlenose and striped dolphins, and one of striped dolphin and sperm whale.

Encounters were principally collected in the summer (59.4%), followed by spring (20.8%), autumn (14%), and winter (5.8%). Two species were observed in all seasons (bottlenose and striped dolphin).

Very little and scattered evidence of the presence of immature individuals was available in this dataset, so this information is not presented here.

3.3 | Stranding dataset

One hundred and ninety-one stranded animals in the study area were recorded between 2008 and 2017 (Table 2). The species was determined in 156 cases (82%). Five different species were represented in the dataset, of which ~78.5% comprised two species (striped and bottlenose dolphins). The stranding events mainly occurred in winter (45%) and were located all along the ~500 km long coast of the study area (Figure 2).

3.4 | Group size

Only species with more than one sighting were included. Differences in overall mean group size between social media and research datasets were statistically significant for fin whale and striped and common dolphins (Table 3).

In the social media dataset, there was evidence of a seasonal difference in group size for fin whale (Poisson GLM; P < 0.05), with significantly higher group sizes in autumn than in spring, and for sperm whale (Poisson GLM; P < 0.05), with significantly higher values in summer than in spring (Table 3).

TABLE 3 Overall and seasonal group sizes of all species recorded within the social and research datasets in the study region (Tyrrhenian Sea, Italy) in 2008–2017

		Social media dataset					Research dataset					*P* (between datasets)
		Mean	Median	*SD*	Range	*n*	Mean	Median	*SD*	Range	*n*	
Fin whale	Spring	1.50	1	1	1–3	4	1.22	1	0.53	1–3	22	
	Summer	1.78	1	1.20	1–4	9	1.22	1	0.49	1–3	31	
	Autumn	2.25	2	1.50	1–4	4	1	1	0	1	2	
	Winter	1	1	0	1	1	0	0	0	0	0	
	Overall	**1.77**	**1**	**1.16**	**1–3**	**18**	**1.21**	**1**	**0.50**	**1–3**	**55**	**<0.05**
Sperm whale	Spring	1	1	0	1	3	1	1	0	1	2	
	Summer	1.90	1	2.51	1–9	10	2.43	1	2.57	1–8	7	
	Autumn	1	1	0	1	1	2.00	1	1.41	1–3	2	
	Winter	0	0	0	0	0	0	0	0	0	0	
	Overall	**1.65**	**1**	**2.13**	**1–9**	**14**	**2.20**	**1**	**4.04**	**1–8**	**11**	**0.774**
Cuvier's beaked whale	Spring	0	0	0	0	0	1.80	0.83	1.40	1–3	5	
	Summer	0	0	0	0	0	1.40	2	1	1–3	10	
	Autumn	0	0	0	0	0	0	0	0	0	0	
	Winter	0	0	0	0	0	0	0	0	0	0	
	Overall	**0**	**0**	**0**	**0**	**0**	**1.53**	**1**	**0.74**	**1–3**	**15**	**n.a.**
Risso's dolphin	Spring	0	0	0	0	0	0	0	0	0	0	
	Summer	4.33	5	3.05	1–7	3	0	0	0	0	0	
	Autumn	0	0	0	0	0	0	0	0	0	0	
	Winter	0	0	0	0	0	0	0	0	0	0	
	Overall	**4.33**	**5**	**3.05**	**1–7**	**3**	**0**	**0**	**0**	**0**	**0**	**n.a.**
Bottlenose dolphin	Spring	5.61	5	4.69	1–28	96	7.30	7	6.13	1–20	13	
	Summer	7.05	6	5.25	1–30	177	6.85	5	5.18	1–20	54	
	Autumn	8.32	7	5.58	1–25	70	9.00	7	7.06	1–20	18	
	Winter	6.27	6	4.48	1–17	49	2.20	1	2.17	1–6	6	
	Overall	**6.83**	**6**	**5.14**	**1–30**	**392**	**7.06**	**6**	**5.71**	**1–20**	**91**	**0.1532**
Common dolphin	Spring	3.50	3.5	0.70	3–4	2	12.00	12	5.65	8–16	2	
	Summer	5.33	5	2.51	3–8	3	9.66	10.5	5.35	1–17	6	
	Autumn	0	0	0	0	0	0	0	0	0	0	
	Winter	0	0	0	0	0	0	0	0	0	0	
	Overall	**4.60**	**4**	**2.07**	**3–8**	**5**	**10.00**	**10**	**5.07**	**1–17**	**8**	**<0.05**
Striped dolphin	Spring	5.65	4	4.55	2–21	37	18.62	4.5	6.62	1–120	26	
	Summer	7.61	6	4.94	1–22	63	10.74	6	1.67	1–100	98	
	Autumn	5.89	4.5	4.30	2–20	18	5.91	4	1.60	1–20	11	
	Winter	4.83	5.5	2.40	1–7	6	3.30	2	0.86	1–10	10	
	Overall	**6.62**	**5**	**4.70**	**1–22**	**124**	**11.27**	**5**	**20**	**1–120**	**145**	**<0.001**

As for seasonal differences in the research dataset, significantly lower average group sizes were recorded in bottlenose dolphin (Poisson GLM; $P < 0.05$) during winter than in other seasons, and in striped dolphin (negative binomial GLM; $P < 0.001$) during winter compared with spring and summer (Table 3).

3.5 | Spatial relative density

KDEs were calculated for the species with at least 15 records using the combined dataset (research and social media; Figure 3), whereas records of the rarer species were just plotted on a map (Figure 4). Overall, fin whales were more spatially concentrated in a wide northern area, with the highest relative density in the pelagic waters just outside of the Italian territorial seas, and a medium relative density zone was located in the south, around Ponza island (Figure 3). Sperm whale records were mainly grouped off the coasts of the island of Ventotene, and a medium relative density zone was recorded in the north in an area partially overlapping fin whale distribution (Figure 3). Cuvier's beaked whales were sighted infrequently in the study area but were concentrated principally in the north in the same region as fin whales and, to a certain extent, sperm whales (Figure 3). Bottlenose dolphin sightings were clustered more evenly in coastal areas, including Giglio and Ponza islands. Striped dolphin records were unevenly distributed across the study area and did not show any patterns, and the 'hotspots' were most likely driven by the methodology, as a higher number of coastal records were likely related to the number of sightings from citizens rather than reliable higher relative densities. This fact has motivated and supported the choice of not modelling habitat suitability for this species.

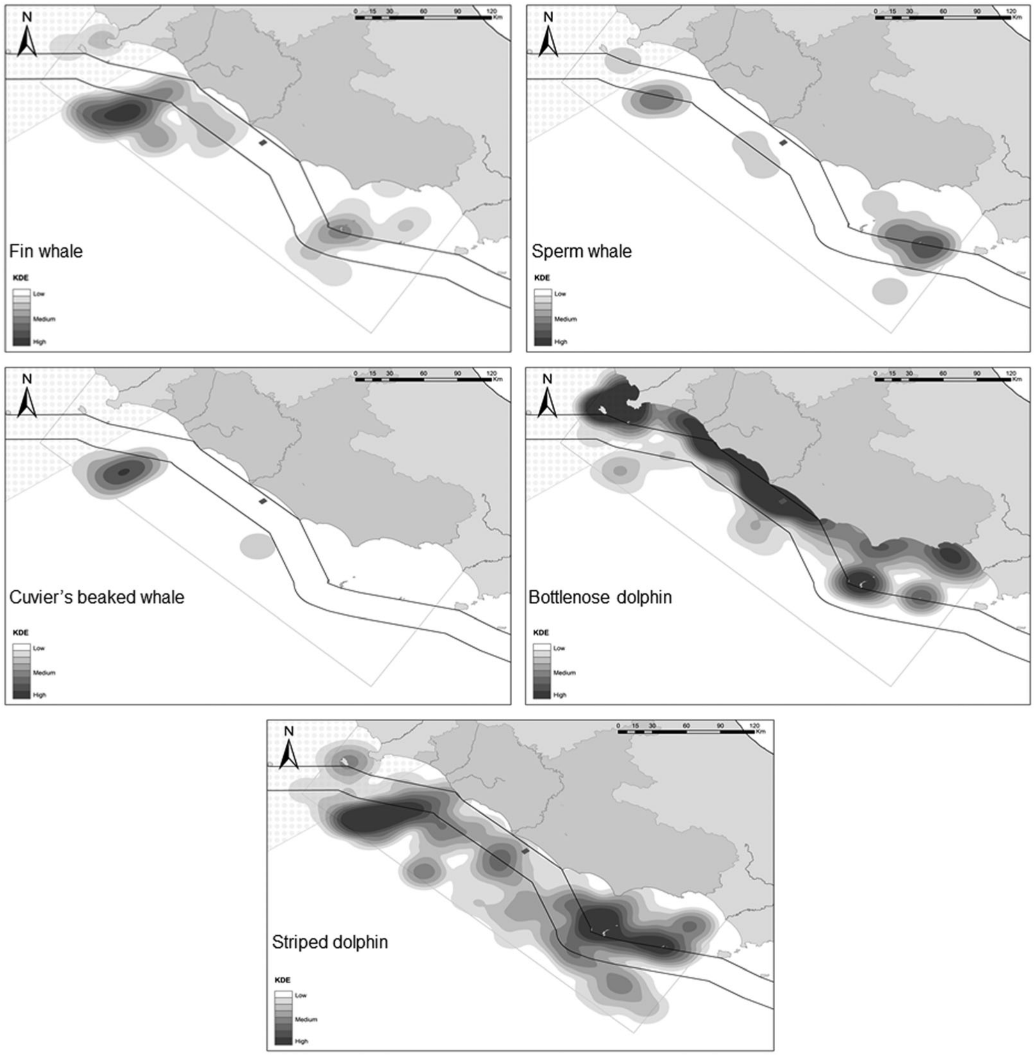

FIGURE 3 Relative densities (kernel density estimate) of different cetacean species (darker shading represents higher density areas) off the Lazio region (central Mediterranean Sea, Italy) in 2008–2017

3.6 | Habitat suitability

To determine the MaxEnt distributions, after the removal of duplicates, a total of 163 presence records out of 486 for the research dataset, 383 presence records out of 597 for the social media dataset, and 533 points out of 1,086 for the combined dataset were used.

All MaxEnt models obtained AUC $\geq$ 0.77 (Table 4), which suggests very good predictive power of the fitted model compared with the value (0.5) expected from a random prediction (Lobo, Jiménez-Valverde, & Real, 2008).

The relative contribution of environmental variables based on both the numerical measures of variable importance (percentage contribution and permutation importance) and the graphical results of jackknife tests is shown in Table 5. Overall, ecological relationships between species presence and environmental conditions showed different predictions across the datasets. In fin whale, the distance from the nearest coast had the greatest explanatory power for both research and combined datasets, and slope (permutation importance) and distance from the river estuaries (percentage of contribution) was greatest for the social media dataset. Models built on the combined dataset demonstrated that fin whales preferred relatively greater distances from the coast in the north and shorter distances in the south (Pontine Archipelago) of the study area with a depth range of 200–1,000 m and some topographic complexity (top of seamounts). Distance from the nearest coast emerged as a relevant variable for sperm whales as well. It had the greatest explanatory power for both social media and combined datasets and for distance from the seamounts for the research dataset. Models developed using the combined dataset revealed that the highest logistic probability for finding sperm whales was at relatively greater distances from the coast in the north and shorter distances in the south (Pontine Archipelago) of the study area at around 800 m in depth. Depth best predicted encounters with Cuvier's beaked whales (research dataset only), with a high probability of prediction at 800–1,000 m depth far from the coast. For bottlenose dolphin, depth emerged as the variable with the greatest explanatory power for both social media and

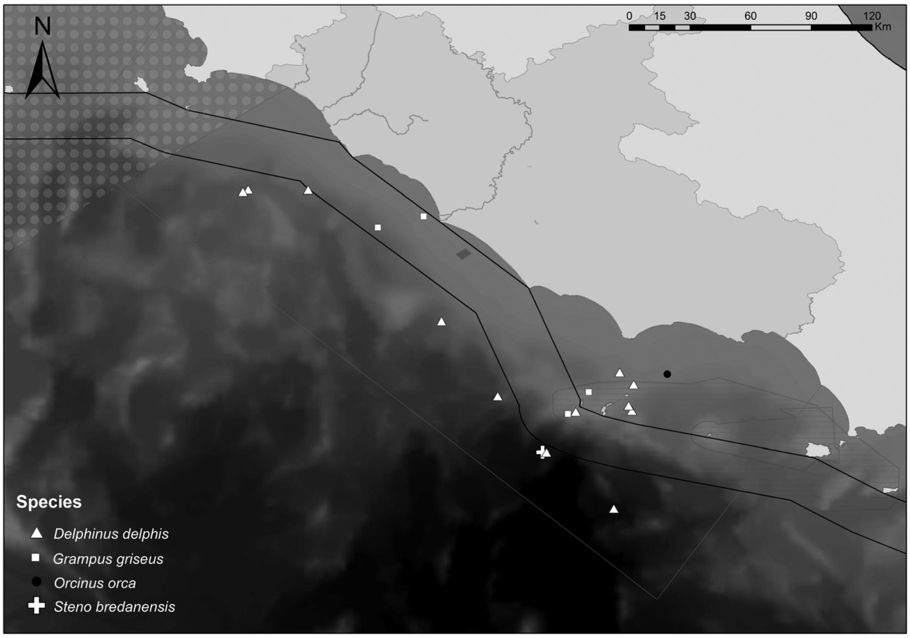

FIGURE 4 Distribution of cetacean rare species (common dolphin, Risso's dolphin, rough-toothed dolphin, and killer whale) recorded off the Lazio region (central Mediterranean Sea, Italy) in 2008–2017

TABLE 4 Area under the curve (AUC) of the receiver operating characteristic plots for MaxEnt models applied to research, social media, and combined datasets of cetacean presence records collected in the central Tyrrhenian Sea between 2008 and 2017

	Research dataset	Social media dataset	Combined dataset
Balaenoptera physalus	0.936	0.770	0.830
Physeter macrocephalus	0.858	0.892	0.923
Ziphius cavirostris	0.975	—	—
Tursiops truncatus	0.943	0.905	0.922

combined datasets combined with distance from estuaries for the research dataset. Overall (combined dataset), bottlenose dolphins favoured shallow waters less than 100 m deep near to the coast.

An extended area in the northern part of the study region outside Italian territorial waters was highly suitable for fin whale, as well as a smaller area between Ponza and Ventotene islands and south-east Ventotene with a predicted wide-ranging pattern of suitability in the area between these two areas (Figure 5). A less evident but similar pattern emerged for sperm whale. The best predicted conditions were in the waters surrounding Ventotene island and between Ponza and Ventotene and in a northern zone bordering the highly suitable area for fin whales. This area in the northern part of the study area emerged as extremely suitable for Cuvier's beaked whales as well, showing a major overlap with fin whales and, to a lesser extent, with sperm whales (Figure 5). Another area for beaked whales emerged in the south far from Ventotene island. The areas of highest suitability for bottlenose dolphin were principally located near estuaries (Tiber in particular) and in the waters surrounding all the islands (Figure 5).

4 | DISCUSSION

Social media is becoming a rich source of data on species occurrence and, therefore, is a new and promising way to assess cetacean distribution, seasonality, or habitat use, particularly in unknown or scarcely investigated areas. The collection of sightings from citizens is a useful first activity for researchers entering a new study region to identify where to focus future intensive research (Alessi, Bruccoleri, & Cafaro, 2019), but the combination of research data and social media information to study cetaceans in a pilot area, like the one presented here, may add value to this new approach.

The proposed scheme uses two global data sources, namely FB and YT, which are freely accessible and available online, making the method attractive for large-scale assessments. With increasing requests to study whale and dolphin population ecology coming from national and international directives, support from citizens to aid research may act as a practical, inexpensive solution to gathering extensive spatial and temporal data for regional-scale population monitoring and for the development of management priorities (Braulik et al., 2018; Hann et al., 2018). The information obtained from social media includes potential limitations, biases, and errors that can reduce its scientific benefits, and these need to be accounted for when utilizing such a methodology. For example, a lack of spatial and temporal effort data complicated the analysis of space use patterns to some extent, and encounters are likely to be skewed in favour of more widely distributed species.

The area in the Tyrrhenian Sea investigated here was used as a case study to test an integrated approach using research and social media data to provide the first assessment on cetacean species in a sea sector where survey effort and scientific information are mostly scattered. Nevertheless, few or no data are available from both

TABLE 5 Estimates of relative contribution (%) and permutation importance of the environmental variables to the MaxEnt models applied to research, social media, and combined datasets of cetacean presence records collected in the central Tyrrhenian Sea between 2008 and 2017. Results related to striped dolphin are not shown (see text)

		Research dataset		Social media dataset		Combined datasets	
Species	Variable	Contribution (%)	Permutation importance	Contribution (%)	Permutation importance	Contribution (%)	Permutation importance
Balaenoptera physalus	Distance from coast	47.11	52.77	12.57	28.81	41.03	44.08
	Distance from rivers	1.08	6.49	45.23	5.27	0.70	4.59
	Distance from harbours	8.30	23.49	31.70	19.02	10.26	12.32
	Distance from seamounts	21.41	13.92	0.00	0.00	18.05	16.47
	Depth	16.05	1.94	0.00	0.00	20.20	22.33
	Slope	6.06	1.39	10.51	46.90	9.76	0.22
Physeter macrocephalus	Distance from coast	17.63	19.11	39.35	41.29	32.13	51.54
	Distance from rivers	0.00	0.00	14.48	15.31	11.17	16.05
	Distance from harbours	0.00	0.00	11.88	0.00	16.93	15.24
	Distance from seamounts	52.05	55.98	4.59	5.77	11.26	9.04
	Depth	23.80	13.58	29.17	37.51	28.26	7.74
	Slope	6.53	11.33	0.52	0.12	0.25	0.39
Ziphius cavirostris	Distance from coast	27.02	23.50				
	Distance from rivers	0.00	0.00				
	Distance from harbours	4.08	14.60				
	Distance from seamounts	6.90	2.56				
	Depth	55.97	45.79				
	Slope	6.03	13.55				
Tursiops truncatus	Distance from coast	8.26	3.11	8.64	5.06	9.86	10.50
	Distance from rivers	48.38	42.74	6.29	4.55	4.99	1.95
	Distance from harbours	14.61	18.74	2.79	4.35	2.60	1.42
	Distance from seamounts	8.67	16.50	6.67	3.70	6.48	5.49
	Depth	18.12	17.26	71.65	78.93	74.75	79.31
	Slope	1.95	1.66	3.96	3.41	1.32	1.34

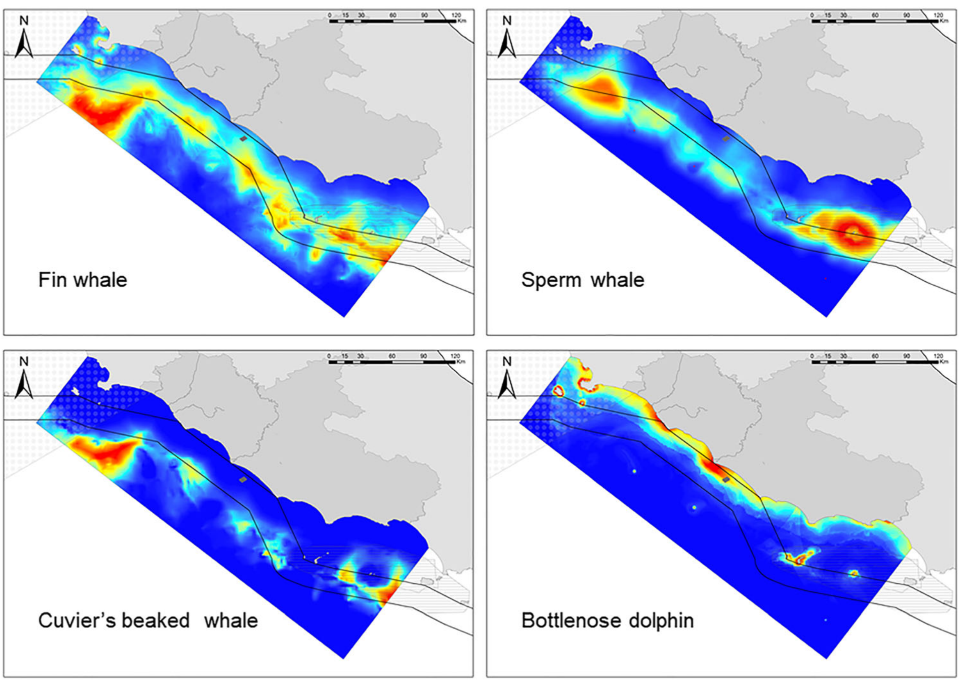

FIGURE 5 Suitability habitat maps for fin whale, sperm whale, Cuvier's beaked whale, and bottlenose dolphin in the central Tyrrhenian Sea (2008–2017). Warmer colours show areas with better predicted conditions

sources for some portions of the region yet, further highlighting the need to investigate gap areas to confirm the inferences of this study.

As far as the accuracy and quality of the reported observations by citizens is concerned, the analysis showed that most cetacean records are reliable, with particular regard to more coastal presences, distribution, and species identification. Experts recorded similar observations, which complemented information in more pelagic waters. A further data check can be made by comparing records with independent observations of stranding events, despite the possibility of carcasses ending up stranding at a point distant from where the animal died, making inferences problematic. Regardless, stranding records often compare well with sightings records (Maldini, Mazzuca, & Atkinson, 2005; Peltier et al., 2012), and results obtained here in terms of species diversity and relative abundance seem to confirm this model.

Possible biases in the social media dataset have to be considered, too. For example, most of the citizens' accounts found in social media platforms were principally associated with human-populated coastal areas, travel routes, and common holiday destinations (e.g. Ponza, Ventotene, and Giglio islands), with a degree of variability connected to the presence of harbours and to seasons (summer is more suitable for recreational activities at sea, thus increasing the potential for cetacean encounters). Moreover, the difference found between research and social media datasets for fin whales and striped and common dolphins mean group size may be due to the fact that videos and pictures could represent only a small part of the observed groups, resulting in an incorrect group size estimation. Additionally, the more coastal nature of the citizens' observations may reflect the habit of striped and common dolphins to form smaller groups than in pelagic areas, and the opportunistic aggregation of fin whales to exploit more coastal, ephemeral food patches (Notarbartolo di Sciara, Castellote, Druon, & Panigada, 2016). The proximity to the coast of the citizens' data and the ecological characteristics of the striped dolphin, such as the plastic and opportunistic behaviour and the absence of a clear preference for any specific physiographic features with an almost homogeneous distribution over different habitats (Arcangeli et al., 2012; Arcangeli, Campana, & Bologna, 2017; Azzellino, Gaspari, Airoldi, & Nani, 2008), could have also affected the relative density estimates, leading to unreliable preconditions for the MaxEnt modelling exercise.

In any case, choosing a consistent analysis method with the appropriate adjustments, such as MaxEnt in this case, could be helpful to minimize the differences in the combination of research, social media, and stranding records, allowing sound information on cetacean relative abundance, distribution, and habitat selection to be analysed.

In the following subsections, the results for each single species are discussed in light of evaluating the reliability of the approach applied here.

4.1 | Fin whales

Several studies have demonstrated the association of fin whales with a range of physical aspects of the marine environment, including depth, bottom slope, and distance from shore, and using them as presence indicators or predictors (Arcangeli et al., 2012; Azzellino et al., 2008; Cañadas, Sagarminaga, & Garcia-Tiscar, 2002; Hastie, Swift, Slesser, Thompson, & Turrell, 2005; Ingram, Walshe, Johnston, & Rogan, 2007; MacLeod, Weir, Pierpoint, & Harland, 2007; Ready et al., 2010; Redfern et al., 2006; Yen, Sydeman, & Hyrenbach, 2004). In this study, distance from the nearest coast was the environmental variable with the greatest explanatory power for fin whales, with a preference for relatively greater distances from the coast in the north and shorter in the south (Pontine Archipelago) with a depth range of 200–1,000 m and some topographic complexity (top of seamounts). Fin whales in the Mediterranean are most common in deep waters (400–2,500 m), but they can occur in slope and shelf waters as well, depending on the distribution of their prey (Gannier, Drouot, & Goold, 2002; Laran & Gannier, 2008; Notarbartolo di Sciara, Zanardelli, Jahoda, Panigada, & Airoldi, 2003; Pace, Miragliuolo, & Mussi, 2012; Panigada, Notarbartolo di Sciara, & Zanardelli, 2006; Panigada et al., 2017). Fin whale summer distribution and its interannual variability are closely linked to spatial and temporal interactions with zooplankton concentrations, demonstrating large-scale fidelity corresponding to the prey spatial and temporal predictable distribution, and mesoscale fidelity with higher density in the areas where northern krill (*Meganyctiphanes norvegica*) tend to concentrate (Cotté, Guinet, Taupier-Letage, Mate, & Petiau, 2009; Littaye, Gannier, Laran, & Wilson, 2004). Previous studies reported the presence of the species in the study area since the beginning of the 1990s, with an increased occurrence after 20 years (Arcangeli et al., 2012; Arcangeli et al., 2014). It was recently suggested that the central Tyrrhenian Sea is an opportunistic feeding ground while transiting to the Ligurian Sea in summer (Arcangeli et al., 2014; Panigada et al., 2017; Santoro et al., 2015), with a marked change in the use of the area from the early 1990s, when it was primarily considered as a transit zone (Marini et al., 1996; Nascetti & Notarbartolo di Sciara, 1996). More fin whales have been observed spending longer periods in the central Tyrrhenian Sea for feeding purposes instead of just moving through this area, possibly due to the presence of two gyres (Barale, Jaquet, & Ndiaye, 2008; Vetrano, Napolitano, Iacono, Schroeder, & Gasparini, 2010), which enhance the productivity in the region. One of these two cyclonic eddies is located close to the most suitable area for fin whales recognized in this study, reinforcing the prediction of local favourable conditions for the species' occurrence. A second hotspot of fin whale relative abundance was highlighted here in the waters surrounding the Pontine Archipelago, and some other zones between the two were also suitable, defining other potential regions of importance for the species and depicting a sort of 'corridor' between the most suitable areas in the north and in the south. Fin whales are considered to be nomadic opportunists that travel between highly concentrated feeding areas (Notarbartolo di Sciara et al., 2016), and these results seem to be in accordance with this model. It is worth noting that even the two stranding records of fin whales correspond with the area where higher densities were found.

4.2 | Sperm whales

Distance from the nearest coast emerged as a relevant environmental variable for sperm whales, as well as the distance from the seamounts. Higher relative abundance was found at 800 m depth in the waters surrounding Ventotene island, between Ponza and Ventotene, and in a northern zone bordering the highly suitable area for fin whales. This seems consistent with previous studies reporting that sperm whales select for two main habitats: one where complex topographical features (escarpments, canyons, seamounts) characterize the sea floor and one where downwelling or upwelling water movements are associated with frontal zones in higher seas (Aïssi, Fiori, & Alessi, 2012; Arcangeli et al., 2017; Carpinelli et al., 2014; Fiori, Giancardo, Aïssi, Alessi, & Vassallo, 2014; Gannier & Praca, 2007; Mussi, Miragliuolo, Zucchini, & Pace, 2014; Pace, 2016; Pace et al., 2018). In this way, sperm whales would be able to capitalize on the food resources (cephalopods) available in both areas by shifting between different prey targets (Gannier et al., 2002). The lower sperm whale relative abundance in the northern part of the study area than in the southern part (Pontine Archipelago) also seems to be consistent with recent published results showing variable encounters in the north (Arcangeli et al., 2017) and regular occurrences near Ponza and Ventotene islands (Mussi et al., 2014; Pace, 2016; Pace et al., 2018; Pace, Miragliuolo, Mariani, Vivaldi, & Mussi, 2014). This is also in line with sperm whale strandings, all located in the southern part of the area where deep waters are closer to the coast.

4.3 | Bottlenose dolphins

Bottlenose dolphin was the most frequently recorded species in the study area. Depth and distance from the river estuaries emerged as the variables with the greatest explanatory power. Overall, bottlenose dolphins favoured shallow waters about 0–100 m deep, near the shore and close to river mouths with sandy–muddy substrates and gentle slopes. This is consistent with the species' general distribution in the Mediterranean Sea (e.g. Gnone et al., 2011; Marini et al., 2015), where shallow water preference of the bottlenose dolphin could be related to its feeding habits of preying mostly on benthic and demersal fish (e.g. Blanco, Salomon, & Raga, 2001; Orsi Relini, Cappello, & Poggi, 1994). In addition, the results of this study are in accordance with the only published account on bottlenose dolphins in the study area (Cafaro et al., 2016), which reported near-shore foraging at the mouths of two river basins (Fiora and Mignone), and with preliminary, unpublished data collected in 2017–2018 by the University of Rome near the Tiber river mouths and Tor Paterno MPA. These findings are also in agreement with other studies that documented high numbers of bottlenose dolphins at the mouths of rivers in the Mediterranean, such as the Magra river in the Ligurian Sea (Alessi & Fiori, 2014), and outside the basin (e.g. the Shannon estuary on the Ireland's west coast; Ingram & Rogan, 2002). Rivers may play an important role, at a local scale, in affecting water temperature, salinity, sediment distribution, and nutrient loads, producing algal blooms and subsequent secondary production processes, which can in turn sustain species at higher trophic levels (Cafaro et al., 2016). According to the peak in immature/calf presence in August–October and to stranding data (mostly spring/summer), a significant fraction of bottlenose dolphins in the study area would dwell in coastal waters for calving and thus would mainly forage on coastal demersal and benthic prey (mainly hake) during the warmer season, whereas in the coldest season, when calves are older, they would switch towards deeper waters where they would have to forage on different species. This hypothesis seems to be supported by pilot observations of feeding bottlenose dolphin groups at the Albano seamount in autumn/winter (University of Rome and Pavia 2017–2018 visual and acoustic surveys; unpublished data), a pelagic area proposed as an attractive site for cetacean species in the Tyrrhenian Sea (Fiori et al., 2016).

The regular coastal presence of bottlenose dolphins at the Tiber river mouths poses relevant conservation issues, since the River Tiber is the most polluted river among the 20 longest rivers in Italy (Crosti, Arcangeli, Campana, Paraboschi, & González-Fernández, 2018; Legambiente, 2006), with high heavy metal concentrations (Inghilesi et al., 2012; Montuori, Aurino, Garzonio, Nardone, & Triassi, 2016), organophosphate pesticides pollution (Montuori, Aurino, Garzonio, Sarnacchiaro, et al., 2016), and solid waste reported around its two mouths.

Areas with higher relative abundance estimates and identified as suitable habitat for bottlenose dolphins are likely to be influenced by other anthropogenic factors as well. In particular, bottlenose dolphins were observed both by citizens and researchers to regularly follow trawlers or move near different fishing gears, resulting in the overlap between recreational, artisanal, and professional fishing operations and the species; this well-known interaction behaviour, already reported in numerous coastal areas in the Mediterranean Sea and worldwide, where animals are attracted towards easily accessible and concentrated food sources (Bonizzoni et al., 2014; Fertl & Leatherwood, 1997; Gonzalvo, Giovos, & Moutopoulos, 2015; Lauriano, Fortuna, Moltedo, & Notarbartolo di Sciara, 2004; Pace, Pulcini, & Triossi, 2003, 2012; Pennino, Rotta, Pierce, & Bellido, 2015; Pulcini et al., 2014), was then documented for the first time in the study area, conferring further support to the reliability of the approach applied.

Bottlenose dolphin strandings were principally recorded near the mouths of the rivers, but they also occurred over other coastal portions of the study area.

4.4 | Cuvier's beaked whales

Depth best predicted encounters with Cuvier's beaked whales, with a high probability at 800–1,000 m depth, far from the coast. This coincides with a number of descriptions of the habitat of this species in the Mediterranean, which report a clear habitat preference for areas at least 1,000 m deep, and complex bottom topographies related to phenomena (upwelling, increased primary production, and aggregation of zooplankton) that could play a role for beaked whales' main prey species (cephalopods) (Azzellino et al., 2008; Cañadas et al., 2018;

Podestà et al., 2006, 2016). Several studies demonstrated the importance of the central Tyrrhenian Sea for the species, showing long-term site fidelity and changes in habitat selection over time (Arcangeli et al., 2012, 2017; Arcangeli, Campana, Marini, & MacLeod, 2016; Cañadas et al., 2018). One of the hypotheses for these changes is related to the possible negative influence of maritime traffic (Campana et al., 2015). Our findings, both in terms of relative abundance and habitat suitability predictions, seem to support this theory, as the species was never recorded in one suitable area recognized by the modelling in the south (near Ventotene island), a highly touristic region heavily affected by boat traffic. The absence of Cuvier's beaked whales in this part of the study area was also reported in a recent Mediterranean assessment (Cañadas et al., 2018). In accordance with the low density of the species, no strandings were registered in the study area.

4.5 | Other species

Other species were relatively rare in the study area, in line with previous research results, and confirmed by the absence of stranding records, except for one species. The common dolphin is in steep decline throughout the Mediterranean (Pace et al., 2016), occurs in few delimitated areas, and is present in low numbers in Italian waters (Arcangeli et al., 2017; Pace et al., 2015, 2016). Mixed groups of common dolphins with other cetacean species were recorded here in both social media and research datasets, reinforcing similar observations already reported in the Mediterranean (Arcangeli et al., 2017; Frantzis & Herzing, 2002; Pace et al., 2015). This kind of association generally involved a few common dolphin individuals within a group of striped dolphins, supporting the hypothesis that the relative low abundance of common dolphin restrains the possibility of forming single-species groups (Arcangeli et al., 2017; Frantzis & Herzing, 2002).

Risso's dolphin was observed three times, and only one stranding occurred over the 10-year period. On average, Risso's dolphin in the Mediterranean appears to be a low-density species, showing a preference for deep waters over steep slopes and submarine canyons (Azzellino et al., 2008, 2012, 2016; Casacci & Gannier, 2000; David & Di-Meglio, 2012; Gannier, 2005; Mariani et al., 2016; Moulins, Rosso, Ballardini, & Würtz, 2008; Pace, Miragliuolo, & Mussi, 2012; Praca & Gannier, 2008). This kind of habitat was less represented in our study area, and results seem to be in accordance with previous studies in the central Tyrrhenian Sea (Arcangeli et al., 2017). One of the regularly encountered Mediterranean species, the pilot whale (*Globicephala melas*), was never observed in this study. This is consistent with previous reports, with no records of the species in the central Tyrrhenian Sea since the 1990s (Arcangeli et al., 2012, 2017), and there were a few sightings of a small group south-west of Ventotene island at a depth of over 400 m until 2006 (Pace, Miragliuolo, & Mussi, 2012).

Finally, rough-toothed dolphins and killer whales were confirmed as sporadic species within the Mediterranean (they are reported as 'visitors'; Notarbartolo di Sciara, 2016).

4.6 | Management implications

This study showed how information gathered from citizens can support scientific research acting as a feasible solution to improve spatial and temporal coverage for cetacean monitoring and research, even across scattered survey areas. Producing robust baseline data on cetacean communities at a wide spatial scale is a critical first step to recognizing threats and prioritizing species/sites that may necessitate conservation actions (Pace, Mussi, Gordon, & Würtz, 2014). This kind of information is of fundamental interest for the development of management priorities (Braulik et al., 2018; Hann et al., 2018), whose aim should be to preserve the most favourable areas for cetaceans and to enable these species to be included in global and regional initiatives, such as IMMAs and MPAs. For example, the central Tyrrhenian Sea was identified as a candidate IMMA in the Mediterranean (https://www.marinemammalhabitat.org/imma-eatlas/), and is waiting for further evidence to qualify as an IMMA. In this regard, results from this study may be used in the future to reassess the status of the entire region.

This study has shown that the area has a year-round presence of cetacean species and more than half of these are reliant on the inshore habitat, which makes them more exposed to human impacts. These extensive urbanized, highly populated coasts are also recognized to provide an important and thriving economic and recreational resource, supporting activities (e.g. tourism and fisheries) that may adversely influence its ecosystems and threaten cetacean species. Thus, the baseline information generated here, at least within the extent of the area surveyed, may enable early recognition of adverse effects from human impacts on cetaceans, highlighting main sources of risks and selecting areas for protection and human activities management (spatial management measures), or more detailed research. For example, bottlenose dolphin, classified as Vulnerable on the IUCN Red List because of its decline in the Mediterranean Sea, may receive targeted attention since the species is exposed to a wide variety of threats (chemical and plastic pollution, bycatch, reduced prey availability caused by overfishing, and habitat degradation, including acoustic disturbances from noise and vessel traffic) in the coastal areas investigated.

Commission Decision (EU) 2017/848 specified that, for marine mammal species, both state and pressure indicators (bycatch, contaminants, and marine litter) should be developed to help interpret changes in the populations. However, the abundance and the conservation status of cetacean populations in the study area, as well as actual pressures and potential threats for each species, fundamental requisites for conservation, still have to be assessed. In the absence of more robust data and management frameworks, the following operational recommendations are offered to cope with cetacean species in the region: (1) Implement further, targeted studies and monitoring activities that branch from the present work. (2) Enforce specific conservation measures (sensu Habitat European Directive 92/43/CEE) to maintain or restore 'favourable conservation status' for the species that are included in Annexes II and IV of the Habitats Directive. (3) Actively regulate, restrict, or prevent disturbance sources that may cause temporary displacement from key habitats, disruption of the animals' natural behaviours, and stress, such as shipping and boat traffic (including nautical tourism) and regulated or unregulated whale/dolphin watching (e.g. adjustment of navigation speed and observance of safety distance to the animals). (4) Develop and implement a code of conduct/guidelines to be promoted among tour boats and nautical tourism companies as well as among the large community of recreational boaters. (5) Develop dedicated initiatives to increase local public awareness and knowledge regarding the effects of human activities on cetacean species. (6) Develop a dedicated mobile phone app (with strict criteria for the input and incorporation of photograph/video, effort time, species, and simple behaviour categories) to obtain more structured sighting data from an engaged public. Regarding point (1), coordinated research activities in the study region due to a networking effort between the universities of Rome, Tuscia, and Pavia, combined with ISPRA, are currently ongoing. In addition, the use of passive acoustic monitoring in some specific areas is taking place, adding opportunities to collect data on cetacean presence and possible noise disturbance.

This study produced valuable scientific data for the cetacean case study offshore of the Lazio region in the central Tyrrhenian Sea. Opportunistic sightings collected by citizens were important to further understand the complexities and level of importance of different areas. Using cetacean research and social media combined datasets, new and interesting occurrence information and distribution results were obtained, especially for the species with more selective habitats (bottlenose dolphins, and fin, sperm, and beaked whales). The approach used here may be opportunistically applied in other regions worldwide, given the necessary caution that must be taken when using and analysing data and interpreting results of such data.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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ECOGRAPHY

Research

Integration of presence-only data from several sources: a case study on dolphins' spatial distribution

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Presence-only data are typical occurrence information used in species distribution modelling. Data may be originated from different sources, and their integration is a challenging exercise in spatial ecology as detection biases are rarely fully considered. We propose a new protocol for presence-only data fusion, where information sources include social media platforms, to investigate several possible solutions to reduce uncertainty in the modelling outputs. As a case study, we use spatial data on two dolphin species with different ecological characteristics and distribution, collected in central Tyrrhenian through traditional research campaigns and derived from a careful selection of social media images and videos. We built a spatial log-Gaussian cox process that incorporates different detection functions and thinning for each data source. To finalize the model in a Bayesian framework, we specified priors for all model parameters. We used slightly informative priors to avoid identifiability issues when estimating both the animal intensity and the observation process. We compared different types of detection function and accessibility explanations. We showed how the detection function's variation affects ecological findings on two species representatives for different habitats and with different spatial distribution. Our findings allow for a sound understanding of the species distribution in the study area, confirming the proposed approach's appropriateness. Besides, the straightforward implementation in the R software, and the provision of examples' code with simulated data, consistently facilitate broader applicability of the method and allow for further validations. The proposed approach is widely functional and can be considered with different species and ecological contexts.

Keywords: cetacean, data fusion, dolphins, Mediterranean Sea, presence-only data, point processes

Introduction

Progress of ecological science is more and more reliant on combining data from diverse sources (Fletcher et al. 2019). This approach can increase the comprehension

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of ecological processes for both research and conservation purposes (Pace et al. 2019). Data availability to model species distribution, for example, is rapidly expanding thanks to the fast development of new technologies (Soranno and Schimel 2014), the growth of citizen science initiatives (Sicacha-Parada et al. 2020, Matutini et al. 2021) and the opportunity of exploiting huge information harvested from social media platforms (Mikula and Tryjanowski 2016, Pace et al. 2019). The latter data types can be intrinsically challenging to merge in with existing, valued and validated data collected via standard research protocols. Yet, if that can be achieved, they can offer enrichment of existing data to generate powerful insights and even reduce the costs of collecting data conventionally (Buchanan and Bryman 2018). Nevertheless, heterogeneous data are complex to manage as they are polymorphic in nature and affected by numerous forms of bias and limitations (Isaac and Pocock 2015). Information on species occurrence collected at sea by sea-users, for example, is characterised by a different spatiotemporal distribution of effort, which can be biased toward easily accessible habitat and times with better weather, or known areas of use (Corkeron et al. 2011, Sicacha-Parada et al. 2020). Hence, a simple data pooling (Fletcher et al. 2019) with data gathered under conventional research methodologies is not enough to reliably model the presence of a species considering different explanatory variables both environmental and anthropogenic and to define its distribution over multiple spatial and temporal scales.

Integrated distribution modelling (IDM), i.e. the practice of fitting species distribution models with more than one observation practice (Isaac et al. 2020), is a new approach to combine different datasets, preserving the strengths of each and adjusting, at least to some extent, their limitations. IDM sets a spatial or spatio-temporal latent state, here statistically defined as a point process, of the sites where the animals were sighted, described by a series of covariates shared by different datasets. Multiple observation sub-models can be estimated from them, each describing a part of the latent state.

Coping with several challenges, we propose a novel path to combine different data sources to provide cohesive summaries of species' potential and realised distribution (Isaac et al. 2020). First, as the available information is presence-only data, we opt for point process as the most natural solution (Miller et al. 2019). Second, as several sources of bias are potentially present in the datasets, we propose models based on a location-dependent thinning of a Poisson process to reduce these biases (Dorazio 2014, and references therein); however, the parameters of these models are not fully identifiable unless the covariates of abundance are distinct and linearly independent of the covariates of detectability (Dorazio 2014). In Yuan et al. (2017), a flexible stochastic partial differential equation (SPDE) model describes the spatial structure that is not accounted for by explanatory variables, and estimation is carried on using integrated nested Laplace approximation (INLA) in a Bayesian inference framework. The latter allows simultaneous fitting of detection and density models and permits prediction at an arbitrarily fine scale. Very recently Sicacha-Parada et al. (2020) adopt a similar approach using citizen science data on moose *Alces alces* occurrence in Norway, accounting for the geographical bias (oversampling of 'accessible' locations). For marine observations, the boat's size, the distance from the coast, policy regulations and weather conditions are just some of the factors that can affect the accessibility of an area. We propose a new protocol for presence-only data fusion, where information sources include social media. We investigate several possible solutions and compare different types of detection function and accessibility explanations. We use the IDM approach on sighting data derived from different data sources (research, monitoring and social media) to predict the distribution of two dolphin species in the central Mediterranean Sea. The study of spatial distribution patterns of dolphin species is incredibly puzzling. They spend much time under the water surface (Redfern et al. 2006), and a lot of visual/acoustic effort for scientists is needed to assess their presence in a specific habitat (Breen et al. 2017, Redfern et al. 2017). We show how variation in the detection function affects ecological findings on two dolphin species with different spatial distribution.

The proposed approach is entirely broad and the selected species are representatives for different habitats. Hence they constitute a good benchmark for the entire proposal. We provide R functions and example code to replicate our work in the online Supporting information (<https://github.com/smar-git/SM-data-merging>).

Material and methods

Study species

Two dolphin species were selected for this study, the bottlenose *Tursiops truncatus* and the striped dolphins *Stenella coeruleoalba*, both widely distributed throughout the Mediterranean Sea. The bottlenose dolphin is reported predominantly coastal or inshore (Bearzi et al. 2012), but its habitat changes depending on the region: it can inhabit shallow waters (less than 50 m) close to the coast and at the mouths of the rivers (Triossi et al. 2013, Pace et al. 2019), around archipelagos or islands (Pace et al. 2012, 2019, Pulcini et al. 2014), and in waters above the continental shelf and slope (Azzellino et al. 2008); less frequent, but still present, in deeper waters and pelagic areas. Bottlenose dolphins feed a wide range of demersal and coastal prey and can forage opportunistically behind trawling vessels (Pace et al. 2012). The striped dolphin is considered pelagic in the Mediterranean Sea, showing a general preference for highly productive, open waters beyond the continental shelf (Aguilar and Gaspari 2012). Although the species is the most abundant cetacean in the Mediterranean, it is not found at uniform densities. The striped dolphin diet is mainly composed by pelagic or bathypelagic schooling-nictemeral fish, squids and even crustaceans (Meissner et al. 2012). There are not exact estimations of the number of bottlenose and striped dolphins living in the Mediterranean Sea. The poor understanding of the status of a population, together with the suspected decline in numbers (both species are listed under the status vulnerable in the IUCN Red List as their populations have been decreasing during the last decades), emphasize the importance of integrating all available information (Pace et al. 2014, 2021b).

Study area

The study area covers about 39 000 km^2, and is located in the central Tyrrhenian Sea (Italy) (Fig. 1); it is characterized by different environmental features (e.g. bathymetries), structures (e.g. seamounts) and types of habitats (Pace et al. 2019, 2021a). Several rivers flow in this region, including the Tiber, and the simultaneous presence of both fresh and salt waters, as well as the geomorphological action of sedimentation and erosion, generate different ecological gradients, making the coastal area highly productive and rich in biodiversity (Ventura et al. 2015, Ardizzone et al. 2018, Casoli et al. 2019). The study area also includes five islands (Giglio and Giannutri at north; Ponza, Ventotene and Santo Stefano at south) and several commercial/touristic harbours generating high-levels of maritime traffic by different vessels. The region hosts seven of the eight cetacean species regularly found in the Mediterranean, with a major presence of bottlenose and striped dolphins (Pace et al. 2019, 2021a).

Data sources and attributes

Dolphin data cover a period of 13 years (2007–2019). Records are from three sources: a) conventional research protocols from motor and sailing boats (non-systematic haphazard, sensu Corkeron et al. 2011) (labelled UNIRM) (Pace et al. 2019); b) standardized monitoring protocols from platforms of opportunity within the project FLT Mediterranean Monitoring Network (labelled FERRY) (ISPRA 2016, Arcangeli et al. 2019, Pace et al. 2019); c) social media reports (Facebook and YouTube) by sea-users (Pace et al. 2019) (labelled SM). Data collection protocols and selection procedures are provided in Pace et al. (2019). As the SM dataset included also details on other cetacean species than the two here investigated (Fig. 1b), we used this information as a proxy to infer boat densities potentially able to record the animals' presence.

These three sources accounted for 283 records of striped dolphin (about 50% from SM) and 579 of bottlenose dolphin (about 80% from SM). The major contribution by SM justified the need for a careful choice of the related model's elements.

We used distance from the coast (i.e. the euclidean distance between a sighting point and the shoreline), depth, slope, temperature and primary productivity as covariates. These are commonly selected in cetacean distribution studies as they may represent good proxies for species' ecological needs (Chavez-Rosales et al. 2019, Stephenson et al. 2020). Temperature and primary productivity were retrieved from COPERNICUS platform <https://marine.copernicus.eu/>. Depth data were downloaded from GEBCO (General bathymetric Chart of the Ocean – <www.gebco.net/>). Slope was computed from depth data through the `terrain()` function in R. Details of the retrieved datasets and covariates handling procedures are reported in the Supporting information.

Modelling approach

Our aim was to integrate data from three main sources. Two are typical approaches adopted in research surveys. We consider the adaptive sampling procedure used by Sapienza University of Rome (UNIRM) (see for instance Dawson et al. 2008, Lennert-Cody et al. 2018, and references therein) and the very well known distance sampling (Buckland et al. 2001) adopted by ISPRA (FERRY) (ISPRA 2016), together with the Social Media (SM) extracted data (below and Pace et al. 2019, for detailed description of the data). We aimed at representing and managing possible detection bias in each dataset adopting a point processes modelling approach. The Supporting information illustrates and summarises the workflow used for building the model.

We followed Yuan et al. (2017) and Sicacha-Parada et al. (2020), expanding their approaches by building a spatial log-Gaussian cox process (LGCP) (Illian 2019) that incorporates different detection functions and thinning for each data source. We assumed that sighting patterns, i.e. locations of dolphin groups in space ($s \in \mathcal{S} \subset \Re^2$) and time ($t \in \mathcal{T}$), are properly described by a point process whose intensity function $\lambda(s,t)$ is additive on the log-scale:

$$\log\left(\lambda(s,t)\right) = \mathbf{X}^T(s,t)\beta + f(\mathbf{z}) + \omega(s) \tag{1}$$

Here $\mathbf{X}(s,t)$ is a set of covariates detected at location s and time t with linear effect β to be estimated. $f(\mathbf{z})$ is a smooth effect (that may be present or not) of some geo-referenced covariates $\mathbf{z}$. A common prior for $f(\mathbf{z})$ is a random walk (RW) model of order 1 (Rue and Held 2005). Finally, $\omega(s)$ is a zero-mean Gaussian process describing the residual spatial variation. As in Yuan et al. (2017) we adopted a Matèrn covariance of order 1 with range ρ and standard deviation σ. Although it would have been, in theory, possible to consider $\omega(s)$ a complex spatio-temporal model (Yuan et al. 2017), the limited number of sightings each year did not provide enough information in practice. Therefore we chose to run $\omega(s)$ a pure spatial model.

We assumed that the above process was observed in three different ways, conditionally independent given $\lambda(s,t)$. Thus, three observed intensities were defined:

$$\lambda^*(s,t) = T_j g_j(s)\lambda(s,t), \qquad j = 1,2,3 \tag{2}$$

where T_j is a time scaling factor and $g_j(s)$ is the detection function (with values between 0 and 1) which determines the thinning of the original process. The form of the detection function depends on the type of observational process. For adaptive sampling (UNIRM)

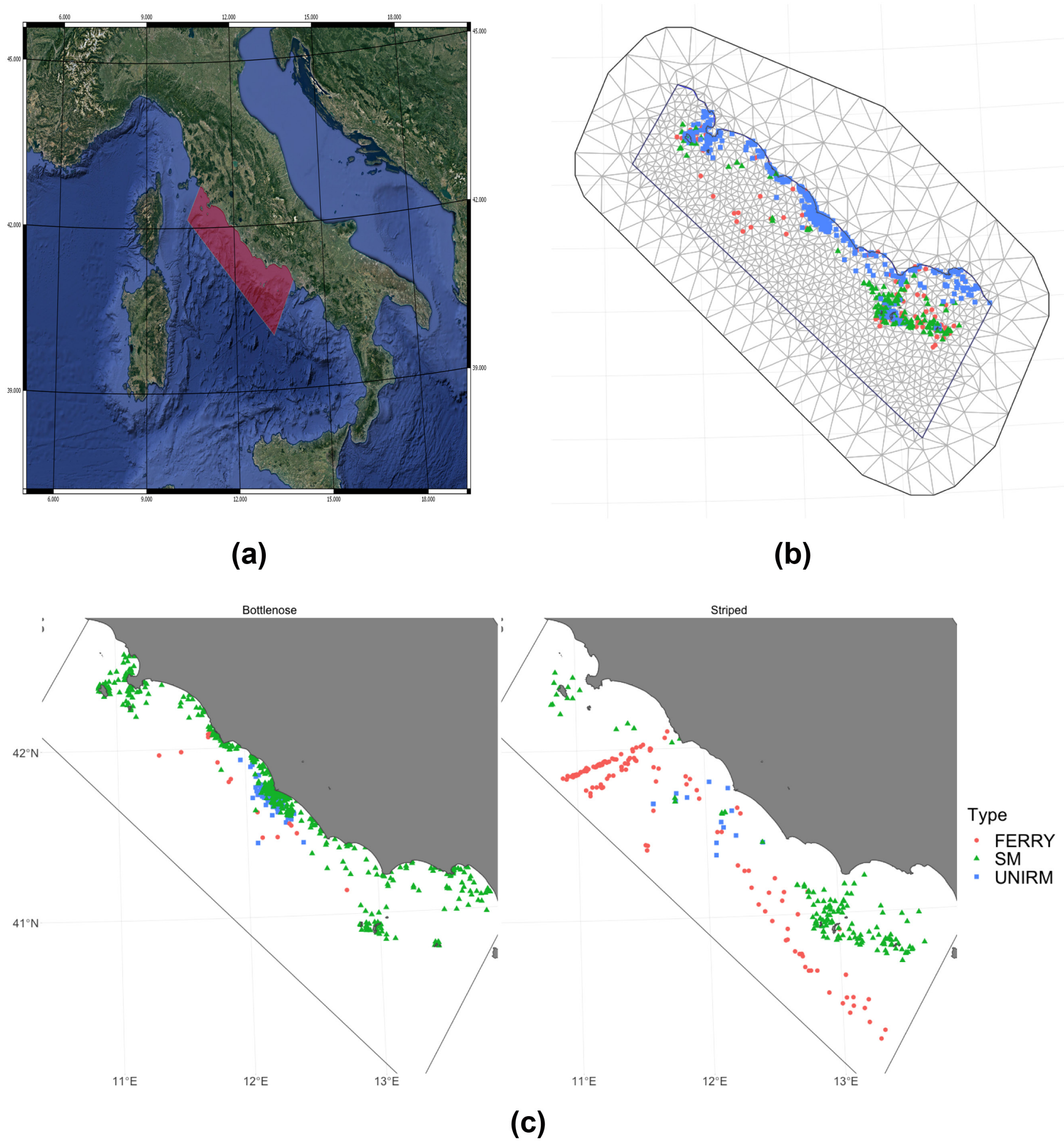

Figure 1. (a) Study area. (b) Study area and SM records for striped (green triangles), bottlenose (blue squares) dolphins and other cetacean species (red dots) superimposed to the mesh chosen for models estimation. (c) Dolphins encounters' locations by source (observation processes): SM (green), FERRY (red) and UNIRM (blue).

$$g_1(s) = \begin{cases} 1 & d_1(s) \le K \\ 0 & d_1(s) > K \end{cases} \quad (3)$$

where $d_1(s)$ is the distance (km) between point s and the position of the boat when the groups were sighted. K was defined as the maximum distance measured between the location of the first visual sight of a dolphin group by researchers (equipped with 7 × 50 and 10 × 50 binoculars) on the boat and the effective location of the group under optimal survey conditions (i.e. sea state ≤ 1 Douglas, wind force ≤ 1 Beaufort, no rain, no fog, no clouds). This measurement was possible because, upon sighting dolphins, researchers marked the GPS point where the animals were first located, the survey effort was suspended and the vessel departed from its route to approach the group to a suitable distance (10–30 m) to correctly identify the species, estimate group size and composition. K was set to 4 km, assuming that researchers can spot animals closer than K.

For the distance sampling (FERRY) data, we used the classical half normal detection function (Thompson and Ramsey 1987) defined as

$$g_2(s) = \exp\left(-\frac{d_2(s)^2}{2\,\xi_2^2}\right) \quad (4)$$

where, $d_2(s)$ is the perpendicular distance (km) to the ferry track and ξ_2 is a scale parameter.

For the SM dataset, the definition of the detection function was carefully considered for biases. Records in this dataset are affected by large uncertainty, as observations are generally a) skewed towards more accessible areas (Monsarrat et al. 2019, Sicacha-Parada et al. 2020) and b) collected from small leisure boats that are difficult to track in a systematic way. To better define 'more accessible' and consider the distribution of the small boats we explored three different possibilities.

First, we reasonably assumed that locations closer to the coast are more accessible to sea-users with small boats. Thus, following Sicacha-Parada et al. (2020), the detection function, labelled as 'detection coastline', was defined as:

$$g_{3,1}(s) = \exp\left(-\frac{d_{3,1}^2(s)}{2\,\xi_{3,1}^2}\right) \tag{5}$$

where $d_{3,1}(s)$ is the Euclidean distance from the coast (Fig. 2a) and $\xi_{3,1}$ a scaling parameter. However, the distance from the coast may not provide an accurate representation of the small boats' density in a given area: locations close to harbours and holiday destinations (e.g. islands) are generally more crowded than other sites at the same distance from the coastline.

To obtain information on the boats density in the study area, we used data from EMODnet (European Marine Observation and Data Network; Martín Míguez et al. 2019), a free-usage platform of vessel density data derived from boats using AIS (automatic identification system, mandatory above 15 m length). The database has a spatial resolution of 11 km and covers 2017–2019 period. We selected two vessels categories (sailboats and pleasure crafts) from the 11 listed, and applied a kernel estimator to ensure a smoothed density surface. The resulting log-density surface (Fig. 2b) was labelled as vessel log-density surface. As expected, higher vessel log-densities were identified near the principal harbours and the islands. Our second detection function for SM data, denoted 'detection Emodnet', was defined as

$$g_{3,2}(s) = \Phi\left(\frac{d_{3,2}(s)}{\xi_{3,2}} - \mu_{3,2}\right) \tag{6}$$

where $d_{3,2}(s)$ is the vessel log-density, and Φ is the normal cumulative distribution function (cdf) with $\mu_{3,2}$ and $\xi_{3,2}$ as location and scale parameters, respectively. The normal cdf was selected as we required the detection function to be close to 1 when the vessel log-density is high, and close or equal to zero when it is small (or null). EMODnet information

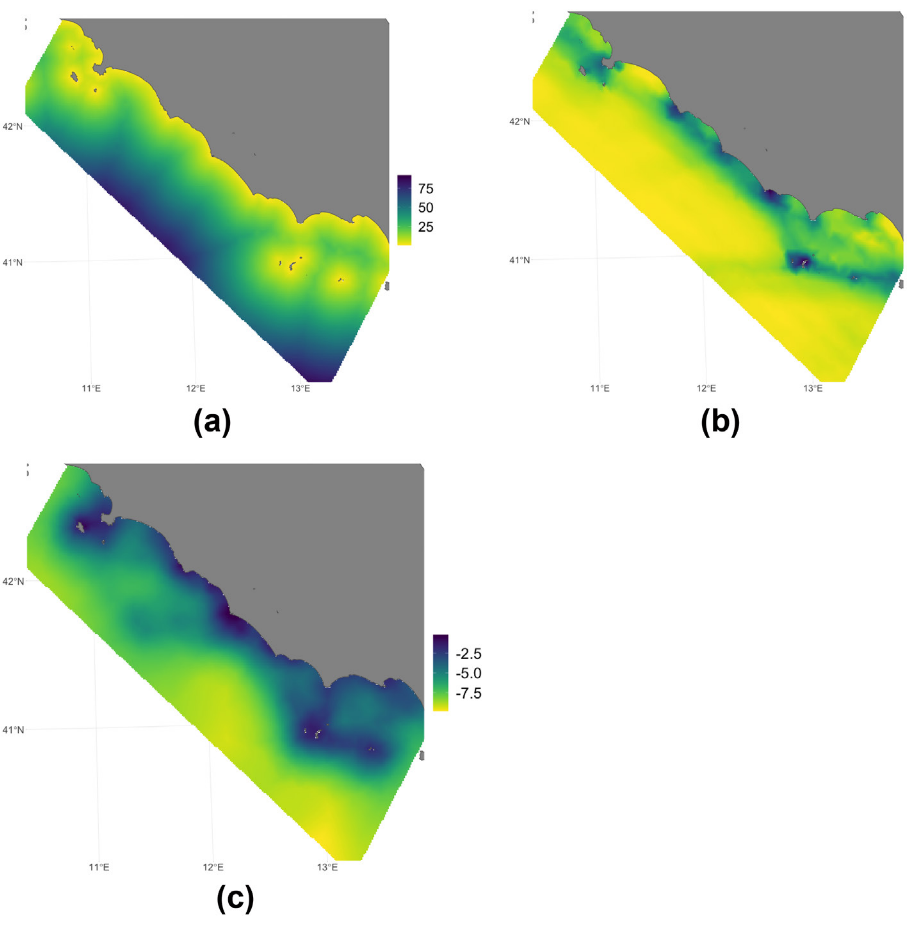

Figure 2. (a) Distance from the coastline (km), (b) log vessels density, (c) estimated log intensity from observations of all species.

accounted for a limited time frame compared to our study and for larger vessels than the ones generally reporting observation records in SM platforms (small recreational boats moving near the coastline). We therefore introduces a third detection function.

We used the entire SM dataset of 581 records (125 striped and 334 bottlenose dolphins, and 122 other cetacean species) to estimate the observation process intensity. We considered the spatial pattern of such observations as a proxy for the small boat density process if we disregard the species. A similar approach was used in occupancy models context, where non-detection records were constructed from sightings of other benchmark species (Kery et al. 2010, Dennis et al. 2017). We applied a spatial LGCP to estimate the (log) intensity of the process. Details of the estimation process can be found in the Supporting information. Figure 2c shows the resulting estimated log-intensity used as input for the detection function, labelled 'detection animals':

$$g_{3,3}(s) = \Phi\left(\frac{d_{3,3}(s)}{\xi_{3,3}} - \mu_{3,3}\right) \quad (7)$$

where $d_{3,3}(s)$ is the estimated log-intensity at point s while Φ, $\mu_{3,3}$ and $\xi_{3,3}$ are defined as in (6).

Eventually, another potential bias affecting the observation processes is the different time (days) spent at sea by each data source. To account for this bias as well, we introduced the t_j parameter in expression (1). T_j is known for both the FERRY and the UNIRM data (311 and 73 days at sea respectively) and undetermined for SM data. We know that SM observations were collected by leisure boats all over the year, with a major number of sightings reported in spring–summer. Thus, we ran estimations with $T_3 = 160, 200, 365$ days, without sensible changes, and selected $T_3 = 360$.

Table 1. Comparison criteria for the four fitted models for both striped (a) and bottlenose (b) dolphins.

Model	DIC	WAIC	MLIK	LMPL
(a) *Stenella coeruleoalba*				
Constant detection	4078.53	4129.16	−2111.81	−2098.13
Detection coastline (Eq. 5)	3895.52	3933.01	−2008.44	−1988.77
Detection emodnet (Eq. 6)	3840.93	3889.51	−2019.20	−1969.47
Detection animals (Eq. 7)	3789.38	3810.08	−1942.07	−1922.56
(b) *Tursiops truncatus*				
Constant detection	4639.94	4874.47	−2375.33	−2568.07
Detection coastline (Eq. 5)	4555.23	4797.14	−2337.60	−2726.91
Detection emodnet (Eq. 6)	4552.61	4810.37	−2344.68	−2658.98
Detection animals (Eq. 7)	4485.78	4624.58	−2281.27	−2351.19

Priors specification

To finalize the model in a Bayesian framework, we needed to specify priors for all model parameters. To avoid identifiability issues when estimating both the animal intensity and the observation process, we used slightly informative priors. For the parameters in the spatial field $\omega(s)$ in (1) we used PC priors (Fuglstad et al. 2019) setting $P(\rho < 150) = 0.5$ and $P(\sigma > 2) = 0.01$, thus we considered a standard deviation above 2 and a range of 150 km likely. We assigned β in (1) and the locations parameters $\mu_{(3,2)}$ and $\mu_{(3,3)}$ Gaussian prior precision 0.01 and means 0, 3 and −5 respectively. Finally, for the scale parameters in (4–6), let $\xi = F_{\alpha}^{-1}(\Phi(\theta))$ where $F^{-1}(\cdot)$ is the inverse exponential cdf with rate α and Φ a normal cdf. This corresponds to attributing an exponential prior to ξ. We assigned θ a standard normal prior. The parameter α is set to 1/20 in (5), and 1 in all other cases. The difference in rate was due to the different scale of the three inputs for the detection function (Fig. 2). The Supporting information illustrates the effect of our prior choice on the detection functions.

Inference and computational approach

The traditional way of fitting point processes is by gridding the space and modelling the intensity on a discrete number of cells. This implies that observations' locations are also approximated. We followed instead the approach introduced in Simpson et al. (2016) and applied in Yuan et al. (2017) and Sicacha-Parada et al. (2020). Such an approach allowed us to use the true sighting locations, thus avoiding loss of information. Besides, the Gaussian field's SPDE representation has several computational advantages (Lindgren et al. 2011). To build a spatial model using the SPDE approach, we used the mesh shown in Fig. 1b.

For computational efficiency, we used INLA (Rue et al. 2009). INLA allowes also to easily combine the three observation model in (2) to form the likelihood. Our model does not directly fall under the latent Gaussian model framework for the INLA estimation software because the parameters in the detection functions in (4–6) do not enter the model in a log-linear way. We used therefore the methodology introduced in Yuan et al. (2017) and implemented in the `inlabru` R package (Bachl et al. 2019) that allows fitting models with some non-linear elements. This is done by linearizing the model via Taylor approximation and using a line search to optimize the linearization point.

Model evaluation was carried out using goodness of fit measures as in Sicacha-Parada et al. (2020), through the deviance information criterion (DIC), Watanabe–AiKaike information criterion (WAIC), marginal likelihood (MLIK) and the logarithm of the pseudo marginal likelihood (LMPL). As a benchmark for the SM detection function choice, we used a constant detection function $g(s) = 1, \forall s$, that is equivalent to not include any thinning for the SM data.

Results

The distribution of the dolphins encounters in the study area is shown in Fig. 1c. Environmental covariates selection was finalized considering several combinations of covariates and detection functions. Two different models have been selected, one for each species

- *S. coeruleoalba* (striped dolphin)
  - Depth: categorized as (< 100, 100–200, 200–1000, > 1000 m)
  - Slope: non parametric with a prior Random walk of order 1
  - Distance from the coast: linear term
- *T. truncatus* (bottlenose dolphin)
  - Depth: linear term
  - Slope: linear term
  - Distance from the coast: linear term

There was no evidence that the sightings intensity was affected by the spatio-temporal covariates, therefore our final models are reduced to purely spatial ones.

The evaluation of SM detection functions was based on model's goodness of fit measures, DIC, WAIC, MLIK and the LMPL, it is reported in Table 1. The selected best performing detection function for all criteria and species is (7) (labelled as 'intensity'). This choice affected model's terms estimate. For striped dolphin model with varying detection functions (Supporting information), the effects of categorized depth were fairly in agreement with the species distribution ranges: it is generally not found in very shallow waters (negative effects), observed at 100–200 m depth, and more often encountered at depths over 200 m.

The effect of the detection function was found in the reduction of uncertainty in the estimates, which is reflected in the smaller size of the credibility intervals (7). Slope showed a significant reduction effect in the encounters where it is steeper. No significant difference was found among the smooth effects with varying detection (overlapping 95% confidence band, not shown). The effect of the Distance from the coast was not significant, and the intercept was larger for detection functions (6) and (7), with the latter showing less uncertainty than the first.

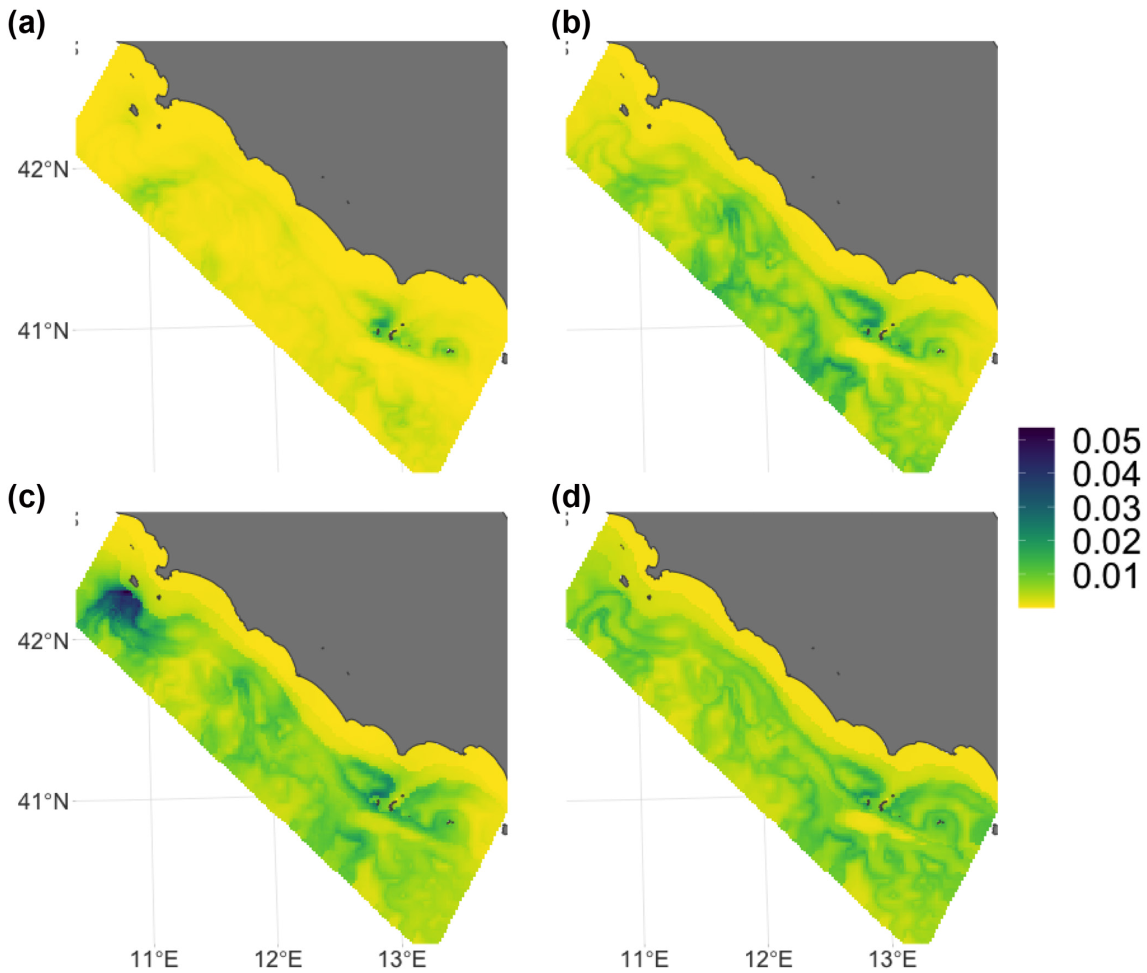

Figure 3. Estimated posterior median for the intensity of striped dolphins using different detection functions for SM data. (a) constant detection, (b) detection coastline (Eq. 5), (c) detection Emodnet (Eq. 6), (d) detection animals (Eq. 7).

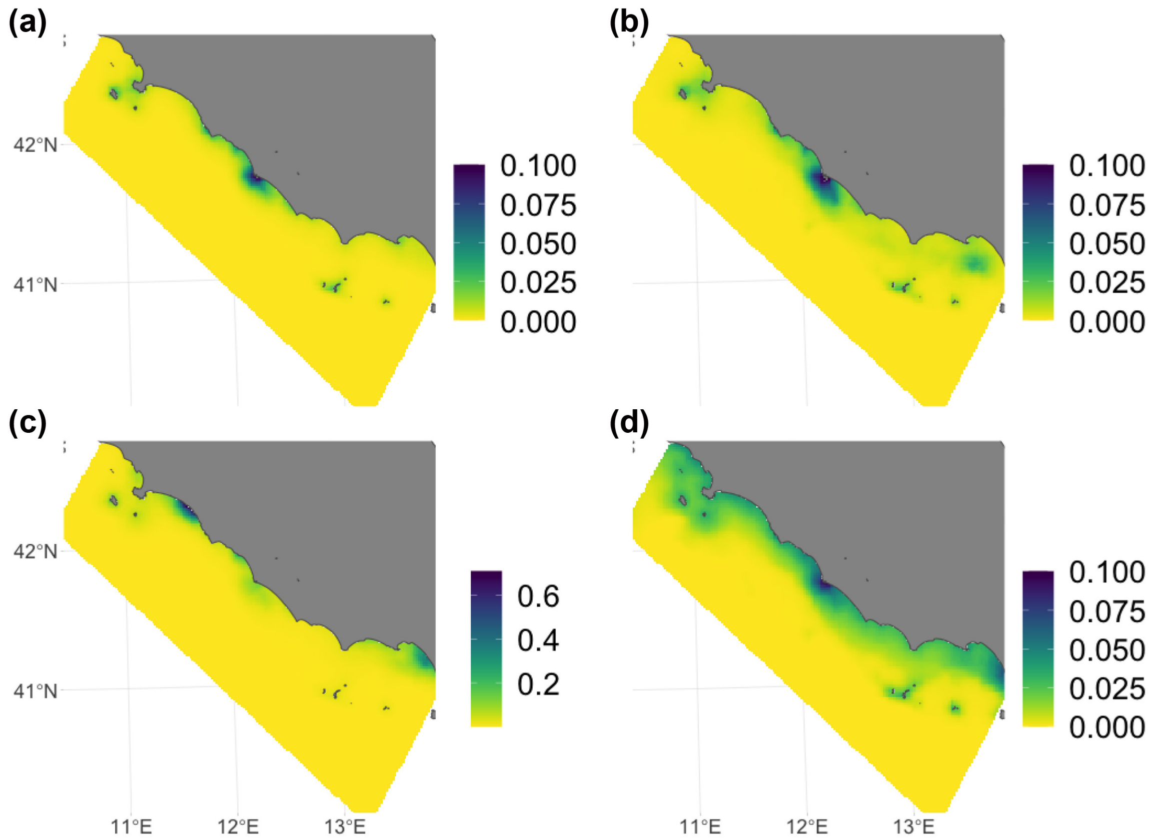

Figure 4. Estimated posterior median for the intensity of bottlenose dolphin using different detection functions for SM data. (a) constant detection, (b) detection coastline (Eq. 5), (c) detection vessels (Eq. 6), (c) detection animals (Eq. 7). Note that the scale in (c) is different from the other three figures.

For bottlenose dolphin model with varying detection functions (Supporting information), both depth and distance from the coast had negative effects on sightings (deeper waters and increasing distance from the coast mean less encounters), with no-significant difference among detection functions. Again, detection (7) induced narrower 95% credible intervals.

Estimates of detection functions parameters for both species are reported in the Supporting information.

As a measure of relative uncertainty for the predicted intensity $\lambda(s)$ we used the relative width of the 50% posterior credible interval (RWPCI) as proposed in Yuan et al. (2017). This measure is defined as the interquartile range divided by the median

$$\text{RWPCI} = (Q_3 - Q_1) / Q_2 \quad (8)$$

The intensity surfaces estimated for the striped and bottlenose dolphins are shown in Fig. 3, 4, respectively; associated RWPCIs (8) are mapped in the Supporting information. The intensity surface for both species changed consistently with the different detection function adopted for SM. For example, the vessel-based detection (6) (EMODnet-log density surface) induced some artifacts for the striped dolphin (Fig. 3c), and in general seems to over-estimate the dolphins encounter probability. This is evident, for example, around the Giglio Island, where detection highlights hotspots for both striped and bottlenose (Fig. 4c) dolphins possibly induced by the presence of few vessels and several encounters. The detection based on distance from the coast (5) and the constant detection, under-estimate the same probability for the striped dolphin and create some artifacts as well. A relevant feature of the detection function (7) is that it allowed for a consistent reduction in the uncertainty associated with the estimated intensity surface.

Figure 5 describes the estimated probability of the average number of sightings in the area over 13-year study period. These distributions represent the potential encounters if the entire area would be surveyed. (d) corresponds to the chosen detection, as it best represents the studied phenomena. Striped dolphin is considered the most abundant and common cetacean in the Mediterranean (Aguilar and Gaspari 2012), but seems to be less represented in the study area than bottlenose dolphins (283 records of striped versus 579 of bottlenose dolphins). Although this may introduce a large uncertainty on the estimates, it is still possible to appropriately capture the species spatial distribution. In panel (c) an over-estimation of the bottlenose dolphin encounters given by the vessel detection seems evident, and in panel (b) the coastline detection apparently induces a distribution of potential sightings only driven by the data.

Discussion

This study demonstrates that methods of spatial data integration able to carefully consider and minimize datasets' biases can be efficiently used to predict species' distribution. Results here obtained may be broadly applicable to other species that require an improvement of spatial knowledge for their conservation and management.

Dorazio (2014) pointed out that several statistical models have been proposed to analyse presence-only data, but they have largely ignored the effects of imperfect detectability and survey bias. The same author showed that proper modelling choices could reduce the bias in SDM estimates induced by these types of errors. Here we do more than just correct for detectability issues; we allow multiple sources of information to be integrated. We defined and estimated source-specific detection functions considering the nature of the data, i.e. presence-only, and the different observation processes, offering a more precise picture of the distribution of two dolphin species in the central Mediterranean. The output is consistent with the ecology of these species, highly supporting a thoughtful usage of spatial data extracted from social media platforms and introducing a novel way to model observation biases. In analysing different detection functions, we optimise distribution models for each species. That is very attractive considering the importance of defining suitable habitats for vulnerable or endangered cetaceans exposed to anthropogenic disturbance or threats, particularly in coastal areas (Pace et al. 2018).

The point process approach allows us to reliably estimate the observation intensity surface. The analysis of intensity surfaces in Fig. 3, 4, gives important insights on the relevance of the detection function in observation intensity estimation. The artefacts around the Tiber river estuary (central part of the area) for the bottlenose dolphin and close to the Giglio island (northern portion of the study area) for the striped dolphin are solved by detection (7). Again, with the same detection function's choice, analysing in the Supporting information, we can observe the reduction of intensity estimates' variability (and hence uncertainty). The proposed 'best' choice is very general and can be adopted whenever social media data are available.

The two species were also studied in Pace et al. (2019) using a presence-only data approach based on MaxEnt (Phillips et al. 2006). While results related to the bottlenose dolphin analysis were ecologically sound and coherent, striped dolphins analysis was unfeasible in that framework, given the relevant number of near-to-the-coast observation by sea-users. In particular, the depth around the Pontine islands rapidly increases with the distance from the coast, playing a misleading role in the MaxEnt modelling approach. The proposed methodology, instead, is fully able of capturing both species behaviour, thus addressing the complex task of finding targeted techniques weighting species' diversity.

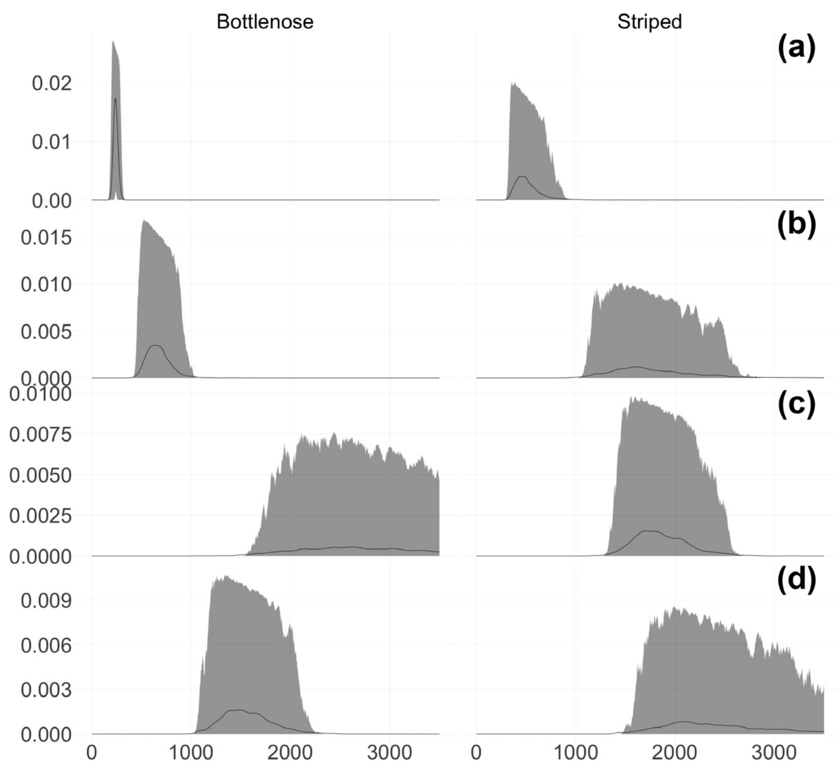

Figure 5. Estimated probability of the average number of sightings in the area over 13-year study period, in the entire study area for bottlenose (left) and striped (right) dolphins for the four fitted models: (a) constant detection, (b) detection (5), (c) detection (6) and (d) detection (7). The grey band indicate 95% credible intervals.

Some limitations are intrinsic to the proposed approach. On the one hand, spatial estimation does not distinguish between land and sea. That implies the use of post-processing to cut the estimated intensity surface. On the other, each analysed detection function is not very flexible. Eventually, the information used to model the observation effort in the SM data can be further improved. Hence, further investigations will be carried out to:

- Develop spatially non-stationary modelling approaches where a barrier can be added at the coastline as in Bakka et al. (2019).
- Develop flexible detection functions.
- Explore the use of satellite data to estimate the density of small boats in the study area (Santamaria et al. 2017).
- Explore the use of biological driving variables (e.g. prey biomass) as predictors.

The implementation of these tasks and the improvement of the models capabilities may further develop a fast-growing research approach and provide innovative insights in marine top-predators distribution patterns. The multiplicity of issues confronting these marine species requires collaborative efforts at all levels to share and merge resources, data and expertise efficiently (Pace et al. 2018, Vella et al. 2021).

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Data availability statement

Data and tutorials for elaboration are available from: <https://github.com/smar-git/SM-data-merging>. (Martino et al. 2021).

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MedSens index: The bridge between marine citizen science and coastal management

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ABSTRACT

Citizen science (CS) projects may provide community-based ecosystem monitoring, expanding our ability to collect data across space and time. However, the data from CS are often not effectively integrated into institutional monitoring programs and decision-making processes, especially in marine conservation. This limitation is partially due to difficulties in accessing the data and the lack of tools and indices for proper management at intended spatial and temporal scales. *MedSens* is a biotic index specifically developed to provide information on the environmental status of subtidal rocky coastal habitats, filling a gap between marine CS and coastal management in the Mediterranean Sea. The *MedSens* index is based on 25 selected species, incorporating their sensitivities to the pressures indicated by the European Union's Marine Strategy Framework Directive (MSFD) and open data on their distributions and abundances, collected by trained volunteers (scuba divers, free divers and snorkelers) using the Reef Check Mediterranean Underwater Coastal Environment Monitoring (RCMed U-CEM) protocol. The species sensitivities were assessed relative to their resistance and resilience against physical, chemical, and biological pressures, according to benchmark levels and a literature review. The *MedSens* index was calibrated on a dataset of 33,021 observations from 569 volunteers (2001–2019), along six countries' coasts. A free and user-friendly QGIS plugin allows easy index calculation for areas and time frames of interest. The *MedSens* index was applied to Mediterranean marine protected areas (MPAs) and the management and monitoring zones within Italian MPAs. In the studied cases, the *MedSens* index responds well to the local pressures documented by previous investigations.

MedSens converts the data collected by trained volunteers into an effective monitoring tool for the Mediterranean subtidal rocky coastal habitats. *MedSens* can help conservationists and decision-makers identify the main pressures acting in these habitats, as required by the MSFD, supporting them in the implementation of appropriate marine biodiversity conservation measures and better communicate the results of their actions. By directly involving stakeholders, this approach increases public awareness and the acceptability of management decisions, enabling more participatory conservation tactics.

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1. Introduction

Community-based environmental monitoring (CBM) is a participatory approach to engage citizen volunteers, through citizen science (CS) programs, to enhance the ability of decision-makers and non-government organisations to monitor and manage natural resources, track at-risk species, and protect biodiversity (Chandler et al., 2017; Conrad and Hilchey, 2011). Thus, CBM involves citizens and other stakeholders in the ecosystem-based management (EBM) of natural heritage, aiming to conserve ecological goods and services by recognising their interactions within an ecosystem (Alexander et al., 2019; Freiwald et al., 2018; Keough and Blahna, 2006). Marine citizen science (MCS) may represent a valuable contribution to CBM in marine environments, given the vastness of the oceans and the world's coastlines and the diversity of their habitats, communities, and species (Garcia-Soto et al., 2017; Thiel et al., 2014). By engaging millions of people around the world, MCS programs are becoming increasingly important to conservation science by influencing and improving the management of marine protected areas (MPAs) and fishery resources (Freiwald et al., 2018). MCS programs also increase observation capacities (Hodgson, 2001; Pattengill-Semmens and Semmens, 2003; Sully et al., 2019). Despite a worldwide increase in the number and extent of MCS programs (Thiel et al., 2014), the collected information is rarely used for institutional monitoring programs or to inform decision-making processes in marine conservation (Conrad and Hilchey, 2011). This disconnect is partially due to persisting scepticism of the reliability of data collected from volunteers (Burgess et al., 2017) and to a co-creation approach that is still not well-integrated in CS processes (Bonney et al., 2015). If the results of a CS project answer research questions that are of low interest to decision-makers, it will inevitably be difficult to integrate the CS data into management strategies. However, many studies demonstrate that well-trained citizens can provide valuable data on marine environmental issues and that suitable protocols for volunteer projects can provide results that are consistent with the methods used by professional researchers (e.g. Done et al., 2017; Forrester et al., 2015; Holt et al., 2013). Still, there are limits to accessing the data, which are not always well-organised and readily available according to the FAIR (findable, accessible, interoperable, and reusable) data principles (Wilkinson et al., 2016). Also, there is a lack of simple analysis tools and indices to summarise the data and extract relevant information for management purposes at the proper spatial and temporal scales.

This study aims to provide a biotic index to environmental managers and decision-makers – the RCMed species sensitivity (*MedSens*) index, based on open data collected under the Reef Check Mediterranean Underwater Coastal Environment Monitoring (RCMed U-CEM) protocol (www.reefcheckmed.org; Cerrano et al., 2017). The *MedSens* index is not purport to replace detailed studies and the indices applied by professional researchers, such as the Coralligenous Assemblage Index (CAI; Deter et al., 2012), the Coralligenous Assessment by Reef Scape Estimate index (COARSE; Gatti et al., 2015), the Ecological Status of Coralligenous Assemblages index (ESCA; Piazzi et al., 2017), the Index Coralligenous approach (INDEX-COR; Sartoretto et al., 2017), the Standardized Coralligenous Evaluation procedure (STAR; Piazzi et al., 2019), and the 3D-complexity index (Valisano et al., 2019). The *MedSens* index is intended to integrate the assessment of the environmental status of coastal Mediterranean areas threatened by multiple stressors (Micheli et al., 2013) while considering the protected and sensitive species and adhering to the requests of the European Union's Habitat Directive (92/43/EEC) and Marine Strategy Framework Directive (MSFD, 2008/56/EC; Borja et al., 2010). A plugin has been specifically developed for the open-source geographic information system QGIS (QGIS Development Team, 2019), allowing index calculations for the areas and time frames of interest.

2. Materials and methods

2.1. The Reef Check Mediterranean U-CEM protocol

The RCMed volunteers (mainly scuba divers, but also free divers and snorkelers; EcoDivers hereafter) collect data on the abundances of selected taxa according to the U-CEM protocol (Cerrano et al., 2017). After a short training course and the verification of their learning and abilities, EcoDivers can make independent observations along random swim (Hill and Wilkinson, 2004). The taxa were selected from a combination of criteria, including ease of identification and being a key indicator of shifts in the Mediterranean subtidal habitats. Before starting the data recording, each EcoDiver have to choose some of the 43 taxa included in the protocol as search targets, according to the expected habitat typology and personal motivations. This freedom of choice ensures greater attention and accuracy by the participants. The EcoDivers select species based on confidence (thereby reducing identification errors), personal interest (increasing satisfaction), and the number of species they feel able to handle (to reduce psychological stress during dive). However, this generates skewed distribution efforts among the taxa. The most-searched taxa are attractive and iconic species, such as the red coral *Corallium rubrum* and sea fans *Paramuricea clavata* and *Eunicella cavolini*. Less conspicuous but highly concerning species, such as invasive algae in the genus *Caulerpa*, are also frequently surveyed (Cerrano et al., 2017).

EcoDivers record the abundance (using numerical or descriptive classes according to the countability of organisms) and observed depth ranges of the searched taxa, along with the prevalent habitat type. Not encountered but actively searched taxa are recorded as absent. The diving sites are localised by global positioning system (GPS) receivers, nautical charts, or known points (e.g. mooring buoys at MPAs). Geographical coordinates (WGS84) are recorded with ± 6 arc-seconds (i.e. 185 m in latitude) accuracy, the usual distance range explored by EcoDivers.

Recorded observations, including absence, site name, geographic coordinates, date and time, underwater visibility, survey depth range (min and max), and observation effort in terms of time dedicated are uploaded to the online database through an internet form[2] or a dedicated app for Android smartphones ('Reef Check Med' app).

Recorded data are subjected to quality assurance and control (QA/QC) procedures, based on automatic filters (e.g. consistency among survey and observation depth ranges) and on manual checks (e.g. matching between the site name and geographic coordinates), and made freely available on a web-based GIS[3].

2.2. Species sensitivity assessment

The marine evidence-based sensitivity assessment (MarESA; Tyler-Walters et al., 2018) has been conducted for 25 taxa inhabiting the Mediterranean subtidal rocky bottoms, especially the coralligenous habitats (Ingrosso et al., 2018), and included in the RCMed U-CEM protocol (Supporting Information S1: Table S1.1). The species assessment is based on evidence from a literature review, complemented by expert judgement, for the possible effects of physical, chemical, and biological pressures listed in the MSFD Annex III (Supporting Information S1: Table S1.2). For each taxon and pressure, resistance (none, low, medium, high, or not relevant) and resilience ranks (very low, low, medium, high, or not relevant) were assigned according to the MarESA standard benchmarks. The quality and applicability of the evidences were also assessed according to the MarESA principles. The species

[2] https://www.reefcheckmed.org/english/underwater-monitoring-protocol/upload-your-data/

[3] https://www.reefcheckmed.org/english/underwater-monitoring-protocol/webgis-map/

Table 1
Abundance classes and their converted scores (*Sc*).

Numerical class	Descriptive class	*Sc*
0	absent	0
1	isolated specimen	1
2	some scattered specimens	2
3–5	several scattered specimens	3
6–10	a crowded area	4
11–50	some crowded areas	5
>50	several crowded areas	6

sensitivity ranks (not sensitive, low, medium, high) to each pressure were established by combining the resistance and resilience ranks using the MarESA combination table. Species sensitivity ranks were converted to numerical scores (0–3), and the mean sensitivity values toward physical (MSV_{phy}), chemical (MSV_{chem}), and biological (MSV_{bio}) pressures and the overall mean (MSV_{tot}) were calculated.

2.3. Territorial units and time frames

The RCMed U-CEM data are unevenly distributed across space and time because of the preferences and behaviour of the volunteers. To reduce conscious and unconscious bias, the data from several EcoDivers within a defined territorial unit (TU) and time frame (TF) were pooled and analysed together. TUs and TFs should be designed according to the aims of the monitoring and management purposes. For instance, TUs may be the cells of a regular grid over the area of interest, a set of management and monitoring zones within MPAs, or the areas surrounding single dive sites. The minimum TU size depends by the exploration ability of the divers and the positioning accuracy they can achieve (Meidinger et al., 2013). Therefore, the recommended minimum TU size is 0.08 km^2 (e.g. within a 6 arc-second radius). TF may span several months or multiple years, depending on the intensity and scale of the monitoring program.

2.4. RCMed species sensitivity (MedSens) index

The *MedSens* index provides the mean sensitivity of the species assemblages recorded by EcoDivers within a TU and TF. It can be calculated for the physical ($MedSens_{phy}$), chemical ($MedSens_{che}$), biological ($MedSens_{bio}$), and overall pressures ($MedSens_{tot}$) on the species, based on the corresponding mean sensitivity values (*MSV*), weighted for the abundance classes of the taxa. For each observation, the abundance class was converted to an abundance score (*Sc*) of 0 to 6 (Table 1). The index is calculated as:

$$MedSens_x = \Sigma(Sc_i \times MSV_{(x)i})/\Sigma Sc_i$$

where *x* is the chosen pressure typology (*phy*, *che*, *bio*, or *tot*), and $MSV_{(x)i}$ refers to the taxon in the *i*th observation having an abundance score Sc_i in the selected TU and TF. The minimum requirements for the index calculation are: TU size ≥ 0.08 km^2, EcoDivers ≥ 3, number of observations (including absences) ≥ 20, and searched taxa ≥ 10. The index values increase with increasing sensitivity means of the species recorded and, to a lesser extent, with their abundance.

2.5. MedSens index classification

The distribution of values assumed by the index was explored by applying the formula through a 15 arc-second grid (i.e. 1/4 of a nautical mile in latitude) covering the coasts of the Mediterranean Sea and the entire time frame of the available data (2001–2019; last access May 18, 2019). The index values distributions ($MedSens_{phy}$, $MedSens_{che}$, $MedSens_{bio}$, and $MedSens_{tot}$) were compared for homogeneity of variances and differences in the means using Bartlett's test and the analysis of variance (ANOVA), respectively (in both cases, α = 0.05). The index

Table 2
Mean sensitivity values of the physical (MSV_{phy}), chemical (MSV_{che}), and biological (MSV_{bio}) pressures, and the overall mean (MSV_{tot}) of the selected taxa.

Taxa	*MSVphy*	*MSVche*	*MSVbio*	*MSVtot*
Caulerpa cylindracea	0.643	0.571	0.333	0.583
Caulerpa taxifolia	0.643	0.571	0.333	0.583
Axinella spp.	1.231	0.714	1.333	1.087
Aplysina spp.	1.538	0.714	1.333	1.261
Geodia cydonium	1.769	1.571	1.667	1.696
Corallium rubrum	2.308	2.333	3.000	2.409
Paramuricea clavata	2.462	2.667	2.750	2.565
Eunicella cavolini	2.462	2.500	2.750	2.522
Eunicella singularis	2.231	2.500	2.500	2.348
Eunicella verrucosa	1.692	2.333	2.750	2.043
Parazoanthus axinellae	1.769	1.833	0.667	1.636
Savalia savaglia	2.385	2.000	2.000	2.217
Cladocora caespitosa	2.154	2.500	2.333	2.273
Astroides calycularis	1.769	2.500	1.000	1.826
Balanophyllia europaea	1.769	2.333	1.333	1.864
Leptopsammia pruvoti	1.692	2.000	1.000	1.682
Pinna nobilis	1.923	1.500	2.750	1.957
Arca noae	1.308	2.167	2.250	1.696
Palinurus elephas	1.214	1.857	2.500	1.600
Homarus gammarus	1.214	1.857	2.750	1.640
Scyllarides latus	1.231	1.857	2.500	1.625
Paracentrotus lividus	1.462	1.429	2.250	1.583
Hippocampus spp.	1.933	1.143	2.250	1.769
Diplodus spp.	1.133	0.714	2.250	1.192
Sciaena umbra	1.267	1.286	2.000	1.385

values were classified into 5 classes, from very low to very high sensitivity, via quintiles.

2.6. MedSens index calculator

To facilitate the application of the *MedSens* index, a plugin for QGIS was developed in Python language and made freely available in the QGIS plugin repository (also linked at the '*MedSens* index' web page[4]). The plugin requires two input datasets in shapefile format (ESRI, 1998), one containing a subset of the data collected using the RCMed U-CEM protocol with the abundances of the 25 assessed taxa (i.e. the open access 'MedSens data' shapefile; Ponti et al., 2020), and a second with polygons representing the TUs of interest. The polygons shapefile may be any file containing one or more enclosed areas ≥ 0.08 km^2. The desired TF can be defined as the starting and ending dates. The output is a new polygons shapefile reporting in the attribute table the values of $MedSens_{phy}$, $MedSens_{che}$, $MedSens_{bio}$, $MedSens_{tot}$, observers, observations, searched taxa and area (km^2) for each assessed area. Colour legends are also provided.

2.7. Case studies

As case studies, the *MedSens* index was calculated for the Mediterranean MPAs reported in the World Database on Protected Areas (WDPA) from UNEP-WCMC and IUCN (2019), and the management and monitoring zones within Italian MPAs, wherever sufficient *MedSens* data were available in the time frame 2001 – 2019. In particular, Italian MPAs are usually organised into management zones with different levels of protection enforcement, as indicated in their management plans and coast guard directives. With some exceptions, A zones (no-entry/no-take areas) allow only scientific activities, B zones (partial protection) allow recreational dives under some circumstances (e.g. a limited number of participants, only guided tours), and C zones (buffer zones) allow dives with no restrictions (Villa et al., 2002).

Used testing polygons shapefiles and their resulting *MedSens*

[4] https://www.reefcheckmed.org/english/underwater-monitoring-protocol/medsens-index/

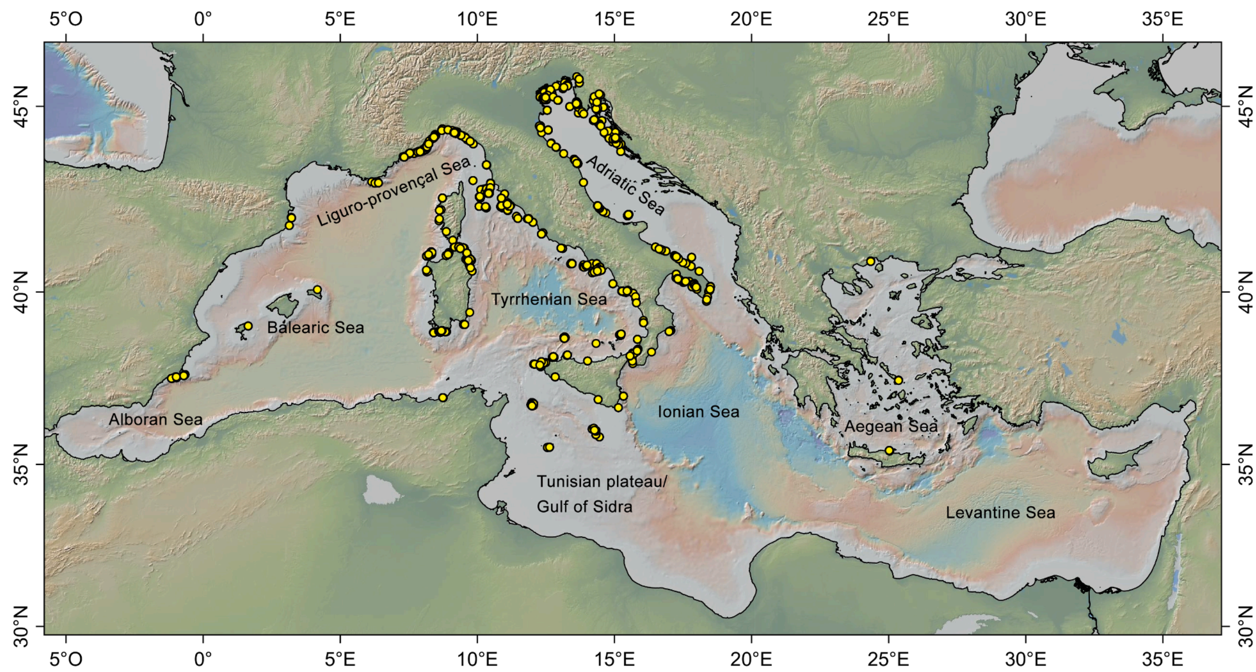

Fig. 1. Geographical distribution of the *MedSens* data points (Ponti et al., 2020). Map is in Mercator projection, datum WGS84.

Table 3
MedSens index classification of the physical, chemical, biological, and overall pressures.

Mean sensitivity	$MedSens_{phy}$	$MedSens_{che}$	$MedSens_{bio}$	$MedSens_{tot}$
Very low	≤1.5106	≤1.4381	≤1.5554	≤1.5305
Low	≤1.6275	≤1.6342	≤1.7908	≤1.6432
Moderate	≤1.7206	≤1.7806	≤1.9168	≤1.7431
High	≤1.8456	≤1.9621	≤2.0594	≤1.8921
Very high	>1.8456	>1.9621	>2.0594	>1.8921

classifications are available at the '*MedSens* index' web page[4].

Possible correlations among *MedSens* index, calculated for different pressure typologies, number of observations, observers, taxa considered, and the size of the investigated areas were analysed by the Pearson correlation coefficient (*r*). Differences of *r*-values from zero were tested with a *t* distribution ($\alpha = 0.05$).

3. Results

The details of the evidence-based sensitivity assessment (including references) for the 25 selected taxa are summarised in the Table S2 (Supporting Information S2). The mean sensitivity values are reported in Table 2.

The *MedSens* data shapefile used to classify the *MedSens* index contained 33,021 observations from 569 EcoDivers (Fig. 1; Ponti et al., 2020). The data came from the Croatian, French, Greek, Italian, Spanish, and Tunisian coasts. The *MedSens* index calculation for 15 arc-second grid cells along the Mediterranean coasts resulted in 137 TUs assessed. When calculated for the physical, chemical, biological, and overall pressures, the index value distributions significantly differed for the variances (Bartlett's test: $p = 1.475 \times 10^{-10}$) and means (ANOVA: $p = 2.347 \times 10^{-6}$). This suggests slightly different classification scales for the different pressure types; the 5-class scheme obtained by quintiles is reported in Table 3.

3.1. MedSens index applied to Mediterranean MPAs

In October 2019, WDPA reported 1504 MPAs (*sensu lato*) in the Mediterranean Sea. This included many coastal areas characterised by rocky bottoms, but also several wetlands, coastal lagoons, estuaries, and pelagic areas that are unsuitable for the *MedSens* index. Overall, 81 MPAs were assessed by the *MedSens* index, and the results are reported in Table S3.1 (Supporting Information S3). The assessed areas ranged from very small rocky outcrops (e.g. Scoglio dell'Argentarola, 0.15 km^2) to vast marine spaces (e.g. Tabarca-Cabo de Palos, 1262 km^2). The protected areas with the most sensitive species assemblages were located in the southern and central Tyrrhenian Sea (e.g. Isole Egadi, Scoglio dell'Argentarola, Isola di Ustica, Scilla, and Costa Viola) and Ligurian Sea (e.g. Punta Manara). The protected areas with the least sensitive species assemblages were characterised by artificial habitats, such as shipwrecks in Malta and the offshore platform wreck 'Paguro' in the northern Adriatic Sea, whose benthic assemblages are simplified compared to natural rocky bottoms (Ponti et al., 2002, 2015). Low to very low mean species sensitivities were also found at 'Tegnùe di Chioggia', a northern Adriatic no-take zone characterised by mesophotic coralligenous banks. These results are consistent with high anthropic disturbance in the area, including several dystrophic crises (Tomašových et al., 2017; Zuschin and Stachowitsch, 2009) and intense trawling (Melli et al., 2017; Ponti et al., 2011) that may limit the abundance of species sensitive to physical and chemical pressures.

The indices for the different pressures were correlated (Table S3.2, Supporting Information S3). However, there were instances where the classifications differed greatly, particularly between assessments of the chemical and biological pressures. The number of taxa considered was correlated to the number of observations and observers, and the number of observations was correlated to the number of observers, but these parameters did not correlate with the size of the area or the sensitivity of their assemblages.

3.2. MedSens index applied to Italian MPA management zones

The *MedSens* index was calculated for 22 management zones belonging to 12 Italian MPAs (Table S3.3, Supporting Information S3). The management zones with the most sensitive species assemblages were in the MPAs Isole Egadi, Tavolara – Punta Coda Cavallo, Isola di Ustica, Punta Campanella, and Portofino. Many A zones were not assessed due to the lack of data; the exceptions being the Cinque Terre,

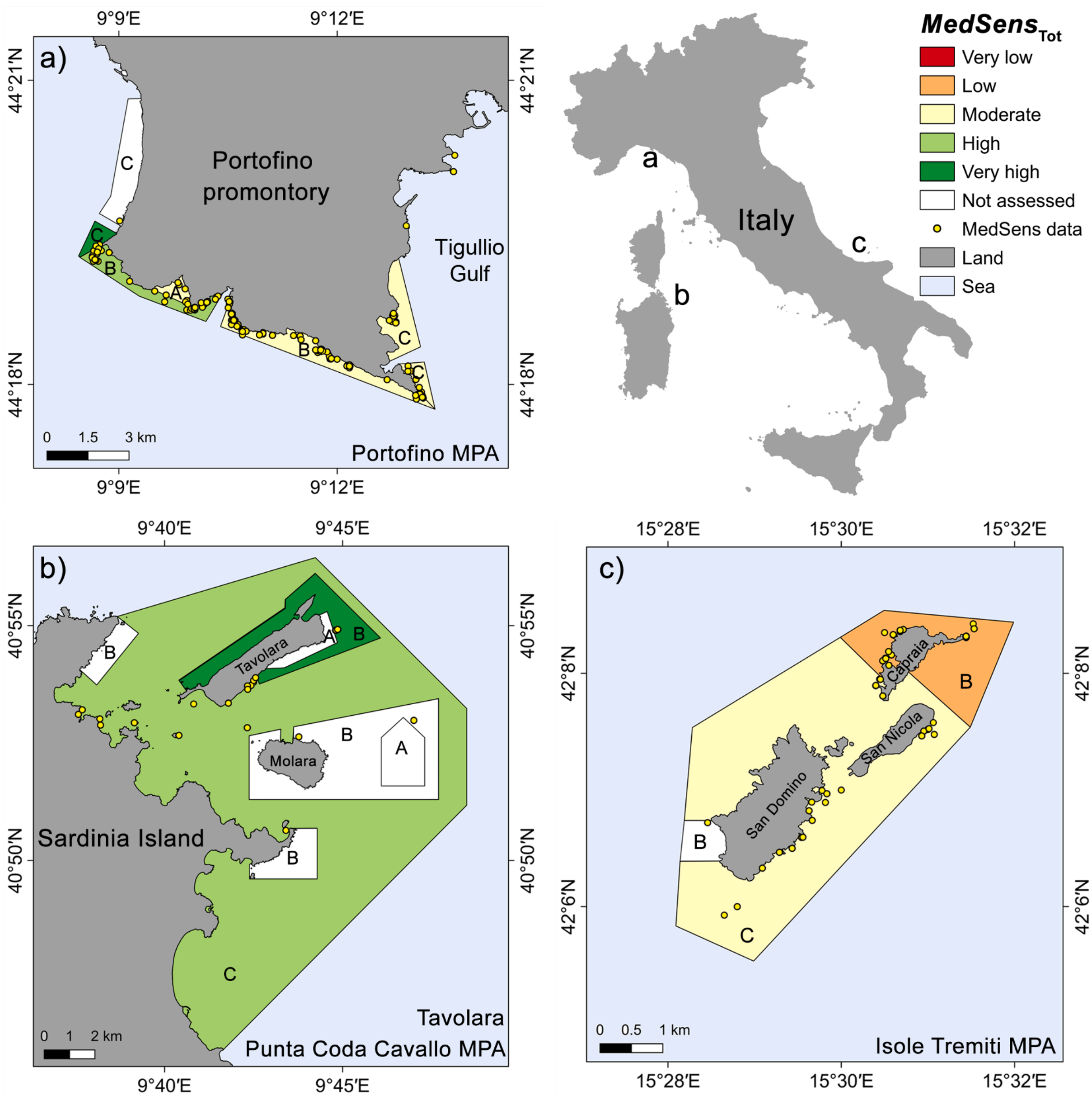

Fig. 2. Examples of sensitivity assessments (*MedSens*$_{tot}$ index) applied to MPAs management zones: a) Portofino, b) Tavolara – Punta Coda Cavallo, and c) Isole Tremiti. Yellow dots display *MedSens* data points. Letters indicate protection levels (Mercator projection, WGS84). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Miramare, Isole Egadi, and Portofino MPAs, where data were collected during authorised dives. However, the *MedSens* index did not detect very sensitive assemblages into these A zones. The A zones of Miramare and Isole Egadi are characterised by seagrass meadows and the *MedSens* index may not provide reliable results in these habitats. At Cinque Terre, the non-indigenous algae *Caulerpa cylindracea* has invaded (Bianchi et al., 2019; Montefalcone et al., 2015) and reduced the sensitivity of the assemblages, especially towards biological pressures.

At Portofino MPA (Ligurian Sea, Fig. 2a), the assemblage sensitivities ranged from moderate to very high. Overall, the mean species sensitivities increased in the west and consistently with distance from the Tigullio Gulf, located upstream of the dominant currents (Doglioli et al., 2004). This is the source of the main local physical and chemical pressures due to increasing urbanisation (Mangialajo et al., 2007) and the fluvial transport of sediments and pollutants (Mateos-Molina et al., 2015).

At Tavolara – Punta Coda Cavallo MPA (northern Tyrrhenian Sea, Fig. 2b), the assemblage sensitivities ranged from high to very high. This is consistent with limited anthropic impacts in a well-managed MPA (Bianchi et al., 2012). Pressure gradients cannot be uniquely defined in this area, but the B zone performed better than the C zone in terms of assemblage sensitivity, as expected from the management and conservation plan.

At the Isole Tremiti MPA (central Adriatic Sea, Fig. 2c), the assemblage sensitivities ranged from low to moderate. The B zone had the lowest mean species sensitivity, especially for biological pressures. This may be related to a decline in the algal assemblages due to increasing pollution (Cormaci and Furnari, 1999) and the growing number of non-indigenous species, such the invasive algae *Womersleyella setacea* (Cormaci et al., 2000) and *C. cylindracea* (Pierucci et al., 2019).

The *MedSens* index may allow even more detailed analysis. The Portofino MPA can be further subdivided into 19 monitoring zones, as designated by the MPA authority. The *MedSens* index revealed that some zones have less-sensitive assemblages than others, which may help to

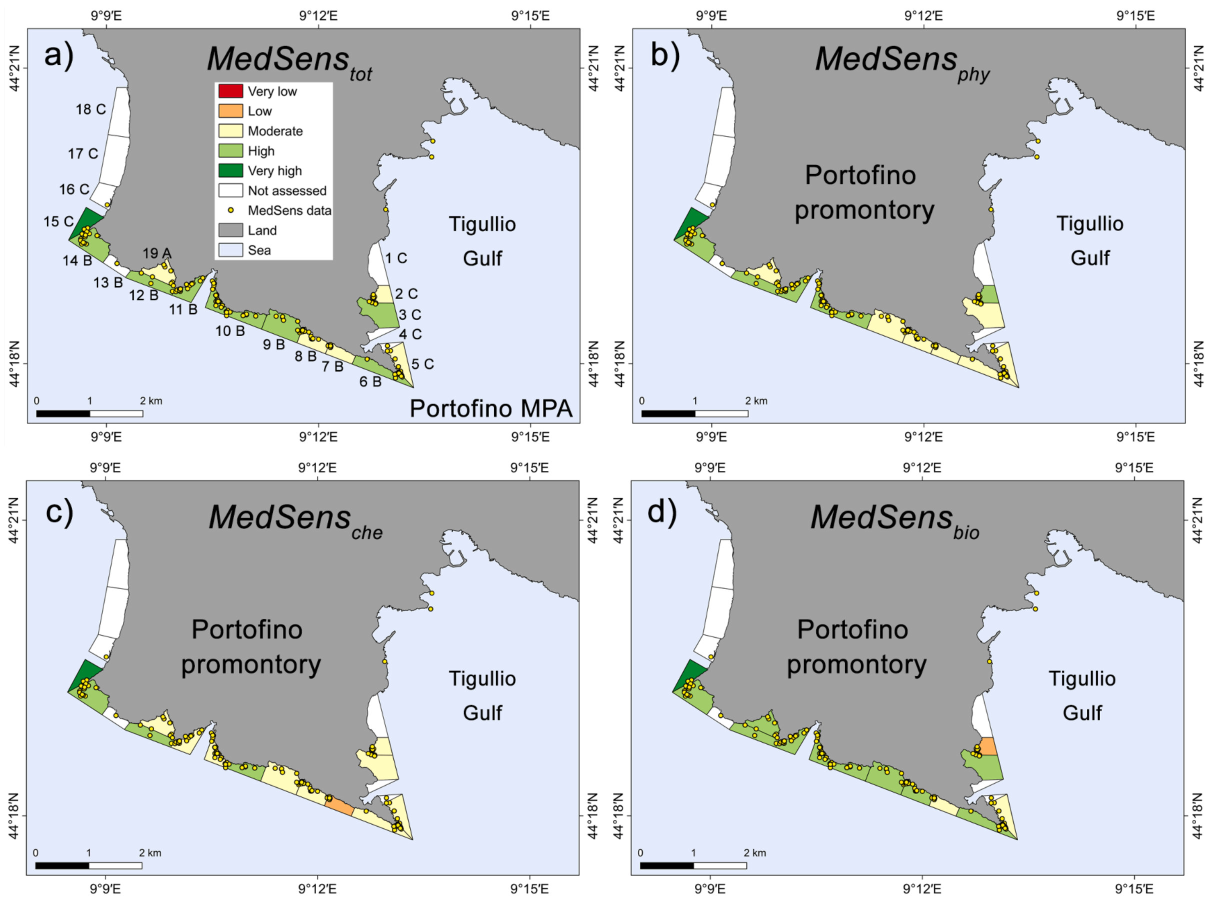

Fig. 3. Sensitivity assessments in the Portofino MPA monitoring zones (from 1 to 19) for the: a) overall assessment ($MedSens_{tot}$), b) physical pressures ($MedSens_{phy}$), c) chemical pressures ($MedSens_{che}$), and d) biological pressures ($MedSens_{bio}$). Yellow dots display *MedSens* data points. Letters indicate protection levels (Mercator projection, WGS84). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

identify local pressures and fine-tune the adaptive management actions (Fig. 3). The south-east side (zones 3–9), in particular, appeared less sensitive to physical disturbances (Fig. 3b), possibly resulting from exposure to sedimentation and water turbidity from the Tigullio Gulf (Mateos-Molina et al., 2015). These zones are also most affected by the mass mortality of gorgonians and other organisms that have frequently occurred since 1999 (Cerrano et al., 2000) and by recreational and artisanal fishing activities (Markantonatou et al., 2014). The results from zones 7 and 2 suggest that the management authorities should develop tailored strategies for their species assemblages that are less sensitive to chemical and biological pressures, respectively (Fig. 3c and d).

3.3. Changes in the MedSens index over time

The *MedSens* index can be calculated for specific time frames (TF). As an example, changes in the mean assemblage sensitivity at Gallinara Island (Ligurian Sea; 44° 1.400′ N 8° 13.700′ E) were analysed annually from 2006 to 2018 (except 2011 due to a lack of data). The assemblages showed very low to moderate sensitivity along the whole study period (Fig. 4a). This result is consistent with the impoverishment of benthic assemblages that occurred after increases in human disturbance since the 1990s and the failure to establish a planned MPA (Bianchi et al., 2018). The mean sensitivity to biological pressures was very low due to the invasion of *C. cylindracea* in 2005 (Cerrano et al., 2017). In 2016, there was an increase in species mean sensitivity, especially to the chemical and physical pressures. However, in the following two years, there was a new decline, likely due to the heatwaves of 2017 and 2018 (Garrabou et al., 2019).

Another case study is represented by the mass mortality of the gorgonian *Paramuricea clavata* at Secca del Papa, Tavolara Island (northern Tyrrhenian Sea; 40° 54.910′ N 9° 44.840′ E) in the late summer 2008 heatwave (Huete-Stauffer et al., 2011). Data collected in 2007, before the crisis, showed a very high mean sensitivity of the assemblages, especially to the chemical and physical pressures (Fig. 4b). Data collected between 2015 and 2017 (after the crisis) indicated a drastic reduction in the sensitivity of the assemblages. Indeed, the loss of *P. clavata* may affect the structure of benthic communities (Ponti et al., 2014, 2018). However, the sensitivity to biological pressures was consistently moderate before and after the crisis.

4. Discussion

4.1. The success of MedSens

The United Nations Decade of Ocean Science for Sustainable Development Goals 2021–2030 (SDG 14, Life Below Water) asks for an urgent improvement of the capacity of marine conservation actions worldwide, and MCS is a promising and powerful tool to enhance engagement in marine conservation worldwide. Following the ten principles of the Citizen Science (Kelly et al., 2020), the RCMed U-CEM open access dataset allows for various uses, e.g. to complement scientific papers on species distribution and abundance, aid distribution modelling, and compare historical series (Lucrezi et al., 2018 and references therein). The *MedSens* index, being based on this dataset, represents a bridge between MCS and coastal management in the Mediterranean Sea,

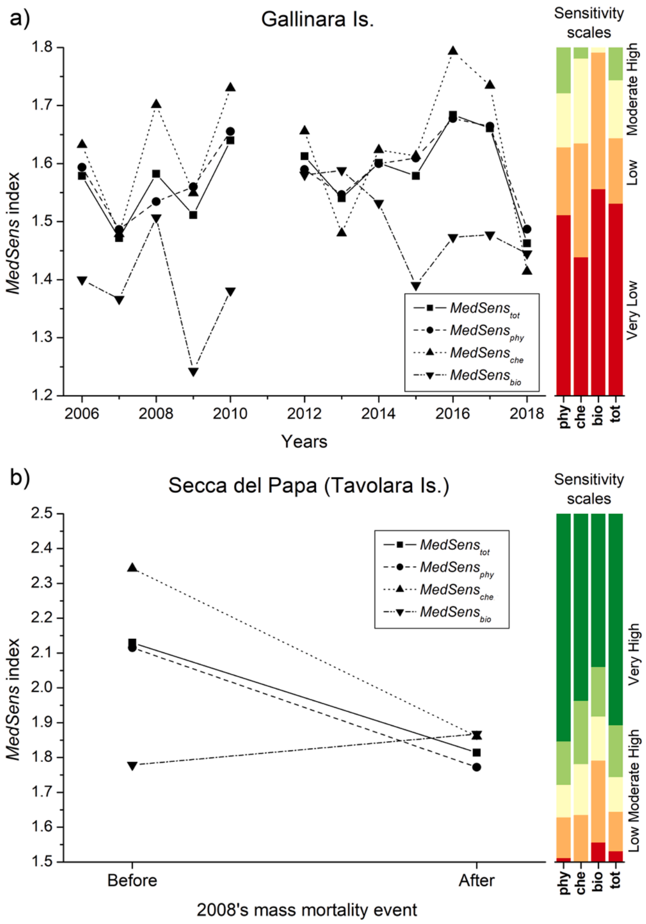

Fig. 4. Temporal change in the *MedSens* index at: a) Gallinara Island from 2006 to 2018 and b) Secca del Papa, Tavolara Island before and after the 2008 mass mortality of *Paramuricea clavata.*

allowing the effective integration of a consolidated community-based environmental monitoring into ecosystem-based management policies. It provides a proxy of the mean sensitivity of the rocky bottom assemblages to natural and anthropic pressures listed by MSFD. Higher average assemblage sensitivities are associated with lower levels of disturbance, thereby indicating good environmental conditions.

The *MedSens* index was calibrated on a large dataset of wide-ranging conditions occurring along the Mediterranean Sea coasts. Case studies showed that the index responds well to the local pressures documented by previous studies. The *MedSens* index may also be applied in a wide range of circumstances; it is particularly suitable for monitoring MPAs and can aid spatial gradients analysis, time series analysis, and before/after-control/impact studies. Moreover, the newly developed QGIS plugin provides an easy freeware tool to calculate the index whenever data are available.

MedSens is a biotic index based on the sensitivities and tolerances of the species to pollution and/or other disturbance sources (for a review see Ponti et al., 2009). Other indices based on a similar approach for the Mediterranean benthic communities include the AZTI' Marine Biotic Index, AMBI (Borja et al., 2000), for soft bottoms, and the Ecological Status of Coralligenous Assemblages index, ESCA (Piazzi et al., 2017), for rocky bottoms. These indices are based on the assumption that sensitive species decrease in abundance and number as the pressures increase, leaving space for the more tolerant species (Hilsenhoff, 1987). While the high abundance of a sensitive species is likely witness of reduced pressures, the high abundance of tolerant species is not necessarily related to poor environmental conditions – this should be considered when interpreting the results. The main strengths of the *MedSens* index are that the sensitivities of the selected species in a wide range of taxonomic groups and biological and ecological features are based on scientific evidence and that these sensitivities were assessed according to the different pressure types. This can help discriminate against local pressures that are likely to act in an area. Conversely, the main weaknesses lie in the reduced number of considered species, which could be increased in the future, and the need for large amounts of data from many well-trained volunteers.

The success of a CS project stems from simple and effective protocols (Bonney et al., 2009; Holt et al., 2013), developed by scientists to include particular aims, proper training and skills assessment of the participants, and timely feedbacks on the progress and efficacy of the participants' actions to keep high their involvement (Devictor et al., 2010). The RCMed U-CEM protocol is a simple but effective visual census, with easy-to-monitor species that encompass the key ecological aspects of the Mediterranean subtidal habitats (Cerrano et al., 2017). This protocol is easy to learn and may provide a large amount of timely, up-to-date geo-referred data, from the Mediterranean Sea coasts. Data quality is assured by rigours participant training (subject to learning tests), numerous surveys by independent observers, and quality control measures.

4.2. Future perspectives

The population of European divers is over 3 million people (data from the European Underwater Federation[5]), many of whom dive in the Mediterranean Sea. The Mediterranean Sea has about 23,000 km of rocky coasts (Furlani et al., 2014) and more than 7000 km^2 of subtidal rocks and biogenic reefs in the scuba diving depth range (EMODnet broad-scale seabed habitat map for Europe, v2019[6]). With the *MedSens* index, volunteers applying the RCMed U-CEM protocol can support researchers and managers to collect and interpret data over larger spatial and temporal scales than would otherwise be possible.

The *MedSens* index provides a free, complementary to professional investigations, and user-friendly tool to evaluate the ecological quality of the Mediterranean subtidal rocky habitats according to the Habitat Directive and the MSFD requirements. This will also help decision-makers as they plan and apply conservation strategies. The *MedSens* index offers a detailed picture of the vulnerability levels of different coasts, allowing tailored measures of conservation in an adaptive management framework. Moreover, this index can enable more opportunities for effective feedback to volunteers involved in the RCMed U-CEM protocol. The *MedSens* index application may represent a way to raise public awareness and enhance the collaboration between coastal management authorities, stakeholders, and researchers. By directly involving stakeholders, the *MedSens* index increases the acceptability of management decisions, including unpopular ones, as they may occur in MPAs where fragile sites and restoration areas are closed to the public.

The RCMed U-CEM protocol and *MedSens* index may also complement ocean observation systems and oceanographic forecast models, helping to develop an early-warning system for mass mortality events in benthic species along the Mediterranean Sea coasts (Turicchia et al., 2018). Thus, their combined application provides an effective strategy to achieve the habitat and species conservation objectives set by the European Union (Borja et al., 2010) and the Mediterranean Regional Activity Centre for Specially Protected Areas (UNEP-MAP-SPA/RAC, 2017).

The *MedSens* index was designed for the Mediterranean subtidal rocky bottoms, but its approach may be applied to other habitats, from temperate to tropical reefs, by including the relevant local species, with appropriate calibration and validation.

[5] https://www.euf.eu
[6] https://www.emodnet-seabedhabitats.eu

5. Authors' contributions

MP conceived the ideas and designed the methodology; ET conducted the evidence-based species sensitivity assessments; MG developed the *MedSens* plugin; ET and MP analysed the data; ET and MP wrote the manuscript. All authors contributed to earlier drafts and approved the final version.

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

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2020.107296.

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