UNIVERSITAT DE BARCELONA

Ecological restoration of cold-water corals on the Mediterranean continental shelf

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ECOLOGICAL RESTORATION OF COLD-WATER CORALS ON THE MEDITERRANEAN CONTINENTAL SHELF

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Facultat de Biologia, Departament de Biologia Evolutiva, Ecologia i Ciències Ambientals.

Programa de Doctorat d'Ecologia, Ciències Ambientals i Fisiologia Vegetal.

Ecological restoration of cold-water corals on the Mediterranean continental shelf.

Restauració ecològica de coralls d'aigua freda a la plataforma continental del Mediterrani.

Memòria presentada per Maria Montseny Cuscó per optar al Grau de Doctora per la Universitat de Barcelona

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A tots els que m'heu acompanyat

en aquesta etapa

“ La utopía está en el horizonte.

Camino dos pasos, ella se aleja dos pasos.

Camino diez y el horizonte se corre diez pasos más allá

¿Entonces para que sirve la utopia?

Para eso, sirve para caminar. “

(EDUARDO GALEANO)

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ADVISOR'S REPORT

Dr. Andrea Gori professor at Departament de Biologia Evolutiva, Ecologia i Ciències Ambientals (Universitat de Barcelona) and Dr.Josep-Maria Gili professor at Institut de Ciències del Mar (ICM-CSIC)**,** advisors of the PhD thesis entitled "**Ecological restoration of cold-water corals on the Mediterranean continental shelf**".

INFORM, that the research studies developed by Maria Montseny Cuscó for her Doctoral Thesis have been organized in four chapters, which correspond to four scientific papers listed below: two of them are already published, one is in review, and the last one will be submitted in the next few weeks;

and CERTIFY, that the work has been carried out by Maria Montseny Cuscó, participating actively in all the tasks: conceiving and setting the objectives, conceiving and performing the analyses, participating actively in the field work, carrying out the experiments and writing the manuscripts.

Finally, we certify that the co-authors of the publications listed below and that conform this doctoral thesis, will not use these manuscripts in another PhD thesis.

Barcelona, 15th September 2020

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SUMMARY

Cold-water coral (CWC) habitats dwell on continental shelves, slopes, canyons, seamounts, and ridge systems around the world's oceans, from 50 m to depths up to 4000 m. CWC species provide heterogeneous habitats supporting a myriad of associated fauna and form highly diverse CWC reefs and CWC gardens. Main threats, currently impacting CWC ecosystems come from anthropogenic stressors, such as fishing activities, oil and gas exploitation and the incipient mining activity. Likewise, climate change, causing changes in the water column, is also affecting these ecosystems. Life-history traits of CWC species (long lifespans, slow growth and limited recruitment) make them very vulnerable to current and potential threats. Given their limited recovery capacity, interest to preserve and restore CWC ecosystem is steadily growing. The creation of Marine Protected Areas and active ecological restoration actions are nowadays the best management tools to conserve native ecosystems and represents an opportunity to revert the anthropogenic damage that has already taken place. Through passive (natural regeneration after the cessation of stressors) and active (human interacts with biotic and/or abiotic ecosystem features) approaches, restoration activities seek to accelerate the recovery of ecosystem structure and functioning relative to a reference model. Contrarily to terrestrial and shallow-water marine ecosystems, ecological restoration in intermediate (50 – 200 m) and deep marine (> 200 m) environments has received lesser attention. To date, only few restoration actions at local scales have been carried out at those depths, mainly due to technical and economic limitations which questions its wide application. Scaling-up restoration actions and make them affordable are the main present challenges for CWC restoration. In this sense, in order to move forward towards the conservation of intermediate and deep-sea ecosystems, the general aim of the present thesis is to assess the impact of fishing activity on CWC gardens as well as to explore the feasibility of novel active ecological restoration techniques.

All the work performed during this thesis has been carried out at the Cap de Creus marine area (North-Western Mediterranean Sea), specifically at the continental shelf (60 – 130 m), where gorgonians, sponges, and sea pen species form CWC gardens supporting a variety of mobile associated fauna. The target species is the yellow gorgonian *Eunicella cavolini* (Koch, 1887) which dominate in the area forming density patches. In the first chapter, the impact of artisanal fishing was quantified to evaluate the threat of this activity on CWC gardens and to provide essential information to mitigate such impact. The rest of chapters (2, 3 and 4) evaluated, for the first time, the viability to actively restore degraded *E. cavolini* populations. Specifically, in the second chapter, gorgonians obtained from bycatch (accidentally caught of non-target species) of local artisanal fishers, were transplanted to artificial structures deployed on the continental shelf (805 m). This pilot study demonstrated, for the first time, the high survival of *E. cavolini* transplants. Following, and going one step forward, in the third chapter, field experiments and modeling approaches were combined to develop and technically validate an innovative large-scale and cost-effective restoration method for CWC gardens. Successful results evidenced the feasibility of recovering bycatch *E. cavolini* and returning them to their natural habitat with this novel method so-called "badminton method". Finally, in the fourth and last chapter, a large-scale restoration action of *E. cavolini* populations was carried out in collaboration with local artisanal fishers during two consecutive fishing seasons by applying the technique previously developed. A large number of gorgonians (460 colonies) were successfully reintroduced and survived at the end of the action (2 years) at 80-100 m depth. The results suggested an initial establishment of a new gorgonian population, which will potentially evolve toward a comparable natural population in terms of size and spatial structure, if natural recruitment also occurs. Moreover, an economic evaluation was performed, also confirming the cost efficiency of this method aimed at enhancing the recovery of impacted CWC gardens.

The lack of knowledge of some key ecological processes of CWC ecosystems as well as the technical limitations hinder a complete evaluation of restoration efforts performed. However, this thesis represents a promising improvement for the conservation and recovery of CWCs that could be extended to other areas and regions.

Els coralls d'aigua freda habiten en les plataformes continentals, talussos, canyons, muntanyes submarines i dorsals oceàniques d'arreu del món, des de 50 a 4000 metres de profunditat. Les espècies de coralls d'aigua freda creen hàbitats heterogenis que donen suport a una infinitat de fauna associada i formen esculls i boscos de coralls altament diversos. Les principals amenaces que actualment impacten aquests ecosistemes de coralls d'aigua freda son d'origen antròpic, com ara l'activitat pesquera, l'explotació de petroli i gas i l'incipient explotació minera. Així mateix, el canvi climàtic, el qual provoca canvis en la columna d'aigua, també està afectant aquests ecosistemes. Les característiques vitals dels coralls d'aigua freda (longevitat, creixement lent i reclutament limitat) els fan molt vulnerables a les amaces tan actuals com futures. Atesa la seva limitada capacitat de recuperació, l'interès per preservar i restaurar els ecosistemes de coralls d'aigua freda està en constant creixement. La creació d'àrees marines protegides i les accions restauració ecològica activa són avui en dia les millors eines de gestió per conservar ecosistemes autòctons i representen una oportunitat per revertir els danys antròpics que ja han tingut lloc. A través d'enfocaments passius (regeneració natural després del cessament del impacte) i actius (l'ésser humà interacciona amb les característiques biòtiques i/o abiòtiques de l'ecosistema), les activitats de restauració busquen accelerar la recuperació de l'estructura i funcionament dels ecosistemes en funció a un model de referència. Contràriament als ecosistemes terrestres i d'aigües someres, la restauració ecològica enfocada a ambients marins intermedis (~50–200 m) i profunds (> 200 m) ha rebut menor atenció. Fins ara, a aquestes profunditats només s'han dut a terme algunes poques accions de restauració a escala local, principalment a causa de limitacions tècniques i econòmiques que qüestionen la seva àmplia aplicació. Els principals reptes actuals per la restauració de coralls d'aigua freda són incrementar l'escala espacial de les accions de restauració i fer-les econòmicament més assequibles. En aquest sentit, per avançar en la conservació dels ecosistemes de fons intermedis i profunds, l'objectiu general de la present tesi és avaluar l'impacte de l'activitat pesquera sobre els boscos de coralls d'aigua freda i explorar la viabilitat de innovadores tècniques de restauració activa.

Tot el treball realitzat en aquesta tesi s'ha dut a terme a la zona marina del Cap de Creus (Nord-Oest del mar Mediterrani), contretament a la plataforma continental (60 – 130 m) on espècies de gorgònies, esponges i plomalls formen boscos de coralls d'aigua freda donant suport a una gran varietat de fauna mòbil associada. L'espècie objectiu és la gorgònia groga *Eunicella cavolini* (Koch, 1887) la qual domina a la zona formant denses agregacions. En el primer capítol, es va quantificar el impacte de la pesca artesanal per avaluar l'amenaça d'aquesta activitat sobre els boscos de coralls d'aigua freda i proporcionar informació essencial per mitigar aquest impacte. La resta de capítols (2, 3 i 4) van avaluar, per primera vegada, viabilitat de restaurar activament les poblacions degradades de *E.cavolini*. Especificament, en el segon capítol és van trasplantar les

gorgònies capturades accidentalment pels pescadors artesanals de la zona, a estructures artificials fondejades a la plataforma continental (85 m). Aquest estudi pilot va demostrar per primera vegada l'alta supervivència dels transplantaments d'*E.cavolini*. Seguidament, i anant un pas més enllà, al tercer capítol es van combinar experiments de camp i modelització per desenvolupar i validar tècnicament una nova tècnica de restauració per als boscos de coralls d'aigua freda, a gran escala i econòmicament assequible. Els exitosos resultats van evidenciar la viabilitat de recuperar les colònies de *E.cavolini* capturades accidentalment i retornar-les al seu hàbitat natural amb aquesta innovadora tècnica , anomenada "el mètode bàdminton". Finalment, al quart i últim capítol, es va dur a terme una acció de restauració de les poblacions de *E.cavolini* a gran escala, amb la col·laboració de pescadors artesanals de la zona i al llarg de dues temporades de pesca consecutives aplicant la tècnica desenvolupada anteriorment. Un gran nombre de gorgònies (460 colònies) van ser re-introduïdes amb èxit i van sobreviure al final de l'acció (2 anys) a 80-100 m de profunditat. Els resultats van suggerir l'establiment inicial d'una nova població gorgònies, que potencialment evolucionarà cap a una població natural comparable en termes d'estructura de talles i estructura espacial, sempre i quan es produeix també un reclutament natural. D'altra banda, es va fer una avaluació econòmica, que va confirmar la rendibilitat d'aquest mètode dirigit a millorar la recuperació dels boscos de coralls d'aigua freda impactats.

El desconeixement d'alguns processos ecològics claus en els ecosistemes de coralls d'aigua freda, així com les limitacions tècniques, dificulten una avaluació completa dels esforços de restauració realitzats. Tanmateix, aquesta tesi suposa una millora prometedora per a la conservació i recuperació dels coralls d'aigua feda, que es podria estendre a altres zones i regions.

GENERAL INTRODUCTION

1 COLD-WATER CORAL ECOSYSTEMS

In the last decades, the increasing interest towards intermediate and deep-water ecosystems and the improvement in underwater exploration technology have led to expand knowledge about benthic communities dwelling on continental shelves (~60 – 200 m depth) and deep-sea bottoms (>200 m depth) (Freiwald et al. 2004; Althaus et al. 2009). At depths of 50 to 4000 m depth, cold-water corals (CWC) are key habitat-forming species, generating complex three-dimensional structures that create hotspots of biodiversity over large areas (Roberts et al. 2009).

CWCs are cnidarians encompassing species from Scleractinia, Octocorallia, Antipatharia and Stylasteridae (ICES and Hall-Spencer 2007; Roberts et al. 2009). Despite the high prevalence of tropical coral reefs at shallow depths, most coral species are found in the world's aphotic zone, in cold waters (Roberts et al. 2009; Bergmark and Jorgensen 2014) on continental shelves and slopes, seamounts, canyons and ridge systems (50 – 4000 m depth) (Freiwald and Roberts 2005; Roberts et al. 2006; Cordes et al. 2016a; Angiolillo and Canese 2018). Under suitable conditions (e.g., appropriate substrate, water current and food supply, there are six species of reef framework-forming CWCs (Scleractinia) (Fig. 1) while many other CWC species form extensive coral gardens (Octocorallia, Antipatharia and Stylasteridae) often mixed with solitary and/or reef framework-forming corals (Fig. 2). CWCs ecosystems increase habitat heterogeneity and support enhanced biological diversity and ecosystem functioning (Roberts et al. 2006, 2009; Cordes et al. 2008; Armstrong et al. 2014). The structures they create can alter current flow, food availability, and sediment resuspension, providing niches, shelter and nursery grounds for an abundant and diverse associated fauna including economically valuable species (e.g., sponges, ascidians, polychaetes, crustaceans, fish species…) (Costello et al. 2005; Henry and Roberts 2007). CWC ecosystems support unique species, providing benefits to adjacent fisheries through the spill-over effect of eggs, larvae, juveniles and adults (D'Onghia et al. 2010; Corbera et al. 2019).

CWCs are mainly sustained by feeding on particulate organic matter and zooplankton, which they capture from the water (Kiriakoulakis et al. 2005; Dodds et al. 2009; Mueller et al. 2014), although more recently, chemosynthesis has been shown to provide a source of energy in addition (Middelburg et al. 2015). Significantly higher levels of biogeochemical cycling, respiration and benthic-pelagic coupling processes take place in CWC ecosystems in comparison to the surrounding seafloor, enhancing the ecosystem functioning of the deep-sea biome (Wild et al. 2009; Cathalot et al. 2015; Rovelli et al. 2015).

CWC species are slow-growing, with potentially long lifespans (Andrews et al. 2002; Roark et al. 2009; De Moura Neves 2016), delayed sexual maturity and limited recruitment success (Roberts et al. 2009; Watling et al. 2011; Girard et al. 2016); thus they are expected to recover slowly after any disturbance. Two separate studies have shown that over a period of 8 to 10 years following the cessation of fishing activities with bottom-contact gears there was no natural recovery of impacted CWC communities, even at depths shallower than 1000 m (Williams et al. 2010; Huvenne et al. 2016). Likewise, CWCs heavily impacted by the Deepwater Horizon oil spill (Gulf of Mexico, April 2010) could take up to three decades to visibly recover, and likely hundreds of years to grow back to their original size (Girard and Fisher 2018). According to their fragility, structural complexity, functional significance, and low recovery capability from impacts, most CWC ecosystems are recognized as Vulnerable Marine Ecosystems (VME) (FAO 2009) and listed in the OSPAR List of Threatened and/or Declining Species and Habitats (Hall-Spencer and Stehfest 2009, OSPAR 2010). Thus, their conservation is internationally recognized as a high priority for the maintenance of marine biodiversity and the ecosystem services provided (Thurber et al. 2014; Cordes et al. 2016a).

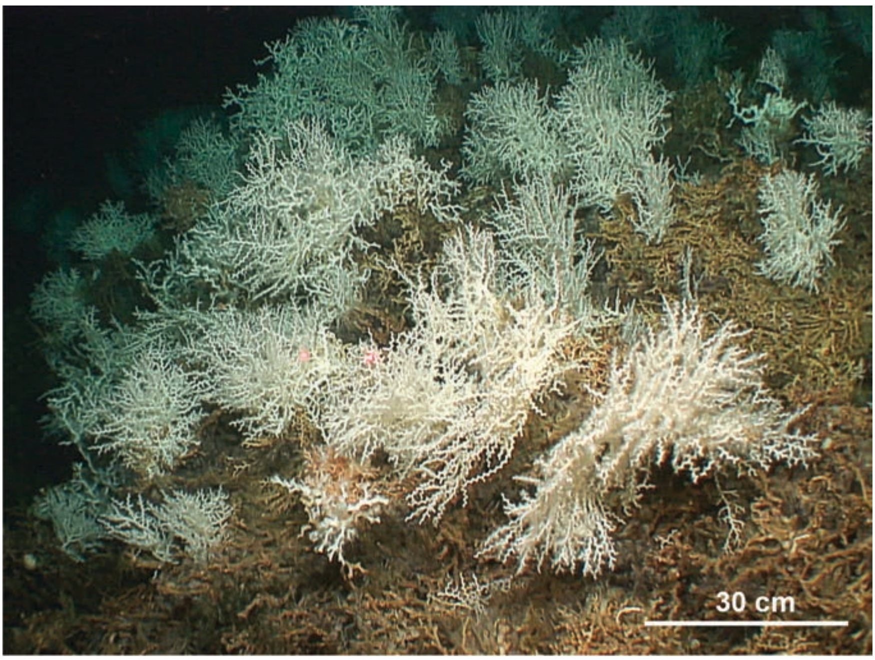

Figure 1. Living CWC reef of *Madrepora ocualta* found in the northern Cabliers Province (Mediterranean Sea). **Source:** Corbera et al. 2019.

1.1 Drivers of degradation of CWC ecosystems

Different antrophogenic pressures can have direct and synergic effects on intermediate and deep-sea habitats and fauna (Fig. 3). The main threats for CWC ecosystems come from fishing, hydrocarbons exploitation and the incipient mining activity, which have direct contact with the seabed causing an immediate impact on CWC communities (Ragnarsson et al. 2017). Likewise, climate change is prone to cause changes in the water column (e.g., increase in water temperature, ocean acidification, deoxygenation, reduced primary productivity), which cause additional stress and will, within the next century, significantly reduce suitable habitat for CWCs (Sweetman et al. 2017; Morato et al. 2020; Puerta et al. 2020).

Figure 2. CWC garden observed in the Irish continental shelf (North Atlantic Ocean). **Source:** ICES 2019.

1.1.1 Fishing impacts

Fishing is one of the most extensive activities that interacts with the seabed and has gradually encroached deep waters (Roberts 2002; Watson et al. 2006; Gross 2015) due to exhaustion of coastal and shelf fish stocks and the improvement in fishing technologies (Pusceddu et al. 2014; Paradis et al. 2018). Worldwide fishing impacts down to 1000 m depth are widespread, with almost all shelf and slopes areas being fished to some extent

(Puig et al. 2012; Pham et al. 2014). In the Porcupine Seabight (North Atlantic), bottom trawling fishing is estimated to have directly impacted ca. 52,000 km^2 at 500 – 1500 m depth, with potential (indirect) impacts extending over 142,000 km^2 (Priede et al. 2011). In European seas, the area swept by trawls at 200 – 1000 m depth ranges from 2% to 77% of the seabed in EU fishery management areas (Eigaard et al. 2017). Whilst the majority of trawling occurs on soft flat sediments, impacted areas also include CWC habitats (Hall-Spencer et al. 2002; Boavida et al. 2016; Huvenne et al. 2016; Buhl-Mortensen and Buhl-Mortensen 2018), given their role as important nursery grounds, for the surrounding fisheries (Baillon et al. 2012).

In general, the impacts from bottom trawling in the deep sea and the sensitive nature of most CWC ecosystems are now quite well known. Bottom trawling can impact the seabed and benthic organisms in different ways including scraping, ploughing and inducing hydrostatic shock (Rijnsdorp et al. 2016). These gears cause direct overall habitat degradation (bulldozing, contact/modification, removal, death and damage to the fauna (Pusceddu et al. 2014; Rijnsdorp et al. 2016) as well as indirect effects including the resuspension of sediment plumes from the trawl track that can smother CWCs as they precipitate over the reefs (Ramirez-Llodra et al. 2011; Pusceddu et al. 2014). The historical scale of direct impacts from bottom trawling is at least an order of magnitude greater than all other impacts combined (Benn et al. 2010). Other bottom-contact fishing gears such as traps, longlines and trammel nets may also impact the seabed mostly through the damage and removal of habitat-forming "bio-engineering" corals, as they are easily entangled in nets, overall contributing to the degradation of CWC communities (Wareham and Edinger 2007; Durán Muñoz et al. 2011; Sampaio et al. 2012). However, it has been estimated that a single deep-sea bottom trawl in a CWC area has the same impact as 296–1,720 longlines, depending on the morphological complexity of the impacted coral species (Pham et al. 2014). Finally, impacts from the fishing industry also include those caused by lost fishing gears and fishing related litter in which corals can get entangled (Bo et al. 2014; Buhl-Mortensen and Buhl-Mortensen 2018).

1.1.2 Impacts from oil and gas industry

Dwindling hydrocarbon reserves in more accessible parts of the ocean have driven exploration and production from the continental shelf into deep (500–1500 m) and ultra-deep (>1500 m) waters (Cordes et al. 2016b). In North Atlantic including the Faroe Shetland Channel and the Rockall and Hatton basins, it is estimated that up to 50% of recoverable hydrocarbons in UK waters alone could come from deep basins (Gray 2013). More than 500 oil and gas platforms, linked to several thousand kilometers of pipelines, are currently present in EU waters below 200 m depth (European comission 2009). In the Gulf of Mexico, reserves deeper than 3000 m are being exploited (Cordes et al. 2016b).

Investment in deep-water drilling technologies is increasing in response to costly operational accidents, as shown by the recent Deepwater Horizon well blow-out in the deep Gulf of Mexico (Watts 2016). Environmental risks are likely to increase as oil and gas operations move into Areas Beyond National Jurisdiction (ABNJ) where the oil and gas industry is a relatively new user group and where the current regulatory framework is only now being formulated (Merrie et al. 2014). During standard operations in offshore oil and gas industries, large contaminated cuttings piles and sediment plumes can impact CWCs in a variety of ways (Cordes et al. 2016b). While adult *Lophelia pertusa* corals can generally withstand a degree of discharged cuttings without significant ecophysiological impacts (Larsson et al. 2013; Baussant et al. 2018) the dispersive larval phase of this CWCs is highly susceptible to mortality from sediment loads, particularly during early developmental phases (Järnegren et al. 2017). Moreover, accidental oil spills can profoundly affect CWCs, with impacts from molecular to the ecosystem scale (White et al. 2012; Fisher et al. 2014; DeLeo et al. 2016; Joye et al. 2016).

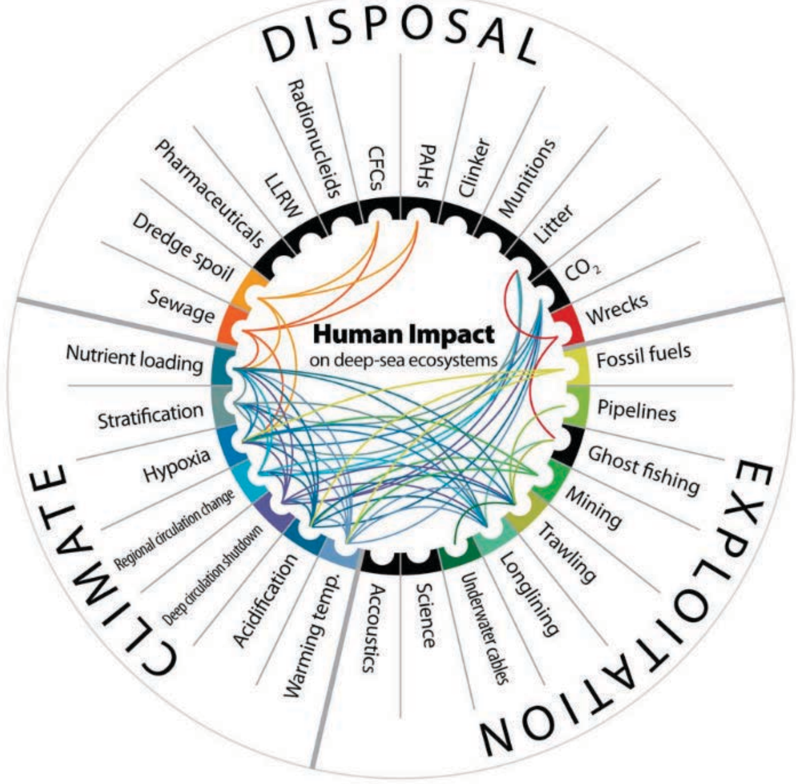

Figure 3. Synergies between human impacts on deep-sea habitats. **Source:** Ramirez-Llodra et al. 2011.

1.1.3 Deep-sea mining impacts

Minerals of commercial interest are found in various sites in the world including CWC habitats in the deep sea. There are three main types of deep-sea mineral resources: polymetallic sulphides, cobalt-rich ferromanganese crusts, and polymetallic nodules containing important metals and rare earth elements (Ragnarsson et al. 2017). Polymetallic sulphides are found on mid-ocean rocky ridges, including sedimentary and hydrothermal vent systems, while cobalt crusts are common on rocky seamounts (Baker and Beaudoin 2013a, b, c; Miller et al. 2018). Polymetallic nodules occur on abyssal plains, one of the most prominent examples being the Clarion-Clipperton Zone in the Pacific. CWCs occur in those habitats, therefore, although mining activity is still in its infancy, they will likely to be exposed to its impacts. The expected scale of physical impacts varies significantly: from a few km^2 in the case of sulphides, (comparable to mines on land), to 10s of km^2 in the case of crusts where the top layer (ca. 25 cm) of an entire guyot might be removed (Weaver and Billett 2019).

Impacts from mining will result in the removal of substrate, plumes of particulate material created by operations at the seabed, and discharge plumes of fine, and potentially toxic material, from the dewatering of the ore at the sea surface. All of these impacts can have deleterious effects on CWCs, as well as on pelagic ecosystems. Effects may occur on the seabed and/or in the water column affecting adult corals and larval dispersal. Some direct impacts will be lethal, but most will impair processes associated with feeding, growth and reproduction (Christiansen et al. 2019). In the case of sulphides, toxic metals may be released when the sulphides are grinded and exposed to oxygen. While the concentrations of the toxic metals may be low, they have the potential for long-term chronic effects through bioaccumulation. Sulphide resources are of interest within the Exclusive Economic Zones (EEZ) of several nations, such as Papua New Guinea (Coffey Natural Systems 2008), Japan (Japan Times 2017) and the South-West Pacific island states (Swaddling 2016), but most sulphide and crust minerals of interest are found in international waters. In the latter case, exploration for resources is managed by the International Seabed Authority (ISA) as the Common Heritage of Mankind. Regulations for the exploration of sulphides and crusts were enacted in 2010 and 2012, respectively (International Seabed Authority 2010, 2012). The ISA is now in the process of drafting new regulations for the exploitation of all deep-sea minerals (International Seabed Authority 2018) and around 1.3 million km^2 of international seabed has been already set aside for mineral exploration in the Pacific and Indian Oceans as well as on the Mid-Atlantic Ridge (Cuyvers et al. 2018).

1.1.4 Global change: ocean acidification and warming

Much of the anthropogenic carbon dioxide released to the atmosphere, from the burning of fossil fuels is absorbed by the world's oceans, altering its chemistry, specifically by decreasing water pH and the saturation state of calcium carbonate (Guinotte et al. 2006). During the 20th century, seawater carbonate concentrations have been depleted by ~30 μmol kg^{-1} of seawater and ocean acidity by 0.1 pH units (Solomon et al. 2007). Future projections estimate that over the next few centuries, ocean pH will undergo larger changes than those inferred from the geological record of the past 300 million years (Caldeira and Wickett 2003). A significant consequence of such changes will be the inhibition of calcification mechanisms of scleractinian corals, which use aragonite (one metastable form of calcium carbonate) to build their skeletons. By the year 2100, projections indicate that about 70% of current CWC reef locations will become undersaturated with respect to aragonite, thus exposing the majority of presently existing reefs to a corrosive environment, likely altering their global distribution and abundance (Guinotte et al. 2006). Although existing evidence suggests that some scleractinians can persist in undersaturated waters (Thresher et al. 2011; Movilla et al. 2014; Baco et al. 2017; Gómez et al. 2018), the metabolic rates of CWC species could be altered due to the suffered stress and therefore vital functions like respiration and food assimilation could be impaired (Hennige et al. 2015; Georgian et al. 2016; Gori et al. 2016).

Global change has been also evidenced through the rise of sea-water temperature, even in deep-sea environments, that in turn contribute to reductions in oxygen concentrations (Ramirez-Llodra et al. 2011; Sweetman et al. 2017). Model analyses in abyssal ecosystems suggest a rise of 0.01 to 0.1°C decade^{-1} (e.g., the southern Ocean; Purkey and Johnson 2010) while O_2 concentrations will decline up to 3.7% together with reduced flux of organic matter to seafloor (Danovaro et al. 2017; Sweetman et al. 2017). Such changes can significantly impact CWC species, by directly affecting growth, survival and recruitment rates (Form and Riebesell 2012; Brooke et al. 2013; Movilla et al. 2014; Hennige et al. 2015; Gori et al. 2016) and leading to sub-lethal responses, such as effects on coral metabolism or disease susceptibility (Dodds et al. 2007; Voss et al. 2010). Given the major stability of deep-sea environments, deep-sea benthic taxa appear to be more sensitive to temperature changes than shallower and pelagic ones, potentially suffering more adverse consequences than expected (Yasuhara and Danovaro 2016; Sweetman et al. 2017).

2 ECOLOGICAL RESTORATION: from terrestrial to marine shallow and deep habitats

Over the past 50 years humans have changed terrestrial and marine ecosystems more rapidly and extensively than in any comparable historic period, with intermediate and deep marine habitats being one of the most threatened ecosystems as commented above. Transformation of the planet has contributed to substantial net gains in human well-being and economic development, but at the same time has resulted in a substantial and largely irreversible loss in the diversity of life on Earth (Millennium Ecosystem Assessment 2005; Cardinale et al. 2012). Many ecosystems and their functioning have been impaired beyond critical points and cannot return to their native states or previous developmental trajectories (Jackson et al. 1995; Jackson 2001; Society for Ecological Restoration International Science & Policy Working Group 2004) (Fig. 4). Conservation measures are crucial for the preservation and recovery of natural and cultural heritage, however, given the current scale of environmental degradation across a wide swath of ecosystems worldwide, protection alone may be nowadays not sufficient (Lotze et al. 2011; Duarte et al. 2020). Earth's habitats are currently calling for active measures to conserve and restore these ecosystems and the services they provide (McDonald et al. 2016).

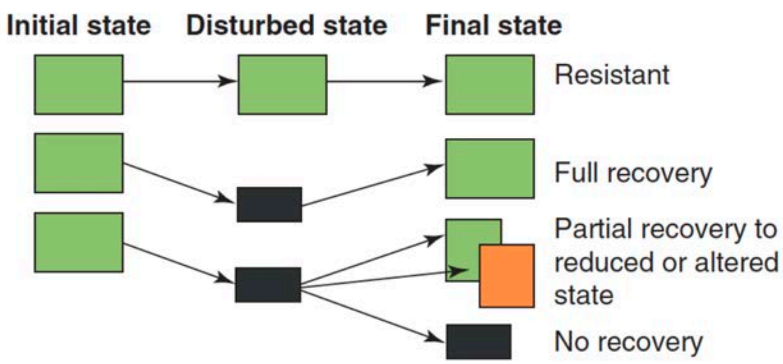

Figure 4. Illustration of possible ecosystems trajectories after disturbance: In the face of external disturbance, ecosystems can be resistant and remain unchanged (green boxes) or they can be disturbed (dark boxes). Afterward, and usually by means of conservation measures, they can either fully recover their initial state, partially recover to altered state (orange box) or remain in the disturbed state. **Source:** Lotze et al. 2011.

Since the end of 1900s, coinciding with the birth of the Society for Ecological Restoration (SER) the practice of ecological restoration, as part of a larger set of ecosystem management practices designed to conserve native ecosystems, is receiving increasing attention worldwide. Ecological restoration has been recognized under different frameworks and conventions, as it offers the opportunity to compensate for anthropogenic damage that has already taken place (Falk et al. 2006). In 2011 the Convention on Biological Diversity (CBD) aimed to restore 15% of degraded ecosystems by 2020 (Aichi Biodiversity Target 15; Convention on Biological Diversity n.d.). More recently, the United Nations General Assembly has proclaimed 2021 – 2030 as the Decade on Ecosystem Restoration, following a proposal for action by over 70 countries worldwide (Waltham et al. 2020). It is predicted that ecosystem restoration will generate clear benefits. From now to 2030, the restoration of 350 million hectares of degraded terrestrial and aquatic ecosystems is expected to generate US$ 9 trillions in ecosystem services worldwide (McDonald et al. 2016).

The SER, founded in 1983, established that ecological restoration actions aim to move a degraded ecosystem to a trajectory of recovery that allows the persistence of its component species, as well as the adaptation to local and global changes (Gann et al. 2019). Ecological restoration activities seek to accelerate the recovery of ecosystem structure and functioning relative to an appropriate reference model (the chosen endpoint of restoration). However, the term "restoration" has always been surrounded by certain ambiguity (Ounanian et al. 2018), and SER has re-evaluated and altered its definition several times in the last decade. The latest standard set by SER in 2019 established three different approaches to restoration that may be applied individually or in combination, as appropriate. We consider these approaches in this thesis: (1) Natural regeneration: this corresponds to so-called 'passive restoration'. It encompasses the removal of stressors through the creation of protected areas where the recovery of the biota arises from natural colonization, dispersal and other in situ process. (2) Assisted regeneration: once the source of degradation has been removed, this type of 'active restoration' necessitates direct human interventions that actively trigger the natural population growth capacity of biota. Interventions include removal of non-native species, installation of resources to prompt colonization, reintroduction of target species, etc. (3) Reconstruction: when ecosystems have been severely damaged, this type of 'active restoration' calls not only for the removal of sources of degradation but also the improvement of the biotic and abiotic ecosystem condition. Here, direct human intervention is needed to reintroduce all, or a major proportion of the biota, which otherwise would not be able to regenerate or recolonize the area within a reasonable time frame (Fig. 5).

Complete recovery of ecosystems is difficult to achieve (Jones et al. 2018), since all identified ecosystem attributes (e.g., physical environment, desirable species) should be reintegrated into a self-organizing system that most resemble the reference model (Gann et al. 2019). Given the current state of most ecosystems, finding reference models close to restoration sites could be challenging. However, alternatively, it can be derived from multiple sources of information, such as records about past and current biota together with site environmental features, including predictions of potential changes in environmental conditions that may lead to altered biological assemblages (McDonald et al. 2016). Emerging formulations of restoration use historical knowledge as a guide, rather than a template, accepting multiple potential ecosystem trajectories, recognize the major importance of ecological process over ecosystem structure and composition and promote the establishment of pragmatic goals to reflect human livelihood needs (Higgs et al. 2014). The three approaches of ecological restoration mentioned are part of a continuum of activities which strive towards recovery of an ecosystem (McDonald et al. 2016; Aronson et al. 2017) (Fig. 5). It is beneficial to consider that some restorative actions (such as reducing human impacts) may not have as an ultimate goal full recovery of an ecosystem at the time, but it will contribute towards a more sustainable use of the ecosystem and facilitate future decisions towards practicing ecological restoration. In order to encourage this difficult task, the active restoration efforts often target key foundation species such as trees, kelps, or corals. These taxa often facilitate the re-establishment of associated species and succession processes that are crucial to restore the functioning of the ecosystems (Bruno et al. 2003).

First publications about ecological restoration appeared since 1984 being more focused on terrestrial ecosystems rather than marine ones (Elliott et al. 2007; Wortley et al. 2013). It was not until the last 20 years when marine restoration initiatives have been increasingly carried out around the world (Swan et al. 2016; Zhang et al. 2018; Basconi et al. 2020). Most marine ecological restoration actions have been performed in shallow waters, mainly focused on the restoration of salt marshes (Bakker et al. 2002; Hughes and Paramor 2004; Laegdsgaard 2006), tropical coral reefs (Rinkevich 2005; Precht and Robbart 2006; Young et al. 2012), oyster reefs (Babcock et al. 1998; Brumbaugh et al. 2006; Baggett et al. 2015), mangroves (Proffitt and Devlin 2005; Bosire et al. 2008; Primavera and Esteban 2008), seagrass meadows (Paling et al. 2009; van Katwijk et al. 2009, 2016) macroalgal forests (Verdura et al. 2018; Layton et al. 2020; Medrano et al. 2020) and temperate gorgonians (Weinberg 1979; Linares et al. 2008; Fava et al. 2010). Recently, marine restoration reviews (Basconi et al. 2020; Duarte et al. 2020) reported high frequency of restoration success, supporting the feasibility and potential of marine restoration techniques (Fig. 6), however neglecting deeper habitats. It is broadly recognized that the deep ocean hosts a large portion of Earth's biodiversity that plays a key role in the functioning of our planet and currently concentrates several anthropogenic stressors (Danovaro et al. 2008, 2017; Ramirez-Llodra et al. 2010). Yet, primarily due to technical challenges and associated high cost in accessing to intermediate (from 50 – 70 to 200 m depth) and deep-sea environments (below 200 m depth), ecological restoration research focusing on those habitats still remains scarce (Van Dover et al. 2014; Da Ros et al. 2019).

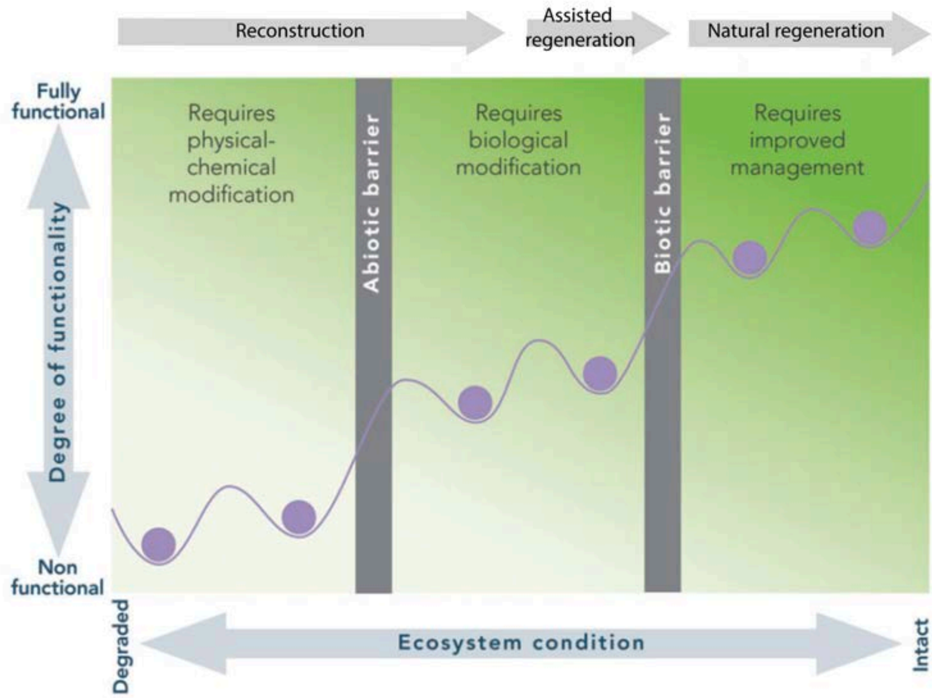

Figure 5. Conceptual model of ecosystem degradation and restoration. Throughs in the diagram represent basin of stability in which an ecosystem can remain prior to being moved toward a restoration (higher functioning state) or degradation event (lower functioning state), helped by the three restoration approaches: reconstruction, assisted regeneration and natural regeneration measures. **Source:** modified from McDonald et al. 2016.

3 ECOLOGICAL RESTORATION OF CWC ECOSYSTEMS

To explore and summarize the scientific efforts done in restoration of CWC ecosystems up to date, a systematic literature search using the biographic database Web of Science was undertaken. Specifically, an advanced search was confined by the combination of the terms “restore*”, “transplant*”, “protect*”, “conserve*”, and “marine protected area” with a second term related to the coral taxa, habitats and target of restoration: “deep sea”, “cold-water coral”, “deep sea coral*”, “deep sea octocoral*”, deep sea scleractinian*” and “deep sea gorgonian*”. Published articles until August 2020, including scientific papers, book chapters, reports and technical summaries were screened and included in the analysis if they contained information regarding restoration of CWC communities and ecosystems. Reference lists of articles, including reviews identified in the literature search, were checked for additional studies.

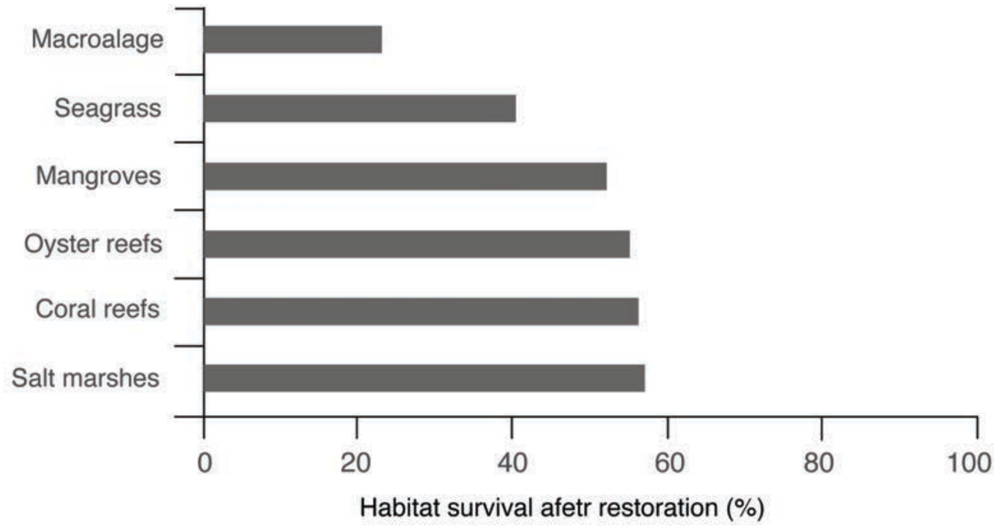

Figure 6. Average success of restoration actions across the six mostly restored coastal marine habitats. **Source:** modified from Medrano 2020 and Basconi et al. 2020.

A total of 81 scientific publications contained data related to CWC restoration, including passive and active restorations approaches. The first studies focusing on the protection and restoration of CWCs were published at the beginning of the 20th century and have been gradually increasing to date (Fig. 7). Given the international concern for the conservation of deep-sea ecosystems (Davies et al. 2007) and the establishment of deep-sea protected areas, passive restoration studies focused on the protection and management of CWCs constitute most of the published papers (48.2%). In contrast, studies on active restoration initiatives (16.0%), complementary studies with implications for restoration (25.9%) and reviews with theoretical considerations (9.9%) have been less frequently published, appearing mainly in the last decade, providing evidence that little work has been done so far and that CWC restoration is a very recent research line (Fig. 7).

Only 13 publications focus on CWC active restoration initiatives (Table 1) since 2001, contrasting with the 221 scientific publications for the case of active restoration of coral ecosystems in shallower waters since 1980 (Boström-Einarsson et al. 2020). Complementary studies encompassed scientific publications exploring natural recovery of CWCs, larval dispersion and recruitment, deep-sea connectivity and in situ coral growth monitoring, with implications for restoring CWC ecosystems. All of them, together with theoretical consideration studies, are crucial for adding scientific and technical knowledge to support restoration actions and improve their success.

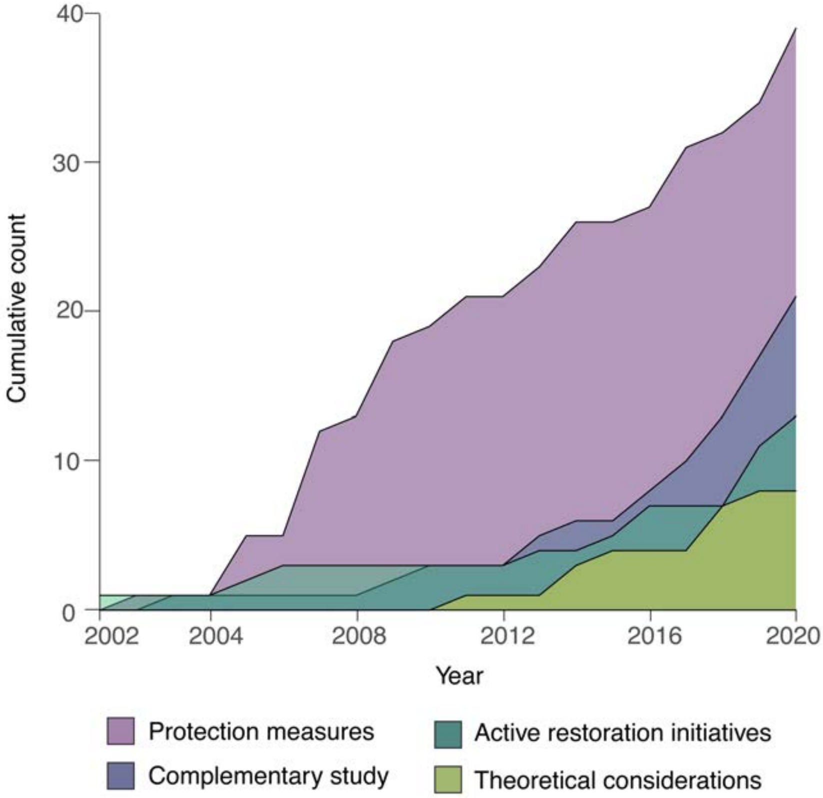

Figure 7. Cumulative count of scientific publications related to CWC restoration over time and according to the study type (N = 81).

Following previous experience in shallow-water ecosystems, scientific efforts to restore CWCs (including protections measures and active restoration publications) were more focused on scleractinian species and CWC reefs (45.7%) rather than on octocoral species and CWC gardens (17.2%) (Fig. 8). There were also publication encompassing both taxa (11.1%) or CWCs in general, without specifying (25.9%). The CWC *L. pertusa* stands out as the most commonly studied coral species appearing in 29.6% of CWC restoration publications (Fig. 8).

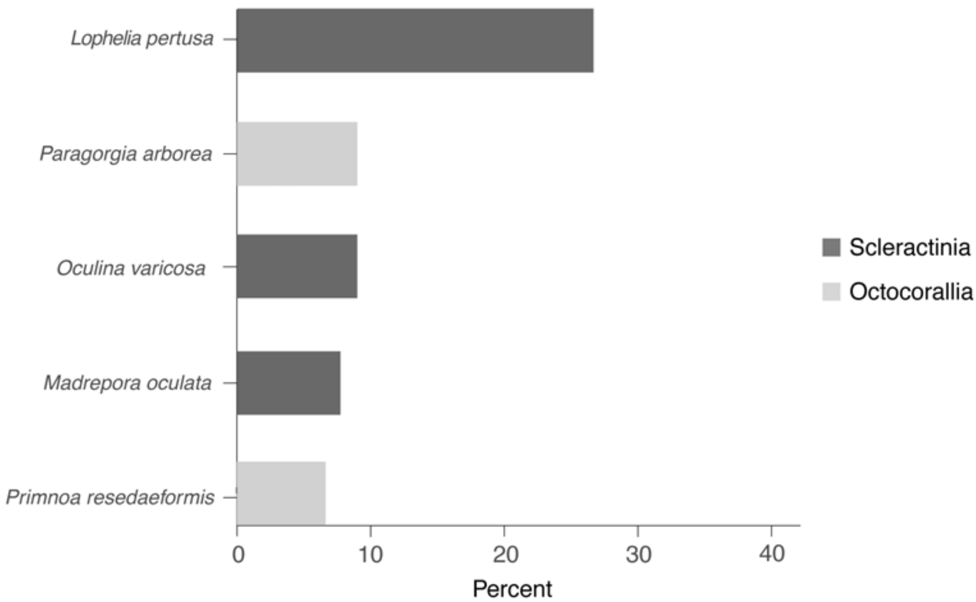

Figure 8. Five most studied coral species in CWC restoration publications (N = 81).

3.1 CWC passive restoration techniques

Since CWCs are extremely vulnerable to anthropogenic activities, given their fragility and life history traits (long-lived and slow-growing species) (Freiwald et al. 2004; Clark et al. 2016), reducing the intensity of key stressors (e.g., fishing closures) is the most sustainable and effective approach for mitigating impacts and protecting CWC ecosystems. International agreements and directives supporting the protection of CWC ecosystems are increasing worldwide (Armstrong et al. 2014). To date, a number of countries have created MPAs for CWC ecosystems: e.g., the Darwin Mounds and Hatton Bank (UK), the Oculina Bank Marine Protected Area, the Davidson Seamount and Aleutian Gorgonian Gardens (USA), the Northeast Channel Coral Conservation Area (Canada), the Commonwealth Marine Reserve (Australia) (DeVogelaere et al. 2005; George et al. 2007; Durán Muñoz et al. 2009; Harter et al. 2009; Huvenne et al. 2016; Althaus et al. 2017; Bennecke and Metaxas 2017). Harter et al. (2009) were the first to quantify positive effects of a deep-sea MPA (Oculina Bank Marine Protected Area). Moreover, high coral abundances and the presence of some large colonies and recruits pointed to a coral population recovery after 12 years following the closure of the North-East Channel Coral Conservation Area to fishing (Bennecke and Metaxas 2017). Likewise, after being a traditional fishery area, the North-Western Hawaiian Ridge and Emperor Seamounts have been protected for 30 years. After this period, evidence for coral re-growth and greater abundances of benthic megafauna were detected (Baco et al. 2019). In some cases, protection alone is however not always successful. For instance, in the eastern Darwin Mounds (UK), very little re-growth and no coral re-colonization have been detected after 8 years of fishing closure (Huvenne et al. 2016). Likewise, CWC community in New Zealand seamounts is not recovering after 15 years from cession of trawling (Clark et al. 2019). Both examples highlighted the low resilience and slow natural recovery capability of CWC ecosystems. Furthermore, MPA designation does not always remove all stressors for CWCs. For instance, oil drilling is still allowed within the Flower Gardens National Marine Sanctuary (US) in the Gulf of Mexico.

3.2 CWC active restoration techniques

In contrast to tropical and temperate shallow-water ecosystems, active restoration in intermediate and deep environments has not yet received much attention, with only sparse knowledge of restoration techniques that may be possible (Van Dover et al. 2014; Da Ros et al. 2019). There are important gaps in understanding the biology and ecology of most target CWC species, including the extent of degradation of their populations, and the potential socio-economic costs and benefits of restoration efforts (Morato et al. 2018). Logistical constraints to access and work in intermediate and deep environments, which in turn increase restoration costs, together with the sensitive nature of CWC species, complicate the development of reliable and efficient restoration techniques (Van Dover et al. 2014; Da Ros et al. 2019). There are currently only a few active restoration actions carried out worldwide, all of them located in the northern hemisphere (13 scientific publications for 7 case studies in 6 different countries): the Säcken Reef in Sweden, the Sur Ridge Seamount in California, CWC Reefs in the Gulf of Mexico and south-eastern off Florida, and CWC gardens in the Azores and the western Mediterranean (this thesis) (Table 1; Fig. 9).

Table 1. Active restoration initiatives carried out to date focused on CWC habitats

Case study	Locality	Restoration technique	Restored Taxa	Depth (m)	Duration time (years)	Results	Reference
1	Swedeen	Transplantats on artifical structures	*Lophelia pertusa*	82 – 87	3 – 4	Mean survival rate of 76% Mean size increase of 39%	Dahl 2013; Jonsson et al. 2015
2	Swedeen	Transplantats on artifical structures	*Lophelia pertusa*	Not specified	Not specified	Still not recorded	Strömberg 2016
3	California	Transplantats	*Corallium sp., Lillipathes sp., Swiftia kofoidi, Isidella tentaculum, Paragorgia arborea, and Sibogagorgia cauliflora*	800 – 1300	1 – 3	Mean survival rate after 1 year of 52% After 3 years coral survival diiffered among species (0% – 100%)	MBARI 2016; Boch et al. 2019; Boch et al. 2020
4	Gulf of Mexic	Artificial structures	Coral species	21 – 400	30 – 34	Attacthment of corals and other sessile invertabrates	Kaiser and Pulsipher 2005; Kaiser et al. 2020
5	SE Florida	Transplantats on artifical structures	*Oculina varicosa*	70 – 100	4 – 8	Mean survival rate 50 – 60%, Larval recuitment episodic Fish abundance enhanced	Koenig 2001; Brooke et al. 2006
6	NW Mediterrean	Transplantats on artifical structures	*Eunicella cavolini*	85	1	Mean survival rate of 87.5%. Feasibility of large-scale and low-cost active restoration method	Montseny et al. 2019; Montseny et al. 2020
7	Azores	Transplantats on artifical structures	*Dentomuricea cf. meteor, Viminella flagellum, Callogorgia verticillata, Paracalyptrophora josephinae and Acanthogorgia armata*	230	10 – 21 months	Coral survival diiffered among species (30% – 100% and 15% – 100% after 10 and 21 months, respectively)	Linares et al. 2019

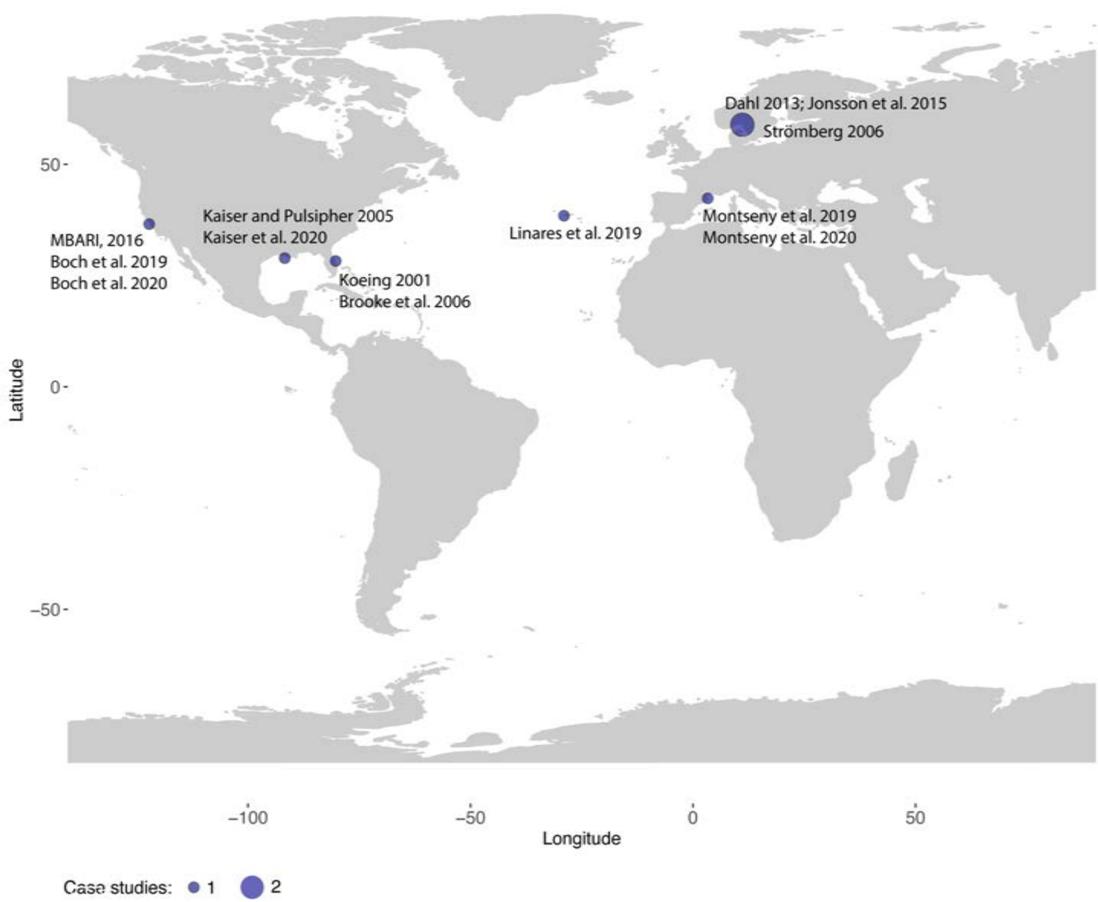

Figure 9. Overview map of the worldwide locations of active restoration projects focused on CWC ecosystems. Circle size corresponds to the number of case studies in each site, up to August 2020 (N= 7).

Most CWC restoration case studies performed to date, where duration time was informed, lasted between 1 and 4 years (Fig. 10A). Exceptionally, longer actions (more than 8 years) are associated with the initiation of the Lousiana Rigs-to-Reefs program in 1986 (The Louisiana Rigs-to Reef Program, Kaiser and Pulsipher 2005; Kaiser et al. 2020) where obsolete oil and gas structures were used as coral colonization substrata. However, short temporal scales are not sufficient to allow for evaluation of full recovery of CWC ecosystems specially bearing in mind the extreme life-history traits of most of deep species (Montero-Serra et al. 2008); therefore, it is still necessary to promote longer monitoring plans (30 – 40 years for CWCs) (Bennecke et al. 2016). Regarding the most used tools for active CWC restoration, transplantation techniques (52%) and artificial structures (44%) were the most widely reported (also including complementary studies and theoretical considerations publications) (Fig. 10B). Transplantation studies were focused on testing different techniques to attach CWC fragments to natural or artificial substrates (e.g., epoxy resin, PVC). While, studies with artificial structures encompassed the deployment or the decommissioning of artificial structures, such as obsolete offshore oil and gas structures and collectors, for CWC larval recruitment. Lastly, mineral accretion through electrolysis (Biorock™) was tested in laboratory conditions with promising results as a suitable restoration method for CWCs (Strömberg et al. 2010).

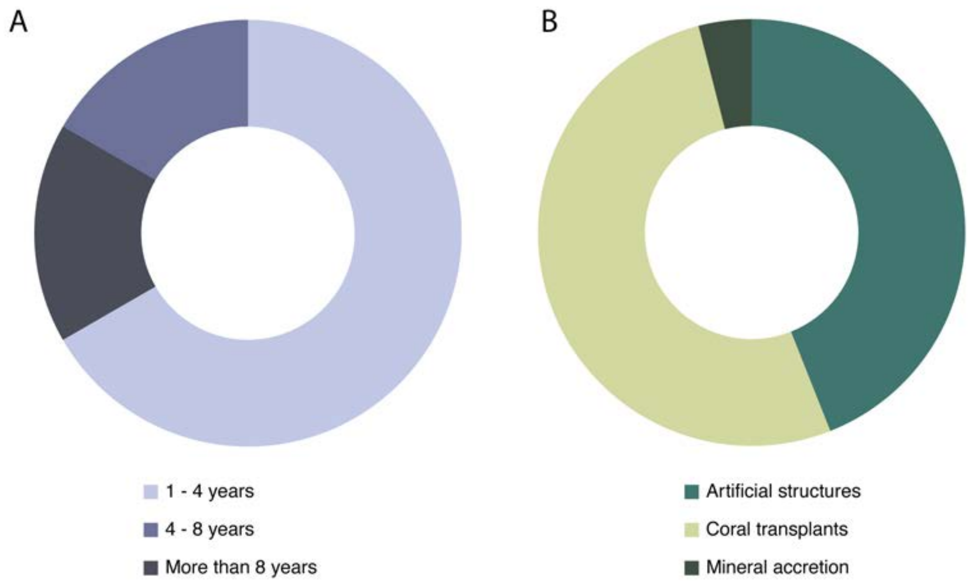

Figure 10. (A) Duration time of the CWC active restoration actions carried out up to date (N=12). (B) Active restoration techniques studied in CWC restoration publications (N = 25).

4 OBJECTIVES

In order to advance towards the conservation of intermediate and deep-sea ecosystems, the general objective of the present thesis was to firstly assess the impact of fishing activities on CWC gardens dwelling on the Mediterranean continental shelf and then to explore novel active restoration measures aimed to enhance the recovery of this vulnerable ecosystems.

The thesis was performed in the Cap de Creus marine area (North-Western Mediterranean Sea; 42° 19′ 12′′ N – 003° 19′ 34′′ E), where CWC gardens dominated by the gorgonian *Eunicella cavolini* (Koch, 1887) are abundant on the extensive continental shelf (60 – 130 m depth; Fig. 11). Gorgonians reach high densities of more than 20 colonies m^{-2} and coexist with a myriad of associated species (sponges, bryozoans, crustaceans, fish) (Dominguez-carrió et al.,2014, 2017; Dominguez-Carrió et al., 2014). Small-scale fishing (bottom longlines and trammel nets) are widespread in this area (Purroy et al. 2014). Since colonies of *E. cavolini* can be easily entangled in nets and longlines due to their branching morphology and erect position, fishing activity could represent a potential driver of degradation to such gorgonian populations.

The thesis is structured in four consecutive chapters, all composed of fieldwork and subsequent data analysis and interpretation.

Chapter 1 aims to evaluate and quantify the small-scale artisanal fishing impact on *E. cavolini* populations on the Cap de Creus continental shelf, through assessing gorgonian bycatch (i.e., the accidentally caught of non-target species in a fishery). During two consecutive fishing seasons the spatial, bathymetric and temporal pattern of gorgonian bycatch was explored to further evidence the need to mitigate the fishing impact, and recommend specific management measures to protect and improve the conservation of CWC gardens.

Chapter 2 explores, for the first time, the feasibility to actively restore gorgonian populations on the continental shelf by using bycatch gorgonian colonies collected from local artisanal fishers. Once collected from fishing nets, gorgonians were transplanted into artificial structures that were deployed at 85 m depth on the inner continental shelf of Cap de Creus. Transplant survival was assessed one year after, by means of a Remotely Operated Vehicle (ROV) monitoring.

Chapter 3 aims to go one step forward, exploring and validating a restoration method to up-scale restoration actions and make them more affordable. Field experiments and modeling approaches were combined to find the best transplant support and gorgonian colony size to ensure the success of the restoration method.

Chapter 4 finally reports the results of the first large-scale active restoration action of CWC gardens carried out involving local fishers using the innovative technique previously developed. During two years, bycatch gorgonians were recovered and returned to their natural habitat. At the end of the restoration action an ecological and economic evaluation was performed in order to quantify the action success.

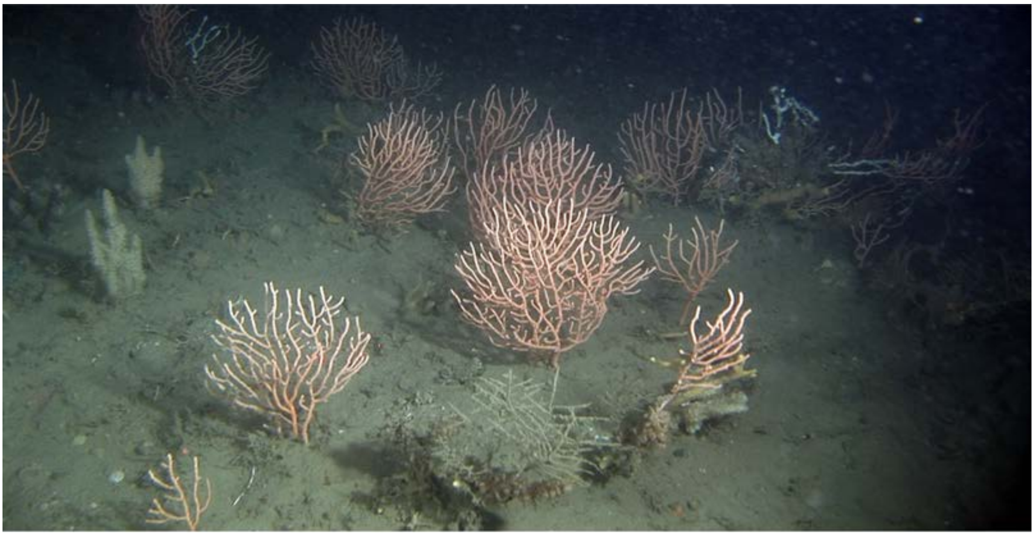

Figure 11. CWC gardens of *E. cavolini* on the Mediterranean continental shelf (60 -130 m depth). **Source:** Dominguez-Carrió, 2018.

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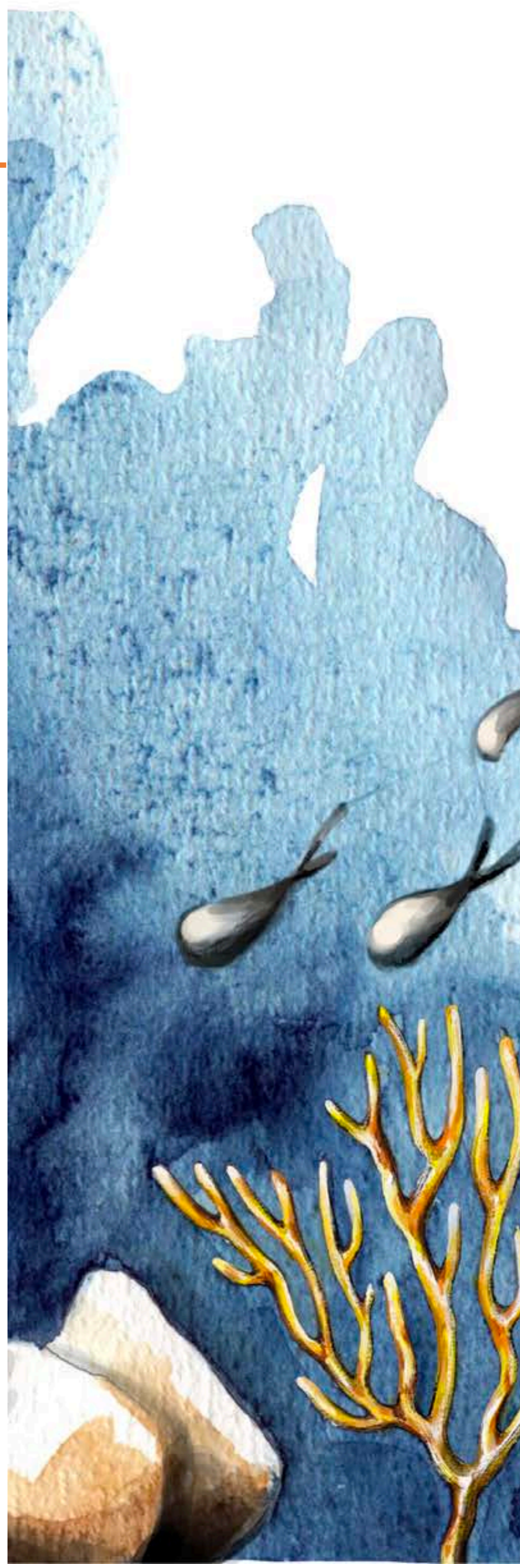
CHAPTER 1

Impact of artisanal fishing on cold-water coral gardens on the Mediterranean continental shelf



1 ABSTRACT

1. Cold-water coral (CWC) gardens, mainly composed of gorgonians, soft corals, sponges and a highly diversified associated mobile fauna, are among the most complex and structuring benthic communities on the Mediterranean continental shelf.
2. These communities have limited recovery capacity and are, consequently, highly vulnerable to disturbances and threatened by bottom-contact fishing activities, widespread on the continental shelf. These fishing activities encompass both bottom trawling but also small-scale fishing using trammel nets or longlines, which although less studied, may also affect benthic communities in a lesser extent.
3. The present study aims to explore and quantify the impact of trammel net fishing on cold-water gorgonian populations dwelling on the continental shelf of Cap de Creus (western Mediterranean Sea) during two consecutive fishing seasons.
4. A high gorgonian bycatch (i.e., the accidentally caught of non-target species in a fishery) was observed in more than half of the assessed nets, with a mean rate of 2.4 ± 0.8 (mean ± sd) gorgonian colonies or fragments for 200 m of trammel net. The higher impact was detected in specific areas between 90–100 m depth and correlated with long soaking time.
5. Trammel nets mostly captured large colonies with a mean height of 20.2 ± 6.6 (mean ± sd), clearly higher than the mean height observed for natural populations in the area, explored with a Remotely Operated Vehicle (ROV). Thus, bycatch effects could lead to changes in population structure and overall ecosystem functioning.
6. Results demonstrated that impact of small-scale trammel net fishing on CWC gardens in the area are small if compared to bottom trawling, however they do not have to be underestimated and can be reduced by promoting the implementation of specific measures.

2 INTRODUCTION

The increase and expansion of fishing activities over time have led to consider them a serious and worldwide threat to marine ecosystems and their living resources (Roberts 2002; Morato et al. 2006). Following the collapse of most shallow-water fish stocks, fishing activity has been moving to deeper waters, concentrating in seamounts, canyons, and continental shelves and slopes, which represent hotspots of biodiversity and production among the deep sea (Roberts 2002). The mean depth of fishing overall increased by 42 m from 1950 to 2001 (Morato et al. 2006), with 40% of the world's trawling grounds located on continental shelves and slopes by the end of the last century (McAllister et al. 1999). Likewise, most of the cumulative anthropogenic impact concentrates on continental shelves (Halpern et al. 2008).

In these environments, between 50 m and 200 m depth, under strong currents, food supply and suitable substrate, CWC species develop elaborate three-dimensional habitats (Freiwald and Roberts 2005; Roberts et al. 2006, 2009) providing shelter, reproductive, nursery and feeding grounds for an abundant and highly diverse associated invertebrate and fish fauna (Henry and Roberts 2007; Price et al. 2019; Rueda et al. 2019; Santín et al. 2020), including several commercial species (Costello et al. 2005; D'Onghia et al. 2012). High richness and abundance of commercially important fish and larvae have been detected in CWC habitats evidencing that their complexity attracts commercial fishes (Costello et al. 2005; Baillon et al. 2012; D'Onghia et al. 2012). CWCs, encompassing reef-forming (Scleractinia) and coral gardens species (Octocorallia, Antipatharia and Stylasteridae) (ICES and Hall-Spencer 2007; Roberts et al. 2009), are classified as Vulnerable Marine Ecosystems (VEMs) due to their functional significance, complexity, fragility and low recovery capability (FAO 2009). These corals are characterized by long lifespan, slow growth, late sexual maturity, and low recruitment rates (Andrews et al. 2002; Orejas et al. 2011; Watling et al. 2011). Thus, impacts can have severe consequences on CWC populations with far-reaching and long-lasting effects, and consequently on the entire associated fauna they support (Williams et al. 2010; Huvenne et al. 2016). Several studies documented that recovery of CWC communities after disturbance is extremely slow and can take decades to centuries (Althaus et al. 2009; Girard et al. 2018).

Consequences of fishing activities on CWC communities have been extensively documented, including habitat destruction, mortality and removing of non-target species, and changes in ecosystem structure and functioning (Hiddink et al. 2006; Durán Muñoz et al. 2011; Pusceddu et al. 2014; Rijnsdorp et al. 2016). Among deep-sea fisheries, bottom trawling is by far the most destructive, causing an overall reduction in habitat complexity and biodiversity (Pham et al. 2014; Oberle et al. 2017), reducing benthic biomass and production (Hiddink et al. 2006), as well as incrementing the near-bottom turbidity and altering sediment properties and sedimentary budgets (Martín et al. 2014; Oberle et al. 2016). On the other hand, although being less studied, it is believed that in areas where bottom trawling is restricted, artisanal fishing (bottom-contact trammel, gill nets, longlines, and traps) also interacts with the benthic ecosystem, with less severe effects but with possible important consequences (Chuenpagdee et al. 2003). As well as impacts on bottom trawling, impacts of small-scale fisheries will depend on the fishing intensity and spatial distribution in relation to sensitive areas, like high structuring CWC areas (Ramirez-Llodra et al. 2011). Structural species with complex morphology such as CWCs can easily get entangled in nets and lines, being common bycatch in several artisanal fisheries (Sampaio et al. 2012; Mytilineou et al. 2014). Additionally, lost fishing gears entangled in corals, broken corals, and scattered skeletons have been extensively observed in artisanal fishing grounds (Buhl-Mortensen et al. 2005; Bo et al. 2014a).

Given the vulnerability of intermediate and deep benthic habitats and the fishing pressure they support, the conservation of CWC ecosystems constitutes nowadays a global priority. The United Nations have recognized the urgency to assess the impact of different types of deep-sea fishing, particularly on CWC ecosystems (Pham et al. 2014). Hence, the aim of this study is to assess the impact of artisanal trammel net fishing on CWC gardens located on the continental shelf. Gorgonian bycatch at 50 – 120 m depth was assessed during two entire artisanal fishing seasons (April – August 2018 and 2019) in a coastal area in the Western Mediterranean Sea. Bycatch records are crucial information to understand the nature and extent of fishing impact (FAO 2018; Carpentieri 2019). Previous studies investigated the potential impact of artisanal fishing in the study area, but quantitative information is still lacking (Purroy et al. 2014; Dominguez-Carrió 2018). The study aims to quantify the bycatch of gorgonian colonies, and to analyze its spatial, bathymetric and temporal pattern, in order to recommend further specific management measures to improve the conservation of CWC gardens and their associated biodiversity.

3. METHODOLOGY

3.1 Study area

The study area is located at the South-western end of the Gulf of Lions in the western Mediterranean Sea, and encompasses the continental shelf on the north and east of the Cap de Creus peninsula, limiting on the north with the adjacent Cap de Creus submarine canyon (42° 19′ 12″ N; 003° – 19′ 34″ E; Fig. 1). The continental shelf located around the cape covers a surface of around 400 km^2, displaying a width ranging from 2.7 km to 12 km. The shelf extends from 60 to 130 m depth with an average slope of 1°, and receives significant terrigenous inputs from the Rhone and many other small rivers resulting in the presence of smooth areas dominated by sandy and muddy sediments alternated with abrupt morphology, such as rocky outcrops (Lo Iacono et al. 2012; Dominguez-Carrió 2018). The area is influenced by the main strong northern winds (Mistral and Tramuntana 28% and 41% of the time, respectively) and the general circulation pattern, dominated by the northern current which flows south-westward (Millot 1990; Estournel et al. 2003; DeGeest et al. 2008). In particular, the north face of the cape is directly exposed to most wave action (Ulses et al. 2008) and to a strong near-bottom current that is accelerated around the cape (DeGeest et al. 2008). Conversely, the eastern face is affected by occasional eastern winds and waves (DeGeest et al. 2008; Ulses et al. 2008). Due to the above-mentioned oceanographic conditions, the Cap de Creus area plays a key role in both sediment and organic matter transport, being considered one of the most productive regions among the Mediterranean Sea. The region hosts a wide variety of benthic ecosystems in a relatively small area: coastal ecosystems, shelf and slope ecosystems, and submarine canyon communities (Orejas et al. 2009; Gili et al. 2011; Sardá et al. 2012). Studies carried out in the area registered around 1,700 species, representing a quarter of all those known in the Mediterranean (Dominguez-Carrió et al. 2014).

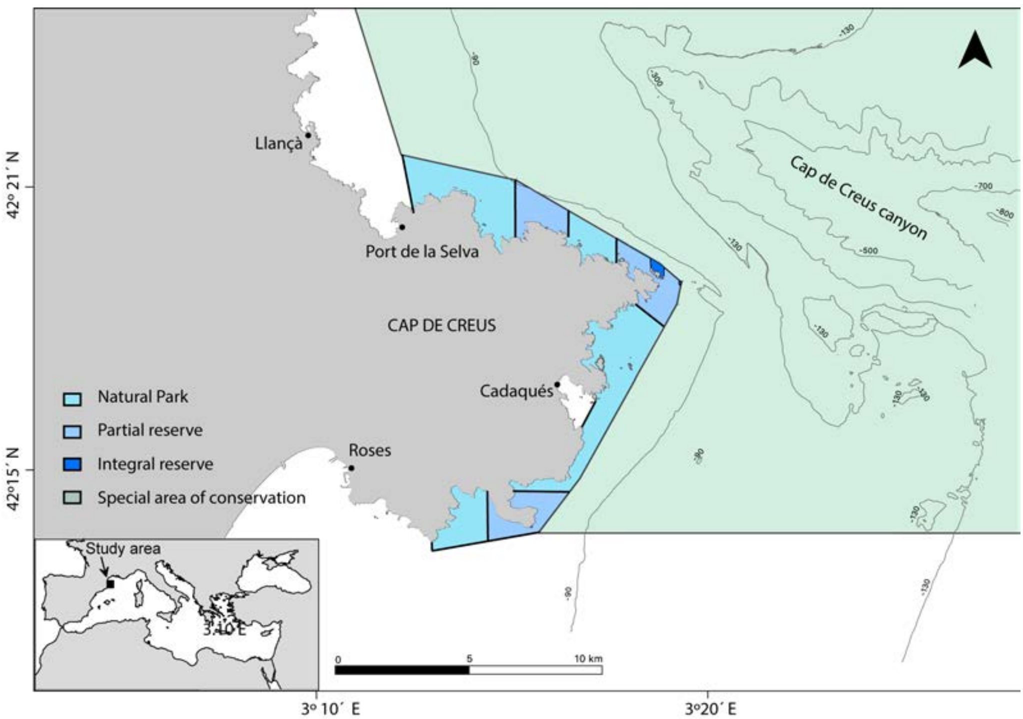


Figure 1: Map of the study area showing boundaries of protection areas (UTM31 WSG84)

Benthic habitats on the continental shelf in the area are mainly dominated by sponges, gorgonians, corals and sea pens, forming CWC garden habitats (Gili et al. 2011; Dominguez-Carrió 2018). Extensive populations of the gorgonian *Eunicella cavolini* (Koch, 1887) dwell on the outcropping rocks at 60 – 130 m depth, being patchily distributed with high-density spots reaching densities of 25 colonies m^{-2}. The presence of *E. cavolini* in the area fosters a high diversity of associated fauna, several sponge's species, soft corals, bryozoans, hydrozoans, polychaetas and species of commercial interest such as lobster, scorpionfish stand out within the gorgonian gardens (Gili et al. 2011; Dominguez-Carrió 2018).

The area is currently under two protection figures: the Natural Park of Cap de Creus, and the South-West Gulf of Lions Canyons System Special Area of Conservation (Fig. 1). The coastal area is protected by the Natural Park declared by the regional government of Catalonia (Spain) since 1998, that covers 13,886 ha, including 10,813 ha on land and 3,073 ha at sea. Regarding the marine region, the park is divided in three protection levels: natural park, partial natural reserve, and integral natural reserve, with protection status running from low to high, respectively (Gómez et al. 2006) (Fig. 1). The special management plan, which is the main instrument for the organization of the protected territory and the use of its resources, forbids several practices as trawling or purse-seining fishing, only allowing artisanal fishing with trammel nets, gillnets or longlines, recreational fishing and diving (Gómez et al. 2006; Dominguez-Carrió et al. 2014). The Natural park limits with the South-West Gulf of Lions Canyons System Special Area of Conservation of the European Natura 2000 network (Fig. 1). The area was recently included in the network due to the presence of CWC gardens and reefs as well as sponge grounds communities, encompassed in the 1170 Reefs habitat from the Annex I of the Habitats Directive (Directive 92/43/CEE). The special area of conservation covers an area of 93,766 ha, of which 1,243 ha are occupied by 1170 Reefs habitat, and includes the extensive continental shelf, as well as a large part of the adjacent submarine canyon (Dominguez-Carrió et al. 2014). Furthermore, despite its vulnerability and high ecological value, there is still no management plan for this special area of conservation.

3.2 Fishing activity in the study area

Being attracted by the abundance of exploitable resources, the marine area of Cap de Creus has been under the influence of professional fishing fleet during decades (Gómez et al. 2006; Lloret and Riera 2008). Within the limits of the natural park, artisanal fishing coexists with other recreational activities as diving and boating, meanwhile outside the park bottom trawling fishing is the main activity (Fig. 2; Dominguez-Carrió et al. 2014). However, strong bottom currents and outcropping rocks in the area between the north coast of the cape and the submarine canyon, limit trawling activities. Regarding artisanal fishing, no specific grounds exists, since the environmental conditions together with fisher's expertise determine the fishing sites. Recently, the fishing activity have diminished due to the decreases of catches together with the conversion of fishers to other tourism-related work sectors (Gómez et al. 2006; Lloret and Riera 2008). Indeed, at present, fishing is mostly a complementary activity for most fishers, with around 30 boats currently working in the area. Some evidence of long lines and ropes entangled around CWCs detected in the area demonstrate the impact of these fishing activities on the littoral zone, continental shelf and slope (Gili et al. 2011; Sardá et al. 2012; Dominguez-Carrió et al. 2014). Although fisher's presence in the area is higher during summer and autumn (June – November) (Gómez et al. 2006), the fishing activity is distributed among all the year, alternating the targeting species according to season and temporary closures. Trammel nets, longlines and gillnets dominate the artisanal fishery in the study area (Purroy et al. 2014). Trammel net fishing is mainly aimed at lobsters (*Palinurus elephas*) and scorpionfish, and are placed, usually on rocky bottoms or close to them, and picked up after some days (BOX 1) during the fishing season from April to August (Gómez et al. 2006; Lloret and Riera 2008).

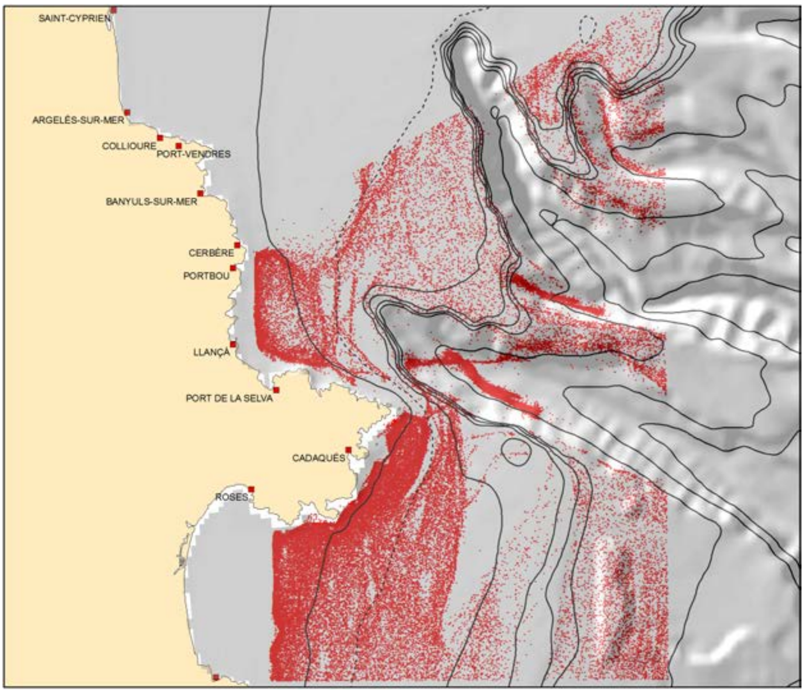


Figure 2: Bottom trawling fishing effort in the study area (Cap de Creus cape), positioning data were emitted by trawlers during fishing activity between 2002 and 2009. **Source:** Dominguéz-Carrió et al 2014.

3.3 Bycatch data collection

The impact of artisanal fishing on gorgonian populations in the study area was evaluated by analyzing gorgonian bycatch of bottom trammel nets. The study was focused on two artisanal fishing associations in the area (Cadaqués and Port de la Selva), accounting for a total of 9 artisanal medium-sized boats operated by one person (6 – 12 m in total length, 2 – 5 gross tons, and 17 – 26 kW) working on the north and north-east coast of the Cap de Creus peninsula. The nine artisanal fishers in the area collaborated in the study allowing scientists to collect and quantify onboard the accidentally fished gorgonians (*E. cavolini*) (Fig. 3). During the entire fishing season (from April to August) of 2018 and 2019 a total 107 trammel net sets during 69 fishing events were evaluated. For each fishing event, location, depth, net longitude, net height, net material, net mesh size, soaking time, and target species were recorded. Moreover, all of the bycatch gorgonian colonies and fragments were photographed on a ruled table in order to record their number and size.

Figure 3: (A) Scientists collecting bycatch gorgonian on board. (B) *E. cavolin***i** colony entangled in a trammel net. **Source:** Laia Sabaté

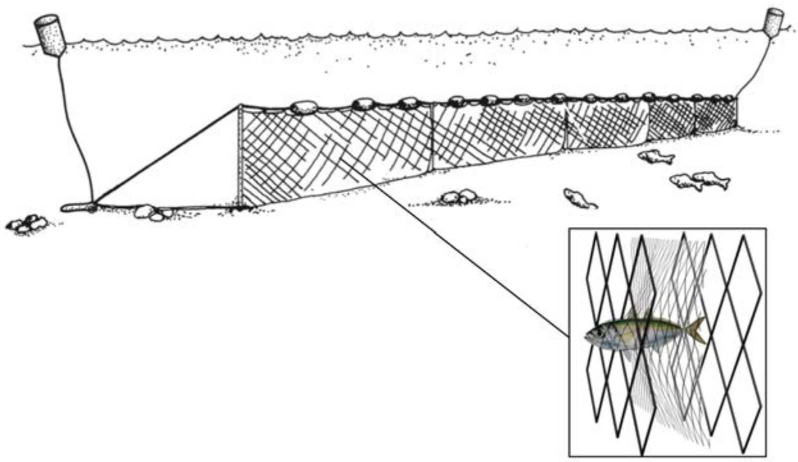
3.4 ROV exploration

At the same time, natural populations of *E. cavolini* in the study area were also investigated by means of the Autonomous Underwater Vehicle (AUV) Girona 500 (Carreras et al. 2016). Three video transects were recorded in September 2018 at 80 – 95 m depth, accounting for approximately 1 hour of analyzed video and covering an area of 314 m^2 of sea bottom at a constant speed of 0.2 m s^{-1} (see Table 1 for detailed information). The AUV was equipped with a set of underwater lights and a high definition video camera pointing forward in order to identify gorgonians colonies. Two parallel laser beams were included to provide a scale and to define a fixed width of the transects (30 cm) for subsequent video analysis. The vehicle counts with a complete navigation suite that includes a attitude sensor, a doppler velocity logger, a pressure sensor and an ultrashort baseline system that allows tracking and correcting the AUV position with respect to a surface vessel.

BOX 1: TRAMMEL NETS

Source: Scheme of a trammel net. Modified from Hubert et al. 2012 and http://www.seafish.org/geardb/gear/trammel-nets/index.html

Trammel net is a traditional small-scale fishing gear placed in contact to the sea-bottom. Buoys at the top and weights at the base of the net allow to fix the net perpendicularly to



the bottom and keep it straight. The gear consists of three walls, each with a smaller mesh size. Fish pass through the outer walls and finally get entangled in the inner small meshed wall. Trammel nets are usually collected two – three days after deployment, but this could vary according to several factors as individually fisher's habits or environmental conditions. Its size is approximately 1.5 m high and 80 – 100 m length. However, one trammel net set is usually composed by more than one consecutive net, achieving overall lengths of 200 – 400 m. It is popular to fish lobsters, cuttlefish, scorpionfish and mullets (Purroy 2010; Dominguez-Carrió et al. 2014).

Table 1: Detail information about video transects recorded on the study area. (Transects position: see Fig. 7).

Id video transect	Start Latitude	Start Longitude	End Latitude	End longitude	Start depth (m)	End Depth (m)	Duration (min)	Length (m)	Width (m)
Tr1	42º 20´10"	3º 19´ 44"	42º 20´08"	3º 19´32"	94.5	92.9	22.5	358.6	0.3
Tr2	42º 20´36"	3º 17´ 45"	42º 20´37"	3º 17´38"	81.1	85.3	11.7	183.5	0.3
Tr3	42º 21´35"	3º 16´ 37"	*42º 21´19"*	3º 16´38"	94.6	93.1	22.7	504.5	0.3

3.5 Data analysis

Gorgonian bycatch was normalized to a standard 200 m length, generally representing the most abundant length of trammel nets used in the area. Given the large size of most collected fragments (10.7 ± 3.7 cm; mean height ± standard deviation (SD)), both entire colonies and fragments were included in the quantification of bycatch. However, only entire colonies were taken into account to analyze the size structure of the bycatch. By analyzing the pictures obtained onboard, the health condition (proportion of necrosis or epibiosis) and the maximum height of each entire gorgonian colony were assessed and measured using the Macnification 2.0.1 software (Schols and Lorson 2008). Similarly, the maximum height of gorgonians (distance from the colony base to the tip of the furthest apical branch) in their natural habitat was measured using pictures extracted from the AUV video transects through the video analysis software Premier Pro CC. The distance between the two laser beams was used for scale calibration, thus measurements were performed only on those colonies in which the laser beams were in the same spatial plane as the gorgonian. Size structure of bycatch gorgonians was determined an compared with the size structure of natural populations in the area, using the non-parametric Wilcoxon signed-rank test, as our data did not follow the normality assumptions of parametric tests (Wilcoxon 1945).

Following, the rank-based non-parametric Kruskal-Wallis test (Kruskal and Wallis 1952, 1953) was used to identify the relationship between the soaking time of trammel nets and the resulting gorgonian bycatch. Furthermore, a free geospatial software (QGIS 3.12) was used to geolocate all the evaluated fishing events obtaining the geographical distribution of gorgonian bycatch in the study area. In addition, their bathymetric and temporal distribution during the fishing season were also determined. These lasts were plotted over the fishing effort performed, established as the number of 200 m trammel nets evaluated for each depth and during each month. A Kruskal-Wallis test was performed to explore differences in gorgonian bycatch among depth ranges and months. All statistical analyses and graphics were performed with R (RCore Team 2018) by means of the RStudio software (RStudio Team 2016) using the ggplot2 package (Wickham 2016).

4 RESULTS

During all the studied period, a total of 107 trammel net sets were evaluated (53 for 2018 and 54 for 2019). They were composed by an aligned series of varying lengths nets (from 1 to 5), with a final length ranging from 100 to 600 m long and from 1 to 3 m height. The nets most frequently employed were 200 m long (43%), 1 – 1.5 m height (95%), of nylon material (86%) and with an inner mesh size of 3 – 3.5 cm (77%). The main target species for fishing was the spiny lobster (84%) followed by the red scorpionfish (19%). Overall, a total of 359 bycatch gorgonians, including fragments and entire colonies, were recorded (213 in 2018 and 146 in 2019). On average and for each fishing season 55.2 ± 4.6 % (mean ± SD) of trammel nets caught at least one gorgonian. Most nets (82.6 ± 2.8 %, mean ± SD) caught between 1 and 4 gorgonians, but 11.9 ± 5.9 % (mean ± SD) fished between 11 and 30 colonies (Fig. 4). The quantification of the bycatch per fishing season (normalized per 200 m of trammel net) revealed that on average 2.4 ± 0.8 gorgonians were accidentally caught for each trammel net (3.0 ± 5.6 in 2018 and 1.8 ± 3.2 in 2019). In comparison with natural gorgonian populations in the area, the mean height of bycatch colonies (20.2 ± 6.6, mean ± SD) was significantly larger than for colonies in their natural habitat (12.9 ± 5.5; mean ± SD) (Wilcoxon test W = 37422, p-value < 0.001). Likewise, the size structure of bycatch colonies was clearly biased towards larger sizes (maximum bycatch colony reached 40.3 cm height) and dominated by 15 – 20 cm size class (Fig. 5). Moreover, among bycatch colonies there were no small young colonies (< 5 cm height).

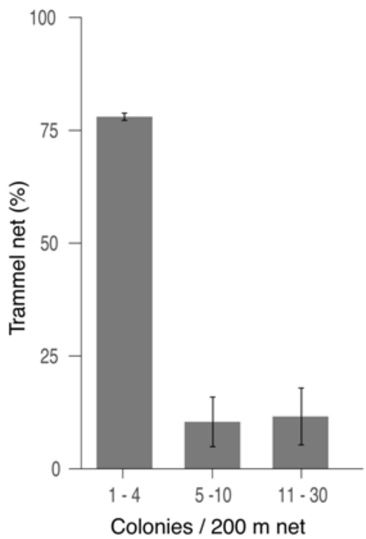


Figure 4: Frequency of gorgonian bycatch among trammel nets (mean ± SD). The number of gorgonian colonies and fragments was counted for 200 m of net.

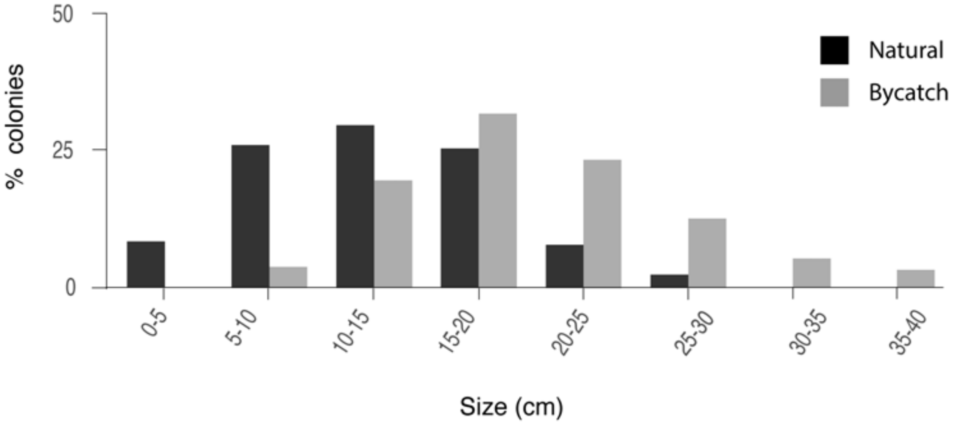


Figure 5: Size structure of bycatch gorgonian colonies (grey; n = 284) and colonies in their natural habitat (black; n = 396).

The 80% of the collected bycatch gorgonians were entire colonies, and 20% were fragments. From all the bycatch gorgonians, 60.9% were in good condition, without any necrotic portions or overgrowth of epibiotic organisms. In this line, 30.3% of the gorgonians showed low necrosis and epibiosis (less tan 30% of their tissue), and only 8.8% showed high necrosis and epibiosis (more than 30% of their tissue). On average, trammel nets had a soaking time of 48h and the gorgonian bycatch significantly increased when soaking time exceeded three days (Kruskal-Wallis chi-squared = 21.047, df = 3, p-value = 0.0001; Fig. 6). Most of the gorgonian catches occurred in the north of the Cap de Creus peninsula (Fig. 7), specifically from 50 m to 120 m depth, with most catches concentrated at 70 – 100 m depth and from June to August (but without significant statistical differences; For the bathymetric distribution, Kruskal-Wallis chi-squared = 10.446, df = 6, p-value = 0.849; for the temporal distribution Kruskal-Wallis chi-squared = 3.718, df = 4, p-value = 0.446), (Fig. 8A and 8B).

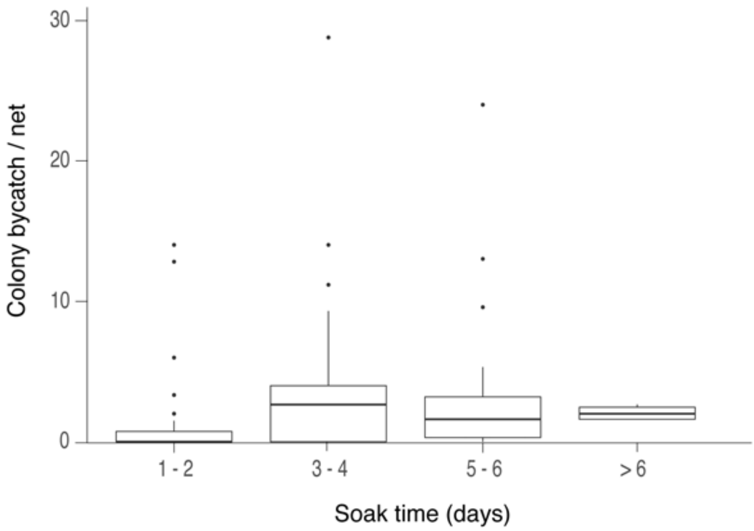


Figure 6: Gorgonian bycatch normalized per 200 m of trammel net according to the soak time of the nets.

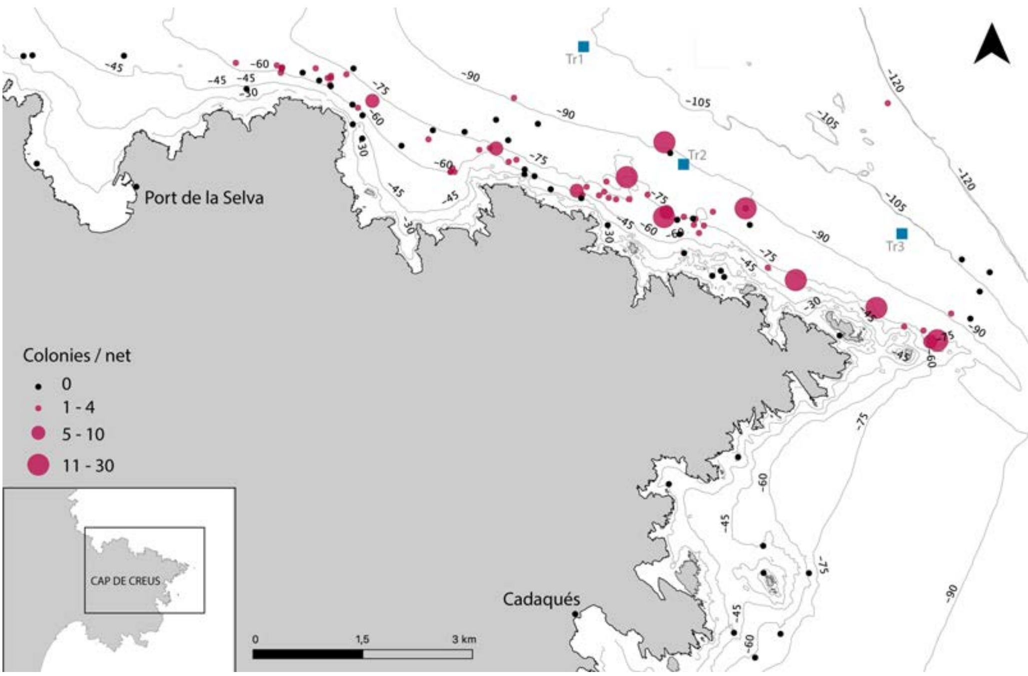


Figure 7: Geographic distribution of the gorgonian bycatch according to the number of colonies and fragments caught for 200 m of trammel net. Blue squares indicate the position of the video transects (UTM31 WSG84).

5. DISCUSSION

The present study evidences that trammel net fishing in the continental shelf of Cap de Creus impact gorgonian populations. Gorgonians were accidentally caught in more than half of the trammel nets evaluated. The frequency of gorgonian bycatch observed was in line with coral bycatch in longlines fishing in CWC banks in the northern Ionian Sea (55%) (D'Onghia et al. 2012), and higher than what was reported in the North-Eastern Atlantic (15%) (Sampaio et al. 2012). On the contrary, the bycatch frequency in the Cap de Creus area was lower in comparison with reports from longlines fishing in the eastern Ionian Sea (72%) (Mytilineou et al. 2014), and trammel net fishing in mesophotic rocky habitats of the Ligurian Sea (80%) (Enrichetti et al. 2019). High bycatch frequency reported by Mytilineou et al. (2014) and Enrichetti et al. (2019) was associated to a lower maximum number of corals per fishing gear (4 and 11 colonies, respectively) than observed in Cap de Creus (30 gorgonian colonies or fragments for 200 m of trammel net), indicating high localized damage due to some trammel nets in our study area. The mean gorgonian bycatch (2.4 ± 0.8 colonies or fragments per 200 m of trammel net) observed in the present study matches with the most frequently bycatch gorgonian (*Eunicella verrucosa*, 3.7 ± 2.1 gorgonians per 200 m of trammel net) in Enrichetti et al. (2019). These differences observed among the studies could be explained by several factors as the type of fishing gear, mechanical and morphological features of coral colonies and its type of aggregation on the sea bottom, the bottom type, or local environmental conditions. For instance, longlines gear, consisting of one main line and a set of hooks, are expected to produce lower impact to coral colonies since they potentially have lower contact surface than trammel nets (Sampaio et al. 2012). Regarding traits of impacted coral species, several studies have already stated that coral catchability rates depend on their shape, size and breakability of the skeleton (Mytilineou et al. 2014; Bo et al. 2015). Gorgonians with elastic skeleton and fan-shaped morphology, such as *E. cavolinii* are highly exposed and have high probabilities to be caught by trammel nets, which easily get entangled between their branches and can remove the entire colony. Enrichetti et al. (2019) found that flexible gorgonians where the second-most caught morphological group by trammel nets after calcareous bryozoans. Contrarily, dense reefs formed by CWC species as *Lopehlia pertusa* are probably more resistant to be caught, and more likely to be broken or damaged *in situ* (Mytilineou et al. 2014). Furthermore, the type of colony aggregation on the sea bottom and the degree of colony exposure would also facilitate incidental catches (Batista et al. 2009; Enrichetti et al. 2019), as well as strong currents that could move the nets incrementing the probability of entanglement. Other species like *Corallium rubrum* or *Paramuricea clavata* found in cavities or sheltered areas present low bycatch rates (Enrichetti et al. 2019).

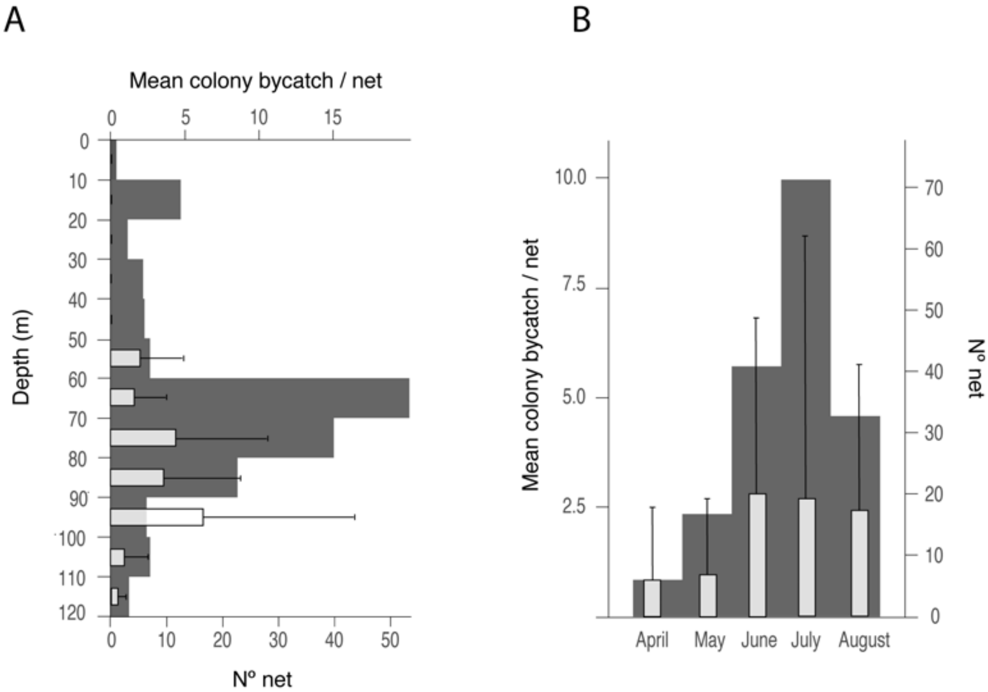


Figure 8: (A) Bathymetric distribution of the mean gorgonian bycatch normalized for 200 m of trammel net (light grey); Dark grey bar plots represent the total number of fishing performed at each depth range. (B) Temporal distribution of the mean gorgonian bycatch normalized for 200 m of trammel net (light grey); Dark grey bar plots represent the total number of fishing carried out each month.

Trammel nets primarily caught medium-large colonies, mainly between 15 – 20 cm size, and not young small colonies (<5 cm size). Although image-based measures of gorgonian size may partially underestimate the real size of the gorgonians, the size structure of bycatch colonies was clearly biased toward larger colonies compared to the size structure of the natural populations (Fig. 6), confirming that nets more easily remove those colonies with major size and complex morphology (Pham et al. 2014). Reducing the abundance of adult large colonies may result in an immediate reduction of the ecosystem functioning, limiting coral gardens role as refugee, foraging and nursery area for sessile and vagile associated fauna (Enrichetti et al. 2019). Moreover, since large colonies are the main contributors in gorgonian sexual reproduction (Coma et al. 1995), these impacts may have delayed effects on the reproductive success and overall long-term survival of gorgonian populations (Linares et al. 2005). Nonetheless, the natural dynamic of *E. cavolini* populations on the Cap de Creus continental shelf is still unknown (recruitment rate, *in* *situ* growth…), therefore key information is missing to fully understand how damaging the impact of trammel nets could be.

Most of the bycatch gorgonians were alive and entire colonies. A large part of them (about 60.9%) shown no necrosis neither overgrowth by epibiotic organisms, and only a small portion (<10%) shown more than 30% of damage. This fact was also reported by Mytilineou et al. (2014), which detected an even higher percentage of alive bycatch colonies (95%). Capturing intact colonies evidences the ease of directly remove entire colonies by trammel nets without causing partial mechanical damage (i.e., tissue abrasion and breakage of branches) to the impacted colonies (Mytilineou et al. 2014). In fact, contrarily to other studies that collected epibiotic and necrotic colonies, and observed a clearly fishing footprint in the habitat (Bo et al. 2014b; Enrichetti et al. 2019), ROV videos in the study did not show lost fishing gears or damaged colonies. Those differences could be explained by major differences between substrate complexity. The more complex substrate, with prominent rocks and vertical walls, in Enrichetti et al. (2019) and Bo et al. (2014b) studies facilitates entanglement and loss of nets, increasing their potential to damage and abrade colonies. In this line, evidences of lost long-line fishing gear were detected deeper in our study area, among *Madrepora oculata* colonies in the Cap de Creus canyon (Orejas et al. 2009).

Gorgonian bycatch in the study area is concentrated in the north-east area of the Cap de Creus peninsula, between 70 and 100 m depth (Fig. 7). Here is where dense populations of *E. cavolini* develop on the outcropping rocky bottom on the continental shelf, with density patches up to 25 colonies $\cdot$ m^{-2} (Dominguéz-Carrió 2018; Dominguéz-Carrió et al. submitted). The substrate nature in this area exclude most bottom trawling fishing (Fig. 2), since nets can get entangled with rocks, reducing the fishing pressure on gorgonian populations to artisanal fishing. Although artisanal fishing effort was primarily concentrated at 60 – 70 m depth, is at 90 – 100 m depth where it had the most impact, since gorgonian bycatch is high despite a much lower fishing effort (Fig. 8A). The bathymetric range between 90 – 100 m depth concentrated only the 4% of all the fishing effort and the 12% of accidental captures. This area is highly exposed to wave action and strong current, and consequently is mainly fished in the last three months of the fishing seasons (June to August) (Fig. 8B). At that time, environmental conditions are good and suitable, and the touristic pressure locally increase the demand of fish and especially lobsters, pushing artisanal fishers to move far from harbours and in deeper waters. Under strong near-bottom currents (Dominguez-Carrió 2018, Dominguez-Carrió et al. submitted; Estournel et al. 2003), trammel nets are likely to move on the sea floor, increasing the probability of entanglement with gorgonians, even more during stormy days (Batista et al .2009). Consequently, long soaking times, as for lobster fishing in the area (since lobsters are attracted by the carcasses of previously trapped fish (Catanese et al. 2018) increase the gorgonian bycatch. If soaking time exceeded three days, the probability of gorgonian bycatch is significantly increased (Fig. 5).

Although our results pointed out to a clearly and non-negligible impact of trammel nets to CWCs of the continental shelf, the studied small-scale artisanal fishery has significantly less impact to benthic ecosystems than bottom trawling fishing (Pham et al. 2014; Catanese et al. 2018). It has been estimated that one single deep-sea bottom trawl in a CWC area has the same impact as 296 – 1,720 longlines, depending on the morphological complexity of the impacted species (Pham et al. 2014). Moreover, representing a traditional and local socioeconomic activity, the viability and suitable development of artisanal fishing should be ensured and promoted (Gómez et al. 2006). The challenge is to find the balance between exploitation and conservation. In this line, our results support that by applying specific measures, the impact of trammel net fishing on CWC gardens could be further reduced. Reducing soaking times (maximum two days) and avoiding adverse weather condition, could significantly decrease fishing discards (Gonçalves et al. 2007; Batista et al. 2009; Catanese et al. 2018) being also favorable for commercial target spices, since longer times could damage captured fish species as well (Acosta 1994). Fishing shallower, avoiding the critical range of 90 to 100 m depth and protecting specific locations, with known high densities of gorgonians, would also help to decrease gorgonian bycatch in the study area. Moreover, modifying and improving fishing nets (e.g., larger mesh sizes, new materials) could be beneficial, both for commercial catch and bycatch (Gonçalves et al. 2007; Szynaka et al. 2018). Finally, through ecological restoration techniques, artisanal fishing impact could also be mitigated. In contrast to the fragments or damaged colonies accidentally collected by trawls, the good health status of bycatch colonies from trammel nets, supports the possibility of restoring them. Recent studies have demonstrated the efficacy of using healthy bycatch gorgonian colonies, recovering and returning them to the natural habitat (Montseny et al. 2019, 2020, submitted). Given the recognized and extraordinary ecological value of the study area (Gili et al. 2011), implementing suitable management plans for the Natural Park and SAC areas (Fig. 1) are a great opportunity to apply the above-mentioned recommendations, contributing to the overall aim to conserve healthy benthic ecosystems.

In conclusion, trammel net fishing in the area interacts with the benthic community, through the bycatch of gorgonians. Overall, the results stress the importance to efficiently manage small-scale fisheries and implement specific conservation measures focused on reducing the fishing impact and protecting CWC gardens ecosystems in the continental shelf. Reducing soaking times, especially in bad weather days, and avoid specific locations stand out as main efficient measures to implement, taking advantage of the legal framework of both the Natural Park and the Special Area of Conservation.

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CHAPTER 2

First attempts towards the restoration of gorgonian populations on the Mediterranean continental shelf

1 ABSTRACT

1. In the Mediterranean Sea, gorgonians are among the main habitat-forming species of benthic communities on the continental shelf and slope, playing an important ecological role in coral gardens.
2. In areas where bottom trawling is restricted, gorgonians represent one of the main fractions of trammel net bycatch. Since gorgonians are long-lived and slow-growing species, impacts derived from fishing activities can have far-reaching and long-lasting effects, jeopardizing their long-term viability. Thus, mitigation and ecological restoration initiatives focusing on gorgonian populations on the continental shelf are necessary to enhance and speed up their natural recovery.
3. Bycatch gorgonians from artisanal fishers were transplanted into artificial structures, which were then deployed at 85 m depth on the outer continental shelf of the marine protected area of Cap de Creus (NW Mediterranean Sea, Spain). After one-year, high survival rates of transplanted colonies (87.5%) were recorded with a Hybrid Remotely Operated Vehicle (HROV).
4. This pilot study shows, for the first time, the survival potential of bycatch gorgonians once returned to their habitat on the continental shelf, and suggests the potential success of future scaled-up restoration activities.

2 INTRODUCTION

Unsustainable and destructive fishing activities have been identified as one of the most pervasive threats to marine benthic ecosystems occurring on continental shelves and slopes (~60 – 1000 m depth), as this area endures the bulk of commercial fishing activity (Watling and Norse 1998; Hall-Spencer et al. 2002). Consequently, the vast majority of benthic communities inhabiting these depths have been degraded for decades (Hall-Spencer 2002). A large amount of the fishing bycatch (the untargeted catch occurring unintentionally in a fishery) of sessile macrofauna comprises coral, gorgonian, and sponge species dwelling on the continental shelf and slope, as they are easily entangled in trammel nets, longlines, and pots, due to their branching morphology and erect structure (Wareham and Edinger 2007; Althaus et al. 2009; Durán Muñoz et al. 2011; Sampaio et al. 2012; Bo et al. 2014a). Additionally, these benthic species are also highly exposed to partial mechanical damage (e.g., breakage and tissue abrasion) from the direct impact of fishing activities (Althaus et al. 2009; Sampaio et al. 2012; Mytilineou et al. 2014) and smothering by sediment suspended by bottom trawling fishing (Grant et al. 2018). The loss of this benthic habitat-forming species can result in overall loss of the associated biodiversity and is comparable to the impact of forest clear-cutting on terrestrial ecosystems (Watling and Norse 1998).

Corals, gorgonians and sponges are among the main engineering species (*sensu* Jones et al. 1994) in marine ecosystems, where they play an important structural and functional role (Wildish and Kristmanson 1997; Gili and Coma 1998). They form complex three-dimensional structures that generate spatial heterogeneity and provide suitable habitat for hundreds of associated species, many of which are of economic importance (Krieger and Wing 2002; Henry and Roberts 2007). Moreover, by capturing plankton and suspended particulate organic matter they influence benthic-pelagic coupling processes and biogeochemical cycles (Gili and Coma 1998). In the Mediterranean Sea, coral gardens dominated by gorgonians are among the main structuring communities in benthic ecosystems on the continental shelf and slope (Bo et al. 2012; Gori et al. 2017; Angiolillo and Canese 2018)(Bo et al. 2012, 2015; Grinyó et al. 2016; Gori et al. 2017). Currently, coral garden distribution on the continental shelf is mostly restricted to areas were bottom trawling does not occur due to the rough topography of the sea bottom (Fabri et al. 2014; Bo et al. 2015; Grinyó et al. 2016). However, since commercial fish species are often associated with these communities, they are largely exploited by artisanal fishers using trammel nets and longlines (Mytilineou et al. 2014; Deidun et al. 2015). The entire removal, or partial damage of corals and gorgonians colonies caused by fishing gears can have far-reaching and long-lasting effects undermining the long-term viability of their populations (Bo et al. 2014b, 2015), since they are long-lived, slow growing species, with delayed sexual maturity and limited recruitment success (Coma et al. 1998; Garrabou and Harmelin 2002; Linares et al. 2007).

Natural recovery of these communities may take centuries, if possible at all (Dayton 2003). In order to enhance their recovery, active intervention to aid the regeneration of these communities is highly desirable (Rinkevich 2005). Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed by human activities, bringing it back as close as possible to its undisturbed state (Society for Ecological Restoration International Science & Policy Working Group 2004). At present, the practice of ecological restoration is receiving increasing attention worldwide as it offers the opportunity to reverse much of the environmental anthropogenic damage caused by mismanagement of natural resources (Falk et al. 2006). While marine restoration practices are widespread, mainly in shallow tropical environments (e.g., Rinkevich 2005; Precht and Robbart 2006; Young et al. 2012) active restoration initiatives focusing on degraded deeper benthic ecosystems are still extremely uncommon (Brooke et al. 2006; Dahl 2013).

The adverse impact of fishing to Vulnerable Marine Ecosystems (VMEs), such as cold-water Coral (CWC) reefs and coral gardens (OSPAR 2010) and the need to conserve them, have become a global concern (Davies et al. 2007; Angiolillo and Canese 2018). Due to the low resilience of coral garden species, they display high vulnerability to disturbing human activities (Roberts and Hirshfield 2004; Montero-Serra et al. 2018b), which has prompted a growing interest in protection and restoration initiatives aimed at mitigating their further degradation and enhancing their recovery (Angiolillo and Canese 2018). Currently, a few deep-sea active restoration initiatives have mainly focused on transplantation actions of CWC species, such as *Oculina varicosa* off the south-eastern coast of Florida (Brooke et al. 2006) and *Lophelia pertusa* in Sweden (Dahl 2013). Nevertheless, restoration techniques for coral gardens on the continental shelf and deeper environments have not yet been validated.

The main goal of this study was to evaluate, for the first time, the feasibility of recovering and returning to their natural environment bycatch gorgonians from the Mediterranean continental shelf in order to the mitigate fishing impact. Bycatch gorgonians collected from artisanal fishers were transplanted onto artificial structures, deployed at the continental shelf (85 m depth) in a marine protected area, and monitored using an HROV. This pilot action could be applicable to deeper ecosystems that require similar technical logistics, and is a first essential step in assessing the feasibility of future large-scale ecological restoration of CWC gardens.

3 METHODOLOGY

3.1 Gorgonian collection and maintenance

Colonies of the gorgonian *Eunicella cavolini* (Koch, 1887) were obtained from artisanal fishers bycatch from Cap de Creus (North-Western Mediterranean Sea, 42° 19′ 12′′ N – 003° 19′ 34′′ E), at a depth range from 70 to 100 m depth, during three fishing sorties in June, and one in August 2015. Fishers picked up gorgonians entangled in trammel nets and kept them in containers filled with surface seawater (~20 – 23ºC). Once back on land (1‑2 hours after collection), gorgonians were transported to the experimental aquarium facilities of the Institute of Marine Sciences (ICM – CSIC) in Barcelona (within 3 – 4 hours after the initial pick up), while seawater temperature was kept at 14 ± 1.0 °C at all times. A total of 120 gorgonians were held in 100 L tanks with continuous seawater flow, filtered through a 50 μm sand filter (Olariaga et al. 2009), fed frozen Cyclops three times a week, and kept at 14 ± 1.0 °C in the dark, thus simulating Cap de Creus continental shelf's natural conditions. The size of the collected colonies ranged from 6.7 cm to 22.4 cm (12.3 ± 4.6 cm, mean ± SD), and they were held under the above-mentioned conditions between a few days and a maximum of 2 months.

3.2 Transplant on artificial structures and deployment on the sea bottom

From June 27th to 30th, 2015, 80 gorgonians were transplanted onto two stainless steel structures (40 gorgonians onto each) (outer diameter = 2 m; inner diameter = 1.5 m), with a base grid (10 x 10 cm) surrounded by four concrete plates and a central 1 m vertical axis holding an acoustic reflector (30 cm in diameter) supported by four stainless steel bars (12 mm in diameter) (Fig. 1). Forty conical supports for the gorgonians (80 mm high, 20 mm diameter) were placed on the grid. The inside of the supports was filled by polyester fiberglass resin and, once dry, 8 mm boreholes were made in order to attach the gorgonians colonies with epoxy putty (Corafix SuperFast, GROTECH®). Each structure weighed 137 Kg in the air. Initially, the structures were deployed at 6 m depth north of the marine protected area of Cap de Creus, where gorgonians (entire colonies) were attached to the supports by scuba divers. Each structure was then raised up to below the water surface by means of a buoy, and transported by boat at a slow and constant speed (~0.5 knots) towards the continental shelf, where they were deployed at 85 m depth (Structure 1: 42° 20.06′ N – 003° 18.67′ E; Structure 2: 42° 20.05′ N – 003°18.67′ E). Since additional 40 gorgonian colonies were collected as bycatch in fishing events in August, they were transplanted later on a third structure on October 23rd and 24th, and deployed on October 25th, 2015 nearby the first two structures (Structure 3: 42° 20.05′ N – 003° 18.64′ E) following exactly the same procedure. The density value of colonies transplanted onto each structure corresponds to ~15 colonies $\cdot$ m^{-2}, and was selected based on data about Mediterranean gorgonian assemblages dwelling at 40 – 300 m depth (10 – 20 colonies $\cdot$ m^{-2}; Bo et al. 2009; Grinyó et al. 2016).

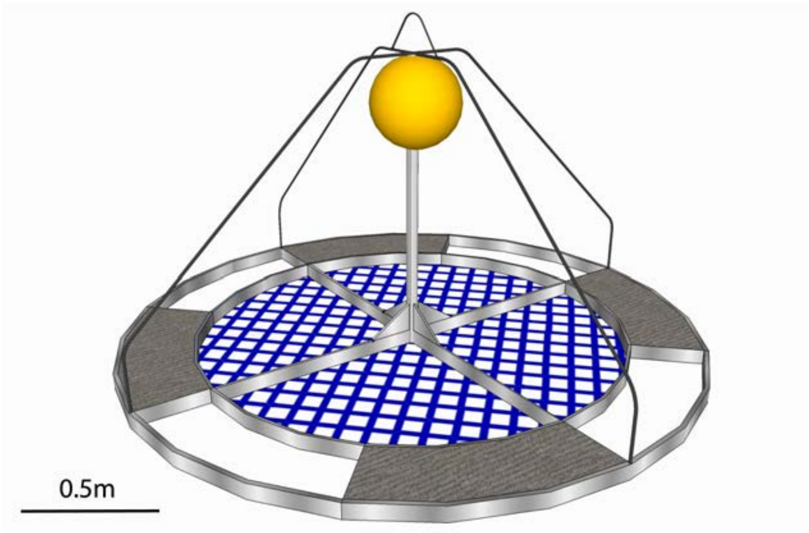

Figure 1. Schematic figure of the stainless-steel structures used in this study.

3.3 Monitoring of transplanted colonies

The structures were monitored through three consecutive surveys using the Girona 500 Autonomous Underwater Vehicle (AUV), equipped with the Bumblebee stereo camera, working as a HROV (Carreras et al. 2016). Surveys were conducted on July 21th, 2015 (21 days after deployment for Structure 1 and 2), December 12th, 2015 (6 months after deployment for Structure 1 and 2; 47 days for Structure 3), and September 2nd, 2016 (14 months after deployment for Structures 1 and 2; 10 months for Structure 3). During each survey, the HROV used the sonar to locate the acoustic reflector and approach each structure. The images, with a resolution of 1024 x 768 px, were subsequently collected by encircling each of the structures, while maintaining the gorgonians in the centre of the view. Robot maintained an approximately constant distance of 2 m between the camera and the centre of the structure, enabling observations of the gorgonians from various directions with sufficient image quality, required for performing successful assessment of their survivorship. Gorgonian survival was assessed by individually observing if each transplanted colony was still in place and alive (with no evidence of necrotic tissue).

The 3D reconstructions of the three structures deployed on the continental shelf with transplanted gorgonians (Fig. 2 and Supporting Information*) were made using an optical 3D reconstruction procedure, as described in Hernández et al. (2016). The final models are obtained through a series of steps, starting with the simultaneous optimization of the pose of the camera (at each moment of the image acquisition) and the sparse geometry of the structure, followed by densification of the geometrical representation, surface estimation and texture mapping.

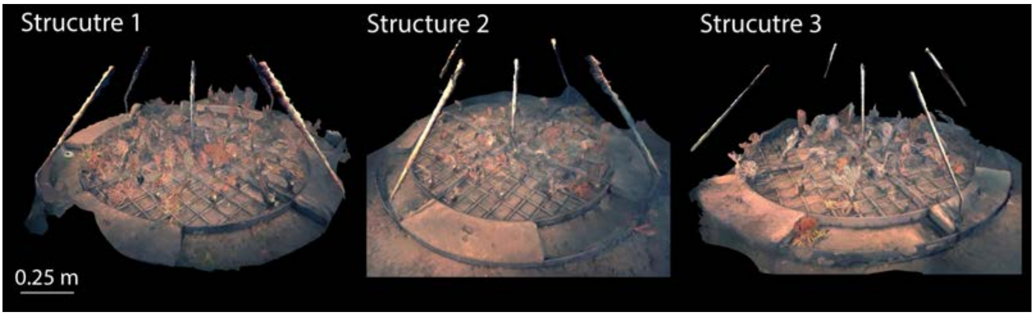

Figure 2. Three-dimensional reconstruction of the three structures deployed on the continental shelf with transplanted gorgonians. The 3D visualization of the Structure 1 can be found at Supporting Information. The 3D computer model was obtained from images acquired by the HROV while circling the structures, using the 3D reconstruction pipeline described in Hernández et al. (2016).

4 RESULTS

Several of the gorgonians collected from fishers showed partial breakage and little evidence of abrasion tissue. Even so, they all recovered and survived while being maintained in aquaria at the Institute of Marine Science prior to re-deployment at sea. On Structures 1 and 2, 98.8 ± 1.8% (mean ± SD) of the transplanted gorgonians were still in place at the time of the first survey (21 days after deployment), and they all were still surviving after 6 months at the time of the second survey. On Structure 3, 85% of the transplanted gorgonians were still in place at the time of the second survey (47 days after deployment). Finally, approximately one year after deployment (14 months for Structures 1 and 2, and 10 months for Structure 3) 87.5 ± 9.0% (mean ± SD) of the gorgonians were still in place and alive on the three structures (Fig. 3).

5 DISCUSSION

This pilot study has assessed, for the first time, the feasibility of successfully returning bycatch gorgonians recovered from artisanal fishery to their natural environment on the Mediterranean continental shelf. Initial results showed that, in spite of some *E. cavolini* colonies suffering partial breakage, tissue abrasion, or both, all colonies survived while being maintained in aquaria. This survival may be attributable to the species' high healing rate (0,085 mm of tissue recovery $\cdot d^{-1}$) (Fava et al. 2010). In contrast, other Mediterranean common gorgonians, such as the red gorgonian *Paramuricea clavata* (which is also frequently collected by artisanal fishers in Cap de Creus), shows low survival rates when recovered from bycatch and maintained in aquaria, with a rapid degradation of living tissues and high colony mortality (M. Montseny, pers. observation.). These observations highlight the importance of understanding the biological and ecological characteristics of each species before engaging in any restoration initiative (Montero-Serra et al. 2018a), and points at *E. cavolini* as a suitable gorgonian species for restoration projects in the Mediterranean continental shelf.

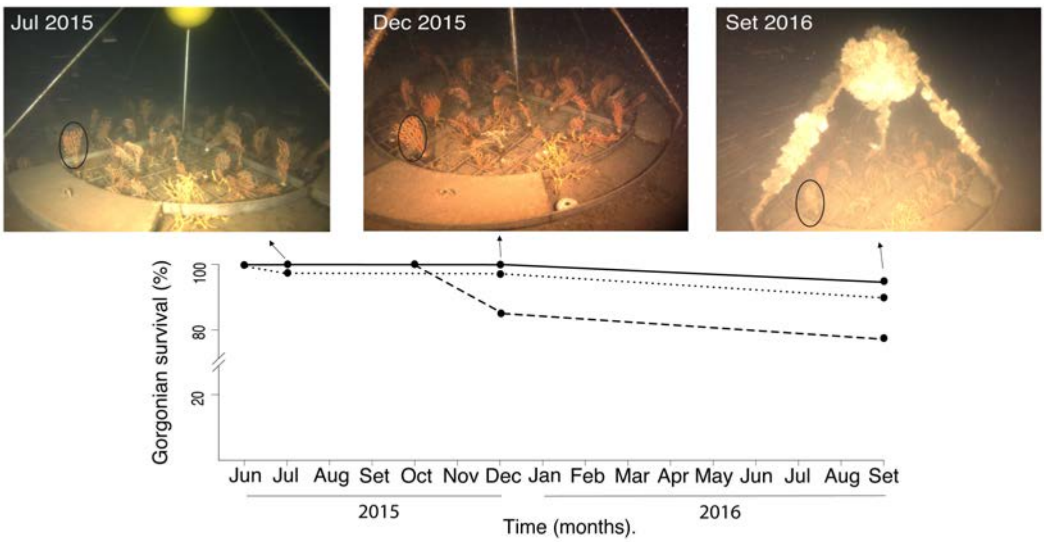

Figure 3. Survival rate (%) of transplanted gorgonians for each structure during the study period. (Solid line corresponds to Structure 1, dashed line to Structure 2, and dotted line to Structure 3). Pictures correspond to Structure 1 during the consecutive surveys.

Monitoring of structures shortly after their deployment (21 days or 47 days, depending on the structure) suggested that initial loss of gorgonians was mainly due to colony detachment during the structure deployment on the continental shelf (Fig. 3). Although we cannot strictly exclude natural mortality causes, the high survival following initial losses (Fig. 3), that is in accordance with previous gorgonian transplantations in Mediterranean shallower habitats (Linares et al. 2008; Fava et al. 2010), suggests that initial securement of a right colony attachment to the substrate is critical to their long-term survival with rather small effect of stress due to transplantation (Linares et al. 2008). Gorgonian transplants in the present study shown high survival (almost 85%) approximately one year after deployment, inline with the high survival observed for *Corallium rubrum* after 4 years from transplantation (about 99.1%) (Montero-Serra et al. 2018a), and much higher compared to transplanted *Eunicella singularis* (35 – 45% survival after 1 year), *Eunicella verrucosa* (30% survival after 1 year) and *P. clavata* (35 – 50% survival after 1 year) (Linares et al. 2008; Fava et al. 2010; Montero-Serra et al. 2018a).

In shallow Mediterranean environments, survival of transplanted gorgonians can be compromised by several environmental parameters generally associated with seasonal fluctuations, such as high-water turbulence, high irradiance, algal competition (Weinberg 1979; Linares et al. 2008) and thermal stress (Fava et al. 2010). Long-term survival of *E. cavolini* transplants on the continental shelf may thus be partially explained by the higher stability of environmental factors in deeper habitats (below ~ 40 m depth) (Garrabou et al. 2002; Grinyó et al. 2018). Indeed, outcomes from shallow restoration studies in tropical ecosystems under high environmental stability, are in accordance with the high gorgonian survivorship detected in the present study (Guzman 1991; Edwards and Gomez 2007). However, tropical corals encompass species with contrasting life history traits, including fast to slow growing species (Darling et al. 2012), which make tropical transplant survival rates highly variable (43% to 95% during the first year) (Yap et al. 1992; Lindahl 2003; Young et al. 2012). Therefore, the high survival rate detected in this study is consistent with the notion that slow-growing species require little initial transplantation effort, since they show high survival rates after transplantation in comparison with fast-growing species, but the period required to fully reestablish habitat complexity will tend to be far longer (Montero-Serra et al. 2018a).

In comparison to shallow water restoration studies, there are only a few instances of ecological restoration attempts in deeper habitats. The first such attempt to restore a deep-sea coral ecosystem was conducted with the CWC *O. varicosa* in Florida, where restoration modules were deployed at 70 – 100 m depth with colonies transplanted that showed moderate survival rates (50 – 60%) after one year (Brooke et al. 2006). In Sweden, a restoration action focused on the CWC *L. pertusa* recorded high survival of the transplants (76%) which increased by 36% in their size after more than three years (Dahl 2013; Jonsson et al. 2015). Similarly, an *in situ* growth study showed high survival (> 90% polyp survival) and active colony growth of *L. pertusa* fragments deployed during 1 year at ~500 m depth in the northern Gulf of Mexico (Brooke and Young 2009).

These first pilot studies (including the present one) demonstrate the feasibility of the active restoration of CWC reefs and coral gardens, which should encourage future initiatives aimed at recovering, preserving, and sustainably managing these VMEs. However, ecological restoration of intermediate depths and deep-sea habitats involves considerable constraints due to the difficult access, requiring for the use of advanced underwater technology entailing high economic cost. Deep-sea restoration cost per hectare has been estimated at two to three orders of magnitude higher than for shallow marine ecosystems (Van Dover et al. 2014). Future availability of accessible cost-effective underwater technology (such as relativity low-cost AUV for monitoring) will be paramount for the wide application and up-scale of corals and gorgonians restoration at depths below conventional or technical scuba diving limits.

The ultimate goal of restoration initiatives should be to achieve the recovery of the structure and ecological functioning of affected ecosystems (Society for Ecological Restoration International Science & Policy Working Group 2004; McDonald et al. 2016). For coral gardens, restoration of sessile engineering species can drastically alter the abiotic system state triggering a consequent response in the biotic state (Byers et al. 2006), such as that transplanted gorgonians not only provide habitat structure, but also enhance the recovery of its associated biodiversity and positively influence ecosystem functioning (Geist and Hawkins 2016). Overall, although restoration is often a long-term investment and its potential results are still highly uncertain (Suding 2011; Van Dover et al. 2014), the results of this pilot project highlight the feasibility of using bycatch gorgonians recovered from artisanal fishery to mitigate the fishing-related degradation by restoring coral gardens on the Mediterranean continental shelf. This is a first essential step that leads to future large-scale and cost-effective restoration actions of coral gardens located on the continental shelves or even in deeper environments. In contrast to most restoration practices using coral transplants obtained from fragmentation of donor colonies (Brooke et al. 2006; Dahl 2013), restoration based on bycatch gorgonians would minimize damage to other colonies or populations. Nevertheless, to be effective, these restoration actions should be accompanied by a reduction of fishing impacts in the restored areas, by partial closures or by improving fishing techniques.

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Supporting Information*:

Video S1: 3D visualization of Structure 1. The 3D computer model was obtained from images acquired by the HROV while circling the structures during the last survey (14 months after its deployment) and using the 3D reconstruction pipeline described in Hernández et al. (2016). Available online in: http://onlinelibrary.wiley.com/action/downloadSupplement?doi=10.1002%2Faqc.3118&file=aqc3118_sup-0001-supplementary+material.mp4.

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CHAPTER 3

A new large-scale and cost-effective restoration method for cold-water coral gardens.

1 ABSTRACT

1. Gorgonians dwelling on the Mediterranean continental shelf are among the most frequent fishing bycatch taxa. These species display several traits, such as long lifespans and slow growth, which make them very vulnerable to the impacts caused by fishing activities with far-reaching and long-lasting effects.
2. Hence, restoration and mitigation actions are crucial to enhance and speed up the natural recovery of damaged cold-water coral (CWC) gardens. Given the growing concern to develop effective and affordable restoration actions, the present study aims to propose and technically validate a new large-scale and cost-effective restoration method.
3. This technique, named "badminton method", consists in attaching bycatch *Eunicella cavolini* (Koch, 1887) colonies to cobbles supports and returning them to the continental shelf by gently throwing the gorgonian transplants directly from a boat.
4. Two consecutive field experiments were conducted in order to find the best cobble type support and gorgonian size to be used: first, to evaluate the landing efficiency of gorgonian transplants at different depths (from 5 to 30 m) and second, to evaluate their capability to maintain a correct position over time.
5. Natural cobbles with large gorgonians attached were the best option. Field results and modelling approaches suggest that the transplants would correctly land on the continental shelf seabed in a predicted area of around 60 m^2. Moreover, they would successfully maintain an upright position ensuring gorgonian survival over time.
6. The success of this method highlights the feasibility of large-scale and low-cost restoration actions with promising results for the conservation and recovery of CWC gardens.

2 INTRODUCTION

Over the last decades, the increasing attention towards deep water ecosystems and the continued advancements in deep sea exploration technology have contributed to expand knowledge about benthic communities dwelling on continental shelves (~60 – 200 m depth) and deep sea bottoms (> 200 m depth) (Freiwald et al. 2004; Althaus et al. 2009). At depths of 60 to 1000 m, CWC and sponges are among the main habitat-forming species, generating complex three-dimensional ecosystems that create hotspots of biodiversity over large areas (Hovland 2008; Roberts et al. 2009). They provide suitable habitat, acting as feeding, reproductive, nursery and refuge areas for a wide variety of associated species, many of which are of commercial interest (Krieger and Wing 2002; Henry and Roberts 2007; Cartes et al. 2013). At the same time, CWC reefs, coral gardens and sponge grounds also play an important role in the benthic-pelagic transfer of energy and matter(Graf 1989; Gili and Coma 1998) as well as in most of the biogeochemical cycles of the deep sea (Van Oevelen et al. 2009; Cathalot et al. 2015; Coppari et al. 2016).

The continuously increasing exploitation of deep sea resources is currently recognized as a major threat (Jackson et al. 2001; Morato et al. 2006). Since the last century, continental shelves and upper slopes have been heavily impacted by fishing activities (Jones 1992; Koslow et al. 2000; Oberle et al. 2017). Bottom trawling represents one of the main threats to benthic ecosystems (Hall-Spencer et al. 2002; Clark et al. 2016), resulting in a severe oversimplification of benthic communities (Watling and Norse 1998; Thrush and Dayton 2002; Reed et al. 2007). However, in rocky areas with complex topography inaccessible to trawling, artisanal fishing can also jeopardise the integrity of benthic ecosystems (Sampaio et al. 2012; Angiolillo and Canese 2018).

In the Mediterranean Sea, gorgonians are among the most frequent species of bycatch of artisanal trammel net and longline fishing in coastal areas and on the continental shelf. Due to their arborescent morphology and erect position, gorgonians are easily entangled in bottom-contact fishing gears (Mytilineou et al. 2014; Deidun et al. 2015). These fishing gears can remove entire gorgonian colonies, or partially damage their tissue, making them more vulnerable to diseases and/or epibiont overgrowth (Bo et al. 2014a, b; Angiolillo et al. 2015). Since most Mediterranean gorgonians are long-lived, slow-growing species, with delayed sexual maturity and limited recruitment (Coma et al. 1998; Linares et al. 2007), fishing activities impacting their populations can have far-reaching and long-lasting negative effects in their long-term viability (Bo et al. 2014b, 2015). As a consequence, Mediterranean CWC gardens located on the continental shelf and slope have recently been internationally recognized as Vulnerable Marine Ecosystems (VMEs) (OSPAR 2010; Fabri et al. 2014), stressing the urgent need for their sustainable management and conservation (Davies et al. 2007; FAO 2009; Aguilar and Marín 2013). The conservation and recovery of benthic engineering species like gorgonians will also help to preserve all their associated fauna, maintaining the ecosystem functioning and the ecosystem services provided (Byers et al. 2006; Geist and Hawkins 2016). It is, thus, highly desirable to actively initiate or improve the slow natural recovery of impacted CWC assemblages, by means of reducing impacts and assisted regeneration with biotic intervention (i.e., active restoration) (Van Dover et al. 2014; Possingham et al. 2015).

Ecological restoration offer the opportunity to redirect the environmental damage caused by anthropogenic impacts and assist the recovery of a natural range of ecosystem composition, structure, and dynamics (Society for Ecological Restoration International Science & Policy Working Group 2004; Falk et al. 2006). However, until now, most of the actions and initiatives carried out in marine ecosystems have been concentrated on the restoration of shallow habitats, particularly in tropical ecosystems (e.g., Guzman 1991; Rinkevich 2005b; Young et al. 2012). Only a few restoration attempts have been carried out in the deep sea, specifically targeting CWC species (Brooke et al. 2006; Dahl 2013) and coral gardens on the Mediterranean continental shelf (60–120 m depth, Montseny et al. 2019). High survival values of coral and gorgonian transplants were found in these studies, highlighting the feasibility of active restoration of CWC reefs and coral gardens, despite the considerable limitations associated with the difficulties of working at intermediate and deep depths (Clauss and Hoog 2002).

Among these constraints, the need for advanced underwater technologies is considered to increase the economic cost of any deep sea restoration action compared to shallow areas (Van Dover et al. 2014). Likewise, the difficult and extremely limited access to these habitats also restricts restoration actions and their monitoring to small spatial extents (Brooke et al. 2006; Dahl 2013). However, since the main stressors impacting most continental shelf and deep sea ecosystems are largely widespread (Halpern et al. 2008), scaling up and reducing economic costs are necessary steps to ensure a long-term viability of deep sea ecological restoration actions.

Based on the demonstrated capability of bycatch gorgonians to survive after their transplantation back into their natural environment (Montseny et al. 2019), the main goal of the present study was to explore the technical viability of a new large-scale and cost-effective restoration method. Gorgonians colonies recovered from artisanal fishery bycatch (trammel nets) were attached to cobble supports, and transplanted to the continental shelf by gently throwing them from the sea surface. The landing efficiency (e.g., successful landing in upright position and dispersion area of transplants) and the capability to maintain the upright position over time were estimated according to depth, the characteristics of the cobbles used, and the size of the transplanted gorgonians. The method has been named "badminton method" because the fan-shaped morphology of the colonies attached to the cobbles makes them act as a badminton shuttlecock, slowing down the fall and facilitating an upright landing. It was inspired by the common practice of artisanal fishers from Menorca (Balearic Islands, Spain) to return to the sea gorgonians that they accidentally catch attached to maërl cobbles, by throwing them from the sea surface.

3 METHODOLOGY

2.1 Studied Species

E. cavolini is one of the most common Mediterranean gorgonian species (Carpine and Grasshoff 1975; Weinberg 1976), showing a wide bathymetric distribution (<10 – 220 m depth) (Russo 1985; Bo et al. 2012). In general, this non-symbiotic gorgonian presents fan-shaped colonies with a variable ramification pattern and growth rate, depending on environmental conditions. Commonly, the ramifications are numerous and tend to be curved, pointing in many directions although lying in one plane, fully exposed to the main current flux (Velimirov 1973; Weinbauer and Velimirov 1995). In the Mediterranean Sea, the mean height of the entire colony is around 15 cm (Sini et al. 2015) with largest colonies reaching 50 cm height (Bo et al. 2012). In the Cap de Creus area (north-western Mediterranean Sea, 42° 19′ 12″ N – 003° 19′ 34″ E), this species is abundant on the outcropping rocks at 80 – 120 m depth on the continental shelf, forming high density patches reaching densities of 25 colonies m^{-2} (Gili et al. 2011; Dominguez-Carrió 2018).

2.2 Gorgonian collection and maintenance

Bycatch colonies of *E. cavolini* (5 to 17 cm height) were collected from trammel nets fishing during summer 2016 with the collaboration of artisanal fishers. The collected gorgonians were kept on board in a bucket filled with surface sea water (~22–25 ºC). Once on land, colonies were kept in a 100 L tank filled with sea water maintained at 14 ± 1.0 °C and with a submersible pump providing continuous water movement. A chiller (Tank chiller line TK 2000) was used to maintain a constant sea water temperature, and the water was filtered using a biological filter (SERA fil bioactive 250+UV). As soon as it was possible (after one or two days) gorgonians were moved to the Institute of Marine Sciences (ICM-CSIC) in Barcelona (Spain) without being adversely affected during the transport. There, gorgonians were held in a 100 L tank with a continuous flow of Mediterranean seawater pumped from a depth of 15 m at a rate of 50 L h^{-1} and filtered with a 100 μm pore size (Olariaga et al. 2009). A submersible pump provided continuous water movement in the tank with a flow rate of 320 L h^{-1}, seawater temperature was constantly maintained at 14 ± 1.0 °C and gorgonians kept in dark conditions, corresponding to the natural conditions of the Cap de Creus continental shelf (Dominguez-Carrió 2018). Gorgonians were held under these conditions during a few weeks up to 5 months, being fed three times a week with frozen *Cyclops* (Crustacea, Copepoda).

2.3 Gorgonian transplants preparation

In order to identify the best support to be used in this gorgonian restoration method, three different types of cobbles were tested: (A) natural cobbles, (B) small artificial cobbles and (C) large artificial cobbles (Fig. 1). Natural cobbles were collected from the coastal area of Cap de Creus (mean width: 9.7 ± 1.6 cm, mean length: 12.4 ± 1.6 cm, mean height: 3.1 ± 1.0 cm, mean weight: 455 ± 139 g). Conversely, artificial cobbles with established measurements were built with concrete: small artificial cobbles had a square shape (width: 8.0 cm, length: 8.0 cm, height: 2.5 cm, weight: 175 g); whereas large artificial cobbles had a semi-spherical shape (diameter: 13.3 cm, height: 4.5 cm, weight: 450 g). Thirty cobbles in total were used (10 per cobble type). In order to identify the most effective gorgonian size to be transplanted, large-sized gorgonians (10 – 17 cm height, 80 – 300 cm total ramification length) were transplanted on five cobbles per type, and small-sized gorgonians (5 – 10 cm height, 20 – 80 cm total ramification length) were transplanted on the other five cobbles per type. Gorgonians were attached to supporting cobbles using epoxy putty (Corafix SuperFast, GROTECH ®). All colonies used as transplants did not show any signal of necrotic tissue before being returned to the sea.

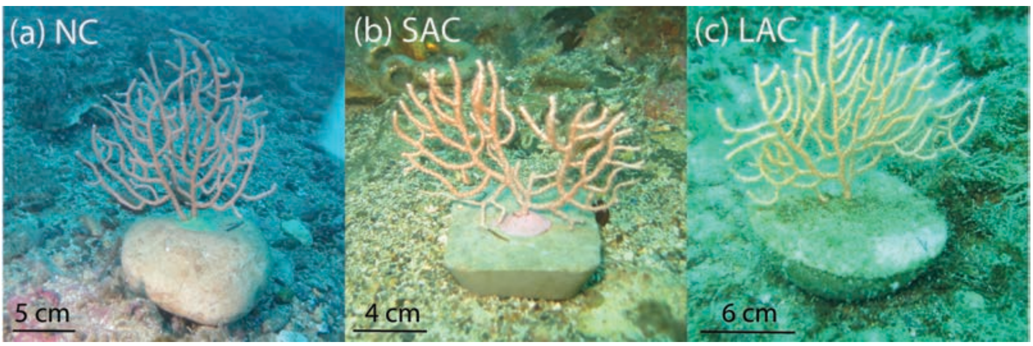

Figure 1. Gorgonian transplants with different types of cobbles tested in the study. (A) Natural cobbles (NC), (B) Small artificial cobbles (SAC) and (C) Large artificial cobbles (LAC).

2.4 Landing efficiency

The first part of the field study consisted of assessing the landing efficiency of the gorgonian transplants gently thrown from the sea surface. The dispersion area on the sea floor and success of upright landing were quantified at 5 m, 10 m, 20 m and 30 m depth (Cap de Creus, 42° 28′ 39″ N – 003° 28′ 31″ E). At each depth, the ten cobbles with five large and five small-sized gorgonians transplanted were gently thrown from a boat. Once at the sea floor, and by means of scuba diving, a hand-drawn map of the ten transplants was produced by measuring *in situ* the perpendicular distance (cm) between the position of each transplant from a central transect established in the area where the transplants had landed. Finally, the landing success of each gorgonian was estimated by recording *in situ* whether the colonies had landed in an upright or overturned position. This procedure was repeated five times for each cobble type and depth.

2.5 Monitoring of transplants

The second part of the field study consisted of assessing whether the transplants, once on the sea floor, maintained the correct upright position over time. To this aim, new gorgonians with no signs of necrosis, were attached to the same three types of cobbles, as previously described. Again, large-sized gorgonians were transplanted on five cobbles per type, and small-sized gorgonians were transplanted on the other five cobbles per type. The resulting thirty cobbles (10 cobbles per type) were placed by (hand in upright position on a horizontal bottom composed of small cobbles and sand at 30 m depth (Cap de Creus, 42° 17′ 03″ N – 003° 17′ 95″ E). During three consecutive months from February to May 2017, a scuba diving monitoring was monthly performed. Colonies were individually photographed on a ruled tablet in order to acquire pictures of the transplants with a size reference. From these pictures, the position of the transplants (upright or overturned) was established, and for each gorgonian the amount of necrotic tissue along the total ramification length was quantified to estimate their survival and health status.

2.6 Data analysis

To analyse the cobbles probability of landing success (response variable) according to depth (explanatory continuous variable), cobble type and gorgonian size (explanatory discrete variables), two model types were fitted: (1) a general linear model (GLM) with a binomial error distribution and a logit link function, (2) a general linear mixed model (GLMM) with a binomial family, identifying individual cobbles as random effect. We finally only plotted the simplest GLM binomial model, which presented the lowest Akaike's Information Criterion with small-sample correction value (AICc) (see Supplementary Material 1). To test the significance effects of depth, cobble type and gorgonian size, an Anova Type II test for non-sequential factors, was applied over the fitted model (GLM). A GLM model with a binomial error distribution and a logit link function was also used to predict the probability of successful landing in a hypothetical situation of throwing the transplants up to 80 m depth according to cobble type. Additionally, given the great variability in shape and weight among the natural cobbles used in the experiment, we explored the relationship between the Sphericity Index (SI) and the weight of cobbles, and their probability of landing upward, using a Spearman rank order correlation (Spearman 1904). The SI of cobbles (Ψp) was calculated from their maximum length, width and thickness measures (Sneed and Folk 1958).

The total dispersion area (m^2) of the transplants on the sea floor was measured from the maps obtained *in situ* by scuba diving. For each cobble type and depth, the area corresponding to the smallest ellipse that included all the transplants was quantified using the Macnification 2.05 software (Orbicule, Leuven, Belgium). To estimate the area of dispersion in which transplants would extend on the continental shelf (80 m) according to cobble type, different statistical models were built. Depth (continuous variable) and cobble type (discrete variable) were the explanatory variables, while the dispersion area was the continuous response variables. Several GLM models, non-linear regression model, mixed linear model and GAM model were adjusted to detect which of them showed the lowest AICc value and therefore better fit the data.
Regarding the monitoring transplant study, the evolution of their position (upright or overturned) according to the cobble type was plotted during the three-month period. Additionally, a Wilcoxon signed-rank statistical test (Wilcoxon 1945) was performed at the end of the monitoring to detect the influence of gorgonian size in overturning the transplants. In order to evaluate the status of transplanted colonies and compare them to natural values (Linares et al. 2008b), all colonies were photographed during the monthly monitoring at 30 m depth and they were analysed with the Macnification 2.05 software (Orbicule, Leuven, Belgium). At the end of the three months of experiment, the percentage of necrosis on the total ramification length ((necrotic tissue length/ total linear length) *100) was calculated for each colony that maintained the upright position. Additionally, a Wilcoxon signed-rank statistical test (Wilcoxon 1945) was used to detect differences in percentage of necrosis between large and small studied gorgonian sizes. Statistical analyses were carried out with R (RCore Team 2018) by means of the R Studio software (RStudio Team 2016) using the "ggplot2" package (Wickham 2016), the "lme4" package (Bates et al. 2015) for the linear mixed model and the general linear mixed model, and the "gam" package (Hastie and Tibshirani, 1990) for the GAM model. The "AICcmodavg" package (Mazerolle 2019) was also used in the model comparison.

3. RESULTS

3.1 Landing efficiency

Depth, cobble type and gorgonian size affected the landing success of transplants (Anova Type II of the binomial model; Depth: chi-squared = 16.045; df = 1; p-value < 0.001; Cobble type: chi-squared = 29.463; d f= 2; p-value < 0.001 Gorgonian size = chi-squared = 22.314; df= 1; p-value < 0.001. Fig. 2 and Supplementary Material 2). For all the cobble types tested, and regardless of the gorgonian size, the probability of landing upright increased with depth (Fig. 2). Transplants with small artificial cobbles displayed the highest probabilities of successful landing at all depths (close to 100% of upright landing; Fig. 2), followed by large artificial cobbles and natural cobbles. According to the gorgonian size, large gorgonian colonies showed a higher probability of upright landing than small colonies (Fig. 2). Regarding the prediction model, a 100% success of gorgonian landing in upright position at 80 m depth (i.e., on the continental shelf), for all the cobble types was forecast (Fig. 3 and Supplementary Material 3). Focusing only on natural cobbles, there was a strong positive relationship (Spearman's r = 0.80; p-value = 0.004; N = 10) between the landing success of natural cobbles (% of upright landing) and their SI (Fig. 4). On the contrary, there was no correlation between the weight of the cobbles and their probability of landing success.

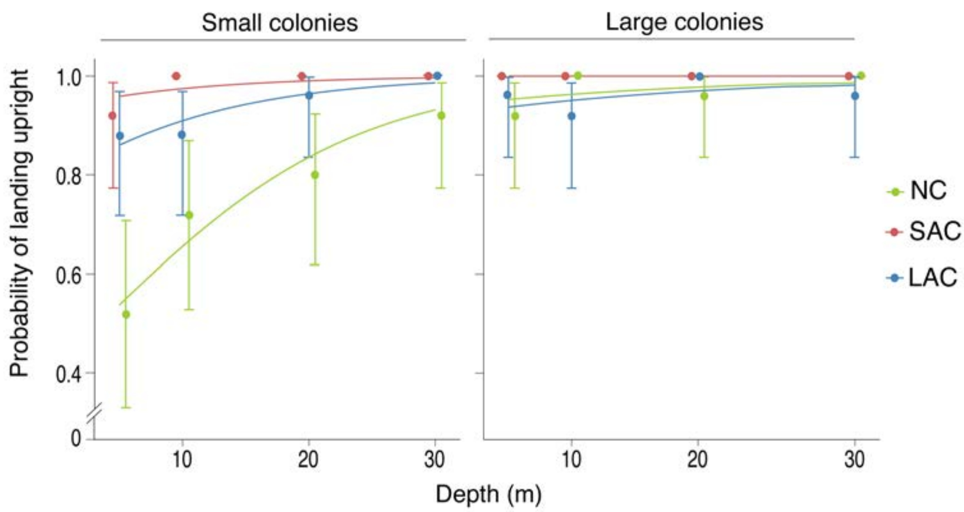

Figure 2. Transplants' probability of landing in upright position according to depth, cobble type and gorgonian size. Dots show real observations with an interval confidence of 95% and lines correspond to the predicted values of the GLM model for binomial data (NC = Natural cobbles, SAC = Small artificial cobbles and LAC = Large artificial cobbles); N = 120.

Regarding the dispersion area reached by the gorgonian transplants on the sea bottom, among all the models tested the one providing the best fit was a GLM with gamma distribution (AICc = 73.9) (Fig. 5 and Supplementary Materials 4 and 5). The model showed that depth and cobble type significantly influenced the dispersion area reached by transplants (Anova Type II; Depth: LR chi-squared = 111.657, df = 1, p-value < 0.001; Cobble type: LR chi-squared = 22.96, df = 2; p-value < 0.001), predicting an increase of the dispersal area with depth (Fig. 5). Natural cobbles showed the largest dispersal area at 80 m depth (60.8 ± 20.6 m^2, mean ± SD), followed by small artificial (47.6 ± 16.3 m^2) and large artificial cobbles (34.3 ± 11.6 m^2).

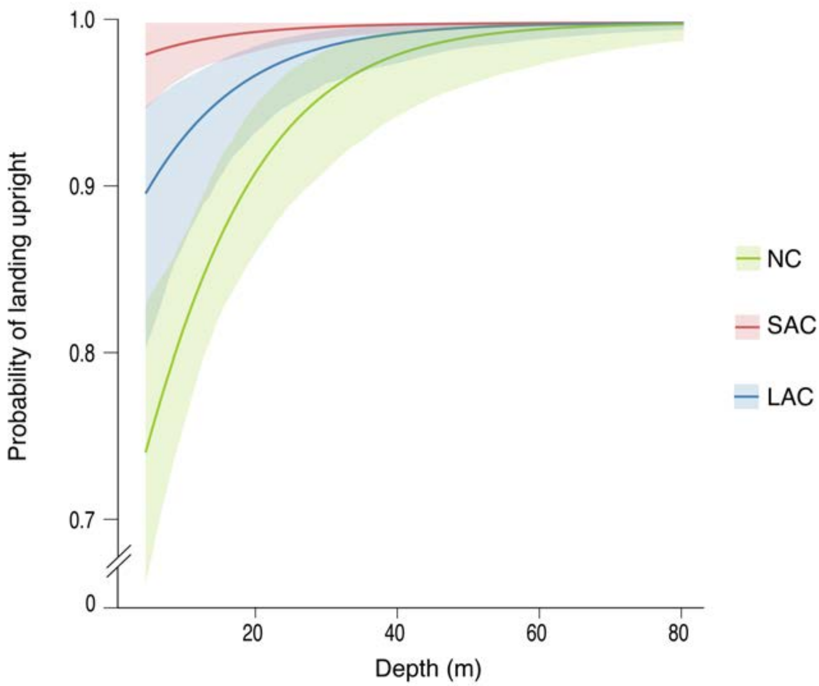

Figure 3. Predicting model of transplants probability of landing in upright position according to depth and cobble type, using a logistic general linear model (GLM) for binomial data. (NC = Natural cobbles, SAC= Small artificial cobbles and LAC= Large artificial cobbles). Colour shadows correspond to 95% intervals confidence corresponding to the GLM model.

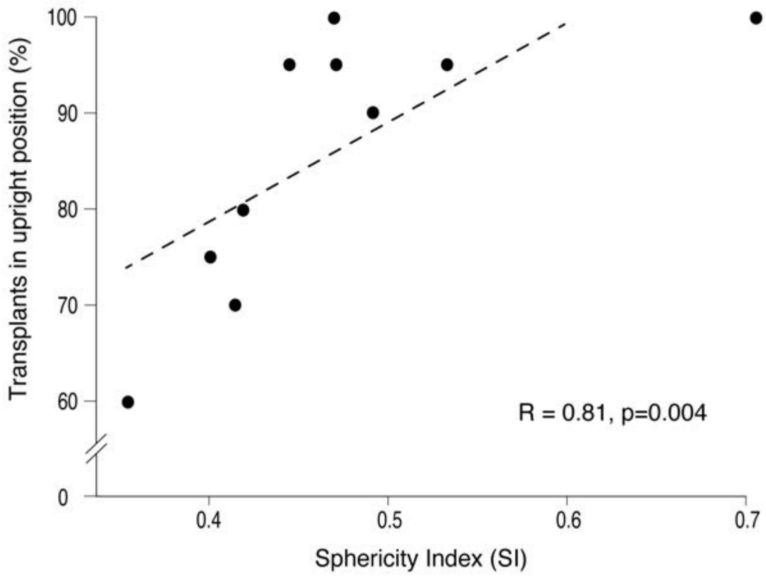

Figure 4. Scatter plot showing the correlation between percentage of upright landing and the Sphericity Index (SI) of natural cobbles (N=10).

3.2 Monitoring of transplants

All the cobbles located at 30 m depth remained in an upright position during the first month. During the second and third month, all the natural cobbles (100%) and almost all the small artificial cobbles (90%) remained in an upright position, contrasting with the lower percentage (60%) of large artificial cobbles that maintained an upright position (Fig. 6). The size of the gorgonian colony did not influence the overturning of the transplants since statistical differences were not detected in the position of the transplants at the end of the monitoring according to gorgonian size ($W = 3.5$, p-value = 0.814). The mean necrosis percentage of the gorgonian transplants at 30 m depth that remained in an upright position during all the monitoring period was very low and not statistically different between gorgonian sizes ($W = 58$, df =1, p-value = 0.281), being $3.65 \pm 6.4\%$ in large colonies and $5.63 \pm 5.7\%$ in small colonies. In the most affected colony, the necrotic tissue represented 17.6% of the total ramification length (Fig. 7).

4 DISCUSSION

The present results have shown that the "badminton method" can be a cost-effective restoration method to potentially restore CWC gardens over large scales.

The probability of successful landing of transplants increased with depth for the three cobble types tested, and by using gorgonians colonies of large size (Fig. 2). Gorgonian colonies attached to cobbles act similar to a badminton shuttlecock, slowing down the fall and forcing an upright landing of the transplants due to the hydrodynamic resistance of the highly branched surface of the gorgonian. Consequently, by using large colonies instead of small ones, this effect is enhanced. Regarding transplant support types, the small artificial cobbles were the most effective at all depths, but all the three support types showed high probabilities of successful landing at 30 m depth, if large colonies were attached (more than 90% of upright landing) (Fig. 2). Accordingly, the model predicted a 100% landing success for all the cobble types returned to the continental shelf (80 m depth; Fig. 3).

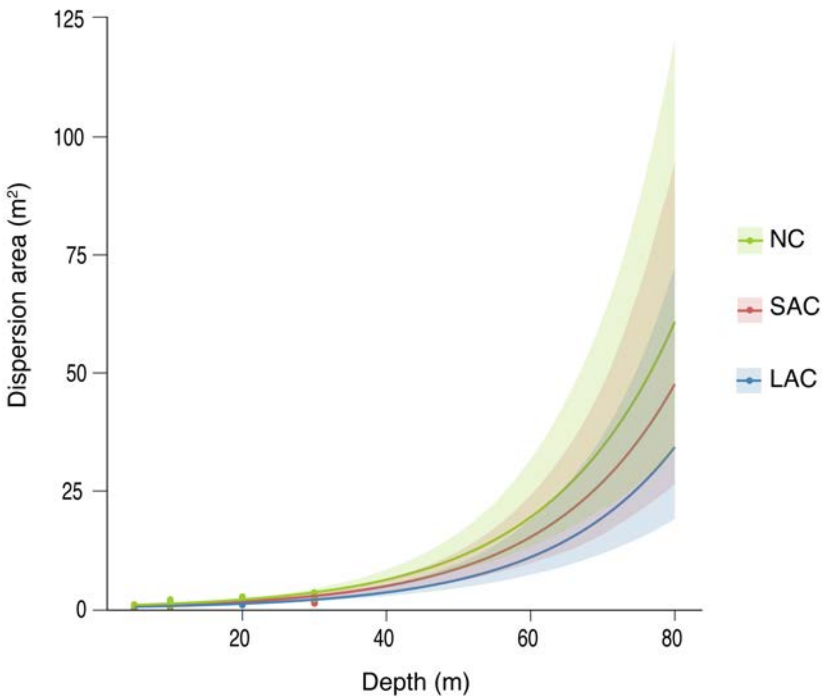

Figure 5. Model predictions of the dispersion area reached by the gorgonians transplants according to cobble type at different depths, using a GLM model with gamma distribution. Dots show real observations and lines with shadow correspond to the predictions of the model and its corresponding 95% intervals confidence. (LAC = Large artificial cobbles, SAC= Small artificial cobbles and NC= Natural cobbles); N = 120.

Once on the sea floor, remaining in an upward position is crucial for the survivorship of the gorgonian transplants. The size of the gorgonian colony did not influence the overturning of the transplants whereas the cobble type did. Natural and small artificial cobbles successfully maintained the upright position after three months at 30 m depth (100% and 90% maintained the upright position, respectively), where hydrodynamic conditions are even stronger than in deeper areas on the continental shelf (Garrabou et al. 2002). In contrast, large artificial cobbles were less stable (60% maintained the upright position) and more easily overturned due to their semi-spherical shape. There were no differences in the amount of necrotic tissue between large and small transplanted colonies that remained upright over the three months. Such results highlight that all cobbles remaining upward were sufficiently heavy to hold the transplanted colonies, without causing any movement that could result in abrasion of the tissue.

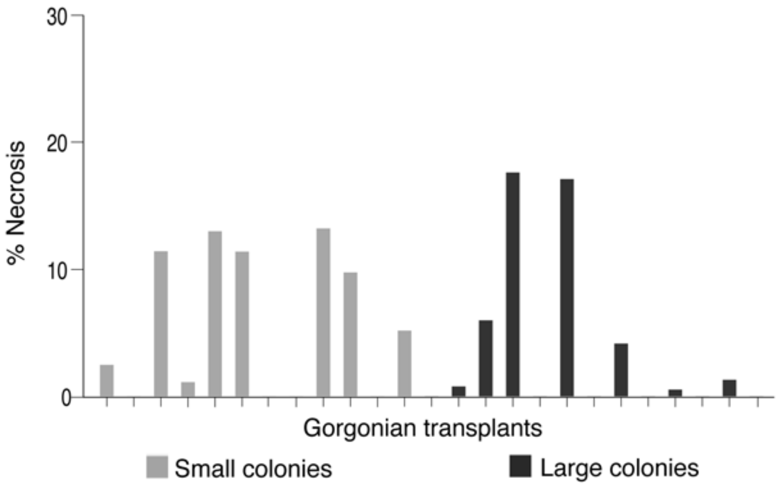

Figure 6. Proportion of cobbles that maintained the upright position at 30 m depth according to cobble type during three months. (LAC = Large artificial cobbles, SAC = Small artificial cobbles and NC = Natural cobbles); N = 30.

Based on these results about the landing efficiency and the capability of the transplants to maintain an upright position once on the sea floor, natural cobbles with large gorgonians attached are to be considered as the best option to ensure the success of this restoration method. Using large colony transplants will be further advantageous, since they have been shown to suffer less natural mortality after transplantation (Brooke et al. 2006), and have higher capacity to regenerate from injuries, being more resilient to further impacts (Henry and Hart 2005; Fava et al. 2010). Likewise, reintroduction of large gorgonians will foster the recovery of the three-dimensional structure of the coral gardens, and their habitat-forming functions, providing habitats for a large number of associated species (Horoszowski-Fridman et al. 2015; Geist and Hawkins 2016). Indeed, the recovery of ecosystem functioning is by definition one of the main goals of ecological restoration (Society for Ecological Restoration International Science & Policy Working Group 2004).

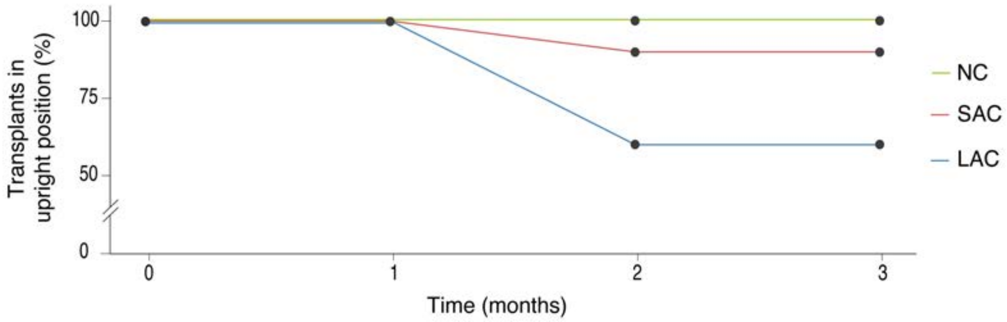

Figure 7. Percentage of necrosis in gorgonian transplants that maintained the upright position at 30 m depth during three months (small colonies N= 12 and large colonies N=13).

By using natural cobbles, no artificial material is introduced to the environment. This is in agreement with the recommendation of using natural substratum, as highlighted for shallow water gorgonian transplant experiments, in which the ineffectiveness of using man-made devices (e.g., PVC-clamps and PVC-racks) was identified (Weinberg 1979). However, when selecting natural cobbles to be used for the restoration, flat cobbles should be avoided. Indeed, our results showed a strong positive relationship between their sphericity of the natural cobbles and their probability of landing upwards, with the flattest cobbles landing worse (less percentage of landings in upright position) than the rounder ones (Fig. 4). During the fall, if the transplants were falling upside-down (i.e., cobbles would land overturned), even large gorgonians do not provide enough hydrodynamic resistance to reverse the trend and overturn the flatter cobbles.

The variability in shape and weight of natural cobbles resulted in the largest predicted dispersion area (60.8 ± 20.6 m^2) of gorgonian transplants on the continental shelf (80 m depth) (Fig. 5). This could reduce the chance of two or more transplants landing on top of each other, enabling to restore large areas in the continental shelf. Restoring large areas has already been successful in tropical shallow environments, where coral reef restoration has been performed at scales from ten square metres to several hectares (Edwards and Gomez 2007). One of the main goals of ecological restoration is to restore a natural range of ecosystem composition, structure and dynamics (Society for Ecological Restoration International Science & Policy Working Group 2004; Falk et al. 2006). In this sense, and taking into account that 60 m^2 would be the mean area reached on the continental shelf by ten gorgonians transplants thrown from the surface, we should consider performing multiple throws at the same site to achieve gorgonian densities similar to natural populations of *E. cavolini* dwelling on the Mediterranean continental shelf (on average 2.8 ± 4.3 colonies m^{-2}, with some high density patches above 25 colonies m^{-2}) (Dominguez-Carrió 2018).

Concerning the survival of the transplants, necrosis observed after the three months monitoring (less than 20% of necrosis in the most affected colonies and less than 6% in average, Fig. 7) is similar to values recorded for other Mediterranean gorgonian species in natural conditions. For instance, the percentage of necrotic tissue among *Paramuricea clavata* colonies ranged from 0 to 22% with a mean of 10% (Coma et al. 2004; Linares et al. 2008b), and maximum 24% of necrosis tissue was reported among *Eunicella singularis* colonies, with a mean of 5% (Linares et al. 2008b). Water turbulence (influenced by strong wave action), high irradiance, and algal competition can compromise the survival of gorgonian transplants in shallow environments (Weinberg 1979; Linares et al. 2008a). During the monitoring period these stressor drivers could have affected our transplants settled much shallower (30 m depth) than the natural distribution range of the species in the study area (60 – 120 m depth, Dominguez-Carrió 2018; Gili et al. 2011). Even so, the amount of necrosis detected after the three months monitoring was still low. The overall efficiency of the "badminton method" is in line with the results obtained from other coral and gorgonian transplantation experiments. A review of 40 studies about hexacoral restoration actions reported a mean annual survival of 60% with a range of 6–98% (Montero-Serra et al. 2018). For Mediterranean gorgonian species, such as *E. singularis, Eunicella verrucosa, P. clavata* and *Corallium rubrum,* the mean annual survival observed was 48% (ranging between 30 and 99%) (Linares et al. 2008a; Fava et al. 2010; Montero-Serra et al. 2018). Referring to the scarce information about CWC restoration experiments, these showed, during the first year, a mean survivorship of the coral transplants around 50–60% (Brooke et al. 2006; Boch et al. 2019).

Due to CWC ecosystems' role as biodiversity hotspots (Henry and Roberts 2007; Baillon et al. 2012), awareness about the need and the importance of their protection and restoration is increasing worldwide (Davies et al., 2007; WWF/IUCN, 2004). First attempts towards the assisted regeneration (active ecological restoration) of these ecosystems demonstrated the high survival of transplants for reef-forming CWCs *Oculina variciosa* (Brooke et al. 2006) and *Lophelia pertusa* (Dahl 2013; Jonsson et al. 2015)) and for cold-water gorgonians (*E. cavolini*) (Montseny et al. 2019). However, technical and logistical difficulties of working below the limit of scuba diving commonly imply the use of underwater technology. Such technological requirements elevate the costs of restoration actions by two to three orders of magnitude compared to shallow areas (Van Dover et al. 2014). Moreover, CWC restoration actions performed up to date have been based on transplantation of coral fragments on artificial structures, which significantly limit the area that can be restored (Brooke et al. 2006; Dahl 2013; Montseny et al. 2019). Thus, in order to increase the scale of restoration actions, many artificial structures would be needed, further increasing the economic cost and the technical constraints. In this sense, and given that the main threats impacting natural habitats occur on large scales (Halpern et al. 2008), it is a foremost challenge to develop effective methods for upscaling ecological restoration actions (Aronson and Alexander 2013; Perring et al. 2018). Indeed, a mismatch between the scale at which ecological restoration can currently be done and the scale at which major impacts act has been highlighted for tropical shallow coral reefs (Edwards and Gomez 2007; Montoya Maya et al. 2016; Pollock et al. 2017). Regarding this aspect, the "badminton method" allows for restoration of high number of gorgonians colonies over extended areas without the need for high-cost underwater technology. Moreover, by using bycatch gorgonians, no additional impact to healthy donor coral gardens will be generated, while a viable output for those gorgonians already fished is provided. Finally, directly involving professional fishers in restoration actions, will also increase the awareness of local society about the need for the protection of CWC gardens and would facilitate the application of this methodology in an extensive manner, which is crucial for the restoration success (Hull and Gobster 2000; Yap 2000).

The current and future perspectives about the degradation of continental shelves (Halpern et al. 2008) and the urgency for conservation and restoration of their benthic communities (Borja 2005) supports the development of new methodologies like the one presented in this work. The present study shows that the "badminton method" can be a reliable cost-effective and large-scale restoration method to assist the recovery of cold-water coral gardens located beyond the limit of scuba depth. It should be highlighted that apart from being a valid technique for continental shelves and potentially for deep environments, it could also be applied shallower for the mesophotic coral ecosystems. While "badminton method" in this study has been demonstrated to be successful for sandy or gravel horizontal bottoms, further studies should explore its applicability to other bottom types, like those in upper-slope habitats or CWC reefs habitats.

The present study is a first evaluation of "the badminton method" that needs now to be followed by a real application in areas of the continental shelf in order to corroborate the long-term survival success predicted by the present results. Furthermore, the limited sample size, arising from the difficulties of working at depths (Clauss and Hoog 2002), results into predicted models with large margin errors constraining broader conclusions of this study. These limitations will be assessed and reduced base on a future monitoring of gorgonians transplanted on the continental shelf. Thanks to a consistent and long-term monitoring the successful of ecological restoration should be generally demonstrable within 10 – 50 years (Jackson et al. 1995; Suding 2011). As well, a general reduction of fishing impacts, together with an effective protection of CWC gardens needs to complement restoration actions. Protecting such habitats, not only would improve the success of any restoration action, but it would also provide benefits at lower costs and without the time delay required for restoration (Possingham et al. 2015).

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CHAPTER 4

Involving fishers in scaling up the restoration of cold-water coral gardens on the Mediterranean continental shelf.

1 ABSTRACT

1. Cold-water gorgonians dwelling on the continental shelf are a common bycatch of bottom-contact fishing practices. Given the slow growth and limited recruitment of cold-water gorgonians, impacts derived from fishing activities may seriously compromise the conservation of the highly complex coral gardens which they generate, as well as the abundant and highly diverse associated fauna.
2. For this reason, the development of effective active and passive restoration methods is nowadays a priority to enhance the natural recovery of impacted cold-water coral (CWC) gardens. However, ecological restoration of intermediate and deep-sea communities remains extremely limited, due to their technological requirements and associated costs bringing their wide-scale and long-term application into question.
3. This study reports the results of the first large-scale active restoration of more than 400 cold-water gorgonians on the Mediterranean continental shelf. By actively involving local fishers during two consecutive fishing seasons, bycatch gorgonians were recovered and returned to the continental shelf (at 80 – 90 m depth).
4. Two-years monitoring performed through Autonomous Underwater Vehicle (AUV) surveys revealed that 460 gorgonian transplants survived over an area of 0.23 ha. This reintroduced cold-water gorgonian population is compared to a reference natural population in terms of size and spatial structure. The cost of the restoration amounted to 140,000 €/ha, which is significantly less than for any deep-sea restoration actions performed to date.
5. The success of this cost-effective active restoration highlights the viability of large-scale restoration of impacted CWC communities, with promising results for the conservation and recovery of intermediate and deep-sea ecosystems.

2 INTRODUCTION

Anthropogenic impacts, which are increasing in terms of magnitude, scale, frequency, and diversity have disrupted ecosystem processes to a large extent and diminished over 60% of the ecosystem services, leading to a serious loss of biodiversity (Jackson et al. 2001; Millennium Ecosystem Assessment 2005; Mooney et al. 2009). Focusing on the marine environment, the escalation of human activities (e.g., fishing, oil and gas extraction, mining) and climate change are seriously imperiling marine ecosystem's biodiversity, functioning, stability, and resilience (Hughes 1994; Dulvy et al. 2008; Ramirez-Llodra et al. 2011). Anthropogenic impacts on the oceans show strong spatial heterogeneity and are mostly concentrated on continental shelf and slope areas (Halpern et al. 2008). In fact, half of the world's continental shelves are continuously being impacted by fishing activities, especially bottom trawling (Watling and Norse 1998; Pusceddu et al. 2014). Fishing practices directly damage benthic fauna, mainly engineering species (*sensu* Jones et al. 1994) such as corals, gorgonians and sponges (MacDonald et al. 1996; Fosså et al. 2002; Hall-Spencer et al. 2002; Reed 2002; Pham et al. 2014). Bottom-contact fishing gears, such as trawling, longlines, gills and trammel nets, get easily entangled in benthic sessile fauna specially corals and gorgonians, directly breaking, tilting their colonies or scattering fragments (Buhl-Mortensen and Buhl-Mortensen 2005; Gage et al. 2005; Martín et al. 2014). Overall, cumulative effects result in fragmented and isolated populations, increasing their vulnerability to further disturbances (Hughes and Connell 1999). Furthermore, the loss of key habitat-forming organisms results in the disappearance of suitable habitat for a significant number of associated species, representing a simplification of the structure and functioning of the entire benthic community (Thrush and Dayton 2002; Althaus et al. 2009; Clark and Rowden 2009; Clark et al. 2010).

Cold-water corals (CWC) are widely distributed in the world's oceans, mostly between 50 and 4000 m depth, playing crucial structural and functional role in mid-depth and deep-sea ecosystems (Roberts et al. 2006, 2009; Orejas and Jiménez 2019). They form complex three-dimensional structures that act as shelter, feeding and nursery areas for a highly-diverse associated fauna, including species of high commercial interest (Roberts and Hirshfield 2004; Henry and Roberts 2007; Miller et al. 2012) while creating hotspots of biodiversity (White et al. 2012; Henry and Roberts 2016). Moreover, these coral assemblages take an active part in most bio-geochemical cycles and benthic-pelagic coupling processes enhancing ecosystem functioning (Wild et al. 2009; Cathalot et al. 2015; Rovelli et al. 2015). CWCs are slow-growing, high-longevity species, with delayed sexual maturity and infrequent recruitment success (Andrews et al. 2002; Reed 2002; Brooke and Young 2003; Orejas et al. 2011; Watling et al. 2011). As a consequence, CWC ecosystems are highly vulnerable to anthropogenic impacts and display reduced recovery capacity, which can jeopardize their long-term viability (Williams et al. 2010; Huvenne et al. 2016). Specifically, several studies have demonstrated that recovery of CWC ecosystems after anthropogenic impacts could take decades to centuries, if recovery is possible at all (Althaus et al. 2009; Williams et al. 2010; Huvenne et al. 2016; Girard and Fisher 2018). Therefore, given their life traits and ecological significance and vulnerability, protection of CWC ecosystems has been stated as a major priority in marine management strategies (Armstrong et al. 2014). In recent years, CWC ecosystems have been recognized as Vulnerable Marine Ecosystems (FAO 2009) and their conservation is now internationally recognized as a high priority for the maintenance of marine biodiversity (Thurber et al. 2014). Conventions, directives and policies (COM 2008; Hall-Spencer and Stehfest 2009; Christiansen 2010; OSPAR 2010; FAO 2016) underline the importance of sustainably managing and protecting CWC ecosystems, addressing both the loss of biodiversity and ecosystem functioning (Armstrong et al. 2014; Bennecke and Metaxas 2017; Otero and Marín 2019).

In this context, ecological restoration assisting the recovery of impacted ecosystems represents a worldwide-recognized strategy to complement protection and management measures (McDonald et al. 2016; Gann et al. 2019). The effectiveness of a passive restoration approach, such as the implementation of deep-sea marine protected areas, has been evidenced by CWCs re-growth, recruitment and recovery of associated megafauna abundance after years of protection (Harter et al. 2009; Bennecke and Metaxas 2017; Baco et al. 2019). However, given the scale of accumulated impacts, protection may not always be sufficient (Huvenne et al. 2016) and additional active ecological restoration may be required to enhance the recovery of impacted habitats (Rinkevich 2005; Lotze et al. 2011). In fact, active restoration actions have been widely developed for terrestrial (Lamb 1998; Harker 1999) and marine ecosystems (Waltham et al. 2020). Nonetheless, the vast majority of actions performed at sea have been heavily skewed toward shallow tropical (e.g., Epstein et al. 2001; Rinkevich 2005; Pizarro et al. 2014) and temperate habitats (e.g., Linares et al. 2008; Verdura et al. 2018; Layton et al. 2020) whereas restoration actions focused on deeper habitats remain scarce (Van Dover et al. 2014; Morato et al. 2018). Despite awareness of the need to protect deep-sea environments, only few studies have addressed the active restoration of CWC habitats (Brooke et al. 2006; Dahl 2013; Jonsson et al. 2015; Montseny et al. 2019), stressing that we are in an initial and pioneering developmental phase for restoration techniques suitable for CWC habitats. Recent studies have successfully evaluated transplantation techniques to restore CWC reef-forming species (Brooke et al. 2006; Dahl 2013; Jonsson et al. 2015) and CWC garden ones (Boch et al. 2019; Montseny et al. 2019). The main challenges for CWC restoration are principally based on our vast lack of knowledge about biodiversity, functioning and resilience of deep-sea ecosystems (Van Dover et al. 2014; Morato et al. 2018; Da Ros et al.

2019). On the other hand, the difficult access to CWC habitats and major expenses related to the required technology, technically and economically limit the spatial scale of restoration actions. Local interventions are far from adequate to match the scale of ecosystem degradation (Bayraktarov et al. 2016; Boström-Einarsson et al. 2020) and scientific efforts are currently focusing on expanding the spatial scale of CWC active restoration actions, making them technologically and economically affordable (Aronson and Alexander 2013; Perring et al. 2018; Da Ros et al. 2019). Integrating ecological data with economic and social aspects is becoming a crucial component in ecosystem management (Hull and Gobster 2000; White et al. 2005). Cost information is essential for ecological restoration planning because it allows for selecting the best approaches (Iftekhar et al. 2017) and identifying aspects that need to be improved (Edwards et al. 2010). However, studies on restoration costs are still limited (Bayraktarov et al. 2016), with less than 5% of studies including an economic evaluation (De Groot et al. 2013; Wortley et al. 2013). To account for all types of costs associated with a restoration action is not an easy task (De Groot et al. 2013; Bayraktarov et al. 2016) due to the difficulties of standardizing cost analysis methods and outputs (Spurgeon and Lindahl 2000; Bullock et al. 2011). Nonetheless, the few studies which have addressed the economic costs of deep-sea active ecological restoration actions have highlighted the fact that economic costs are two to three orders of magnitude higher than for shallow areas (Boch et al. 2019; Da Ros et al. 2019).

Given this situation, the present study aims to go one step further in the restoration of CWC gardens, by scaling up the restoration of gorgonian populations on the Mediterranean continental shelf applying a low-cost method (Montseny et al. 2020) and involving local fishers. The active participation of local actors and stakeholders in ecological restoration actions may play a decisive role in their successful development (Hull and Gobster 2000; Yap 2000). The restoration method consists of reintroducing bycatch gorgonians to their natural habitat by attaching them to cobble supports, and gently throwing them from the sea surface. A two-year restoration study to evaluate the ecological and socio-economic effectiveness has been carried out.

3 METHODOLOGY

3.1 Target habitat and species

The restoration action was conducted on the continental shelf of the marine protected area of Cap de Creus (North-Western Mediterranean Sea, 42° 19' 12" N - 03° 19' 34" E) (Fig. 1). In this area, outcropping rocks and coarse-grained sediments support an extensive population of the gorgonian *Eunicella cavolini* (Koch, 1887) at 80 – 120 m depth (Gili et al.

2011; Lo Iacono et al. 2012). Gorgonians are patchily distributed, with spots dominated by medium to large sized colonies, reaching densities up to 20 colonies m^{-2} (Dominguez-Carrió et al. 2014; Dominguez-Carrió 2018). *E. cavolini* is a common azooxanthellate Mediterranean gorgonian species occurring in a wide bathymetric distribution range (< 10 – 220 m depth) (Russo 1985; Bo et al. 2012; Grinyó et al. 2016). Colonies usually display a fan-shaped morphology with a varied branching pattern, depending on environmental conditions, but mainly lying on a single plane oriented perpendicularly to the dominant current (Velimirov 1973; Weinbauer and Velimirov 1995). The size of *E. cavolini* colonies reported in the Mediterranean continental shelf is quite variable ranging from 9 ± 7 to 15 ± 10 cm in the Menorca Channel (Grinyó et al. 2016) and from 18 ± 2 to 25 ± 3.5 cm in the South Tyrrenian Sea (Bo et al. 2012). The largest colonies can reach 50 cm height (Bo et al. 2012; Grinyó et al. 2016). *E. cavolini* has slow growth rates (a few cm $year^{-1}$), and low recruitment success with lifespans around two decades (Weinbauer and Velimirov 1995; Sini et al. 2015). Moreover, its populations hold a great diversity of associated species such as sponges, soft corals, bryozoans, hydrozoan, polychaetes and some species of high commercial interest such as spiny lobsters or scorpionfishes (Dominguez-Carrió et al. 2014). For this reason, artisanal fishing with trammel nets, longlines and traps are extended and permitted in the area. Due to their arborescent morphology, gorgonians are highly susceptible to being entangled by nets. As a consequence, colonies of *E. cavolini* are among the more accidentally caught species, which represents a significant threat to these populations(Dominguez-Carrió et al. 2014; Enrichetti et al. 2019).

3.2 Restoration action

The restoration action was carried out in close collaboration with artisanal fishers from fishing associations in Cadaqués and Port de la Selva (Fig. 1). During the 2018 and 2019 fishing seasons (from March to August), a total of 9 fishers worked in collaboration with scientists to recover the *E. cavolini* colonies entangled in their nets. Collected colonies derived from trammel net fishing targeting lobster at 70 – 100 m depth. Once disentangled from the net, gorgonians were kept on board in seawater-filled buckets (∼22 – 25°C) until their transport to land (within 2 hours, at most) where they were held in aquaria installed at both harbours (Cadaqués and Port de la Selva), under environmental conditions similar to those on the continental shelf. Aquaria were composed of 100-L tanks (4 in Port de la Selva and 2 in Cadaqués) filled with seawater filtered using a biological filter (EHEIM 1500XL) and maintained at 13 ± 1.0°C by chillers (Teco TK 2000). A submersible pump (Sicce Nano 2000) provided continuous water movement in each tank. Seawater was partially changed and renewed at least twice a week (approximately 1/3 of the water at each water change). Gorgonians were held under these conditions for a minimum of a few weeks to a maximum of three months and then were prepared for their reintroduction to the continental shelf. During this time, no additional food was added to the tanks to prevent nutrient increase, and gorgonians fed on the particulate organic matter incoming with the regular water changes. Colonies were fragmented into medium size nubbins (16.6 ± 0.6 cm height, mean ± SD), according to the size that showed the highest probability of success by the restoration method used (see details in Montseny et al. 2020). Additionally, necrotic portions were discarded. Natural cobbles and artificial concrete ones were used as supports for gorgonian fragments in the restoration. Natural cobbles (approximately 9 – 1 0 cm width, 12 – 13 cm length, 3 – 5 cm height, and 400 – 500 g weight) were collected from the coastal area of Cap de Creus, whereas small artificial cobbles were produced in concrete using a square mould (width: 8.0 cm, length: 8.0 cm, height: 2.5 cm, weight: 175 g) (see details in Montseny et al., 2020). Cobbles were painted with white water-resistant and non-toxic paint to enhance visibility once returned to the continental shelf (Fig. 2A). A hole (1 cm diameter, 2 cm depth) was drilled in each cobble in order to allow attachment of gorgonian fragments using an epoxy putty (Corafix SuperFast, GROTECH®) (Fig. 2B). All the obtained transplants were maintained in the aquaria facilities installed at both harbours and under the same condition as described above. Once approximately 50 transplants were ready in the tanks, they were reintroduced to the continental shelf. Before their return, transplants were individually photographed on a ruled table in order to record gorgonian size and to allow for future growth monitoring after their reintroduction on the continental shelf. Three locations on the continental shelf within the Cap de Creus Natural Park area were selected as restoration sites: "Golfet" (42° 20' 42´´ N – 03° 15' 02 ´´ E; 64 – 68 m depth), "Cala Sardina" (42° 20' 54´´ N – 03° 16' 12´´ E; 82 – 86 m depth), and "Portaló" (42° 20' 23´´ N – 03° 17' 35´´ E; 82 – 90 m depth) (Fig. 1). These locations were selected based on the presence of horizontal bottoms in the natural bathymetric range of the species, and because natural populations of *E. cavolini* were known to be located nearby (Dominguez-Carrió, 2018). Even if artisanal trammel net fishery is allowed inside the Natural Park area, regulation strictly forbids bottom trawling fishing, providing at least a partial protection of the restored sites. A total of 9 return events were performed from June to August 2018, and 8 return events from June to August 2019. During each event, transplants were kept in portable plastic fridges (75 x 40 x 30 cm) filled with seawater (∼13°C) and transported by boat to the restoration sites where they were gently thrown from the sea surface (Fig. 2C and D).

Transplants on the continental shelf were monitored in order to assess the success of the restoration action through two consecutive surveys (November 2018 and September 2019) by means of the Girona 500 Autonomous Underwater Vehicle (AUV). The AUV records videos and acquires geo-referenced photo-mosaics of the three restored sites, and of an adjacent natural gorgonian population to be used as control site. The vehicle was equipped with two high definition cameras: one pointing down to acquire a geo-referenced photo-mosaic of the area and the other pointing forward to identify the gorgonians. Two parallel lasers were also included to provide accurate measurements of the forward-looking camera, as well as a set of underwater lights to illuminate the area. The Girona 500 AUV is equipped with a complete navigation suite that includes a MEMS-based attitude sensor, a Doppler velocity logger, a pressure sensor and an ultrashort baseline system that allows tracking and correcting the AUV position with respect to a surface vessel. The photo-mosaics were generated using image registration (Elibol et al. 2016) combined with a pose-graph optimization step which takes into account the navigation information of the AUV (Campos et al. 2016). Given that the seafloor was essentially flat, a 2D image registration approach was chosen, instead of a full 3D reconstruction, as it allowed better handling of cases of low overlap between images (Gracias et al. 2017).

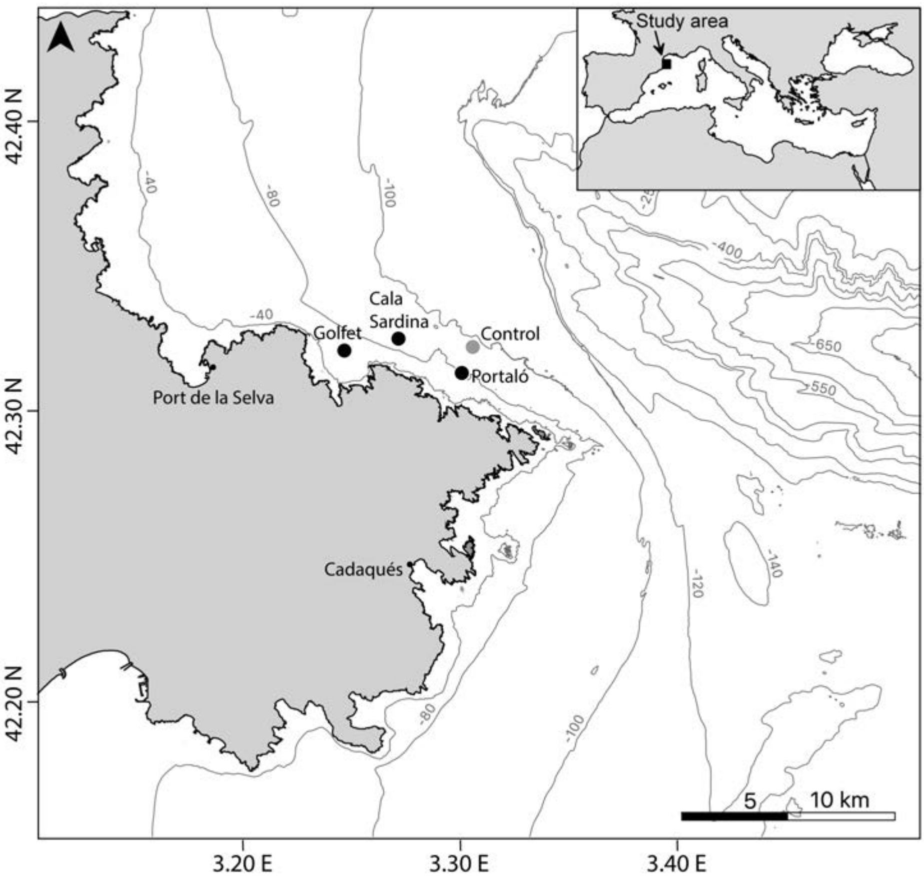

Figure 1. Study area with the three restoration locations in black (Golfet - Cala Sardina - Portaló) and the control site in grey (close to Portaló) (UTM31 WSG84).

3.3 Ecological evaluation

During each year, the total number of gorgonians recovered from artisanal fishers and their survival in the aquaria were quantified, as well as the total number of transplants obtained from the surviving gorgonians and returned to the continental shelf in each restoration site. By analysing the pictures of the transplants prior their reintroduction, the maximum height of each gorgonian fragment was measured using the Macnification 2.0.1 software (Schols and Lorson 2008). Subsequently, the size structure of the reintroduced gorgonians was determined for each site and analysed in terms of descriptive statistics using distribution parameters such as skewness and kurtosis. Statistical analyses and graphics were performed with R (RCore Team 2018) by means of the R Studio software (RStudio Team 2016) using the ´Ggplot2´ (Wickham, 2016) and the ´Moments´ packages (Komsta and Novomestky 2015).

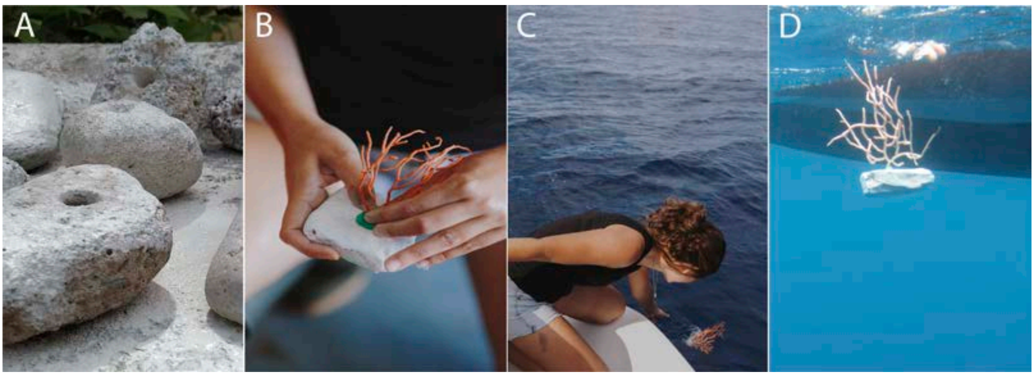

Figure 2. Restoration action images. (A) Drilled and painted natural cobbles (photo credit ICM-CSIC); (B) Gorgonian fragment attached to a natural cobble using epoxy potty (photo credit L. Sabaté); (C and D) Gorgonian transplant gently thrown from a boat (photo credit ICM-CSIC and N. Viladrich).

The area covered by transplants (m^2; ha) and the restoration success at each site was determined through the analysis of the videos and photo-mosaics recorded with the AUV. The restored area was quantified from the photo-mosaics, whereas the restoration success was evaluated by quantifying the percentage of upright and overturned transplants. In addition, the spatial structure of the gorgonians was assessed and compared between restored and control sites from the analysis of the geo-referenced photo-mosaics. The spatial distribution of the gorgonians and their corresponding coordinates were obtained by using a geographic information system software (QGIS 3.12.0). From these coordinates the gorgonian spatial structure was analysed by applying spatial statistics with Passage 2.0 software package (Rosenberg 2008). The distances between pairs of gorgonians were quantified and plotted with histograms. The restored and control areas were divided into 2 x 2 m grids and the mean colony density in each square plus the percentage of occupancy (percentage of occupied squares) were calculated. Finally, the gorgonian distribution pattern was evaluated using Ripley's K-function, a second-order spatial statistic which was plotted as an L-function ($L(t) = t - K(t) / 2$) (Fortin and Dale 2005). In Ripley's K-function, the number of neighbouring colonies within a distance (t) of each gorgonian colony is counted, and an edge correction is applied to colonies near the border of the photomosaic (Fortin and Dale 2005). Following this, the null hypothesis of a complete spatial randomness in the distribution of gorgonian colonies was tested by comparing with distributions generated by randomly repositioning all the observed colonies. For statistical significance a 95% confidence interval was set, and 999 randomizations were used. If the sample statistic was found within the bounds of the confidence interval at any point, then the null hypothesis could not be rejected. A significant positive deviation of the sample statistic indicates overdispersion of the colonies, whereas a significant negative deviation indicates a clumped distribution (Fortin and Dale 2005).

3.4 Economic evaluation

The economic cost of the restoration action and the local fisher's collaboration was evaluated, including the installation and operational costs (Edwards et al. 2010; Medrano et al. 2020; Pagès-Escolà et al. 2020). The restoration action was divided into 5 different phases (Edwards et al. 2010; Chamberland et al. 2017), and estimated costs were broken down into: (1) collection of the bycatch gorgonians, (2) set-up of aquaria facilities for gorgonian maintenance, (3) transplant preparation, (4) transfer and deployment of transplants to the restoration sites, and (5) monitoring of the restoration sites. Salaries of the scientific staff that supported all the phases of the restoration action were accounted separately and according to the base salary for research technician personnel, established by the Spanish Government (2018). Labour was expressed in terms of person-hours only including the time invested in the restoration action. Additionally, a monetary contribution per year was paid to each artisanal fisher for their commitment to collect all the accidentally fished gorgonians during the entire fishing season (6 months, every year).

4 RESULTS

4.1 Ecological evaluation

A total of 805 colonies of *E.cavolini* were recovered from trammel nets during the two studied fishing seasons (468 colonies in 2018 and 337 colonies in 2019). While being maintained in aquaria installed in both harbors, several gorgonian colonies recovered from partial breakage and tissue abrasion they had initially suffered due to the fishing impact. Even so, those gorgonians presenting severe signs of necrosis (22.4%) were rejected and not used for transplant preparation. As a result of this selection, 625 gorgonians (77.6%) were considered suitable for transplantation and were cut into medium-sized fragments, thus increasing the number of nubbins transplanted on supporting cobbles to 864 (representing a 27.7% increase compared to the initial number of colonies). Of these transplants, 38 were discarded (4.4%) which showed additional necrosis, thus resulting in a total of 826 transplants reintroduced to the continental shelf. In total, 693 transplants were placed on natural cobbles and 133 onto artificial small concrete cobbles (see details in Table 1). Based on the experience from 2018 (see below), only natural cobbles were used in 2019 and all transplants were reintroduced at "Portaló". Analyzing the size structure of the reintroduced gorgonian fragments, a dominance of medium-sized colonies (10–20 cm) was observed at all sites. More specifically, skewness and kurtosis values indicated that reintroduced populations were significantly positively skewed, indicating the prevalence of smaller sizes at "Golfet" and "Portaló", while those at "Cala Sardina" were clearly dominated by 15 – 20 cm height colonies (Table 1 and Fig. 3).

Table 1. Number of reintroduced transplants and size structure (height, skewness and kurtosis) for each restoration site and year. Significant skewness or kurtosis are indicated with asterisks. NC = natural cobbles, SAC = small artificial cobbles.

Site	Year	Nº colonies		Height (cm)			Skewness			Kurtosis		
		NC	SAC	Mean± SD	Max	Min	Skew	*p-value*	Sig.	Kurt	*p-value*	Sig.
Golfet	2018	100	50	15.48±3.63	26.77	7.12	0.809	<0.001	***	3.687	0.092	
Cala Sardina	2018	117	33	16.26±4.06	28.64	8.48	0.373	0.056		2.885	0.960	
Portaló	2018 and 2019	476	50	17.23±4.61	34.95	6.2	0.246	0.021	*	2.924	0.827	

The AUV surveys revealed significant differences in the three locations selected for the restoration action in 2018. The restoration failed at "Golfet" (where an area of 2 339.5 m^2 was inspected) because the bottom was found to be covered by seagrass leaves (*Posidonia oceanica*), completely covering the reintroduced gorgonians (only some branches were visible coming out in-between the leaves). Likewise, at "Cala Sardina" (where 2 937.2 m^2 were prospected) the majority of the detected gorgonian transplants were partially or completely buried in fine sediment, hampering their proper identification. In contrast, "Portaló" (where 596.1 m^2 were inspected) turned out to be the most appropriate location for the reintroduction, since 146 gorgonian transplants (out of 151 reintroduced) were detected in 2018, representing 96.7% of all the reintroduced transplants at that site (97.0% on natural cobbles and 88% on small artificial cobbles). An 88.8% of gorgonians transplanted on natural cobbles were landed in a correct upright position, compared to only 72.7% of transplants on small artificial cobbles. In total, the 83.8% of fragments transplanted were correctly landed (Fig. 4). Given the failure in "Golfet" and "Cala Sardina" and the lower success of upright landing shown by small artificial cobbles only natural cobbles were used, and all transplants were devolved to "Portaló" in 2019. The AUV survey in 2019 (area inspected 2,330.30 m²) detected 460 gorgonian transplants, which represented 87.5% of all the reintroduced transplants during the two consecutive years. The majority of the detected transplants were in upright position (416; 90.4%) covering a restored area of 0.23 ha (Fig. 4).

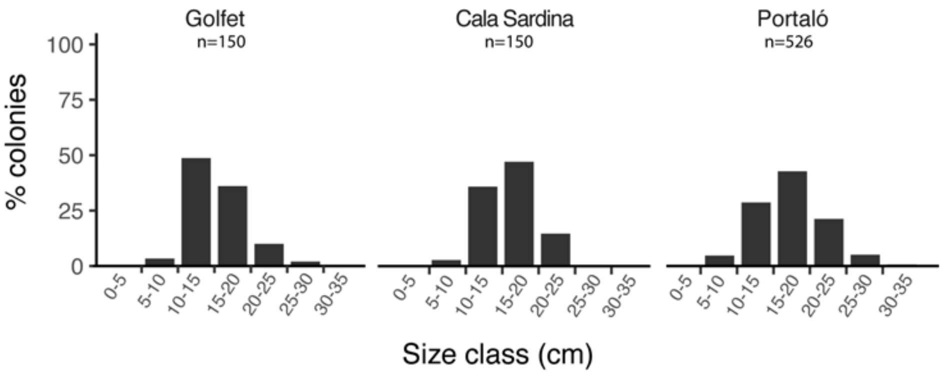

Figure 3. Size frequency distribution of *E. cavolini* transplants reintroduced at each restoration site (n = number of transplants). Note that Portaló includes all the cumulative transplants reintroduced in 2018 and 2019.

The photo-mosaic acquired at “Portaló” allowed detection of a total of 116 transplants, 16 of them were overturned and 100 maintained a correct upright position (86.2%) (Fig. 5). Due to technical difficulties in positioning of the AUV under the strong current conditions encountered on the continental shelf, part of the restored area was left uncovered, preventing the identification of all the transplants. From the 100 upright detected transplants, their spatial structure was analysed and compared to the control site (Fig. 6), where 799 natural *E. cavolini* colonies were detected in an area similar to “Portaló” (2,365 m²). Transplants in “Portaló” were more dispersed than in the control site, where the distances between pairs of colonies were shorter (Fig. 6B). The mean colony densities per square (2 x 2 m) were 5.3 ± 5.4 (mean± SD) and 1.2 ± 0.6 (mean± SD) at the control site and “Portaló”, respectively (Fig. 6A). In accordance, the percentage of occupancy was also higher in the control site (23.7%) than in “Portaló” (13.5%). The distribution pattern displayed a clumped distribution of colonies from a scale of 10 cm distance, at both sites (Fig. 6C).

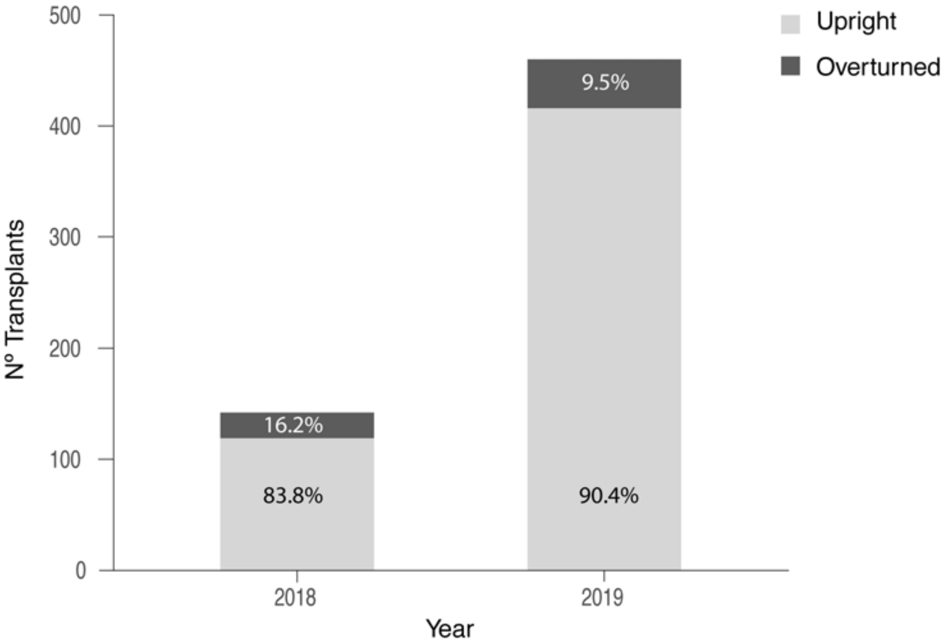

Figure 4: Number of upright (grey) or overturned (black) transplants detected at Portaló (82 – 90 m depth) during the AUV monitoring surveys in 2018 and 2019. Note that 2019 includes all the cumulative transplants reintroduced in 2018 and 2019.

4.2 Economic evaluation

A total cost of approximately 106 783 € was calculated for the whole restoration action of 826 gorgonian transplants reintroduced to the continental shelf of Cap de Creus (Table 2 and Supplementary Material Table A1 and A2). Nevertheless, taking the sum of the three inspected areas as the total restored area, the standardized cost per hectare was of 140 504 € ha^{-1}. The highest costs were related to the collection of bycatch gorgonians, and the monitoring of the restored sites (accounting for >80% of the total cost). Conversely, the setup and maintenance of the aquaria, transplant preparation and reintroduction only accounted for 3.5% of total costs (without including scientists' salaries, which accounted for a considerable 14.2% of total cost) (Table 2A). Focusing only on expenses of the transplant preparation and reintroduction stages, the cost of restoring a single gorgonian colony attached to a natural cobble (1 €) was half of the cost when using transplants with small artificial cobbles (2 €) (Table 2B).

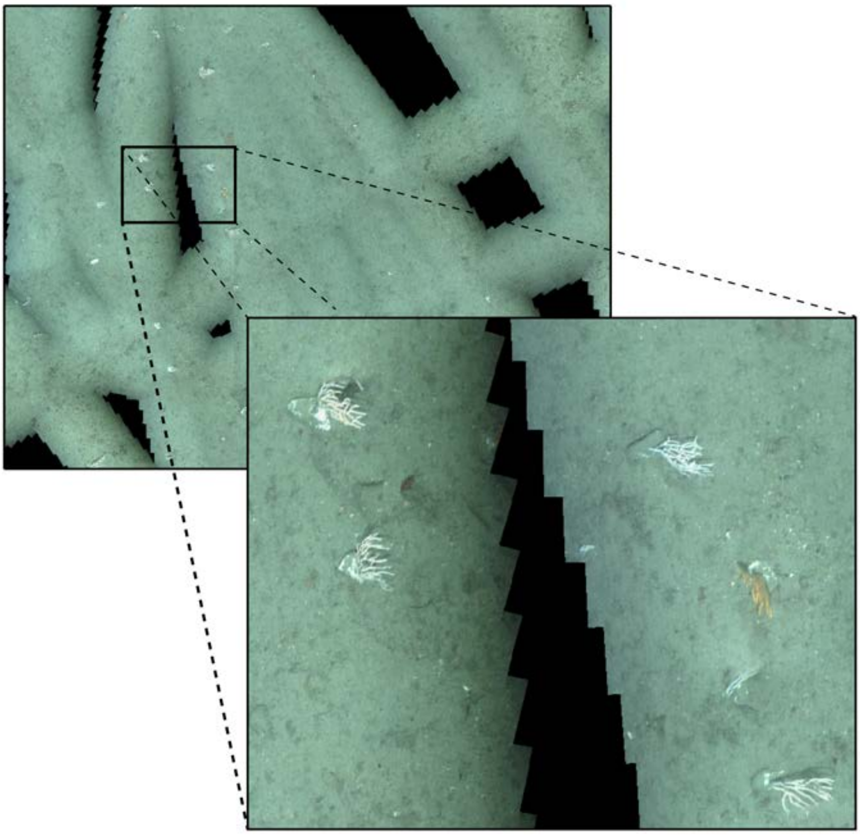

Figure 5: Photo-mosaic section obtained during the AUV monitoring at Portaló in 2019.

5 DISCUSSION

The present study demonstrated, for the first time, the feasibility of restoring a large number of cold-water gorgonians (about 400 colonies) at 80 – 90 m depth at a low-cost and working in close cooperation with local artisanal fishers. The results represent a first step to achieving comparable spatial and size structure to natural reference populations of *E. cavolini* in a similar bathymetric range (Bo et al. 2012). In the successfully restored site ("Portaló") the dominance of medium-sized colonies (10 – 20 cm height) will drive the faster recovery of the ecosystem functioning, and services that gorgonian populations provide (Horoszowski-Fridman et al. 2015; Geist and Hawkins 2016). The area covered at this site was about 0.23 ha, which exceeds most of the current coral restoration projects, mostly conducted at relatively small spatial scales with a mean restored area of 100 m^2 (Boström-Einarsson et al. 2020) However, these results are still far from matching the scale of anthropogenic degradation of ecosystems (10 – 1,000,000 ha.) (Bayraktarov et al. 2016).

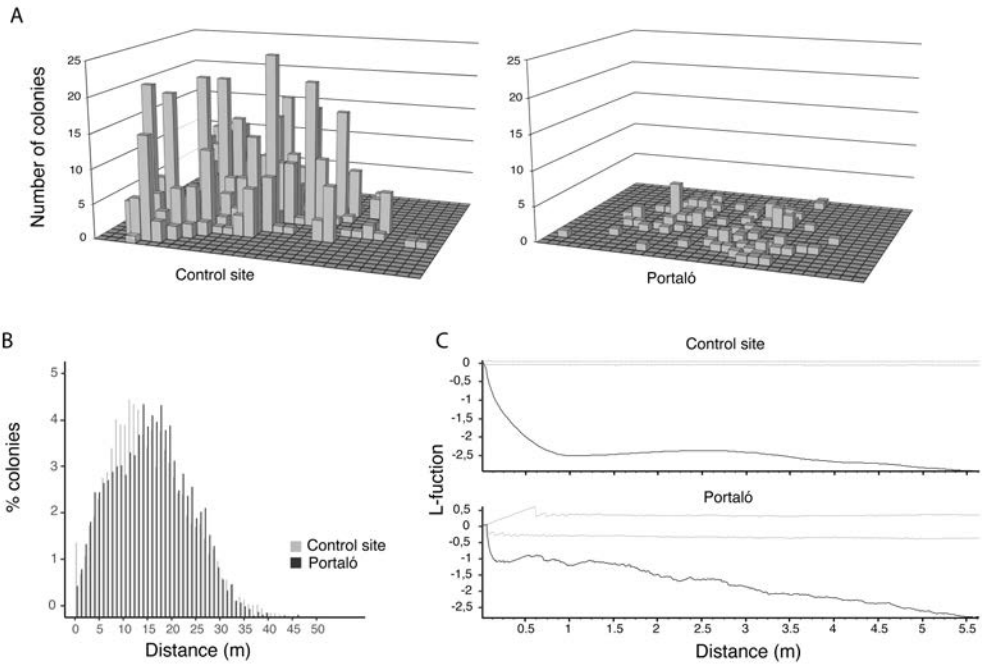

Figure 6. Comparison of colonies' spatial structure between the control site and the restored Portaló site in 2019. (A) Colonies density pattern in control site (2 365 m^2) and Portaló (2 330 m^2). (B) Distribution of distances between pairs of colonies. (C) Colonies distribution pattern by the L-function (derived from Ripley's K-function).

During the initial phase of the restoration action, recovered gorgonians successfully overcame the mechanical damage and stress suffered after being accidentally fished and transported to aquaria facilities. During transport on fishing boats the gorgonians were exposed to high temperatures, suffering an abrupt thermal change in a short time. However, after their transfer and maintenance in aquaria kept at their normal habitat temperature (∼13°C), a large proportion of gorgonians recovered from mild signs of necrosis. This contrasts with the generally recognized complexity of *ex situ* maintenance of CWC species (Orejas et al. 2019), at least for some species. In our case, with relatively simple, low-cost (2,762€) and easy to maintain aquaria installations, only 23% of the collected gorgonians failed to recover, and thus were discarded for transplant preparation. This latter supports the previously demonstrated high recovery capacity of *E. cavolini* (Fava et al. 2010; Montseny et al. 2019, 2020) and proves the possibility of taking advantage of bycatch colonies that otherwise would be discarded. Moreover, since fishing activity generally covers an extensive area, there would potentially be a high genetic diversity of transplants, increasing the success probability for long-term viability of restored populations (Reynolds et al. 2012).

The selection of restoration sites was determined by suitable local conditions for the development of *E. cavolini*, including depth range, bathymetric profile, proximity of natural gorgonian populations, and degree of protection. The restoration sites were located within the Natural Park area where bottom trawling is restricted. Even so, the first monitoring highlighted that the restoration action failed at two out of the three selected sites, due to the presence of fine sediment and dead seagrass leaves making it impossible to properly detect the reintroduced transplants. Contrarywise, at the "Portaló" site the two-years AUV monitoring allowed us to successfully detect more than 85% of the reintroduced transplants. These results underline the importance of considering the environmental conditions for a proper selection of restoration locations, since environmental conditions display a critical role in shaping the outcomes of restoration projects (Suggett et al. 2019; Boström-Einarsson et al. 2020). Several ecological restoration actions have failed due to the complexity of accounting for all the stressors influencing the system (Zedler and Callaway 2000; Bruckner et al. 2008; MBARI 2016). However, most of those failures are often unreported (Precht and Robbart 2006). Selection of proper sites for the restoration actions is especially challenging for deep-sea locations, where limited knowledge of environmental conditions and spatial and temporal dynamics, together with the difficulties in predicting future scenarios, can contribute to unexpected consequences affecting restoration efforts (Abelson 2006).This is the example for our study case, where key information about natural population dynamic of *E. cavolini* in the continental shelf is still lacking to corroborate whether restoration efforts could modify the population recovery capacity within reliable time frames.

The high percentage of transplants found alive and in upright position is in close accordance with forecasts from the previous evaluation study of the used technique (Montseny et al. 2020). The arborescent morphology of the gorgonian colonies leads to a successful landing in upright position on the continental shelf when attached to a cobble. Once there, transplants are likely to survive in the long-term, as previously suggested by small-scale trials for *E. cavolini* (Montseny et al. 2019) and other Mediterranean shallower gorgonians (Linares et al. 2008; Fava et al. 2010). High survival rates of transplants were also observed in the few other active CWC restoration attempts performed to date, with coral survival ranging from 52% to 87.5% after 1 to 3 years (Brooke et al. 2006; Dahl 2013; Jonsson et al. 2015; Strömberg 2016; Boch et al. 2019). Transplants attached to small artificial cobbles in 2018 showed higher probability of landing overturned than transplants on natural cobbles, thus reaffirming the use of local natural cobbles as the best option for this kind of active ecological restoration (Montseny et al. 2020), as well as avoiding the introduction of artificial material (Weinberg 1979).

Table 2: (A) Summary table of the estimated costs for the restoration action resulting in 826 restored transplants. (B) Calculated costs for preparation and reintroduction of a single gorgonian transplant, according to the cobble type.

A

Concept	*Cost (€) / NC transplant*	*Cost (€) / SAC transplant*
Transplants preparation	0.54	0.94
Transfer and deployment of transplants to restoration sites	0.53	1.07
TOTAL	1.07	2.01

B

Concept	*Total (€)*	*% of the total costs*
Collection of bycatch gorgonians	54,000	50.57
Set up of aquarium facilities	2762,58	2.59
Transplants preparation	460.27	0.43
Transfer and deployment of transplants to restoration site	514.5	0.48
Monitoring of the restoration sites	33,88	31.37
Scientists' salaries	15,165.36	14.20
TOTAL RESTORATION COST	***106,782.71***	***100***

To properly assess restoration success over time it is crucial to establish a reference site for comparison (Falk et al. 2006; McDonald et al. 2016; Aronson et al. 2017). This site should ideally be nearby, undamaged, analogous and pristine (or near pristine), serving to evaluate the success of the performed restoration action over time (Falk et al. 2006; McDonald et al. 2016). However, the fact that artisanal fishing traditionally occurs over the entire study area prevented us from identifying a pristine reference site for our study case. This is commonly the case for most deep-sea areas, for which there is still very limited information about ecosystem baseline conditions (Da Ros et al. 2019), and reference ecosystems have to be inferred from the best ecological knowledge available (Morato et al. 2018; Gann et al. 2019). In our study, we selected a nearby control site with a natural *E. cavolini* population to compare with the restored population at the "Portaló" site. From the photo-mosaics comparison we were able to detect a first establishment of a reintroduced gorgonian population that may trend to a natural population in terms of distribution and density patterns, if natural recruitment occurs. Although current values at the "Portaló" site are far from those in natural control sites, the methodology presented here allowed us to set up a conceptual framework for the monitoring of ecological restorations at intermediate and deep-sea habitats. Consistent with our results, the successful use of photo-mosaics for the evaluation of CWC habitats has also been proven in very recent studies (Boolukos et al. 2019; Prado et al. 2019). Long-term monitoring (15 – 20 years) has been highlighted as paramount to proper evaluating success of restoration actions in shallow waters (Bayraktarov et al. 2016), and this is even more crucial for CWC species given their slow population dynamics (Roberts and Hirshfield 2004; Bennecke et al. 2016; Orejas and Jiménez 2019). Moreover, applying an adaptive management based on proper monitoring leads to the opportunity to improve restoration results by incorporating lessons from failures (Hackney 2000; Precht and Robbart 2006), such as the correct selection of the restoration sites in our study case. The short duration of our monitoring period (two years) allowed for a proper assessment of the initial rate of transplant survival, as well as for comparing the restored population with natural ones in terms of population size and spatial structure (establishing a paramount baseline of information for future comparisons). Nonetheless, our monitoring period precluded the detection of any possible recruitment or growth. Most coastal marine restoration projects, even for shallower environments, are performed during short period times (less than two years). This throws into doubt their adequacy for assessing recovery of ecosystem functioning, since outcomes of restoration are directly related with the monitored time period (Bayraktarov et al. 2016). Therefore, enlarging the time scale of monitoring, especially in the deep sea, is a necessity for obtaining reliable evaluation of restoration success.

Given the sophisticated technologies and infrastructures (e.g., oceanographic vessels, Remotely Operated Vehicles (ROVs) and AUVs) involved in the whole process of restoring and monitoring deep-water environments, these actions still nowadays are a costly effort. Restoration costs usually exceed millions of dollars, ranging from US$ 1.2 to 4.4 M ha^{-1} during the first year (Da Ros et al. 2019; Van Dover et al. 2014). For the present restoration action, the fisher's monetary contribution, the monitoring of restoration sites, and scientists' wages, required more than 80% of the project budget (Table 2; and Supplementary Material Table A1 and A2). Although these latter costs are highly dependent on local conditions such as fuel prices, distance to the restoration sites and country salaries, they could be significantly reduced by applying several improvements towards a more routine application, reducing the involvement of scientists and increasing local participation of fishers and stakeholders. Furthermore, improving technological development to obtain specialized, cheaper and easier-to-use underwater tools would also reduce restoration costs (Van Dover et al. 2014). Indeed, once bycatch colonies have been collected, maintenance in aquaria, preparation of the transplants, and reintroduction to the continental shelf only amounted to 3.5% of the total expenses (3 737 € in total; 4.5 € transplant^{-1}; Table 2). Keeping aside costs related with setting-up aquarium facilities, the cost of restoring a single gorgonian colony attached to a natural cobble is about 1 € (Table 2B). Setting-up aquarium facilities requires an initial investment but has a low annual maintenance cost which reduces overall costs for years to come. Overall, the total costs for the 2-yr restoration action reported here accounted for about 140,000 € ha^{-1} (US$ 170,000 ha^{-1}), which is surprisingly more in accordance with the cost of restoring one hectare of marine coastal habitats (from US$ 13,000 to US$ > 1 M ha^{-1}, with a median cost of ~US$ 500,000 ha^{-1}; Spurgeon and Lindahl 2000; Edwards et al. 2010), than the cost estimated for deeper habitats (Van Dover et al. 2014, Da Ros et al. 2019). After one or two years of adaptation, the used method (Montseny et al. 2020) could be a promising cost-effective technique that in itself would not cost more than few euros for each transplant restored. In this sense, the involvement of local communities in the restoration action is key for the success of long-term application. As in the present example, through their local knowledge and meaningful sensitivity to existing conditions, cooperation of local fishers is a great opportunity to enhance the effectiveness of restoration actions (Hull and Gobster 2000; Yap 2000). From the experience of these two years of project implementation, we perceived a growing interest of fishers in CWC gardens and a greater willingness to protect them and reduce the fishing impact. Restoration actions, involving local actors (fishers, managers and stakeholders) could also be an advantage for connecting civil society with the natural environment. From the local actors' point of view, being part of restoration activities can offer an opportunity to participate in the sustainable management of the habitats that guarantees their current and future source of income and resources, while prompting personal growth by achieving the satisfaction of making a difference (Miles et al. 1998).

In conclusion, the low-cost, low-tech and wide-scale applicable methodology presented here could be potentially extended to other CWC gardens, fostering a society-based implementation by involving local actors and using by-catch gorgonians. However, the importance of combining this active restoration with passive restoration measures such as marine protected and managed areas (Gubbay et al. 2003; Davies et al. 2007) to prevent or reduce impacts from anthropogenic disturbances and to ensure habitat recovery should also be noted. A total protection of restored areas would be ideal (Huvenne et al. 2016;

Bennecke and Metaxas 2017), but in turn challenging to apply in every situation. In fact, artisanal fishing practices impacting gorgonians populations will continue in the Cap de Creus Natural Park and Special Area of Conservation. Therefore, a complementary measure would be to search for alternative fishing gears that ensure a commercial catch while reducing the bycatch.

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GENERAL DISCUSSION AND CONCLUSION

1 MAIN CHALLENGES IN RESTORING CWC ECOSYSTEMS

1.1 Searching for the best restoration techniques

First attempts of active cold-water coral (CWC) restoration emerged in the early 21st century, targeted on the CWC reef forming species *Oculina varicosa* (Florida; Koenig 2001; Brooke et al. 2006) and *Lophelia pertusa* (Sweeden; Dahl 2013; Jonsson et al. 2015; Strömberg 2016). These actions obtained coral fragments from a healthy donor reef to transplant them to a degraded one, using ROV (Remote Operate Vehicle) technology. Because of the difficulties associated with the *in-situ* manipulation of coral fragments using ROVs, transplants were first attached to artificial structures (concrete modules or racks), and then deployed at 70 – 100 m depth in impaired areas. After more than two years, transplanted coral fragments showed high survival (>76%) and growth, with associated fauna being re-established, but there was little evidence of new larval settlement or juvenile recruitment (Brooke et al. 2006; Dahl 2013; Jonsson et al. 2015). However, transplantation of coral fragments may not be easily applicable to all CWC species. Handling, collecting, transporting and maintaining some CWCs in aquarium facilities before returning them to their natural habitat can be an issue. The stress suffered by corals during the collection (pressure and thermal changes) and the complexity of replicating their natural environment in the laboratory, can compromise the survival of corals in aquaria (Orejas et al. 2019). In a first attempt to restore CWC gardens at the Sur Ridge Seamount (California, USA), none of the coral fragments that were maintained in aquaria prior to transplantation were found alive after one year from their reintroduction (MBARI 2016). This was overcome by avoiding long-term maintenance of corals in aquaria and improving the attachment of coral fragments to transplantation structures on board the ship at sea, resulting in a mean coral survival of 52% after one year (Boch et al. 2019). Likewise, the good status of the bycatch gorgonians in Cap de Creus, with very low necrosis and epibiosis (chapter 1) together with a strong resistance of *Eunicella cavolinii* to collection and maintenance in aquaria, allowed for a high survival of restored gorgonians (90.4%) in our study (chapters 2 and 4). Gorgonian survival after one year coincides with the survival of some of the cold-water gorgonian species transplanted in the Condor Seamount (survival ranging from 30 – 100% per species; *Dentomuricea cf. meteor*, *Viminella flagellum*, *Callogorgia verticillata, Acanthogorgia armata* and *Paracalyptrophora josephina*) (Linares et al. 2019). The main advantage of transplanting coral fragments (usually with branching forms) is the faster recovery of the three-dimensional structure of coral populations, facilitating the recovery of their habitat-forming function for a large number of associated species (Horoszowski-Fridman et al. 2015; Geist and Hawkins 2016). Conversely, the main disadvantage of fragment translocation is the requirement for coral fragments collections, which usually impact healthy coral assemblages. Moreover, by using large transplants (which suffer less natural mortality after transplantation, Brooke et al. 2006), more material is required from the donor site. To partially overcome this issue, working with local fishers to recover already impacted corals (chapter 3 and chapter 4) could be a reliable alternative, at least in some areas. By using bycatch corals, the fishing impact on the natural populations is also mitigated, as bycatch corals are being returned to the environment and part of the loss is reversed. The overall cost of the restoration action is also reduced, with no need for expensive technology for coral collection. Furthermore, since fishing activity generally covers a wide spatial extent, high genetic diversity of transplanted coral fragments is also increased, which would provide more potential for habitat shelf-support (Reynolds et al. 2012). However, this requires an active, deep-water fishing activity near or around the area designated for restoration, which is not always the case.

The use of artificial structures for restoration is common in ecological restoration actions of shallow-water corals (Clark and Edwards 1994; Bachtiar 2000; Spieler et al. 2001) and, as above-mentioned, has been recently used as support for CWC transplants in Sweden (Dahl 2013; Jonsson et al. 2015; Strömberg 2016) and Florida (Koenig 2001; Brooke et al. 2006). Artificial structures may also result in the promotion of natural coral colonization. Colonization of artificial reefs is determined by the arrival of larvae and propagules and the subsequent local survival of adults (Dannheim et al. 2018). New recruits that will settle on artificial structures will be probably better adapted to the new conditions at the site. The capability of *L. pertusa* larvae to disperse over long distances and to potentially survive for long time confirms the potential of this CWC to colonize artificial structures (Strömberg and Larsson 2017; Henry et al. 2018). The type of structure, age, and depth of artificial reefs used all influence colony density and growth as observed in *L. pertusa* on 10 artificial structures in the Northern Gulf of Mexico (Larcom et al. 2014). Finding the best larval settlement surface is a first key step for the use of artificial structures. Complex substrates have been shown to promote higher colonization by deep-sea benthic invertebrates than simple substrates (Girard et al. 2016). However, *in situ* larval settlement experiments targeting CWCs have had mixed results. While a study found high recruitment rates for *Primnoa resedaeformis* on artificial substrate deployed for four years in the North-East Channel Coral Conservation area off Canada, very few *Paragorgia arborea* recruits were recovered on these same substrates (Lacharité and Metaxas 2013). Differences in the number of recruits of the two gorgonians could be due to differences in the reproductive strategies (broadcast spawning vs brooding). Recent work with larvae of *L. pertusa* has shown that they probably prefer cryptic spaces when settling (Strömberg et al. 2019), indicating that settling substrates must be specific to target species. Moreover, very little coral recruitment or even no recruitment at all, was also observed for *O. varicosa* after 5 years from the deployment of concrete modules in Florida (Brooke et al. 2006). Similarly, we were not able to observe any recruitment on the artificial structures deployed in Cap de Creus, but our monitoring was probably too short for this (chapter 2). The varying recruitment success in these studies highlight the importance of increasing the knowledge on CWC larval ecology, especially about factors affecting dispersal, settlement and recruitment success. The more is known for larvae of a species, the better the artificial substrates can be modeled to promote its settlement and recruitment. One applied example of using artificial structure as colonization substrate for CWCs are the obsolete oil and gas industry platforms which represent *de facto* no-trawling zones, thus providing a suitable habitat for recolonization of corals and other epifaunal species (Macreadie et al. 2011; Bergmark and Jorgensen 2014; Larcom et al. 2014). Structures may be toppled *in situ* or they may be transported to locations where coral restoration is required and where their disposal makes ecological sense (Macreadie et al. 2011). Through the Louisiana and Texas Rigs-to-Reef Programs (RTR), established respectively in 1986 and 1991, a total of 97 artificial platforms have been transported to deep locations (>120 m) in the Gulf of Mexico until 2008 (Kaiser et al. 2020). Shortly after offshores structures were installed, corals and others sessile invertebrates such as oysters, sponges, hydroids, mussels and barnacles attached to the structures, attracting a number of mobile invertebrates and fish species, forming highly complex communities (Kaiser and Pulsipher 2005; Kaiser et al. 2020). However, it should be noted that the potential use of disused structures of the offshore oil and gas industry as artificial structures to prompt CWC reef formation is not free of discussion. There are scenarios where unexpected recolonization of man-made structures may assist in restoring ecosystems at a faster rate than would occur through natural recolonization events, but at the same time, a range of factors need to be considered including health and safety, technology readiness, social factors, and the spread of invasive species (Fowler et al. 2018). Indeed, the invasive species of coral *Tubastrea micranthus* has already colonized RTR structures in the Gulf of Mexic (Sammarco et al. 2010).

As shown for shallow-water corals, deploying artificial substrates in areas receiving larvae during coral spawning, and their subsequent transfer to degraded areas where coral substrata may be limiting or lost, could be another approach to be tested for CWCs to enhance genetic variability and restoration success (Guest et al. 2010). Likewise, introducing large amounts of coral larvae in degraded CWC gardens or reefs could also increase the recruitment rate locally, as shown in tropical shallow-coral restoration (Heyward et al. 2002; Omori and Iwao 2014; Doropoulos et al. 2019). This strategy though requires cultivation of coral larvae at a larger scale in the laboratory, which still represent a challenge for CWCs. Conversely, culturing and rearing coral larvae and juveniles are widely applied techniques in shallow-coral reef restoration actions (Rinkevich 1995; Shafir et al. 2006; Mbije et al. 2010). Culturing larvae reduces the damage to healthy donor reefs caused by collections of coral fragments for translocation, and also enhances the genetic diversity of the restored coral populations. However, the scant knowledge concerning CWC reproduction and larvae ecology and culturing (Brooke and Young 2003; Strömberg 2016; Strömberg and Larsson 2017) together with the slow grow rates of CWCs (Andrews et al. 2002; Risk et al. 2002; Prouty et al. 2016) hinders the application of rearing techniques for CWC restoration.

Mineral accretion through electrolysis technique (Biorock™) was developed by Hilbertz and Goreau (1996) and has also been widely used in shallow tropical coral reefs rehabilitation programs to enhance the growth of coral transplants. During the active phase of the mineral accretion, when the cathode is getting a trickle current and accretion of aragonite is ongoing, the coral transplants are budding and branching more frequently. But it is when the electricity is turned off that the accreted material is attracting coral larval recruits (Kihara et al. 2013). The accreted material has the same composition of minerals as the coral skeleton itself (e.g., calcium carbonate in the form of aragonite). However, using Biorock™ technology in CWC restoration programs would be challenging and potentially expensive, with all the electrical installations necessary to provide the trickle currents for the cathode. For restoration at deep-sea sites it would be more feasible to pre-fabric Biorock™ structures and deploy them on site with the accretions already in place. To protect the structures from corrosion, they could be fitted with small pieces of sacrificing anodes of zinc as an exchange for the attached electrical cables.

1.2 Performing the restoration

The main technical constraint in CWC restoration are the difficulty and associated costs in accessing remote intermediate depths and deep-sea ecosystems (Van Dover et al. 2014). Technical diving with mixed gases can only allow divers to access the shallowest part (50 – 150 m depth; Pyle and Copus 2019) of CWC bathymetric distribution (50 – 4000 m depth; Freiwald and Roberts 2005; Roberts et al. 2006, 2009). Divers could easy transplant coral fragments on the natural substrate following the same techniques used for shallow coral reefs, employing cement, epoxy resins, nails, stainless steel wire or cable-ties (Rinkevich 1995; Edwards and Gomez 2007; Edwards et al. 2010). However, the working-time for deep technical diving is extremely limited, and the risks associated with the activity are severely enhanced (Fock and Millar 2008; Sayer et al. 2008; Pyle and Copus 2019). As a consequence, a long overall time will be needed to perform a restoration action by means of technical diving, and advanced safety measures (continuous access to decompression chambers and medical personnel) would be necessary to support the activity at all times. In contrast, ROVs are the main and most widespread alternative to access to intermediate depth and the deep sea (Van Dover et al. 2014). ROVs allow for long bottom-working time and significantly reduce risks to human life. However, technological hurdles involved in manipulating corals in the deep sea are still significant (Thresher et al. 2015). In particular, the main technical challenge is the dexterity of an ROV's manipulators arms to attach fragile coral fragments on rocky substrates using reattachment materials such as epoxy resins. This is a difficult task to be performed with fragments of stony corals, but even more so with gorgonians or black corals, whose flexible axial skeletons may be moved by seabed currents while the reattachment material hardens, compromising the stability of the transplanted fragments (Collier et al. 2007). Thus, gorgonian and black coral fragments need to be first fixed with epoxy resin to a small sturdy base in aquaria under no water-current conditions, and subsequently be transplanted at field once the resin has dried (Clark and Edwards 1995; Jaap 2000; Young et al. 2012). For this reason, to ensure proper attachment, the base of all gorgonians fragments transplanted in this thesis were first covered with a little amount of epoxy resin. Once the resin was dried, fragments were attached to the artificial structures at shallow depth (6 m depth) (chapter 2) or to cobbles in aquaria conditions (chapter 3 and 4).

The employment of Autonomous Underwater Vehicles (AUVs) to autonomously transplant coral fragments to natural substrata at intermediate depths and in the deep sea is still in the distant future and will require substantial technological development. The possibility for an AUV to move and automatically locate and place small structures or modules supporting transplanted coral fragments on the seabed, such as cobbles with gorgonian transplants, could be a first step in this direction. In this sense, promising advancement are undergoing on the capability of AUVs to autonomously locate and manipulate objects (Galloway et al. 2016; Mura et al. 2018; Sahoo et al. 2019). In addition, machine-learning approaches including deep nets, are helping to advance the automated detection of CWCs and coral habitat where active restoration activities would be suitable (Henry et al. 2016).

1.3 Selecting donor sites

The selection of donor populations, from where to collect fragments to be used in any restoration action, is also of increased difficulty for CWCs compared to shallow-water coral species, owing to still quite limited knowledge on CWC distribution and connectivity at intermediate depths and in deep-sea environments. Even less information is available on CWC population structure and dynamics, which are basic information required to evaluate potential impacts of any collection on source populations and to properly select them (Edwards et al. 2010). In the absence of additional information, donor sites should be selected as close as possible to the restored sites, since these would have a higher probability of containing corals adapted to the environmental and ecological conditions of the area. For the restoration actions developed in the present thesis, donor gorgonians were obtained from artisanal fishing bycatch in the area, avoiding in this way any additional impact to natural populations and giving a second chance to bycatch gorgonians removed from the sea-bottom. However, there is still a lack of knowledge about the natural population dynamic of E. *cavolini* in the area, to understand and determine how damaging the impact of artisanal fishery could be (chapter 1), as well as how a restoration action (chapter 4) could enhance the population recovery capacity.

1.4 Life-history traits of CWCs

Longevity is strongly and positively correlated with maximum depth occurrence in marine sessile species (Montero-Serra et al. 2018). Hence, it is not surprising that CWC species live for hundreds to thousands of years (Roark et al. 2009; Bennecke et al. 2016). Slow growth rates have been reported for both scleractininan corals (Reed 1981; Rogers 1999; Freiwald et al. 2004; Orejas et al. 2008, 2011; Lartaud et al. 2014) and gorgonians (Andrews et al. 2002; Risk et al. 2002; Sherwood and Edinger 2009; Watling et al. 2011; Bennecke et al. 2016), (Table 1). The general trend shows exponentially decreasing in growth rates with increasing colony size (Bennecke et al., 2016), and no increase in size for the largest colonies (Buhl-Mortensen and Buhl-Mortensen 2005; Watanabe et al. 2009). As above mentioned, natural population dynamic of the gorgonian *E. cavolini* studied in this thesis, still remains unknown, nevertheless, aquaria experiments reveal slow linear extension growth rates (Table 1). Overall, the range of CWCs growth rates are a magnitude lower than observed for tropical shallow-water corals (Buddemeier and Kinzie III 1976; Bongiorni et al. 2003; De'ath et al. 2009). For this reason, CWC habitats are generally considered to have low recovery potential, which does not promotes an immediately successful restoration (Bekkby et al. 2020). Long time spans (tens of years) will be required for transplanted CWC fragments to growth up to medium sized colonies, and even longer to grow up to functional reef habitats.

Table 1. Growth rates of several CWC species.

Species	*Growth rate (cm yr^{-1})*	*References*
Scleractina		
Madrepora oculata	0.3 – 1.8	Orejas et al. 2008
Lophelia pertusa	0.4 – 1.7	Orejas et al. 2008; Lartaud et al. 2004
Oculina variciosa	1.1 – 1.6	Freiwald et al. 2004
Octacorallia		
Primnoa resedaefromis	1.6 – 2,7	Andrews et al. 2002; Bennecke et al. 2006
Paragorgia arborea	0.1 – 4.1	Bennecke et al. 2006
Eunicella cavolini	**0.05 – 1.3**	**Dominguéz-Carrió personal communication**

Little is known about the reproductive biology of most CWCs, with some information available for the scleractinian corals *L. pertusa*, *Madrepora oculata*, *O. varicosa, Enallopsammia rostrata, Solenosmilia variabilis* and *Goniocorella dumosa* (Brooke and Young 2003; Burgess and Babcock 2005; Waller 2005), and a number of CWC gorgonians (Cordes et al. 2001; Orejas et al. 2007; Mercier and Hamel 2011; Watling et al. 2011). While the knowledge is quite extensive considering fecundity, our understanding of dispersal processes and population connectivity is still hampered by the lack of knowledge about other reproductive traits (Watling et al. 2011). Such factors, as sexual condition (gonochorism or hermaphroditism), reproductive mode (broadcast spawning, internal or surface brooding), reproductive timing (continuous, periodic or seasonal reproduction) as well as larval ecology strongly affect dispersal potential (Waller 2005; Treml et al. 2015; Reynaud and Ferrier-Pagès 2019). Despite there is still limited knowledge about reproductive biology of *E. cavolini* on the Mediterranean continental shelf, a first study suggested lower fertility and reproductive potential compared with the shallower Mediterranean gorgonian *Paramunicea clavata* (Dominguéz-Carrió, personal communication). When information on larval ecology and behavior becomes available, it can be of high value to improve the predictions of larval dispersal (Fox et al. 2016; Strömberg and Larsson 2017; Henry et al. 2018). Likewise, information on genetic population structure in CWCs is limited (Watling et al. 2011). This limited available knowledge on both CWC reproductive ecology and genetic connectivity significantly precludes or hinders a proper spatial planning for CWC conservation (Cudney-Bueno et al. 2009), as well as the identification of priority locations for restoration actions.

1.5 Restoring at appropriate population densities

Density dependence in population dynamics may act as a negative force (inducing density-dependent mortality), but, at the same time, minimum densities are necessary for population persistence or growth (Halpern et al. 2007). Positive population-level interactions include minimum population sizes (avoiding Allee effects) and conspecific cues that enhance recruitment and survival of juveniles (Halpern et al. 2007). This is extremely important for corals, as density may control reproductive success and feeding efficiency (Levitan 1991; Coma and Lasker 1997; Wildish and Kristmanson 1997). Yet, limited research on the role of transplants density have been performed even for shallow-water coral transplantation (Griffin et al. 2015; Ladd et al. 2016). Whenever possible, the coral density of a non-impacted population nearby the restored site could be used as a restoring target. However, in deep-sea environments, given the current extent of anthropogenic impacts and the very limited information about ecosystem baseline conditions, the establishment of such pristine reference sites is usually difficult (Da Ros et al. 2019). For these cases, as in the Cap de Creus, where artisanal fishing is allowed over the study area, nearby natural populations would be a reliable option as reference areas. Transplant density pattern and spatial distribution at "Portaló" site are still far from matching with reference control site, therefore adequate monitoring over time is required to reveal the effectiveness of the restoration action at the long-term (chapter 4).

1.6 Limited knowledge on species interactions

As habitat-forming engineers (Jones et al. 1994), CWCs create physical structures that enhance space, resources, and refuges for hundreds of associated species (Henry and Roberts 2007; Buhl-Mortensen et al. 2010), facilitating the coexistence of species in highly diverse communities (Bruno and Bertness 2001; Stachowicz 2001). Consequently, transplanting corals into impacted environments will provide habitat for other species, facilitate their return and re-establishment, and ultimately aid ecosystem recovery (Abelson 2006; Halpern et al. 2007). In the Cap de Creus area, transplanting *E. cavolini* colonies should result, in the long-term, in positive effects on the associated species richness and diversity (Dominguez-Carrió et al. 2014; Corbera et al. in prep). However, very limited information is available on intra- and interspecific interactions, including predation, competition, symbiosis and facilitation processes in CWC ecosystems, locally in Cap de Creus, as well as worldwide (Buhl-Mortensen and Buhl-Mortensen 2004). These interactions may significantly impact the success of CWC restoration actions. One such example could be the up to four times enhanced calcification rate in *L. pertusa* when living in association with the polychaete *Eunice norvegica* (Mueller et al. 2013). Overall, positive interactions have been recognized as important factors shaping population and community structure (Stachowicz 2001; Bruno et al. 2003), and should be considered in ecological restoration actions to achieve the most effective results, accelerating recovery while reducing restoration times (Bruno and Bertness 2001; Halpern et al. 2007). For the Cap de Creus restoration case study, could be interesting to study the interaction between *E. cavolini* and one of the most abundant sponge coexisting in the area, *Suberites syringella*. Interspecific interactions may result in indirect facilitation processes in trophic cascades (Halpern et al. 2007), as recently highlighted in CWC habitats where sponges play a key role in transferring energy and nutrients to higher trophic levels in the community by transforming dissolved organic matter into particulate organic matter (DOM to POM) (Rix et al. 2016). The presence of such trophic 'sponge loop' contributes to high levels of biogeochemical cycling enabling CWC reefs to develop in deep-sea energy-limited environments (Cathalot et al. 2015; Rix et al. 2016). In this sense, multispecific restoration actions (e.g., including corals and sponges) may be a significant step forward for the successful recovery of functional CWC communities.

1.7 Monitoring the restoration actions

The success of most coral restoration actions is mainly evaluated in terms of transplant survival. In tropical shallow-water coral reefs, highly successful restoration entail survival of more than 85% of restored corals, while failure occurs if less than 10% of restored corals survived after 5 years (Bayraktarov et al. 2016). However, beyond survival of restored organisms, there is active discussion to define appropriate metrics to properly evaluate restoration success (Fonesca et al. 2002; Elliott et al. 2007). Indeed, success of ecological restoration should be measured in terms of recovery of ecosystem function after the restoration effort, monitoring recovery trajectories and comparing with reference control sites (Kaly and Jones 1998; Ruiz-Jaen and Mitchell Aide 2005; Bayraktarov et al. 2016). Monitoring must be standardized, holistic, and linked to the objectives/goals set at the begging of the restoration project. Thus, providing biological, ecological, and physical assessments such as the success of transplanted organisms, changes in overall population and community structure, and changes in key abiotic factors. It is crucial to establish realistic restoration objectives in order to not fail in the evaluation of success, which could reduce public and academic support (Ferse et al. 2010; Boström-Einarsson et al. 2020) and lead to inaction (McAfee et al. 2019). Monitoring allows for the improvement of transplantation techniques and provides guidance for future restoration efforts (Collier et al. 2007). Most active restoration projects, including coral reefs, seagrasses, mangroves, saltmarshes, and oyster reefs, are generally short-term projects, limited to one or two years of duration (Bayraktarov et al. 2016). Of 362 case studies on shallow-water coral restoration, 60% reported less than 18 months of monitoring of restored sites (Boström-Einarsson et al. 2020). Contrarily, a long-term monitoring (15 – 20 years) has been highlighted as paramount to properly evaluate success of tropical shallow-water coral restoration actions (Bayraktarov et al. 2016). For CWC populations, minimum time spans of 30 – 40 years should be considered for a proper monitoring of restored sites, due to their common slow growth rates and population dynamics (Bennecke et al. 2016). Moreover, to achieve the more general recovery of ecosystem functioning, small-scale restoration actions and tests already carried out need to be properly scaled up (Elliott et al. 2007). In fact, there is a mismatch between the scale at which deep-sea ecological restoration can currently be performed and the scale at which major impacts act, as it has already been highlighted for tropical shallow-water coral reefs (Edwards and Gomez 2007; Montoya Maya et al. 2016; Pollock et al. 2017). This is of particular concern for CWCs, because of the logistical challenges and limitations in performing restoration actions in the deep sea.

A combination of AUV and ROV inspection can allow for repeated monitoring of restored areas over time (Armstrong et al. 2010; Morris et al. 2014; Benoist et al. 2019). The possibility for an AUV to autonomously acquire high-resolution images allow to obtain a reference map for relocating transplanted corals. Several studies have already efficiently applied photomosaic for surveys of coral ecosystems (Pedersen et al. 2019), including CWC reefs sites (Boolukos et al. 2019). Applied to restoration research, the development of reference maps is imperative and should occur concurrently with restoration actions (Collier et al. 2007), as in the large-scale restoration action performed in this thesis (chapter 4). The photo-mosaics carried out in the Cap de Creus area allowed to compare the restoration site with a control site, in terms of density and spatial patterns, with initially promising results. Nonetheless, troubles with AUV positioning system together with strong sea-bottom currents dragging the AUV, could lead to uncovered areas, leaving some sections without information. Future studies should promote AUV technical development to overcome current constraints. Indeed, recent advances in sonar imaging point to the possible future use of automatic classification of sonar images acquired by AUVs to regularly monitoring CWC populations (including restored ones). These surveys have the capability to distinguish between dead and live corals on the sonar images or with the assistance of optical cameras (Williams et al. 2010; Huvenne et al. 2011; Sture et al. 2018). Such an extensive monitoring performed with AUVs can be complemented with intensive survey performed with ROVs to acquire images of all, or a subsample of, the restored coral fragments, to monitor through time their survival and growth. High-resolution images of the same individual coral fragment allow for the detection of small changes in the health of coral colonies as well as the measurement of *in situ* growth rates (Hsing et al. 2013; Girard et al. 2018, 2019). Up to date, the limited monitoring period hindered the detection of transplant growth or new colony recruitment in Cap de Creus restored area (chapter 4), thus it would be interesting to obtain high-resolution images of restored transplants. Since pictures of all the transplanted gorgonians were acquired prior their return to the continental shelf, we initially thought to be able to individually recognize each transplant and monitor its growth. However, the quality of video and images acquire were good to detect and map all the transplants on the seafloor, but not to identify each colony and its growth. Moreover, the recent development in 3D photogrammetry for quantitative measurements and its application to ROV-acquired images of deep-sea fauna, including CWCs, is highly promising to represent a step forward in the near future (Bennecke et al. 2016; Thornton et al. 2016; Prado et al. 2019).

Finally, fixed-point deep-sea seafloor observatories with *in situ* cameras and instruments are now functioning in several areas of the world's oceans (Favali et al. 2010), allowing for continuous monitoring and observation of deep-sea fauna, including CWCs (Doya et al. 2014; Van Engeland et al. 2019), and could be also applied to the monitoring of restored CWC populations. Even so, their employment for monitoring CWC restoration action is probably unnecessarily complex and expensive (for installation and maintenance) compared to a wider periodic monitoring by combining AUVs and ROVs (Armstrong et al. 2008).

1.8 Protecting the restoration effort

Restored CWC ecosystems might continue to be exposed to further impacts from anthropogenic activities and ongoing global change (Ramirez-Llodra et al. 2011; Thresher et al. 2015). Consequently, appropriate selection of locations targeted to CWC restoration actions should prioritise refugia areas forecasted to be sheltered from future impacts and extreme changes in environmental features driven by global change (Thresher et al. 2015; Sweetman et al. 2017; Morato et al. 2020). Active restoration may assist CWC habitats recovery once management is in place, but it will certainly fail without effective protection (Edwards et al. 2010). Conservation measures should precede restoration to avoid any direct impacts (Davies et al. 2007) that may compromise the restoration effort. Nonetheless, this is not always possible, for instance, in our study area artisanal fishing is permitted within the Natural Park and the SAC (Special Area of Conservation), therefore impacting gorgonian populations (chapter 1). Management plans of both protection figures (currently in discussion) should regulate fishing practices, for example by implementing recommendations derived from chapter 1. Identifying key ecological valued areas and depth ranges where artisanal fishing should be excluded and limiting the soaking time of nets would significantly reduce the local fishing impact on the studied CWC gardens (chapter 1). Effective management of deep-sea areas can be a challenge, particularly since most are located far offshore (Davies et al. 2007). To this aim, global positioning surveillance of fishing vessels (Marr and Hall-Spencer 2002; Deng et al. 2005) and remotely-imaging with large spatial coverage (Kourti et al. 2001, 2005) may become more cost effective for the monitoring of deep-sea protected areas (Davies et al. 2007).

1.9 Restoration costs

The infrastructures and sophisticated technology required when working on intermediate and deep environments significantly increase the overall cost of CWC restoration (Van Dover et al. 2014; Da Ros et al. 2019). The median cost of shallow-water restoration initiatives is of ~US$ 0.5 M ha^{-1}, ranging from US$ 13,000 to US$ > 1 M ha^{-1} (Spurgeon and Lindahl 2000; Edwards et al. 2010). Contrarily, estimations cost for various hypothetical restoration actions in the deep sea (US$ 1.2 – 4.4 M ha^{-1} only during the first year) concluded that they may be two to three orders of magnitude greater per hectare than costs for restoration in shallow-water ecosystems (Van Dover et al. 2014; Da Ros et al. 2019). In the scenario presented by Van Dover et al. (2014) for the active restoration of CWC reefs following bottom trawling impacts in the Darwin Mounds, the direct costs of only implementing a laboratory propagation-transplant protocol were estimated to be about US$ 75M ha^{-1}. This estimate did not include the additional costs of geoengineering the seabed to reconstruct the mounds on which the corals were first found (Bett 2001; Wheeler et al. 2004; Huvenne et al. 2016). When all of the costs are considered, including socio-economic, ecological and technological considerations, the conclusion was still that the overall balance would be moderately in favour of a (limited) restoration with estimated cost of restoring 0.06 ha in the order of US$ 4.8 M (Van Dover et al. 2014). However, not all active restoration actions in deep waters necessarily incur great costs. Our restoration carried out at intermediate depth (80 – 90 m depth) in Cap de Creus using bycatch CWC gorgonians accounted for about US$ 170,000 ha^{-1} (140,000 € ha^{-1}), a significantly lower cost compared to what was estimated by Van Dover et al. (2014) and more in accordance with shallow-water restoration actions (chapter 4). Indeed, this cost could be significantly reduced through the increased implication of local stakeholders, reducing scientist's involvement and developing cheaper and efficient technology to monitor restored sites. Keeping aside monitoring costs, the cost of preparing and deploying a single gorgonian attached to a natural cobble (the best support) only accounted for about 1€ (chapter 4). Thus, confirming the possibility of developing cost-effective techniques, even targeting intermediate depth and deep-sea environments.

2 CONCLUSIONS

Overall, despite many challenges facing CWC restoration, outcomes from the few active restoration actions performed to date (including the one performed in this thesis) confirm the feasibility of restoring CWC reefs and coral gardens under certain circumstances. Transplanting fragments of coral colonies on artificial or natural substrates or deploying artificial structures to promote colonization are the more viable and reliable methods which have been developed. However, the variable recovery capabilities of CWC species together with the limited knowledge of CWC larval ecology and recruitment point out to the need of combining assisted and natural regeneration approaches. Assisted regeneration (such as transplantation or deployment of artificial reefs) may be useful for some species at local scales, while natural regeneration (through fishery closures and MPAs) at large scales will promote the recovery of individual native species that cannot be transplanted and may take longer to recover. Furthermore, restoration actions should act in conjunction with proper ecosystem management including all local stakeholders, in order to reduce impactful activities that can further degrade restoration sites and compromise restoration efforts, and to ensure long-term monitoring. In the specific case of the Cap de Creus area, the impact of artisanal fishing on CWC gardens on the continental shelf should first be reduced by implementing conservation measures (recommendations in chapter 1). Then, in cases where damage still occurs, the "badminton method" developed in chapter 3 and applied in chapter 4 proved to be a reliable tool to revert and mitigate such local impact. In this sense, it is likely that the combination of well-designed passive and active restoration approaches will be the most effective, as currently employed in shallow-water ecosystems (Mitsch 2014; Possingham et al. 2015).

The present thesis contributes to the significant advances that have been done in CWCs restoration in the very recent years. The restoration methods described here suggest that cost-effective restoration of CWC communities is possible and practical. Ecological restoration of CWC ecosystems is a field of study in an early stage of development and the associated technologies required are at low levels of readiness. The long-term survival of transplanted CWCs and how restoration effort would influence the recovery capacity of CWC ecosystems over decades to centuries is yet to be determined. However, active restoration is a promising tool for the future conservation and management of marine intermediate and deep-sea areas. Wide-scale and long-term restoration actions combining a variety of methods should be tested to determine the ways in which simple interventions might enhance the rate of natural regrowth and recolonization. As opposed to the restoration of terrestrial habitats, marine restoration initiatives are often criticized for being too small and too expensive to compensate for the extent of anthropogenic impacts (Hughes et al. 2017; Bellwood et al. 2019), which undervalues other outcomes, such as the capability to speed up the recovery of local biodiversity or offset the economic and socio-cultural impacts of habitat degradation (Gordon et al. 2020). A more explicit focus on the positive outcomes of restoration would aid their adoption and success, providing a way to connect and engage local communities, scientists, and industry (McAfee et al. 2019).

2.1 Future directions

Several recommendations to move forward the restoration of CWC reefs and CWC gardens arise from this thesis:

- **Establishing a reference baseline to properly evaluate the current state of conservation of CWC populations and communities**. Characterization and monitoring of *E. cavolini* populations in the Cap de Creus continental shelf is needed to evaluate their state of conservation and the effects of fishing impacts at population and community level.

- **Filling the gaps in our knowledge regarding the ecological, biological and genetic features of CWC species and populations**. Information about the natural dynamic of *E. cavolini* populations in the Cap de Creus area is required to advance in understanding how these populations may respond to restoration actions.

- **Testing novel restoration techniques for CWC species, such as the feasibility of culturing and rearing coral fragments and larvae with the goal of scaling-up the extent of restoration actions**. *E. cavolini* has proven to be a resistant gorgonian species, easily to maintain in aquaria. Enhancement of coral fragments growth and larvae recruitment in aquaria and under specific conditions (e.g., temperature and food) should be explored to advance in the capacity to obtain coral colonies for restoration actions, with no impacts to donor populations. Furthermore, it would be interesting to identify which coral species show more adaptive potential and resilience capacity facing the ongoing climate change, thus ensuring a greater success of restoration actions.

- **Exploring multi-specific restoration actions, taking advantage of positive interactions among species**. In CWC gardens of the Cap de Creus area, positive interactions between the most abundant sponge *Suberites syringella* and *E. cavolini* transplants should be investigated in order to enhance restoration outputs.

- **Improving ROV and AUV technology to perform restoration actions and their monitoring**. Develop efficient ROVs and AUVs under high current condition would be a significant improve for the monitoring of restoration action in Cap de Creus. As an alternative, as long as possible, video transects should be oriented parallel to the main water current and multiple transects should be performed in order to ensure covering the entire study area.

- **Performing longer, standardized and holistic monitoring programs which include assessing the recovery of biological and physical attributes, as well as ecosystem function**. For the Cap de Creus restoration action, continuous monitoring overtime will be necessary to detect potential growth of transplanted gorgonians, as well as evidence of recruitment and recovery of associated fauna.

- **Seeking new strategies to involve local stakeholders and society in all phases of restoration actions**. Engagement of local fishers has been paramount for the restoration action in Cap de Creus, specifically in the collection of donor gorgonians. In addition, they should also be involved in the return of gorgonians transplants or in the design and development of lesser impacting fishing nets.

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SUPPLEMENTARY MATERIAL

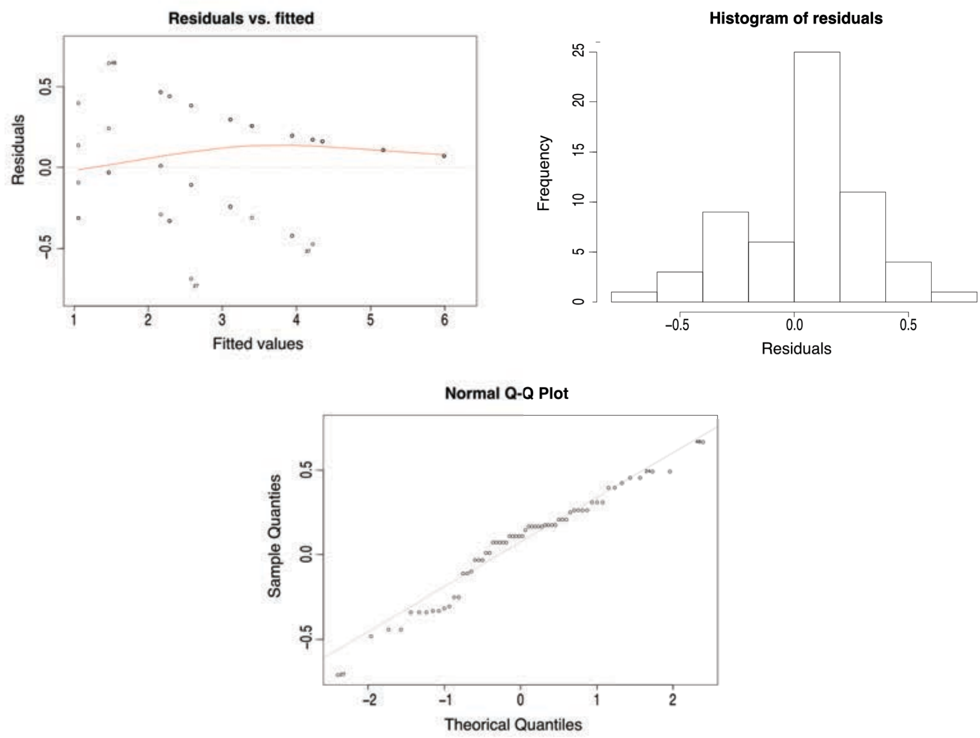
CHAPTER 3

Supplementary Material 1: Comparison of AICc's values between the binomial models (GLM and GLMM) for probability of landing upright according to depth, cobble type and gorgonian size. In the GLMM individual cobble factor (ID) was used as a random effect.

Model	AICc value
glm ǀTransplant position ~Depth + Cobble type + Gorgonian sizeǀ	244.047
glmm ǀTransplant position ~Depth + Cobble type + Gorgonian size + (1ǀID)ǀ	246.078

Supplementary Material 2: A) Diagnostic outputs of the generalized linear model, with a binomial distribution and a logit link function, for probability of landing upright according to depth, cobble type and gorgonian size: (glm ǀTransplant position ~Depth + Cobble type + Gorgonian sizeǀ). **B)** Coefficients table of the GLM.

A)

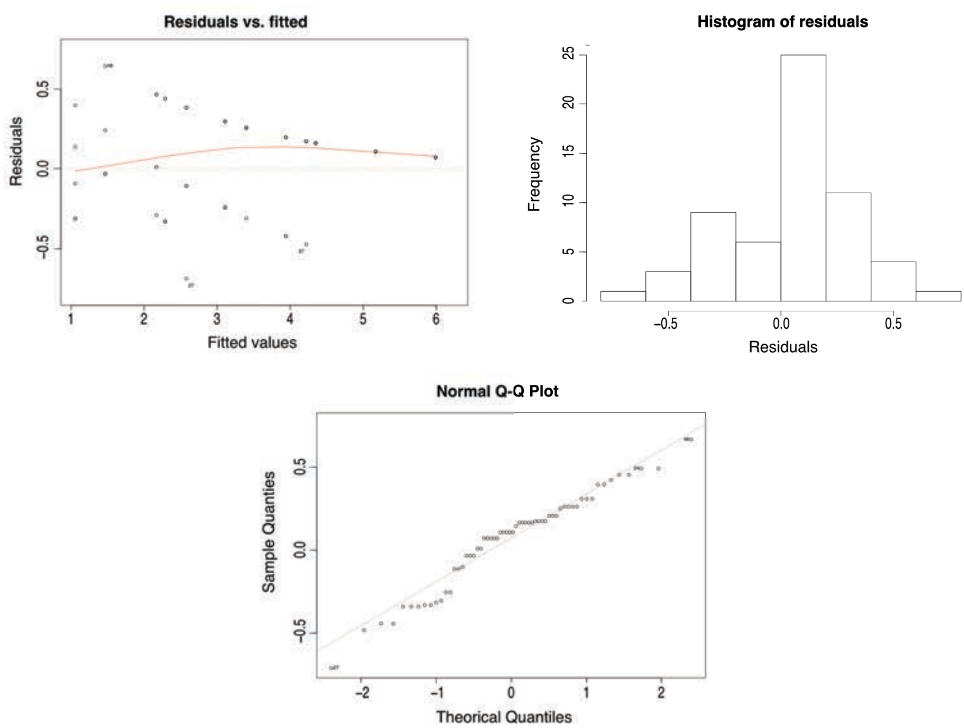


B)

	Estimate	Std. Error	z value	P-value
(Intercept)	4.804	0.832	5.775	<0.001*
Depth	0.080	0.022	3.616	<0.001*
Cobble type LAC	-1.902	0.778	-2.444	<0.001*
Cobble type NAT	-2.887	0.750	-3.850	<0.001*
Small size	-1.804	0.436	-4.134	<0.001*

Supplementary Material 3: A) Diagnostic outputs of the generalized linear model, with a binomial distribution and a logit link function, for probability of landing upright according to depth and cobble type: (glm |Transplant position ~ Depth + Cobble type|). **B)** Coefficients table of the GLM.

A)



B)

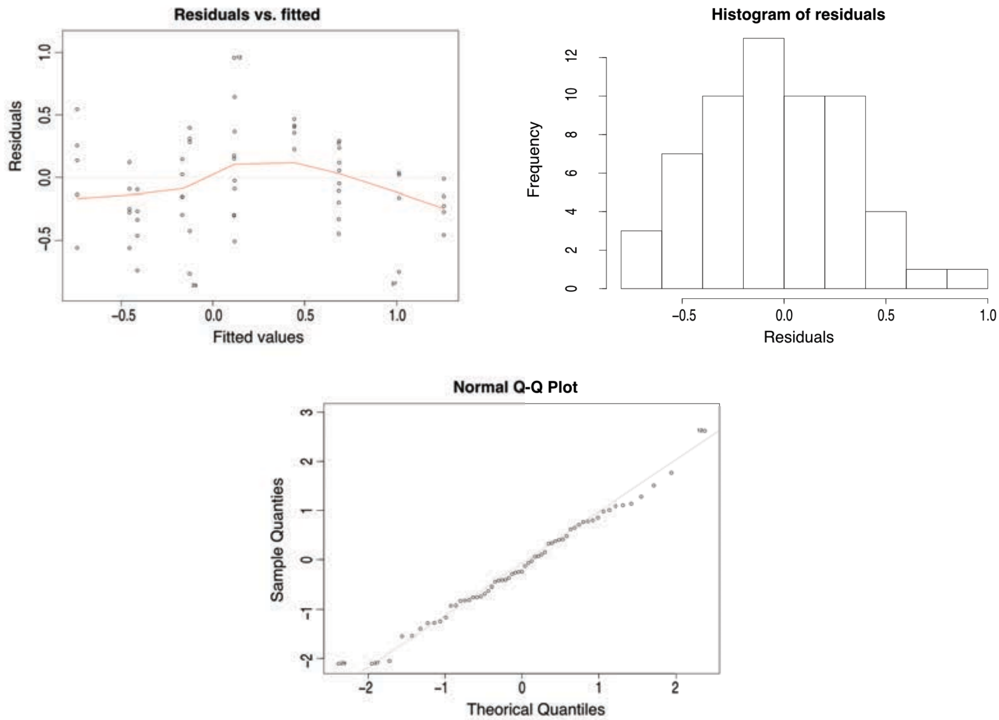
	Estimate	Std. Error	z value	P-value
(Intercept)	3.529	2.363	1.494	0.135
Depth	0.082	0.069	1.182	0.237
Cobble type LAC	-1.771	2.461	-0.720	0.472
Cobble type NAT	-2.881	2.346	-1.228	0.219

Supplementary Material 4: Statistic models fitted to dispersion area (m^2), depth and cobble type data. Probability distribution, link function, Rstudio function and AICc values are specified for each model. Models are ranked from minor to major AICc value. (μ = expected value; $\chi\beta$ = linear predictor).
* Random effect = cobble type

Statistic model	Probability distribution	Link function	Rstudio function	AICc
GLM Gamma	Gamma	$\chi\beta = -\mu^{-1}$	glm	73.884
Non linear regression	Gaussian	$\chi\beta = \mu$	nls	77.035
Linear mixed model*	–	–	lmer	98.523
GAM	Gaussian	$\chi\beta = \mu$	gam	101.581
GLM Gaussian	Gaussian	$\chi\beta = \mu$	glm	101.582

Supplementary Material 5: A) Diagnostic outputs of the generalized linear model, with gamma distribution for dispersion area reach by the transplants and depth: (glm |Dispersion area ~ Depth + Cobble type|) **B)** Coefficients table of the GLM.

A)



B)

	Estimate	Std. Error	z value	Pr(<|z|)
(Intercept)	-0.695	0.117	-5.775	2.13e-07 ***
Depth	0.057	0.005	11.089	1.21e-15 ***
Cobble type LAC	-0.328	0.120	-2.729	0.00852 **
Cobble type NAT	0.244	0.120	2.031	0.04710 *

CHAPTER 4

Table A1: Estimated costs for the different concepts required to apply the Badminton method during two consecutive years (826 restored transplants)

Concept	*Rate*	*Cost*	*Total (€)*
Collection of bycatch gorgonians			
Fishers salary	9 fhisers x 12 moths	500 €/ month/pers	**54 000**
Set up of aquarium facilities (Cadaqués and Port de la Selva)			
100 L Tank	6 units	50 €/unit	300
Biological filter	2 units	87.34 €/unit	174.68
Water pump	6 units	30 €/unit	180
Chiller	2 units	970.95 €/unit	1941.9
PVC and silicone pipes	8 m	2 € /m	16
PVC fittings and other accessories *[1]			~ 150
			2 762.58
Transplants preparation			
(A) For natural cobbles (693)			
Paint	7 units	10.33 €/unit	72.31
Driller	1 unit	40 €/unit	40
Screw	6 units	1.74 €/unit	10.44
Epoxy putty	6 units of 2 components	42 €/unit	252
			374.75
			0.54 €/NC transplant
(B) For small artificial cobbles (133)			
Cement	1 unit	9.38 €/unit	9.38
Plastic moulds	20 units	0.5 €/unit	10
Paint	2 units	10.33 €/unit	20.66
Driller*[2]	1 unit	40 €/unit	40
Screws	2 units	1.74 €/unit	3.48
Epoxy putty	1 units of 2 components	42 €/kg	42
			125.52
			0.94 €/SAC transplant
Transfer and deployment of transplants to restoration sites			
(A) For natural cobbles (693)			
Portable plastic fridge	1 unit	74.95	74.95
Plastic Ice-packs	2 units	2.75 €/unit	5.5
Boat from the Natural Park (gas/oil)	224 km (14 events)	1.3 €/L	291.2
			371.65
			0.53 €/NC transplant
(B) For small artificial cobbles (133)			
Portable plastic fridge	1 unit	74.95	74.95
Plastic Ice-packs	2 units	2.75 €/unit	5.5
Boat from the Natural Park (gas/oil)	48 km (3 events)	1.3 €/L	62.4
			142.85
			1.07 €/SAC transplant
Monitoring of the restoration sites (4 days of monitoring for each year)			
AUV Girona 200 team	2 years	16940 €/year	**33 880**
Scientists' salaries *3	2 persons x 918h*[3]	8.26 € /h/person	**15 165.36**
TOTAL RESTORATION COST			~ 106 782.71

*[1] Since fittings and other pieces were purchased in pre-packed sets depending on the manufacturer it was difficult to determine a cost per unit used. Nevertheless, we were able to
*[2] For the computation of the total cost, the drill was accounted only once.
*[3] Labor time is given in more detail in Table A2 of the supplementary material.

Table A2: Estimated time (h) spent by the scientific staff for the development of the restoration project during both years

Task	***Rate***	***Duration for both years (2018 and 2019)***	***Invested time (h)***
Collaboration with fishers	4h/day	90 days	360
Set up aquarium facilities	6h/day	6 days	36
Transplants preparation	1h/day	360 days	360
Transfer of transplants to restoration si	2h/day	17 days	34
Restoration sites monitoring	16h/day	8 days	128
		TOTAL	**918**

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RESEARCH ARTICLE

WILEY

First attempts towards the restoration of gorgonian populations on the Mediterranean continental shelf

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Abstract

1. In the Mediterranean Sea, gorgonians are among the main habitat-forming species of benthic communities on the continental shelf and slope, playing an important ecological role in coral gardens.
2. In areas where bottom trawling is restricted, gorgonians represent one of the main components of trammel net bycatch. Since gorgonians are long-lived and slow-growing species, impacts derived from fishing activities can have far-reaching and long-lasting effects, jeopardizing their long-term viability. Thus, mitigation and ecological restoration initiatives focusing on gorgonian populations on the continental shelf are necessary to enhance and speed up their natural recovery.
3. Bycatch gorgonians from artisanal fishermen were transplanted into artificial structures, which were then deployed at 85 m depth on the outer continental shelf of the marine protected area of Cap de Creus (north-west Mediterranean Sea, Spain). After 1 year, high survival rates of transplanted colonies (87.5%) were recorded with a hybrid remotely operated vehicle.
4. This pilot study shows, for the first time, the survival potential of bycatch gorgonians once returned to their habitat on the continental shelf, and suggests the potential success of future scaled-up restoration activities.

KEYWORDS

benthos, conservation evaluation, coral, fishing, new techniques, recovery

1 | INTRODUCTION

Unsustainable and destructive fishing activities have been identified as one of the most pervasive threats to marine benthic ecosystems occurring on continental shelves and slopes (~60–1,000 m depth), as these areas endure the bulk of commercial fishing activity (Hall-Spencer, Allain, & Fossa, 2002; Watling & Norse, 1998). Consequently, the vast majority of benthic communities inhabiting these depths have been degraded for decades (Hall, 2002). A large amount of the fishing bycatch (the untargeted catch occurring unintentionally in a fishery) of sessile macrofauna comprises coral, gorgonian, and sponge species dwelling on the continental shelf and slope, as they are easily entangled in trammel nets, longlines, and pots due to their branching morphology and erect structure (Althaus et al., 2009; Bo, Bava, et al., 2014; Durán Muñoz et al., 2011; Sampaio et al., 2012; Wareham & Edinger, 2007). Additionally, these benthic species are also highly exposed to partial mechanical damage (i.e. breakage and tissue abrasion) from the direct impact of fishing activities (Althaus et al., 2009; Mytilineou et al., 2014; Sampaio et al., 2012) and smothering by sediment suspended by bottom-trawling fishing (Grant, Matveev, Kahn, & Leys, 2018). The loss of this benthic habitat-forming species can result in overall loss of the associated biodiversity and is comparable to the impact of forest clear-cutting on terrestrial ecosystems (Watling & Norse, 1998).

Corals, gorgonians, and sponges are among the main engineering species (sensu Jones, Lawton, & Shachak, 1994) in marine ecosystems, where they play an important structural and functional role (Gili & Coma, 1998; Wildish & Kristmanson, 1997). They form complex three-dimensional structures that generate spatial heterogeneity and provide suitable habitat for hundreds of associated species, many of which are of economic importance (Henry & Roberts, 2007; Krieger & Wing, 2002). Moreover, by capturing plankton and suspended particulate organic matter they influence benthic–pelagic coupling processes and biogeochemical cycles (Gili & Coma, 1998). In the Mediterranean Sea, coral gardens dominated by gorgonians are among the main structuring communities in benthic ecosystems on the continental shelf and slope (Angiolillo & Canese, 2018; Bo et al., 2012; Gori et al., 2017). Currently, coral garden distribution on the continental shelf is mostly restricted to areas were bottom trawling does not occur due to the rough topography of the sea bottom (Bo et al., 2015; Fabri et al., 2014; Grinyó et al., 2016). However, since commercial fish species are often associated with these communities, they are largely exploited by artisanal fishermen using trammel nets and longlines (Deidun et al., 2015; Mytilineou et al., 2014). The entire removal or partial damage of coral and gorgonian colonies caused by fishing gears can have far-reaching and long-lasting effects, undermining the long-term viability of their populations (Bo, Cerrano, et al., 2014, Bo et al., 2015), since they are long-lived, slow-growing species, with delayed sexual maturity and limited recruitment success (Coma, Ribes, Zabala, & Gili, 1998; Garrabou & Harmelin, 2002; Linares, Doak, Coma, Díaz, & Zabala, 2007).

Natural recovery of these communities may take centuries, if possible at all (Dayton, 2003). In order to enhance their recovery, active intervention to aid the regeneration of these communities is highly desirable (Rinkevich, 2005). Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed by human activities, bringing it back as close as possible to its undisturbed state (Society for Ecological Restoration International Science & Policy Working Group Restoration [SER], 2004). At present, the practice of ecological restoration is receiving increasing attention worldwide as it offers the opportunity to reverse much of the environmental anthropogenic damage caused by mismanagement of natural resources (Falk, Palmer, & Zedler, 2006). Although marine restoration practices are widespread, mainly in shallow tropical environments (e.g. Precht & Robbart, 2006; Rinkevich, 2005; Young, Schopmeyer, & Lirman, 2012), active restoration initiatives focusing on degraded deeper benthic ecosystems are still extremely uncommon (Brooke, Koenig, & Shepard, 2006; Dahl, 2013).

The adverse impact of fishing to vulnerable marine ecosystems, such as cold-water coral (CWC) reefs and coral gardens (OSPAR Commission, 2010) and the need to conserve them, have become a global concern (Angiolillo & Canese, 2018; Davies, Roberts, & Hall-Spencer, 2007). Owing to the low resilience of coral garden species, they display high vulnerability to disturbance from human activities (Montero-Serra, Linares, Doak, Ledoux, & Garrabou, 2018; Roberts & Hirshfield, 2004), which has prompted a growing interest in protection and restoration initiatives aimed at mitigating their further degradation and enhancing their recovery (Angiolillo & Canese, 2018). Currently, a few deep-sea active restoration initiatives have mainly focused on transplantation actions of CWC species, such as *Oculina varicosa* off the south-eastern coast of Florida (Brooke et al., 2006) and *Lophelia pertusa* in Sweden (Dahl, 2013). Nevertheless, restoration techniques for coral gardens on the continental shelf and deeper environments have not yet been validated.

The main goal of this study was to evaluate, for the first time, the feasibility of recovering and returning to their natural environment bycatch gorgonians from the Mediterranean continental shelf in order to mitigate fishing impact. Bycatch gorgonians collected from artisanal fishermen were transplanted onto artificial structures, deployed at the continental shelf (85 m depth) in a marine protected area, and monitored using a hybrid remotely operated vehicle (hybrid ROV). This pilot action could be applicable to deeper ecosystems that require similar technical logistics and is a first essential step in assessing the feasibility of future large-scale ecological restoration of CWC gardens.

2 | METHODS

2.1 | Gorgonian collection and maintenance

Colonies of the gorgonian *Eunicella cavolini* (Koch, 1887) were obtained from artisanal fishermen's bycatch from Cap de Creus (north-western Mediterranean Sea, 42°19.12′N; 03°19.34′E), at a depth range of 70 to 100 m, during three fishing sorties in June and one in August 2015. Fishermen picked up gorgonians entangled in trammel nets and kept them in containers filled with surface sea water (~20–23°C). Once back on land (1–2 hr after collection), gorgonians were transported to the experimental aquarium facilities of the Institute of Marine Sciences (ICM–CSIC) in Barcelona (within 3–4 hr after the initial pick up), while seawater temperature was kept at 14 ± 1.0°C at all times. A total of 120 gorgonians were held in 100 L tanks with continuous seawater flow, filtered through a 50 μm sand filter (Olariaga, Gori, Orejas, & Gili, 2009), fed frozen *Cyclops* three times a week, and kept at 14 ± 1.0°C in the dark, thus simulating Cap de Creus continental shelf's natural conditions. The size of the collected colonies ranged from 6.7 to 22.4 cm (12.3 ± 4.6 cm, mean ± *SD*), and they were held under the aforementioned conditions between a few days and a maximum of 2 months.

2.2 | Transplant on artificial structures and deployment on the sea bottom

From June 27 to 30, 2015, 80 gorgonians were transplanted onto two stainless steel structures (40 gorgonians onto each; outer diameter: 2 m; inner diameter: 1.5 m), with a base grid (10 × 10 cm^2) surrounded by four concrete plates and a central 1 m vertical axis holding an acoustic reflector (30 cm in diameter) supported by four stainless steel bars (12 mm in diameter) (Figure 1). Forty conical supports for the gorgonians (80 mm high, 20 mm diameter) were placed on the grid. The inside of the supports was filled by polyester fibreglass resin and, once dry, 8 mm boreholes were made in order to attach the gorgonians colonies with epoxy putty (Corafix SuperFast, GROTECH®). Each structure weighed 137 kg in the air. Initially, the structures were deployed at 6 m depth north of the marine protected area of Cap de Creus, where gorgonians (entire colonies) were attached to the supports by scuba divers. Each structure was then raised up to below the water surface by means of a buoy and transported by boat at a slow and constant speed (~0.5 kn) towards the continental shelf, where they were deployed at 85 m depth (structure 1: 42°20.06′N; 03°18.67′E; structure 2: 42°20.05′N; 03°18.67′E). Since an additional 40 gorgonian colonies were collected as bycatch in fishing events in August, they were transplanted later on a third structure on October 23–24, 2015, and deployed on October 25, 2015, nearby the first two structures (structure 3: 42°20.05′N; 03°18.64′E) following exactly the same procedure. The density value of colonies transplanted onto each structure corresponds to ~15 colonies/m^2, and was selected based on data about Mediterranean gorgonian assemblages dwelling at 40–300 m depth (10–20 colonies/m^2; Bo et al., 2009; Grinyó et al., 2016).

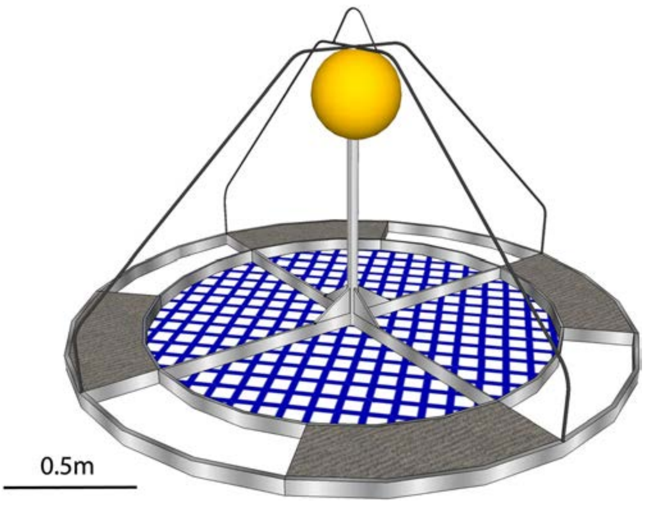

FIGURE 1 Schematic figure of the stainless steel structures used in this study

2.3 | Monitoring of transplanted colonies

The structures were monitored through three consecutive surveys using the Girona 500 autonomous underwater vehicle, equipped with the Bumblebee stereo camera, working as a hybrid ROV (Carreras et al., 2016). Surveys were conducted on July 21, 2015 (21 days after deployment for structures 1 and 2), December 12, 2015 (6 months after deployment for structures 1 and 2; 47 days for structure 3), and September 2, 2016 (14 months after deployment for structures 1 and 2; 10 months for structure 3). During each survey, the hybrid ROV used sonar to locate the acoustic reflector and approach each structure. The images, with a resolution of 1,024 × 768 px^2, were subsequently collected by encircling each of the structures, while maintaining the gorgonians in the centre of the view. The robot maintained an approximately constant distance of 2 m between the camera and the centre of the structure, enabling observations of the gorgonians from various directions with sufficient image quality to allow successful assessment of their survivorship. Gorgonian survival was assessed by individually observing if each transplanted colony was still in place and alive (with no evidence of necrotic tissue).

The three-dimensional (3D) reconstructions of the three structures deployed on the continental shelf with transplanted gorgonians (Figure 2 and Supporting Information) were made using an optical 3D reconstruction procedure, as described in Hernández et al. (2016). The final models are obtained through a series of steps, starting with the simultaneous optimization of the pose of the camera (at each moment of the image acquisition) and the sparse geometry of the structure, followed by densification of the geometrical representation, surface estimation, and texture mapping.

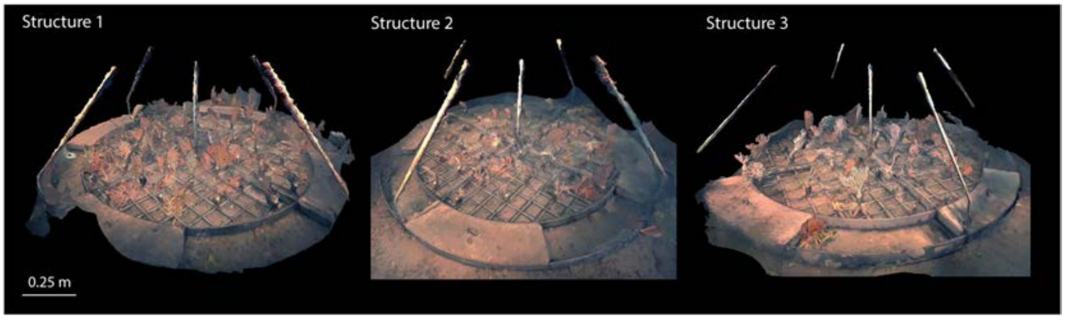

FIGURE 2 Three-dimensional (3D) reconstruction of the three structures deployed on the continental shelf with transplanted gorgonians. The 3D visualization of structure 1 can be found in Supporting Information. The 3D computer model was obtained from images acquired by the hybrid remotely operated vehicle while circling the structures, using the 3D reconstruction pipeline described in Hernández et al. (2016)

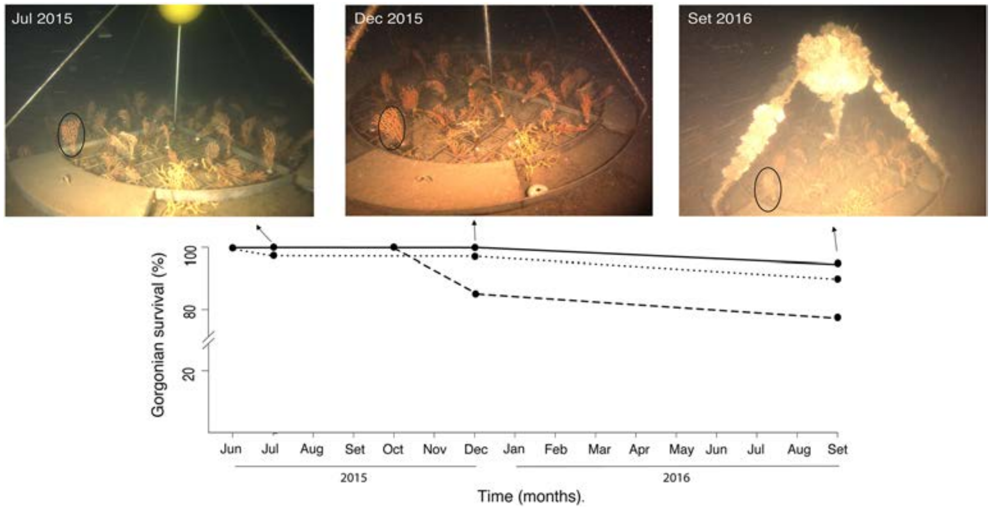

FIGURE 3 Survival rate of transplanted gorgonians for each structure during the study period. Solid line corresponds to structure 1, dashed line to Structure 2, and dotted line to structure 3. Pictures correspond to structure 1 during the consecutive surveys

3 | RESULTS

Several of the gorgonians collected from fishermen showed partial breakage and a little evidence of tissue abrasion. Even so, they all recovered and survived while being maintained in aquaria at ICM–CSIC prior to redeployment at sea. On structures 1 and 2, 98.8 ± 1.8% (mean ± *SD*) of the transplanted gorgonians were still in place at the time of the first survey (21 days after deployment), and they all were still surviving after 6 months at the time of the second survey. On structure 3, 85% of the transplanted gorgonians were still in place at the time of the second survey (47 days after deployment). Finally, approximately 1 year after deployment (14 months for structures 1 and 2, and 10 months for structure 3) 87.5 ± 9.0% (mean ± *SD*) of the gorgonians were still in place and alive on the three structures (Figure 3).

4 | DISCUSSION

This pilot study has assessed, for the first time, the feasibility of successfully returning bycatch gorgonians recovered from artisanal fishery to their natural environment on the Mediterranean continental shelf. Initial results showed that, in spite of some *E. cavolini* colonies suffering partial breakage, tissue abrasion, or both, all colonies survived while being maintained in aquaria. This survival may be attributable to the species' high healing rate (0.085 mm of tissue recovery·per day; Fava, Bavestrello, Valisano, & Cerrano, 2010). In contrast, other common Mediterranean gorgonians, such as the red gorgonian *Paramuricea clavata* (which is also frequently collected by artisanal fishermen in Cap de Creus), show low survival rates when recovered from bycatch and are maintained in aquaria, with a rapid degradation of living tissues and high colony mortality (M. Montseny, personal observation). These observations highlight the importance of understanding the biological and ecological characteristics of each species before engaging in any restoration initiative (Montero-Serra, Garrabou, et al., 2018), and points at *E. cavolini* as a suitable gorgonian species for restoration projects in the Mediterranean continental shelf.

Monitoring of structures shortly after their deployment (21 days or 47 days, depending on the structure) suggested that initial loss of gorgonians was mainly due to colony detachment during the structure deployment on the continental shelf (Figure 3). Although natural mortality cannot strictly be excluded, the high survival rate following initial losses (Figure 3), which is in accordance with previous gorgonian transplantations in Mediterranean shallower habitats (Fava et al., 2010; Linares, Coma, & Zabala, 2008), suggests that the initial successful securement of a colony to the substrate is critical to its long-term survival with a relatively minor effect of stress due to transplantation (Linares et al., 2008). Gorgonian transplants in the present study showed high survival (almost 85%) approximately 1 year after deployment, in line with the high survival observed for *Corallium rubrum* 4 years after transplantation (about 99.1%) (Montero-Serra, Garrabou, et al., 2018), and much higher when compared with transplanted *Eunicella singularis* (35–45% survival after 1 year), *Eunicella verrucosa* (30% survival after 1 year) and *Paramuricea clavata* (35–50% survival after 1 year) (Fava et al., 2010; Linares et al., 2008; Montero-Serra, Garrabou, et al., 2018).

In shallow Mediterranean environments, survival of transplanted gorgonians can be compromised by several environmental parameters generally associated with seasonal fluctuations, such as high water turbulence, high irradiance, algal competition (Linares et al., 2008; Weinberg, 1979), and thermal stress (Fava et al., 2010). Long-term survival of *E. cavolini* transplants on the continental shelf may thus be partially explained by the higher stability of environmental factors in deeper habitats (below ~40 m depth) (Garrabou, Ballesteros, & Zabala, 2002; Grinyó et al., 2018). Indeed, the outcomes from shallow restoration studies in tropical ecosystems with high environmental stability are in accordance with the high gorgonian survivorship detected in the present study (Edwards & Gomez, 2007; Guzmán, 1991). However, tropical corals encompass species with contrasting life history traits, including fast- and slow-growing species (Darling, Alvarez-Filip, Oliver, Mcclanahan, & Côté, 2012), which make tropical transplant survival rates highly variable (43–95% during the first year) (Lindahl, 2003; Yap, Alino, & Gomez, 1992; Young et al., 2012). Thus, the high survival rate detected in this study is consistent with the notion that slow-growing species require little initial transplantation effort, since they show high survival rates after transplantation in comparison with fast-growing species, but the period required to fully re-establish habitat complexity will tend to be far longer (Montero-Serra, Garrabou, et al., 2018).

In comparison with shallow-water restoration studies, there are only a few instances of ecological restoration attempts in deeper habitats. The first such attempt to restore a deep-sea coral ecosystem was conducted with the CWC *Oculina varicosa* in Florida, where restoration modules were deployed at 70–100 m depth with colonies transplanted that showed moderate survival rates (50–60%) after 1 year (Brooke et al., 2006). In Sweden, a restoration action focused on the CWC *Lophelia pertusa* recorded high survival of the transplants (76%), which increased in size by 36% after more than 3 years (Dahl, 2013; Jonsson et al., 2015). Similarly, an in situ growth study showed high survival (over 90% polyp survival) and active colony growth of *L. pertusa* fragments deployed during 1 year at ~500 m depth in the northern Gulf of Mexico (Brooke & Young, 2009).

These first pilot studies (including the present one) demonstrate the feasibility of the active restoration of CWC reefs and coral gardens, which should encourage future initiatives aimed at recovering, preserving, and sustainably managing these vulnerable marine ecosystems. However, ecological restoration of intermediate depths and deep-sea habitats involves considerable constraints due to the difficult access, requiring the use of advanced underwater technology entailing high economic cost. Deep-sea restoration cost per hectare has been estimated at two to three orders of magnitude higher than for shallow marine ecosystems (Van Dover et al., 2014). Future availability of accessible cost-effective underwater technology (such as relativity low-cost autonomous underwater vehicles for monitoring) will be paramount for the wide application and upscaling of coral and gorgonian restoration at depths below conventional or technical scuba diving limits.

The ultimate goal of restoration initiatives should be to achieve the recovery of the structure and ecological functioning of affected ecosystems (McDonald, Gann, Jonson, & Dixon, 2016; SER, 2004). For coral gardens, restoration of sessile engineering species can drastically alter the abiotic system state and trigger a consequent response in the biotic state (Byers et al., 2006), such as that transplanted gorgonians not only provide habitat structure but also enhance the recovery of associated biodiversity and positively influence ecosystem functioning (Geist & Hawkins, 2016). Overall, although restoration is often a long-term investment and its potential results are still highly uncertain (Suding, 2011; Van Dover et al., 2014), the results of this pilot project highlight the feasibility of using bycatch gorgonians recovered from artisanal fisheries to mitigate the fishing-related degradation by restoring coral gardens on the Mediterranean continental shelf. This is an essential first step that leads to future large-scale and cost-effective restoration actions of coral gardens located on the continental shelves or in even deeper environments. In contrast to most restoration practices using coral transplants obtained from fragmentation of donor colonies (Brooke et al., 2006; Dahl, 2013), restoration based on bycatch gorgonians would minimize damage to other colonies or populations. Nevertheless, to be effective, these restoration actions should be accompanied by a reduction of fishing impacts in the restored areas, by partial closures, or by improving fishing techniques.

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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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RESEARCH ARTICLE

A new large-scale and cost-effective restoration method for cold-water coral gardens

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Abstract

1. Gorgonians dwelling on the Mediterranean continental shelf are among the most frequent fishing bycatch taxa. These species display several traits, such as long lifespans and slow growth, which make them very vulnerable to the impacts caused by fishing activities with far-reaching and long-lasting effects.
2. Hence, restoration and mitigation actions are crucial to enhance and speed up the natural recovery of damaged cold-water coral gardens. Given the growing concern to develop effective and affordable restoration actions, the present study aims to propose and technically validate a new large-scale and cost-effective restoration method.
3. This technique, named 'badminton method', consists of attaching bycatch *Eunicella cavolini* colonies to cobble supports and returning them to the continental shelf by gently throwing the gorgonian transplants directly from a boat.
4. Two consecutive field experiments were conducted in order to find the best cobble type support and gorgonian size to be used: first, to evaluate the landing efficiency of gorgonian transplants at different depths (from 5 to 30 m) and second, to evaluate their capability to maintain a correct position over time.
5. Natural cobbles with large gorgonians attached were the best option. Field results and modelling approaches suggest that the transplants would correctly land on the continental shelf seabed in a predicted area of around 60 m^2. Moreover, they would successfully maintain an upright position ensuring gorgonian survival over time.
6. The success of this method highlights the feasibility of large-scale and low-cost restoration actions with promising results for the conservation and recovery of cold-water coral gardens.

KEYWORDS
benthos, conservation evaluation, coral, fishing, new techniques, recovery

1 | INTRODUCTION

Over recent decades, the increasing attention towards deep water ecosystems and the continued advancements in deep sea exploration technology have contributed to expand knowledge about benthic communities dwelling on continental shelves (~60–200 m depth) and deep sea bottoms (>200 m depth) (Althaus et al., 2009; Freiwald, Fossa, Grehan, Koslow, & Roberts, 2004). At depths of 60–1000 m, cold-water corals and sponges are among the main habitat-forming species, generating complex three-dimensional ecosystems that create hotspots of biodiversity over large areas (Hovland, 2008; Roberts, Wheeler, Freiwald, & Cairns, 2009). They provide suitable habitat, acting as feeding, reproductive, nursery, and refuge areas for a wide variety of associated species, many of which are of commercial interest (Cartes, LoIacono, Mamouridis, López-Pérez, & Rodríguez, 2013; Henry & Roberts, 2007; Krieger & Wing, 2002). At the same time, cold-water coral reefs, coral gardens, and sponge grounds also play an important role in the benthic–pelagic transfer of energy and matter (Gili & Coma, 1998; Graf, 1989) as well as in most of the biogeochemical cycles of the deep sea (Cathalot et al., 2015; Coppari et al., 2016; Van Oevelen et al., 2009).

The continuously increasing exploitation of deep sea resources is currently recognized as a major threat (Jackson et al., 2001; Morato, Watson, Pitcher, & Pauly, 2006). Since the last century, continental shelves and upper slopes have been heavily impacted by fishing activities (Jones, 1992; Koslow et al., 2000; Oberle, Puig, & Martín, 2017). Bottom trawling represents one of the main threats to benthic ecosystems (Clark et al., 2016; Hall-Spencer, Allain, & Fossa, 2002), resulting in a severe oversimplification of benthic communities (Reed, Koenig, Shepard, & Gilmore, 2007; Thrush & Dayton, 2002; Watling & Norse, 1998). However, in rocky areas with complex topography inaccessible to trawling, artisanal fishing can also jeopardise the integrity of benthic ecosystems (Angiolillo & Canese, 2018; Sampaio et al., 2012).

In the Mediterranean Sea, gorgonians are among the most frequent species of bycatch of artisanal trammel net and longline fishing in coastal areas and on the continental shelf. Due to their arborescent morphology and erect position, gorgonians are easily entangled in bottom-contact fishing gears (Deidun et al., 2015; Mytilineou et al., 2014). These fishing gears can remove entire gorgonian colonies, or partially damage their tissue, making them more vulnerable to diseases and/or epibiont overgrowth (Angiolillo et al., 2015; Bo, Bava et al., 2014; Bo, Cerrano et al., 2014). Since most Mediterranean gorgonians are long-lived, slow-growing species, with delayed sexual maturity and limited recruitment (Coma, Ribes, Zabala, & Gili, 1998; Linares, Doak, Coma, Díaz, & Zabala, 2007), fishing activities impacting their populations can have far-reaching and long-lasting negative effects in their long-term viability (Bo et al., 2015; Bo, Cerrano, et al., 2014). As a consequence, Mediterranean cold-water coral gardens located on the continental shelf and slope have recently been internationally recognized as Vulnerable Marine Ecosystems (Fabri et al., 2014; OSPAR Commission, 2010), stressing the urgent need for their sustainable management and conservation (Aguilar & Marín, 2013; Davies, Roberts, & Hall-Spencer, 2007; FAO, 2009). The conservation and recovery of benthic engineering species such as gorgonians will also help to preserve all their associated fauna, maintaining the ecosystem functioning and the ecosystem services provided (Byers et al., 2006; Geist & Hawkins, 2016). It is, thus, highly desirable to actively initiate or improve the slow natural recovery of impacted cold-water coral assemblages, by means of reducing impacts and assisted regeneration with biotic intervention (i.e. active restoration) (Possingham, Bode, & Klein, 2015; Van Dover et al., 2014).

Ecological restoration offers the opportunity to redirect the environmental damage caused by anthropogenic impacts and assist the recovery of a natural range of ecosystem composition, structure, and dynamics (Falk, Palmer, & Zedler, 2006; Society for Ecological Restoration International Science & Policy Working Group, 2004). However, until now, most of the actions and initiatives carried out in marine ecosystems have been concentrated on the restoration of shallow habitats, particularly in tropical ecosystems (i.e. Guzman, 1991; Rinkevich, 2005; Young, Schopmeyer, & Lirman, 2012). Only a few restoration attempts have been carried out in the deep sea, specifically targeting cold-water coral species (70–1300 m depth, Boch et al., 2019; Brooke, Koenig, & Shepard, 2006; Dahl, 2013) and coral gardens on the Mediterranean continental shelf (60–120 m depth, Montseny et al., 2019). High survival values of coral and gorgonian transplants were found in these studies, highlighting the feasibility of active restoration of cold-water coral reefs and coral gardens, despite the considerable limitations associated with the difficulties of working at intermediate and deep depths (Clauss & Hoog, 2002).

Among these constraints, the need for advanced underwater technologies is considered to increase the economic cost of any deep sea restoration action compared to shallow areas (Van Dover et al., 2014). Likewise, the difficult and extremely limited access to these habitats also restricts restoration actions and their monitoring to small spatial extents (Brooke et al., 2006; Dahl, 2013). However, since the main stressors impacting most continental shelf and deep sea ecosystems are largely widespread (Halpern et al., 2008), scaling up and reducing economic costs are necessary steps to ensure a long-term viability of deep sea ecological restoration actions.

Based on the demonstrated capability of bycatch gorgonians to survive after their transplantation back into their natural environment (Montseny et al., 2019), the main goal of the present study was to explore the technical viability of a new large-scale and cost-effective restoration method. Gorgonian colonies recovered from artisanal fishery bycatch (trammel nets) were attached to cobble supports, and transplanted to the continental shelf by gently throwing them from the sea surface. The landing efficiency (i.e. successful landing in upright position and dispersion area of transplants) and the capability to maintain the upright position over time were estimated according to depth, the characteristics of the cobbles used, and the size of the transplanted gorgonians. The method has been named 'badminton method' because the fan-shaped morphology of the colonies attached to the cobbles makes them act as a badminton shuttlecock, slowing down the fall and facilitating an upright landing. It was inspired by the common practice of artisanal fishermen from Menorca (Balearic Islands, Spain) to return to the sea gorgonians that they accidentally catch attached to maërl cobbles, by throwing them from the sea surface.

2 | METHODS

2.1 | Studied species

Eunicella cavolini (Koch, 1887) is one of the most common Mediterranean gorgonian species (Carpine & Grasshoff, 1975; Weinberg, 1976), showing a wide bathymetric distribution (<10–220 m depth) (Bo et al., 2012; Russo, 1985). In general, this non-symbiotic gorgonian presents fan-shaped colonies with a variable ramification pattern and growth rate, depending on environmental conditions. Commonly, the ramifications are numerous and tend to be curved, pointing in many directions although lying in one plane, fully exposed to the main current flux (Velimirov, 1973; Weinbauer & Velimirov, 1995). In the Mediterranean Sea, the mean height of the entire colony is around 15 cm (Sini, Kipson, Linares, Koutsoubas, & Garrabou, 2015) with largest colonies reaching 50 cm height (Bo et al., 2012). In the Cap de Creus area (north-western Mediterranean Sea, 42° 19′ 12″ N – 003° 19′ 34″ E), this species is abundant on the outcropping rocks at 80–120 m depth on the continental shelf, forming high-density patches reaching densities of 25 colonies m^{-2} (Dominguez-Carrió, 2018; Gili et al., 2011).

2.2 | Gorgonian collection and maintenance

Bycatch colonies of *E. cavolini* (5–17 cm height) were collected from trammel nets fishing during summer 2016 with the collaboration of artisanal fishers. The collected gorgonians were kept on board in a bucket filled with surface sea water (~22–25°C). Once on land, colonies were kept in a 100-L tank filled with sea water maintained at 14 ± 1.0°C and with a submersible pump providing continuous water movement. A chiller (Tank chiller line TK 2000) was used to maintain a constant seawater temperature, and the water was filtered using a biological filter (SERA fil bioactive 250+UV). As soon as it was possible (after 1 or 2 days), gorgonians were moved to the Institute of Marine Sciences in Barcelona (Spain) without being adversely affected during the transport. There, gorgonians were held in a 100-L tank with a continuous flow of Mediterranean sea water pumped from a depth of 15 m at a rate of 50 L h^{-1} and filtered with a 100-μm pore size (Olariaga, Gori, Orejas, & Gili, 2009). A submersible pump provided continuous water movement in the tank with a flow rate of 320 L h^{-1}, seawater temperature was constantly maintained at 14 ± 1.0 °C and gorgonians kept in dark conditions, corresponding to the natural conditions of the Cap de Creus continental shelf (Dominguez-Carrió, 2018). Gorgonians were held under these conditions during a few weeks up to 5 months, being fed three times a week with frozen *Cyclops* (Crustacea, Copepoda).

2.3 | Gorgonian transplants preparation

To identify the best support to be used in this gorgonian restoration method, three different types of cobbles were tested: (A) natural cobbles, (B) small artificial cobbles, and (C) large artificial cobbles (Figure 1). Natural cobbles were collected from the coastal area of Cap de Creus (mean width: 9.7 ± 1.6 cm, mean length: 12.4 ± 1.6 cm, mean height: 3.1 ± 1.0 cm, mean weight: 455 ± 139 g). Artificial cobbles with established measurements were built with concrete: small artificial cobbles had a square shape (width: 8.0 cm, length: 8.0 cm, height: 2.5 cm, weight: 175 g); and large artificial cobbles had a semi-spherical shape (diameter: 13.3 cm, height: 4.5 cm, weight: 450 g). Thirty cobbles in total were used (10 per cobble type). To identify the most effective gorgonian size to be transplanted, large gorgonians (10–17 cm height, 80–300 cm total ramification length) were transplanted on five cobbles per type, and small gorgonians (5–10 cm height, 20–80 cm total ramification length) were transplanted on the other five cobbles per type. Gorgonians were attached to supporting cobbles using epoxy putty (Corafix SuperFast, GROTECH®). All colonies used as transplants did not show any signal of necrotic tissue before being returned to the sea.

2.4 | Landing efficiency

The first part of the field study consisted of assessing the landing efficiency of the gorgonian transplants gently thrown from the sea surface. The dispersion area on the sea floor and success of upright landing were quantified at 5, 10, 20, and 30 m depths (Cap de Creus, 42° 28′ 39″ N – 003° 28′ 31″ E). At each depth, the 10 cobbles with five large and five small gorgonians transplanted were gently thrown from a boat. Once at the sea floor, and by means of scuba diving, a hand-drawn map of the 10 transplants was produced by measuring *in situ* the perpendicular distance (cm) between the position of each transplant from a central transect established in the area where the transplants had landed. Finally, the landing success of each gorgonian was estimated by recording *in situ* whether the colonies had landed in an upright or overturned position. This procedure was repeated five times for each cobble type and depth.

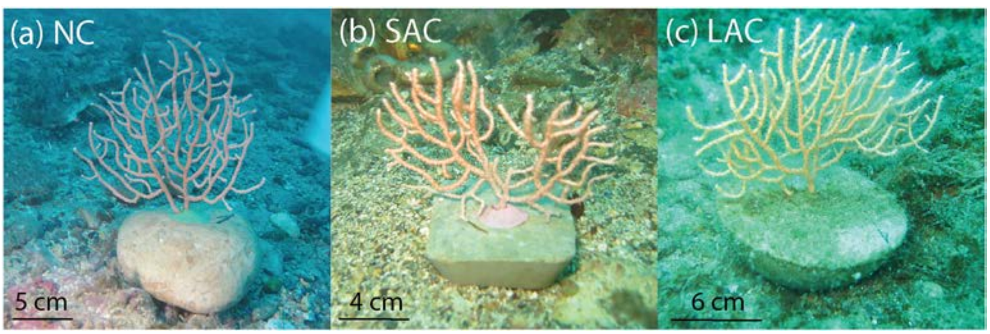

FIGURE 1 Gorgonian transplants with different types of cobbles tested in the study. (a) NC = natural cobbles; (b) SAC = small artificial cobbles; and (c) LAC = large artificial cobbles

2.5 | Monitoring of transplants

The second part of the field study consisted of assessing whether the transplants, once on the sea floor, maintained the correct upright position over time. To this aim, new gorgonians with no signs of necrosis, were attached to the same three types of cobbles, as previously described. Again, large gorgonians were transplanted on five cobbles per type, and small gorgonians were transplanted on the other five cobbles per type. The resulting 30 cobbles (10 cobbles per type) were placed by hand in an upright position on a horizontal bottom composed of small cobbles and sand at 30 m depth (Cap de Creus, 42° 17′ 03″ N – 003° 17′ 95″ E), and monitored by scuba divers in 3 consecutive months. Colonies were individually photographed on a ruled tablet in order to acquire pictures of the transplants with a size reference. From these pictures, the position of the transplants (upright or overturned) was established, and for each gorgonian the amount of necrotic tissue along the total ramification length was quantified to estimate their survival and health status.

2.6 | Data analysis

To analyse the cobbles probability of landing success (response variable) according to depth (explanatory continuous variable), cobble type and gorgonian size (explanatory discrete variables), two model types were fitted: (1) a general linear model (GLM) with a binomial error distribution and a logit link function, (2) a general linear mixed model with a binomial family, identifying individual cobbles as random effect. Finally, only the simplest binomial GLM was plotted, which presented the lowest Akaike information criterion with small-sample correction value (AICc; see Supplementary Material 1). To test the significance effects of depth, cobble type, and gorgonian size, an ANOVA Type II test for non-sequential factors, was applied over the fitted model (GLM). A GLM with a binomial error distribution and a logit link function was also used to predict the probability of successful landing in a hypothetical situation of throwing the transplants up to 80 m depth according to cobble type. Additionally, given the great variability in shape and weight among the natural cobbles used in the experiment, we explored the relationship between the sphericity index (SI) and the weight of cobbles, and their probability of landing upward, using a Spearman rank order correlation (Spearman, 1904). The SI of cobbles (Ψp) was calculated from their maximum length, width and thickness measures (Sneed & Folk, 1958).

The total dispersion area (m^2) of the transplants on the sea floor was measured from the maps obtained *in situ* by scuba diving. For each cobble type and depth, the area corresponding to the smallest ellipse that included all the transplants was quantified using the Macnification 2.0.5 software (Orbicule, Leuven, Belgium). To estimate the area of dispersion in which transplants would extend on the continental shelf (80 m) according to cobble type, different statistical models were built. Depth (continuous variable) and cobble type (discrete variable) were the explanatory variables, while the dispersion area was the continuous response variables. Several GLM models, non-linear regression model, mixed linear model and GAM model were adjusted to detect which of them showed the lowest AICc value and therefore better fit the data.

Regarding the monitoring transplant study, the evolution of their position (upright or overturned) according to the cobble type was plotted during the 3-month period. Additionally, a Wilcoxon signed-rank statistical test (Wilcoxon, 1945) was performed at the end of the monitoring to detect the influence of gorgonian size in overturning the transplants. To evaluate the status of transplanted colonies and compare them to natural values (Coma, Pola, Ribes, & Zabala, 2004; Linares, Coma, Garrabou, Díaz, & Zabala, 2008), all colonies were photographed during the monthly monitoring at 30 m depth and they were analysed with Macnification 2.0.5 software. At the end of the 3-month experiment, the percentage of necrosis on the total ramification length ([necrotic tissue length/total linear length] × 100) was calculated for each colony that maintained the upright position. Additionally, a Wilcoxon signed-rank statistical test (Wilcoxon, 1945) was used to detect differences in percentage of necrosis between large and small studied gorgonian sizes. Statistical analyses were carried out with R (RCore Team, 2018) by means of the R Studio software (RStudio Team, 2016) using the ‘ggplot2' package (Wickham, 2016), the ‘lme4' package (Bates, Mächler, Bolker, & Walker, 2015) for the linear mixed model and the general linear mixed model, and the ‘gam' package (Hastie & Tibshirani, 1987) for the GAM model. The ‘AICcmodavg' package (Mazerolle, 2019) was also used in the model comparison.

3 | RESULTS

3.1 | Landing efficiency

Depth, cobble type, and gorgonian size affected the landing success of transplants (Anova Type II of the binomial model; Depth: χ^2 = 16.045; df = 1; $P < 0.001$; Cobble type: χ^2 = 29.463; df = 2; $P < 0.001$ Gorgonian size: χ^2 = 22.314; df = 1; $P < 0.001$. Figure 2 and Supplementary Material 2). For all the cobble types tested, and regardless of the gorgonian size, the probability of landing upright increased with depth (Figure 2). Transplants with small artificial cobbles displayed the highest probabilities of successful landing at all depths (close to 100% of upright landing; Figure 2), followed by large artificial cobbles and natural cobbles. According to the gorgonian size, large gorgonian colonies showed a higher probability of upright landing than small colonies (Figure 2). Regarding the prediction model, a 100% success of gorgonian landing in upright position at 80 m depth (i.e. on the continental shelf), for all the cobble types was forecast (Figure 3 and Supplementary Material 3). Focusing only on natural cobbles, there was a strong positive relationship (Spearman's r = 0.80; P = 0.004; n = 10) between the landing success of natural cobbles (% of upright landing) and their SI (Figure 4). On the contrary, there was no correlation between the weight of the cobbles and their probability of landing success.

Regarding the dispersion area reached by the gorgonian transplants on the sea bottom, among all the models tested the one providing the best fit was a GLM with γ-distribution (AICc = 73.9; Figure 5

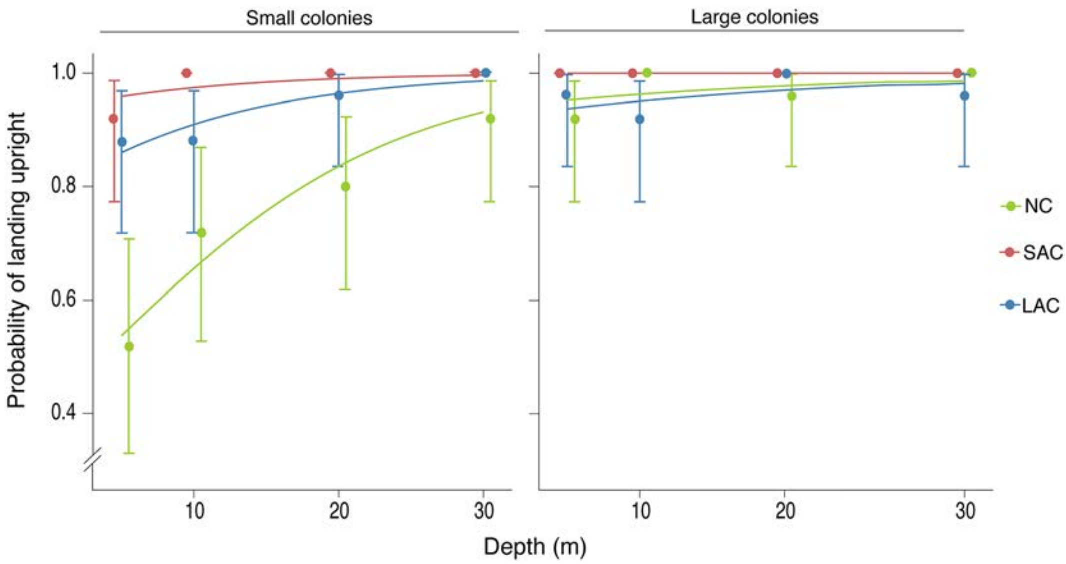

FIGURE 2 Probability of transplants landing in upright position according to depth, cobble type, and gorgonian size. Dots show real observations with a confidence interval of 95% and lines correspond to the predicted values of the GLM model for binomial data. NC = natural cobbles; SAC = small artificial cobbles; LAC = large artificial cobbles (n = 120)

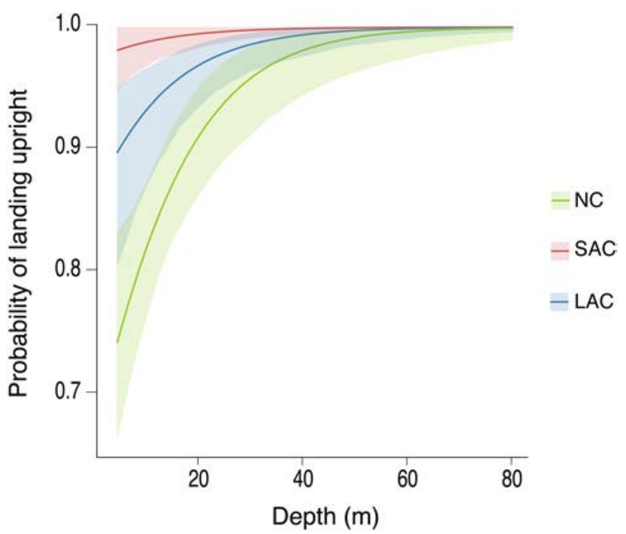

FIGURE 3 Predicting model of probability of transplants landing in upright position according to depth and cobble type, using a logistic general linear model for binomial data. NC = natural cobbles; SAC = small artificial cobbles; LAC = large artificial cobbles. Colour shadows correspond to 95% confidence intervals corresponding to the general linear model

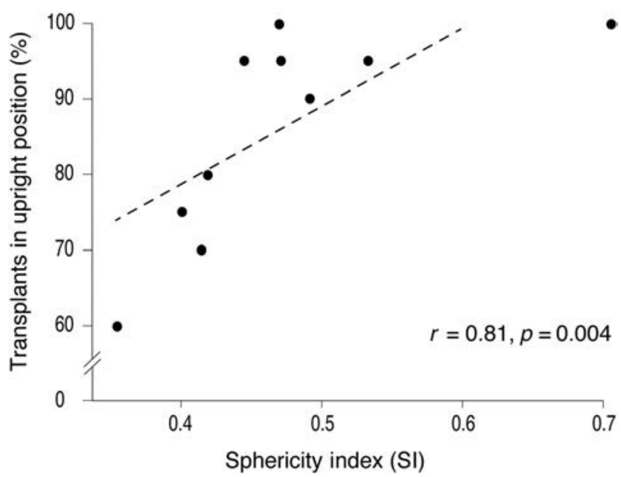

FIGURE 4 Scatter plot showing the correlation between percentage of upright landing and the sphericity index of natural cobbles (n = 10)

and Supplementary Materials 4 and 5). The model showed that depth and cobble type significantly influenced the dispersion area reached by transplants (ANOVA Type II; Depth: LR χ^2 = 111.657, df = 1, P < 0.001; Cobble type: LR χ^2 = 22.96, df = 2; P < 0.001), predicting an increase in the dispersal area with depth (Figure 5). Natural cobbles showed the largest dispersal area at 80 m depth (60.8 ± 20.6 m^2, mean ± SD), followed by small artificial (47.6 ± 16.3 m^2) and large artificial cobbles (34.3 ± 11.6 m^2).

3.2 | Monitoring of transplants

All the cobbles located at 30 m depth remained in an upright position during the first month. During the second and third months, all the natural cobbles (100%) and almost all the small artificial cobbles (90%) remained in an upright position, contrasting with the lower percentage (60%) of large artificial cobbles that maintained an upright position (Figure 6). The size of the gorgonian colony did not influence the overturning of the transplants since statistical differences were not detected in the position of the transplants at the end of the monitoring according to gorgonian size (W = 3.5, P = 0.814). The mean necrosis percentage of the gorgonian transplants at 30 m depth that remained in an upright position during all the monitoring

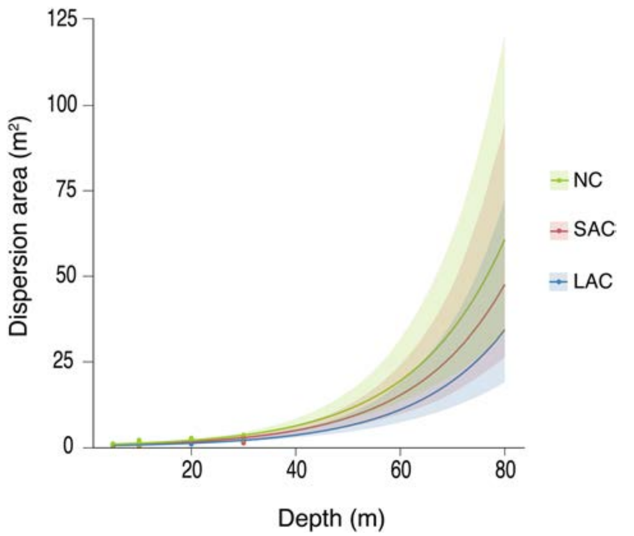

FIGURE 5 Model predictions of the dispersion area reached by the gorgonians transplants according to cobble type at different depths, using a general linear model with γ-distribution. Dots show real observations and lines with shadow correspond to the predictions of the model and its corresponding 95% confidence intervals. NC = natural cobbles; SAC = small artificial cobbles; LAC = large artificial cobbles (n = 120)

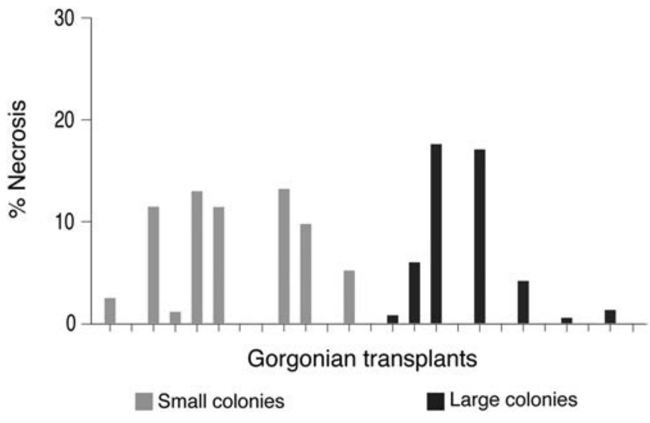

FIGURE 7 Percentage of necrosis in gorgonian transplants that maintained the upright position at 30 m depth during 3 months (12 small colonies and 13 large colonies)

period was very low and not statistically different between gorgonian sizes (W = 58, df =1, P = 0.281), being 3.65 ± 6.4% in large colonies and 5.63 ± 5.7% in small colonies. In the most affected colony, the necrotic tissue represented 17.6% of the total ramification length (Figure 7).

4 | DISCUSSION

The present results have shown that the 'badminton method' can be a cost-effective restoration method to potentially restore cold-water coral gardens over large scales.

The probability of successful landing of transplants increased with depth for the three cobble types tested, and by using gorgonian colonies of large size (Figure 2). Gorgonian colonies attached to cobbles act similar to a badminton shuttlecock, slowing down the fall and forcing an upright landing of the transplants due to the hydrodynamic resistance of the highly branched surface of the gorgonian. Consequently, by using large instead of small colonies, this effect is enhanced. Regarding transplant support types, the small artificial cobbles were the most effective at all depths, but all the three support types showed high probabilities of successful landing at 30 m depth, if large colonies were attached (>90% of upright landing; Figure 2). Accordingly, the model predicted a 100% landing success for all the cobble types returned to the continental shelf (80 m depth; Figure 3).

Once on the sea floor, remaining in an upward position is crucial for the survivorship of the gorgonian transplants. The size of the gorgonian colony did not influence the overturning of the transplants whereas the cobble type did. Natural and small artificial cobbles successfully maintained the upright position after 3 months at 30 m depth (100% and 90% maintained the upright position, respectively), where hydrodynamic conditions are even stronger than in deeper areas on the continental shelf (Garrabou, Ballesteros, & Zabala, 2002). In contrast, large artificial cobbles were less stable (60% maintained the upright position) and more easily overturned due to their semi-spherical shape. There were no differences in the amount of necrotic tissue between large and small transplanted colonies that remained upright over the 3 months, suggesting that all cobbles were sufficiently heavy to hold the transplanted colonies, without causing any movement that could result in abrasion of the tissue.

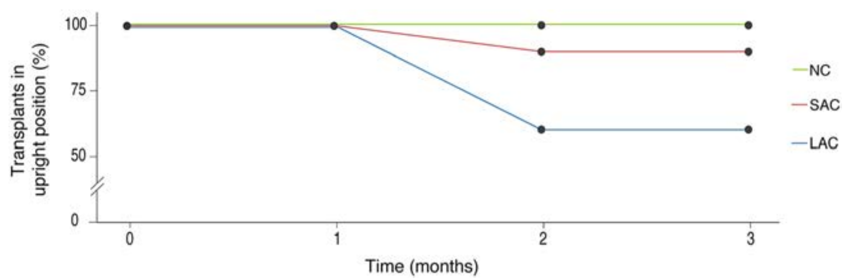

FIGURE 6 Proportion of cobbles that maintained the upright position at 30 m depth according to cobble type during 3 months. NC = natural cobbles; SAC = small artificial cobbles; LAC = large artificial cobbles (n = 30)

Based on these results about the landing efficiency and the capability of the transplants to maintain an upright position once on the sea floor, natural cobbles with large gorgonians attached are considered as the best option to ensure the success of this restoration method. Using large colony transplants will be further advantageous, since they have been shown to suffer less natural mortality after transplantation (Brooke et al., 2006), and have greater capacity to regenerate from injuries, being more resilient to further impacts (Fava, Bavestrello, Valisano, & Cerrano, 2010; Henry & Hart, 2005). Likewise, reintroduction of large gorgonians will foster the recovery of the three-dimensional structure of the coral gardens, and their habitat-forming functions, providing habitats for a large number of associated species (Geist & Hawkins, 2016; Horoszowski-Fridman, Brêthes, Rahmani, & Rinkevich, 2015). Indeed, the recovery of ecosystem functioning is by definition one of the main goals of ecological restoration (Society for Ecological Restoration International Science & Policy Working Group, 2004).

By using natural cobbles, no artificial material is introduced to the environment. This is in agreement with the recommendation of using natural substratum, as highlighted for shallow water gorgonian transplant experiments, in which the ineffectiveness of using man-made devices (i.e. PVC-clamps and PVC-racks) was identified (Weinberg, 1979). However, when selecting natural cobbles to be used for the restoration, flat cobbles should be avoided. Indeed, our results showed a strong positive relationship between the sphericity of the natural cobbles and their probability of landing upwards, with the flattest cobbles performing worse (i.e. a smaller percentage of landings in upright position) than the rounder ones (Figure 4). During the fall, if the transplants were falling upside-down (i.e. cobbles would land overturned), even large gorgonians did not provide enough hydrodynamic resistance to reverse the trend and overturn the flatter cobbles.

The variability in shape and weight of natural cobbles resulted in the largest predicted dispersion area (60.8 ± 20.6 m^2) of gorgonian transplants on the continental shelf (80 m depth; Figure 5). This could reduce the chance of two or more transplants landing on top of each other, enabling the restoration of large areas in the continental shelf. Restoring large areas has already been successful in tropical shallow environments, where coral reef restoration has been performed at scales from 10 m^2 to several hectares (Edwards & Gomez, 2007). One of the main goals of ecological restoration is to restore a natural range of ecosystem composition, structure and dynamics (Falk et al., 2006; Society for Ecological Restoration International Science & Policy Working Group, 2004). In this sense, and taking into account that 60 m^2 would be the mean area reached on the continental shelf by 10 gorgonian transplants dropped from the surface, we should consider performing multiple drops at the same site to achieve gorgonian densities similar to natural populations of *E. cavolini* dwelling on the Mediterranean continental shelf (on average 2.8 ± 4.3 colonies m^{-2}, with some high density patches above 25 colonies m^{-2}) (Dominguez-Carrió, 2018).

Concerning the survival of the transplants, the degree of necrosis observed after the 3 months of monitoring (<20% of necrosis in the most affected colonies and <6% on average, Figure 7) was similar to values recorded for other Mediterranean gorgonian species in natural conditions. For instance, the percentage of necrotic tissue among *Paramuricea clavata* colonies ranged from 0 to 22% with a mean of 10% (Coma et al., 2004; Linares, Coma, Garrabou, et al., 2008), and a maximum of 24% of necrotic tissue was reported among *Eunicella singularis* colonies, with a mean of 5% (Linares, Coma, Garrabou, et al., 2008). Water turbulence (influenced by strong wave action), high irradiance, and algal competition can compromise the survival of gorgonian transplants in shallow environments (Linares, Coma, & Zabala, 2008; Weinberg, 1979). During the monitoring period, these stressor drivers could have affected the transplants settled in much shallower water (30 m depth) than the natural depth range of the species in the study area (i.e. 60–120 m depth, Dominguez-Carrió, 2018; Gili et al., 2011). Even so, the amount of necrosis detected after the 3 months of monitoring was still low. The overall efficiency of the 'badminton method' is in line with the results obtained from other coral and gorgonian transplantation experiments. A review of 40 studies about hexacoral restoration actions reported a mean annual survival of 60% with a range of 6–98% (Montero-Serra et al., 2018). For Mediterranean gorgonian species, such as *E. singularis*, *Eunicella verrucosa*, *P. clavata*, and *Corallium rubrum*, the mean annual survival observed was 48% (ranging between 30 and 99%) (Fava et al., 2010; Linares, Coma, & Zabala, 2008; Montero-Serra et al., 2018). The limited information from cold-water coral restoration experiments showed that during the first year, a mean survivorship of the coral transplants of around 50–60% (Boch et al., 2019; Brooke et al., 2006).

Due to cold-water coral ecosystems' role as biodiversity hotspots (Baillon, Hamel, Wareham, & Mercier, 2012; Henry & Roberts, 2007), awareness about the need and the importance of their protection and restoration is increasing worldwide (Davies et al., 2007; WWF/IUCN, 2004). First attempts towards the assisted regeneration (active ecological restoration) of these ecosystems demonstrated the high survival of transplants for reef-forming cold-water corals (*Oculina variciosa* [Brooke et al., 2006] and *Lophelia pertusa* [Dahl, 2013; Jonsson et al., 2015]) and for cold-water gorgonians (*E. cavolini*) (Montseny et al., 2019). However, technical and logistical difficulties of working below the limit of scuba diving necessitate the use of underwater technology. Such technological requirements increase the costs of restoration actions by two or three orders of magnitude compared to shallow areas (Van Dover et al., 2014). Moreover, cold-water coral restoration actions performed to date have been based on transplantation of coral fragments on artificial structures, which significantly limit the area that can be restored (Brooke et al., 2006; Dahl, 2013; Montseny et al., 2019). Thus, in order to increase the scale of restoration actions, many artificial structures would be needed, further increasing the economic cost and the technical constraints. In this sense, and given that the main threats impacting natural habitats occur on large scales (Halpern et al., 2008), it is a foremost challenge to develop effective methods for upscaling ecological restoration actions (Aronson & Alexander, 2013; Perring, Erickson, & Brancalion, 2018). Indeed, a mismatch between the scale at which ecological restoration can currently be done and the scale at which major impacts act has been highlighted for tropical shallow coral reefs (Edwards & Gomez, 2007; Montoya Maya, Smit, Burt, & Frias-Torres, 2016; Pollock et al., 2017). Regarding this aspect, the 'badminton method' allows for restoration of a high number of gorgonians colonies over extended areas without the need for high-cost underwater technology. Moreover, by using bycatch gorgonians, no additional impact to healthy donor coral gardens will be generated, while a viable output for those gorgonians already fished is provided. Finally, directly involving professional fishers in restoration actions, will also increase the awareness of local society about the need for the protection of cold-water coral gardens and would facilitate the application of this methodology in an extensive manner, which is crucial for the restoration success (Gobster & Hull, 2000; Yap, 2000).

The current and future perspectives about the degradation of continental shelves (Halpern et al., 2008) and the urgency for conservation and restoration of their benthic communities (Borja, 2005) supports the development of new methodologies like that presented in this work. The present study shows that the 'badminton method' can be a reliable cost-effective and large-scale restoration method to assist the recovery of cold-water coral gardens located beyond the limit of scuba depth. It should be highlighted that apart from being a valid technique for continental shelves and potentially for deep environments, it could also be applied to shallower mesophotic coral ecosystems. While the 'badminton method' in this study has been demonstrated to be successful for sandy or gravel horizontal bottoms, further studies should explore its applicability to other bottom types, like those in upper-slope habitats or cold-water coral reefs habitats.

The present study is a first evaluation of the 'badminton method' that needs now to be followed by a real application in areas of the continental shelf in order to corroborate the long-term survival success predicted by the present results. Furthermore, the limited sample size, arising from the difficulties of working at depths (Clauss & Hoog, 2002), results in model predictions with large margin errors constraining broader conclusions of this study. These limitations will be assessed and reduced based on future monitoring of gorgonians transplanted on the continental shelf; the success of ecological restoration, however, should only be generally demonstrable within 10–50 years (Jackson, Lopoukhine, & Hillyard, 1995; Suding, 2011). A general reduction of fishing impacts, together with effective protection mechanisms for cold-water coral gardens are greatly preferable to avoid the need for potentially expensive and long-term restoration projects (Possingham et al., 2015).

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SUPPORTING INFORMATION

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

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ANNEX

El método bádminton

María Montseny, Andrea Gori y Josep-Maria Gili

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El método bádminton

Inspirado en la pesca artesanal, este método sencillo e innovador facilita la restauración de los fondos marinos continentales

En las últimas décadas, los fondos marinos de la plataforma continental se han degradado y empobrecido debido al impacto de la pesca y otras actividades humanas. Una de las consecuencias más destacadas de esta alteración ha sido la destrucción de los hábitats bentónicos. En estos, numerosas especies sésiles (gorgonias, corales y esponjas, entre otras) configuran estructuras tridimensionales que son esenciales para que entre sus colonias vivan muchas otras especies, algunas de ellas de interés comercial. Estas especies bentónicas «constructoras» constituyen el refugio temporal de estadios larvarios y juveniles de peces y crustáceos, que encuentran entre sus colonias alimento y protección. La desaparición de estos organismos supone una pérdida de biodiversidad y de su función ecológica, esencial para la supervivencia de las especies asociadas. Asimismo, la fragilidad, longevidad, crecimiento lento y reclutamiento reducido, característicos de corales y gorgonias, dificultan su recuperación.

La restauración de los hábitats bentónicos es, pues, urgente. Las estrategias de restauración pasiva (conservación y protección) ya no bastan para contrarrestar los efectos de la actividad humana. Necesitamos una restauración activa. Desde hace unos años se ha empezado a aplicar el trasplante de colonias de animales sésiles en zonas litorales accesibles con técnicas de escafandra autónoma, sobretodo en ambientes tropicales [*véase* «¿Es posible salvar los corales?», por Rebecca Albright; Investigación y Ciencia, marzo de 2019]. Pero en las zonas más profundas, a 60 metros y más, el trasplante es difícil, a la vez que muy costoso, debido a que solo puede accederse a ellas mediante robots submarinos.

A partir de la observación de los pescadores artesanales de la isla de Menorca, hemos desarrollado un método de restauración activa para los fondos de gorgonias de la plataforma continental (en el marco de los proyectos de investigación MITICAP y RESCAP del Programa PLEAMAR de la Unión Europea). Los pescadores de Menorca limpian in situ sus redes de pesca y devuelven al mar las colonias de gorgonias que han pescado accidentalmente. Adheridas a una base de rodolitos (algas

calcáreas), caen y se depositan, de pie, en el fondo. Inspirándonos en esta práctica, hemos desarrollado un método similar: las gorgonias atrapadas por los pescadores **1** se extraen de las redes y se trasladan a unos acuarios en las cofradías para que se recuperen **2**; luego se adhieren a un sustrato rocoso que las mantiene erectas **3** (no pueden sobrevivir tumbadas); por fin, los trasplantes de gorgonias se devuelven a su hábitat natural, la plataforma continental, lanzándolos al mar directamente desde las embarcaciones **4**.

Pese a su sencillez, esta técnica ha demostrado ser muy eficaz: más del 80 por ciento de colonias sobreviven un año después de ser trasplantadas y devueltas al fondo marino. Ello permitirá abarcar un gran número de gorgonias, lo que facilitará el restablecimiento de los bosques de gorgonias, acelerando la recuperación global de los fondos. Si, además, implicamos a los pescadores locales, el proceso es todavía más eficiente.

—María Montseny, Andrea Gori y Josep-Maria Gili, Instituto de Ciencias del Mar de Barcelona, del CSIC, en colaboración con la Fundación Biodiversidad y el Parque Natural del cabo de Creus

IMÁGENES DEL ICM-CSIC


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