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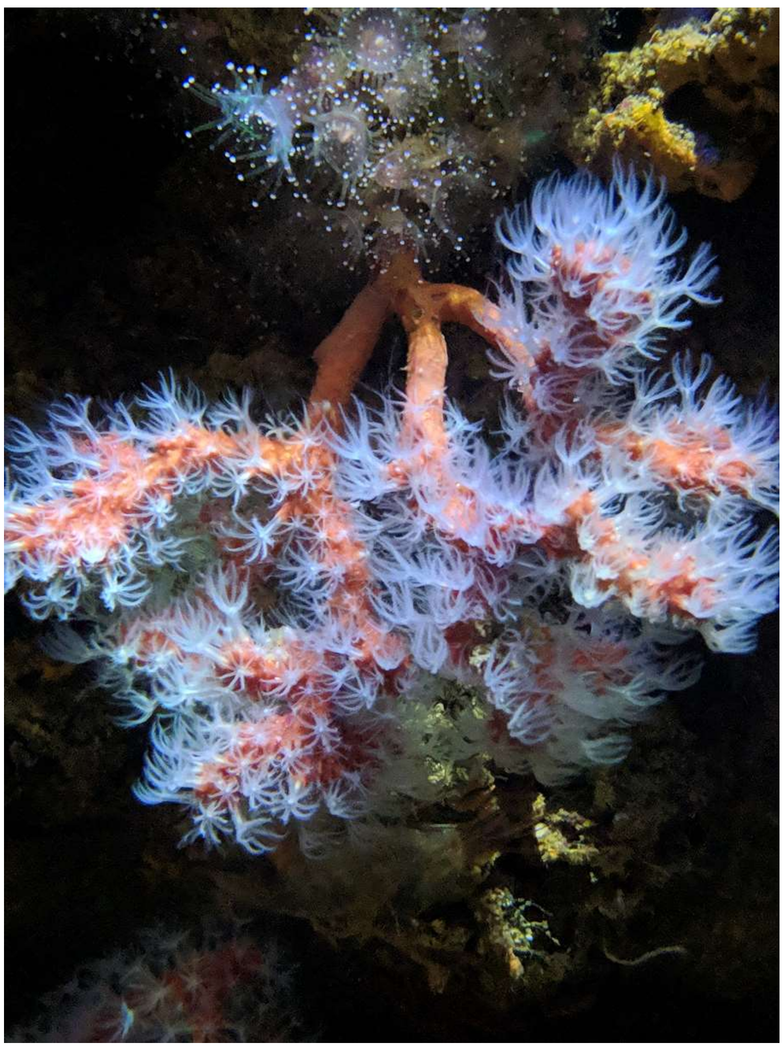
GLOBAL REPORT ON THE BIOLOGY, FISHERY AND TRADE OF PRECIOUS CORALS



Cover photograph: *Corallium rubrum* (Kim Friedman)

GLOBAL REPORT ON THE BIOLOGY, FISHERY AND TRADE OF PRECIOUS CORALS

by
Rita Cannas
Associate Professor of Zoology
Università degli studi di Cagliari
Cagliari, Italy

Maria Cristina Follesa
Associate Professor of Zoology
Università degli studi di Cagliari
Cagliari, Italy

Alessandro Cau
Junior Researcher of Ecology
Università degli studi di Cagliari
Cagliari, Italy

Angelo Cau
Emeritus Professor of Ecology
Università degli studi di Cagliari
Cagliari, Italy

and

Kim Friedman
Senior Fishery Resources Officer
Marine and Inland Fisheries Branch
FAO, Rome, Italy

FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
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PREPARATION OF THIS DOCUMENT

This document presents an overview of information on the global understanding of the biology, fishery and trade of precious corals. The document includes a global overview of coral fisheries and trade compiled from a series of inputs, in particular the integration of two regional studies: i) Mediterranean regional report on the biology, fishery and trade of precious corals (GFCM Secretariat, April 2018); and ii) and Asian regional report on the biology, fishery and trade of precious and semi-precious corals (Nozomu Iwasaki, April 2019). Other relevant information was also considered, e.g. from peer-reviewed papers, grey literature and data derived from CITES documents AC29 Doc.24, AC29 Doc 22 and its annexes, plus inputs from voluntary and professional reviews. The University of Cagliari, Italy, was responsible for the integration and discussion of information contained in the Mediterranean and Asian regional studies that were also prepared as part of this initiative. The request for the study came from CITES Parties and was kindly funded by the government of the United States of America. Work by FAO comes under the CITES–FAO MoU signed in 2006, and was carried out in close collaboration with CITES Parties and the CITES Secretariat.

ABSTRACT

This document has been prepared by the Food and Agriculture Organization of the United Nations (FAO), in accordance with a request from CITES (CoP Decision 17.191 on Precious corals, for consideration at the 30th meeting of the Animals Committee). The report concerns precious (red, pink, white and black) coral species within the hexacoral order Antipatharia, and the octocoral family Coralliidae. According to the requirements of CITES Decision 17.191, the study considers all available data and information on the biology, population status, use and trade in each species, including the identification of gaps in such data and information. It contains information on the management and harvest regulation schemes for these coral species, with the aim of considering the effectiveness of their management and conservation. The report intends to inform the CITES parties of the status of the management and trade of precious corals, in order to provide guidance on the actions needed to enhance the conservation and sustainable use of precious corals.

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Terminology for corals

The term "precious coral" generally refers to any coral species used for jewellery and other high-value objects, including coral species used to imitate the more valuable species (Cooper *et al.*, 2011). Actually, a distinction can be made between "precious" vs. "semi-precious" corals (Cooper *et al.,* 2011) as follows:

- **precious corals** are those that have a hard, solid skeleton that can be readily polished;
- **semi-precious corals** are those that have a porous skeleton that does not achieve a high polish without the use of resin or other filler.

All coral species of the hexacoral order Antipatharia and the octocoral family Coralliidae included in the present report are considered 'precious corals'.

Black corals: for the purpose of this report they correspond to the taxa included in the hexacoral order Antipatharia.

Red, pink and white coral: for the purposes of this report they correspond to the taxa of commercial interest belonging to the octocoral family Coralliidae.

Abbreviations and acronyms

ACL	annual catch limit
AM	accountability measures
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
CoP	conference of parties
CPC	Contracting Parties and Cooperating non-Contracting Parties of GFCM
DGR	diametric growth rate
EEZ	exclusive economic zone
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
FMP	Fisheries Management Plan (for precious corals)
FTIR	fourier-transform infrared spectroscopy (FTIR)
GFCM	General Fisheries Commission for the Mediterranean
GPS	global positioning system
GSD	genetic sex-determination
IUCN	International Union for Conservation of Nature
IUU	illegal, unreported and unregulated (fishing).
LGR	linear growth rate
MCS	monitoring, control and surveillance
MEA	multilateral environmental agreement
MPA	marine protected area
MSY	maximum sustainable yield

NDF	CITES non-detriment finding
NGS	next generation sequencing
NMFS	National Marine Fisheries Service (United States of America)
NPFC	North Pacific Fisheries Commission
NOAA	National Oceanic and Atmospheric Administration (United States of America)
NWHI	Northwest Hawaiian Islands
OY	optimum yield
PEI	population-by-environment interaction
PLD	pelagic larval duration
RAS	Regione Autonoma della Sardegna (Italy)
RFMO	regional fisheries management organization
ROV	remotely operated vehicle
SAC	GFCM Scientific Advisory Committee on Fisheries
SAP BIO	Strategic Action Programme for the Conservation of Biological Diversity in the Mediterranean
SCUBA	Self-Contained Underwater Breathing Apparatus
SDGs	Sustainable Development Goals
SNP	single nucleotide polymorphisms
SPA/BD	(Protocol concerning) the Specially Protected Areas and Biological Diversity in the Mediterranean of the Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean (Barcelona Convention)
VME	vulnerable marine ecosystem
VMS	vessel monitoring system
WGSAD	Working Group on Stock Assessment of Demersal Species (GFCM)
WPFMC	Western Pacific Regional Fishery Management Council (WPCouncil)
XRF	X-ray fluorescence spectrometry

SUMMARY

Precious corals in the study

This study covers precious corals of the order Antipatharia, otherwise known as black corals, and the family Coralliidae, otherwise known as red, pink and white corals.

The order Antipatharia includes about 265 currently accepted species (in 44 genera and seven families), commonly known as black corals. At least 13 species in 11 genera are used in jewellery; however, except for those produced in Hawaii, the names of most species are unknown.

The family Coralliidae comprises 43 valid species. Apart from *Corallium rubrum*, another nine species are known to be harvested. This includes three species from the Hawaiian Archipelago that have not been collected commercially since 2001, although they have been fished in the past. In addition, several undescribed species are still reported to exist within the Coralliidae. In particular, recent molecular data confirm the possible occurrence of cryptic species and species-complexes even among recognized cosmopolitan species. For instance, according to a recent genetic study the boundaries between three commercial species (*Pleurocorallium carusrubrum,*[1] *Pleurocorallium elatius* and *Pleurocorallium konojoi*) were deemed to be ambiguous, and these species were placed together and indicated as the '*P. elatius* species-complex'.

Distribution of precious corals

Black corals have a wide geographic distribution that ranges from tropical to polar regions. However, most of the currently described species are found in tropical and subtropical waters. Black corals are found across a wide depth gradient (from 2 to 8 900 m of depth). Despite this wide bathymetric range, over 75 percent of described antipatharian species are restricted to depths below 50 m.

Species in the family Coralliidae inhabit tropical, subtropical and temperate oceans. Hotspots of coralliid species are located in the west and central Pacific, including the surrounding seas of New Caledonia, Taiwan Province of China, Japan and the Hawaiian Archipelago. Species diversity appears lower in the Atlantic, Indian and eastern Pacific oceans. Nevertheless, historically only two areas had large populations and are commercially exploited: the Mediterranean Sea and the adjacent Atlantic, together with the northern Pacific Ocean.

In the Mediterranean the only species is *C. rubrum*, dwelling at depths ranging from 5 to over 1 000 m. However, it is more common in the 30–200 m depth range. *C. rubrum* inhabits subtidal rocky substrates and is one of the most important components of Mediterranean "coralligenous" animal-dominated assemblages. Within the Pacific context, commercially valuable species of the family Coralliidae are distributed within two depth zones: 50–400 m and 1 000–1 500 m in the waters around the Hawaiian Archipelago and Emperor Seamounts, Japan, the Philippines and Taiwan Province of China.

Biology of precious corals: reproduction, growth, mortality and connectivity

With the exception of a few studies on shallow-water species (< 50 m), very little information is available on reproduction of antipatharians. In general, individual polyps are strictly gonochoric, and colonies are either female or male, with the exception of one species (*Stichopathes saccula*) which has mixed colonies with both male and female polyps. Fertilization and larval development most likely occur externally in the water column. Very little is known about the larval biology and reproductive seasonality of antipatharians. To date, larvae

[1] Recently discovered *P. carusrubrum* has been reported to be circulating on the market as pink coral (Jeng, 2015). Therefore *P. elatius* commercialized in Taiwan Province of China may contain *P. carusrubrum* (see following sections and Annex 2).

have only been observed for members of a single shallow-water species from New Zealand (*Antipathella fiordensis*) in laboratory cultures.

Only few studies have examined the reproductive seasonality of black corals, and all of these have been conducted in shallow waters (< 70 m). All of these studies report the seasonal appearance and disappearance of gametes, which has been correlated to seasonal temperature fluctuations in some cases, with peak maturities occurring when temperatures are warmest. It is currently not known whether the reproductive cycle of deepwater black corals is seasonal and this aspect should be examined by future studies.

Various methodologies have been used to estimate the growth rates and longevities of several species of black coral occurring over a wide depth range. These studies indicate that growth rates vary greatly across different species and environments, with the fastest growth observed in shallow water species, and slowest growth in deepwater species. The fastest growing antipatharians are shallow-water, tropical wire corals, with vertical growth rates ranging between 3 and 7 cm/year for *Stichopathes* spp. from Puerto Rico, up to 159 cm/year for *Stichopathes* cf. *maldivensis* from Indonesia. At the other end of the spectrum, the slowest growing antipatharians belong to the genus *Leiopathes*, with radial growth rates ranging from 0.005 to 0.022 mm/year.

For red corals, *C. rubrum* is the most studied; it is a gonochoric species that undergoes internal fertilization and broods larvae internally (planulator). Gonadal development follows an annual cycle with a synchronized release in summer. Larvae remain in the water column for a period ranging from a few hours to days, before settling from the plankton near parent colonies. The actual age of first reproduction has been estimated to be 6 years for males and 10 years for females. Very little is known about the basic life history of most species of Pacific Coralliidae species, however those that have been studied appear to be gonochoric broadcast spawners, unlike the brooding *C. rubrum*. Data concerning the age at which corals attain sexual maturity and the relationship between the size of a colony and its level of maturity are currently lacking for most species in the family. Estimates for the age at maturity for the three investigated Pacific species range from 10 to 80 years.

Longevity studies are also not well represented in the literature, although *C. rubrum* is a slow-growing (0.21–0.35 mm/year in basal diameter in shallow populations), long-life species. Even slower growth rates are recorded in deeper populations. Known growth rates for Pacific Coralliidae are similar. The natural mortality of *C. rubrum* occurs due to competition over space with the sessile biota. Phenomena of mass mortality events have been observed in shallow-water populations since the late 1990s, linked to elevated temperature anomalies; in some cases these have also been associated with fungal and protozoan diseases.

A few genetic studies on black corals indicate significant genetic variation between sites, suggesting that larval dispersal is restricted even at distances of 10–15 km. The dispersal of black coral larvae is highly philopatric and most larvae appear to settle very close to parent colonies. However, even within the same species there appear to be mixed strategies, with most larvae settling close to their parents, and few larvae dispersing over long geographic distances.

In the case of *C. rubrum,* several studies have confirmed the occurrence of genetic differentiation at spatial scales of tens of metres and across depth. The strong genetic differentiation between nearby samples implies that the recovery of overexploited populations should mainly be the result of self-recruitment, and this has implications for the need for the localised management of red corals. Genetic population studies performed on two Pacific species to date (*Hemicorallium laauense* and *Pleurocorallium secundum*) confirm that recruitment was from local sources with only occasional long-distance dispersal events.

<u>Precious coral fisheries</u>

Because of their beauty, durability and high economic value, precious corals have been exploited since ancient times. The harvesting of black coral is known to occur/have occurred in several areas: in the Indo-Pacific (especially the Philippines), the Caribbean (notably Cayman Islands and the Dominican Republic), Latin America, the Red Sea, and very sporadically in the Mediterranean Sea. In addition to targeted commercial harvesting, black corals are also inadvertently caught in bottom trawls. Southeast Asia and the South Pacific islands continue to be an important source of black coral for international markets – however, very limited information is available on the fishery or the amount harvested. The only black coral fishery in the United States of America, fished by a limited number of divers that collect black corals in shallow waters of the Hawaiian Islands, has been profitable and continuous since its inception in 1958. Three black coral species have been harvested commercially in the Hawaiian Islands (*Antipathes grandis, Antipathes griggi,* and to a lesser extent *Myriopathes* cf. *ulex*). In Mexico, black coral is an important resource for jewellery and handicrafts, providing economic support for authorized fishermen, craftsmen and merchants. Black coral harvests are reported to occur without any control or any management in other regions and countries (e.g. Madagascar), where illegal trade is expanding.

The fishery for Coralliidae, harvests about ten species of red, pink and white coral. Currently, the harvest of *H. laauense* or probably *H. regale,* and *P. secundum* from Hawaii has been on hiatus since 2001, while the collection of corals around the Emperor seamounts (*P. secundum* and *C.* sp. nov.) was suspended at an earlier date. The remaining six species are harvested in the Mediterranean and Atlantic Ocean (*C. rubrum*), and in remaining areas of the Pacific Ocean (*P. konojoi, P. elatius, P. carusrubrum,* [2] *Hemicorallium sulcatum,* and *Corallium japonicum*).

Countries primarily involved in harvesting Coralliidae are in the Pacific region: Japan, Taiwan Province of China, and the United States of America. The methods of harvesting currently used are coral nets (Japan and Taiwan Province of China) and remotely operated vehicles (ROVs) or submersibles (Japan and the United States of America). Experiments carried out to investigate the impact of these dragging nets on the sea floor underlined their negative impact on coral populations and their habitats. The fishery in the Mediterranean involves vessels from five Mediterranean countries, where it is regulated by national law (Croatia, France, Italy, Spain and Tunisia) while it is temporally closed in the Mediterranean waters of three countries (Algeria, Greece and Morocco). Today in the Mediterranean region, SCUBA (Self-Contained Underwater Breathing Apparatus) divers use manual picks as the only legal method for harvesting red coral, with dredging banned throughout the Mediterranean Sea since the mid-1980s. As robotic equipment has become more technically advanced and accessible this method offers very targeted harvesting; however, due to sound reservations as to its sustainability, the use of this extraction method is not permitted for harvesting Mediterranean red coral in the General Fisheries Commission for the Mediterranean (GFCM) competence area, unless for scientific purposes within permitted scientific projects.

<u>Use and trade</u>

The skeleton, or fragments of the skeleton of larger black coral species has been used for jewellery and religious articles from at least the time of the ancient Greeks. Black corals have long been used for a variety of purposes ranging from jewellery to their presumed ability to fend off evil and health-related ailments. With regard to the order Antipatharia, according to the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) trade statistics black coral is frequently traded as Antipatharia spp.. This is because in most cases the identification of genus or species is possible only if an entire colony is available, and even when this is the case, taxonomic understanding at the species level is difficult and requires microscopic examination. Therefore, processed black coral products often cannot be identified beyond the order. CITES guidance (2017) suggests that trade in raw black coral should be reported at the species level.

Species of red, pink and white corals, Coralliidae are traded as whole, dried colonies, unworked branches and branch fragments, as well as beads and manufactured jewellery. A few species within the family Coralliidae are/were most likely encountered in international trade: four non-CITES species (*C. rubrum, H. regale (H. laauense*), *Corallium* sp. nov. and *H. sulcatum*) and four CITES-listed Appendix III species (*P. elatius, P. secundum, P. konojoi,* and *C. japonicum*). Records of products in trade for the four CITES species is

[2] See Footnote 1.

available since their listing in Appendix III of CITES (2008), while data on trades of the three non-CITES species is more difficult to obtain. In the CITES trade database, most of the records in recent years (2008–2016) refer to *P. elatius*, then in decreasing order to *P. secundum*, *P. konojoi*, and *C. japonicum*. It is worth highlighting that a new species, recently discovered by Tu *et al.* (2012) has been reported to be circulating on the market as pink coral. Therefore *P. elatius* commercialized in Taiwan Province of China may contain *P. carusrubrum*. In the Mediterranean and Atlantic, C. *rubrum* has been a precious commodity since prehistoric times. Nowadays, the skeletons of precious corals still provide the raw materials used in jewellery, sculptures, figurines, amulets, clothing, ornamentation, and for a variety of art objects. In general, raw and whole dried specimens of Coralliidae can be identified to the species level using an electron microscope. Where worked specimens are concerned, it may only be possible to identify them to the family Coralliidae level.

Apart from the data on the CITES database, other sources of trade data for Coralliidae harvested species are not available in the public domain. In Japan, for instance, data on trade at the species level are not available. The Japanese trade statistics aggregate all coral species together, and it is difficult to distinguish processed and manufactured items because they are reported under the same code with semi-precious corals, bone, tortoise-shell, horn, antlers, mother-of-pearl and other animal carving materials. On a spatial basis, similar difficulties exist. Official and comprehensive data on the trade of *C. rubrum* are not available for any Mediterranean country.

In general, for most precious corals, data on trade are not species-specific (worked products often cannot be identified beyond the order or the family), and sales can amalgamate reef-building and precious corals in the amounts reported. Unlike fresh fish, precious corals can be preserved after they are landed, which means sellers can hold on to corals when market prices are low; this means there are often differences in the timing of when corals are landed and when they are traded, and the amounts held by traders or in trade can include both live and fossilized or dead corals, thus complicating the identification of precious corals coming from living resources in trade further.

<u>Population status of precious corals</u>

Population densities are highly variable among antipatharian corals, but at least some black coral species can reach particularly high densities, to the point of becoming ecologically dominant. Such aggregations are considered as marine forests that are composed of animals rather than plants, enhancing the complexity of benthic habitats.

Black coral communities, both in shallow and deep waters, are under increasing threat from a range of direct and indirect anthropogenic impacts. In particular, black corals are considered vulnerable because they display life-history characteristics often associated with a susceptibility to local extirpation (e.g. longevity and slow growth) but also because of the threats to their survival associated with fishing, pollution (including nutrient pollution), climate change and ocean acidification. The few studies that exist for the most important commercial black coral species, which investigate shifts in the status of black corals impacted by fisheries, reveal a shift to a younger age frequency in fished populations. In a recent assessment of the conservation status of Mediterranean anthozoa, five species of Antipatharia were included in the Mediterranean International Union for Conservation of Nature (IUCN) Red List as Data Deficient, Near Threatened or even Endangered. On the contrary, black corals have still not been classified as endangered on the Red Lists of Taiwan Province of China, the Republic of Korea, the Philippines, India and Sri Lanka.

Similarly, the harvesting of red corals is just one of the pressures in a range of anthropogenic disturbances that threaten *C. rubrum* populations, including water temperature anomalies, pollution, tourism, recreational diving, incidental take and habitat degradation. The population structure of *C. rubrum* differs significantly between shallow and deepwater locations. Possibly owing to intense harvesting, shallow-water populations

(< 50 m) are now also dominated by small colonies at a high density. In deepwater locations, the percentage of large colonies is greater and the densities lower, but it varies between studies and locations.

Reliable data on the fishing of precious corals is incomplete, and it is difficult to identify historical and current coral capture production, as in most areas the data collected by national authorities were not systematically recorded, while in some areas they were not disclosed. In some cases such as the Mediterranean, a systematic collection of capture production data has begun only recently, after the implementation of new management systems. Historical FAO statistics of precious coral capture production has related shortcomings. In addition, mixed live and dead coral weights are often recorded in landings, thus complicating the identification of the total impact of harvesting on living resources.

The incomplete datasets on the fishing of *C. rubrum* are a case in point. The shortcomings of the historical data available in the FAO database have been recognized by GFCM (Annex 1). This is exacerbated by the understanding that recovery from past depletion can require extended protection periods. For instance, after 20–30 years of protection within French and Spanish marine protected areas (MPAs), red coral colony sizes had not returned to values close to those of pristine populations, suggesting that the full recovery of precious coral beds requires decades of effective protection. This knowledge has further encouraged efforts to collect more systematic recordings of official data from national sources. Trend analysis of recent GFCM data per country (both yield/year and yield/dive) indicate that total landings of red coral have been decreasing in some countries, while remaining stable in all others. However, information on the species is generally not sufficient to provide an overall assessment on the status of red coral populations. An improvement of the quality of data is expected as a result of the entry into force of Recommendation GFCM/41/2017/5, with the establishment of a Regional Management Plan for red coral.

Less information is available on Coralliidae from the Pacific area. In Japan the size differences between non-harvested and harvested populations suggest that harvested populations reach harvestable size after 10–20 years. No data on abundance, density or size structure are available for Coralliidae in the waters of Taiwan Province of China.

Considering local depletions of red corals observed, and an estimated decline in shallow populations, *C. rubrum* has been listed as 'Endangered' by IUCN specialists in the recent assessment of Mediterranean anthozoa. However, doubts have been raised concerning the data used for this assessment, especially the reference to the decline in yields in recent years, because precise figures are not available on this aspect and the FAO data (which they possibly relied on) have recognized shortcomings. For Pacific Coralliidae, the Japanese Ministry of the Environment included red, pink and white corals on their Red List of Threatened Species under the classification "Near Threatened" in 2017. While the risk of extinction of these species is not considered significant at present, this classification was made on the basis of increased pressure on fisheries as a result of dwindling fishing resources, greater numbers of permitted fishing vessels, and poaching by foreign boats.

Management of precious corals

Information on the management of black coral species worldwide is not well documented or easily accessed, with the most detailed management found in the Hawaiian Islands, United States of America. This fishery occurs in waters that fall within the jurisdiction of both the State of Hawaii, as well as American federal waters. The fishery is therefore managed through both state and federal regulations, which are set by the Hawaii Department of Land and Natural Resources, and National Oceanic and Atmospheric Administration (NOAA) National Marine Fisheries Service (NMFS) in consultation with the Western Pacific Regional Fishery Management Council (WPFMC). Fishing regulations include both minimum colony size limits, as well as biannual quotas. Additionally, even though the fishery targets at least three species, it has historically been managed as a single stock, in large part owing to difficulties in differentiating the targeted species *in situ*.

Information listing the progression of implementation of management controls for black coral fisheries are also available from Madagascar.

Coralliidae fishing is regulated in several Mediterranean countries, in the United States of America, Japan and Taiwan Province of China. Japan and Taiwan Province of China have management measures in place, and regulations have been getting stronger in recent years. The 2009 "Regulations Governing Fishing Vessels that Also Engage in Coral Harvesting" established in Taiwan Province of China have tightened controls and restricted the number of vessels fishing, fishing zones, catch quotas and designated landing ports, and have implemented the requirements of the use of vessel monitoring systems (VMS). These regulations require fishers to keep a logbook of fishing operations and to have an observer on board the boat, as well as to centralize auction markets. Fishing in Japan has also been curtailed recently.

WPFMC's Precious Corals Fisheries Management Plan (FMP) has regulated the harvest of *P. secundum* since 1983 within the 3–200 miles exclusive economic zone (EEZ) of the United States of America. The FMP imposes permit requirements valid for specific locations, harvest quotas for precious coral beds (only live corals are included in the quota), a minimum size for pink coral, gear restrictions, area restrictions and fishing seasons. The harvesting of precious corals using non-selective gear is banned. The plan also recognizes differing categories of coral beds: established beds, conditional beds, exploratory areas and refugia.[3] Yield estimates (maximum sustainable yield, MSY) were estimated for *P. secundum* for a single bank for which biological data were available. Data on Coralliidae coral beds show stability of fished stocks.

Management of *C. rubrum* fisheries includes a variety of national measures within territorial waters (See Annex 1). Limits on licenses, temporal closures, minimum legal size for landings, allowed depth and areas are commonly imposed. With regards to monitoring, control and surveillance (MCS), apart from the logbook, other MCS measures are not fully in place in many countries. Regional management cooperation is a reality for the Mediterranean stocks of precious corals. In 2017 red coral became part of the GFCM 'mid-term strategy (2017–2020) towards the sustainability of Mediterranean and Black Sea fisheries,' and the adoption in 2017 of Recommendation GFCM/41/2017/5 (on the establishment of a regional adaptive management plan for the exploitation of red coral in the Mediterranean Sea), which formally entered into force on 1 April 2018.

Protection of precious corals

All international trade in black coral species are regulated on a global scale by CITES, an intergovernmental treaty that controls the international trade of animals and plants under its Appendices. Since 1981, all species of black coral belonging to the order Antipatharia have been listed in CITES Appendix II, which contains species that are vulnerable to becoming overexploited but that are not necessarily yet at risk of extinction. Additionally, in the Mediterranean Sea five antipatharian species are protected by international conventions – in Annex II of the SPA/BD Protocol (Protocol concerning the Specially Protected Areas and Biological Diversity in the Mediterranean) of the Barcelona Convention (list of endangered or threatened species), and Appendix III of the Bern Convention (list of the protected fauna species). Protection is also granted by national laws in several countries. In India and Indonesia measures for the protection and conservation of antipatharians have recently been introduced, while countrywide laws operate in Madagascar.

The Mediterranean red coral, *C. rubrum* is included in several international legal instruments aimed at the conservation and protection of species and their habitats: Annex III of the SPA/BD Protocol of the Barcelona Convention – i.e. the list of species whose exploitation is regulated; Appendix III of the Bern Convention (list

[3] Established beds are ones for which appraisals of MSY are reasonably precise. Conditional beds are ones for which estimates of MSY have been calculated by comparing the size of the beds to that of Makapuu Bed and then multiplying the ratio by the yield from the Makapuu Bed. Refugia beds are areas set aside for baseline studies and possible reproductive reserves. Exploratory areas are the unexplored portions of the EEZ.

of protected fauna species); Annex V of the European Union Habitats Directive (List of the "Animal and plant species of Community interest whose taking in the wild and exploitation may be subject to management measures"). Protection for *C. rubrum* is also granted by national laws in several countries in the Mediterranean region, establishing spatial controls (MPAs) for the protection of the species in some areas or even totally forbidding harvesting. In the Pacific region, China has classified precious corals as a Class 1 Protected Species under the Wildlife Protection Law, so fishing is prohibited, while in the Philippines, according to Section 91 of The Philippine Fisheries Code of 1998 (Republic Act No. 8550), the collection, possession, sale and exportation of precious corals are all prohibited, except for research purposes.

Recommendations for the management and conservation of the fishery and trade of precious corals

The information collated on precious corals in this study highlighted a number of opportunities for enhancing management and conservation of these vulnerable natural resources. Below are five summaries for topics for further consideration to improve the status of these natural renewable resources.

1) An improved understanding of taxonomy and life history is needed

Considering the lack of knowledge on many (most of) precious coral species of commercial importance, priority should be given to enabling their correct identification through systematic studies, and biological investigations conducted on the life history of commercially exploited species, in order to gather useful information for the proper identification and management of harvested species. In the Mediterranean context, the need to carry out scientific research studies on red coral has been raised on several occasions, with a particular focus on the need for data on the biomass, recruitment, and mortality rate of commonly fished species – necessary to construct a population dynamics models to estimate projected resource allocations. Today data are limited by a paucity of landings information, while the growth rate and recovery studies have mainly been conducted on a few 'shallow' water populations. A Mediterranean research programme on red coral which is part of the GFCM regional management plan for red coral (Recommendation GFCM/41/2017/5) has been launched with a programme of work currently being formulated by scientific institutions that are part of the GFCM Scientific Advisory Committee on Fisheries (SAC).[4] The programme prioritizes the collection of useful data for the provision of advice to support management. The combination of fishery-dependent (e.g. analysis of catch) and fishery-independent sources of information (e.g. surveys on a multi-annual basis) will ensure a regular monitoring of the resource.

Information on scientific research studies on precious coral resources in the Pacific are not well represented in literature. In particular, the data on the Asian species – including biomass, recruitment, and mortality rate – necessary to construct a population dynamics model to estimate future resources and landings is almost non-existent, with only growth rates defined. Therefore, such studies should be prioritized in the future.

2) Fishery-dependent data are urgently needed, and fishing methods must be improved

Apart from some countries, data on fishing effort and the amounts of precious corals harvested are either generally inaccurate, amalgamated across a range of species or life-form groupings, or lacking entirely. The priority should be data collection on fisheries worldwide, with detailed information per species/area. GFCM started to collect such data in 2013, moving from a dataset of sporadic submissions of information that included trade data (raw corals and commodities together) to data on coral capture. In some cases, compulsory data, such as the overall quantities of red coral caught per year, or the percentage of undersized colonies has not yet been made available to GFCM. In some Asian countries data are collected, but are also not disclosed or available for general assessment. When such data are not made public, they should be made available to government experts for formal stock assessments. In addition, further studies

[4] Details are available in Appendix 10 of the Report of the twentieth session of the GFCM Scientific Advisory Committee on Fisheries, Tangiers, Morocco, 26–29 June 2018.

are required on the use of damaging fishing methods, such as coral nets, that are known to indiscriminately impact precious coral habitats. More selective and less impactful fishing gears and practices need to be identified and promoted as alternatives to dragging gears.

3) Improvement of science-based management planning needed in many harvesting countries

Apart from a few sporadic cases, comprehensive management plans (including stock assessments) do not exist for the management and conservation of precious coral species. The priority should be given to the development, implementation and enforcement of national management plans for precious coral species in countries where these are exploited, with a greater focus on research and controls of commercially exploited species. The full implementation of the GFCM measures by member states including the newly established management plan (Recommendation GFCM/41/2017/5) is expected to be sufficient to counteract or prevent overfishing, with a view to ensuring long-term yields while maintaining the size of *C. rubrum* populations at biologically sustainable levels. Alongside this, actions to eliminate illegal, unreported and unregulated (IUU) fishing of red coral should also be foreseen. In the Mediterranean, countries actively harvesting red coral should consider the full range of fishery management, including spatial management, to introduce fishing regimes on the basis of the scientific advice available since the end of 2018.

Pacific Coralliidae corals are distributed across different countries and international waters (the United States of America, Japan, Taiwan Province of China and southern China). International cooperation is essential for the conservation and management of precious coral resources, which extends management beyond the boundaries of each nation's territorial waters or EEZ. In order to properly manage precious coral resources, and to eliminate IUU fishing, the establishment of a regional fishery management organization (RFMO) composed of representatives from the different countries is highly recommended. Similar to what has been done in the Mediterranean region, the development, implementation and enforcement of a Pacific management plan for Coralliidae is one option to consider.

4) Improvement of trade statistics is required

Trade data for black corals are available since their inclusion in Appendix II of CITES. However, in many cases the records refer to the genus or even the order, because of the above-mentioned difficulties in the identification of specimens (both raw colonies and finished products). Regardless of whether a species is listed on CITES, data on trade should be obtained and strictly monitored by all the harvesting countries within the framework that is connected to fishery management plans. Traceability mechanisms should be envisaged: from the time the coral is landed and sold as raw material to the manufactures until it reaches the retailer. These mechanisms would allow certification that the precious coral was collected in compliance with regional or national regulations (which would be effective in helping to curtail IUU fishing).

In the Pacific, given the difficulty of distinguishing among different types of corals in imports/exports (precious corals, semi-precious corals, stony corals or even shells) a convenient method of data collection needs to be devised in order to desegregate products from different species, and allow the compilation of trade statistics that can be compared against fishery activity and stock status.

5) Consideration of the likely effectiveness of the conservation of precious corals if they are placed under the provisions of multilateral environmental agreements (MEAs) and national control processes

At present CITES includes black corals (Antipatharia) in CITES Appendix II, without distinguishing between species, given their ecological importance and the range of threats they face. Furthermore, CITES already includes *P. elatius, P. secundum, P. konojoi,* and *C. japonicum* specifically for China in Appendix III. With regard to *C. rubrum*, strict protection was in place in Germany for a limited time (January 1987–June 1997) through the listing of the species in Annex 1 of Germany's Federal Ordinance on Species Conservation.

As early as 1987 at CITES Sixth Conference of Parties (CoP6, Ottawa 1987), a proposal for listing *C. rubrum* in the CITES Appendices was first presented and was rejected by CITES Parties. In 2007 (CITES CoP14), species in the genus Corallium were proposed for inclusion in CITES Appendices (Appendix II) and in 2010 (CITES CoP15) the proposal was extended to all the species of the family Coralliidae. In both these cases the proposals were made under CITES Criteria Annex 2aB (CITES Conf. 9.24 (Rev. CoP17)) that states:

> It is known, or can be inferred or projected, that regulation of trade in the species is required to ensure that the harvest of specimens from the wild is not reducing the wild population to a level at which its survival might be threatened by continued harvesting or other influences.

On both of these later occasions the proposals were assessed by the FAO Expert Advisory Panel[5] with an outcome that differed from that of the CITES Secretariat. In both cases CITES Parties rejected the listing proposals when they came to be voted on at the CITES CoP. The advice from the FAO Expert Advisory Panel concluded that the available evidence did not support the proposal to include all species in the family Coralliidae (*Corallium* spp. and *Paracorallium* spp.) in CITES Appendix II. The FAO Expert Panel considered the available data to be not very reliable, but useful to observe the extreme "boom-and-bust" cycles characteristic of this fishery: for example, where new beds were discovered these yielded large returns when first fished, but quickly depleted. The Expert Panel noted declines in catches of precious corals, maximum size of colonies, mean height and proportion of older colonies per stock, and recognized clear over-exploitation of shallow-water beds which had led to a shift in harvesting to deeper water coral beds. The Expert Panel also considered the difficulty in identifying products in trade, the substantial administrative burden of issuing CITES trade documents and recording the large number of individual pieces of precious coral in trade, as key issues affecting the potential effective implementation of CITES regulations. This advice was supported by experience in Germany, which had significant (irresolvable) identification problems at the species level for the enforcement of their Annex 1 Federal Ordinance on Species Conservation.

Today in addition to black corals being listed on CITES Appendix II, four species of Coralliidae (*P. elatius, P. secundum, P. konojoi,* and *C. japonicum*) are listed under Appendix III by China (as of July 2008).[6] Recent reports have suggested that demand for *Corallium* corals has increased in mainland China in recent years, leading to higher prices for unworked corals. However, available trade data does not reflect this. According to CITES trade data, mainland China has reported fewer imports of *Corallium* corals. According to exporting countries, the total amount of unworked coral exports to mainland China did not show any increase between 2008 and 2016. In assessments of trade, the mixing of products (across species, differentiating *Corallium* corals from non-*Corallium* corals) is further acknowledged to have hampered understanding and compliance of trade controls. Finally, consideration of how counterproductive a CITES listing could be is needed, as it may be perceived as evidence of rarity and may therefore stimulate the desire to purchase – thus boosting demand and increasing fishing pressure on resources, potentially jeopardizing the good fishery management efforts which are in progress.

With regard to the possibility of amending the listing status of precious corals in CITES Appendices, this report offers readers access to data collated across regional studies and summaries of the current situation for their general consideration, both for species listed in the CITES Appendices and not. This information goes some way to inform the reader on whether the conservation of precious corals has been affected by their CITES listing and the likely outstanding local and international management needs to ensure the long-term sustainability of precious coral resources.

[5] http://www.fao.org/fishery/cites-fisheries/ExpertAdvisoryPanel/en

[6] http://www.cites.org/eng/disc/how.php

In general, the use of multiple tools for the management of precious coral resources must continue to be the reality, and the report stresses that any effective management intervention requires the relevant authorities to collaborate on any management framework adopted.

1. INTRODUCTION

1.1. Corals that are the subject of this report

The order Antipatharia includes 7 families, 44 genera, and 265 species (Molodtsova and Opresko, 2018), the majority (> 75 percent) in waters deeper than 50 m (Cairns, 2007; Daly *et al.*, 2007). At least 13 species in 11 genera are used in jewellery, except for those produced in Hawaii;[7] the names of most species are unknown (Annex 2, Table 12 and references therein; Bruckner, 2016).

The family Coralliidae comprises 43 valid species (Tu *et al.,* 2016). Apart from *Corallium rubrum*, harvesting is known for other nine species. Three species distributed in the Hawaiian Archipelago are no longer collected commercially, although they have been fished in the past. Moreover, two new species, recently discovered by Tu *et al*. (2012) have been reported to be circulating on the market as pink coral (Jeng, 2015) (see Annex 2, Table 1 for details).

Coral species of the order Antipatharia and family Coralliidae, included in the present report, are all considered 'precious corals' (Table 1).

Table 1 Precious corals

COMMON NAME	BLACK CORALS	WHITE, PINK, RED CORALS
PHYLUM	Cnidaria Hatschek, 1888	Cnidaria Hatschek, 1888
CLASS	Anthozoa Ehrenberg, 1834	Anthozoa Ehrenberg, 1834
SUBCLASS	Hexacorallia Haeckel, 1896	Octocorallia Haeckel, 1866
ORDER	Antipatharia Milne-Edwards and Haime, 1857	Alcyonacea Lamouroux, 1816
SUBORDER		Scleraxonia Studer, 1887
FAMILY		Coralliidae Lamouroux, 1812

Information on the biology, fishery, management and trade of both Antipatharia and Coralliidae is reported, though mainly for the commercialized species. Other precious corals, for example bamboo and gold corals from the order Alcyonacea (families Isididae and Primnoidae) and order Zoantharia (family Gerardiidae) are also used for jewellery, but they are not the primary concern of this study and are therefore not included in the report. Similarly, semi-precious species such as stony corals (order Scleractinia), blue corals (order Helioporacea) and sponge corals (order Alcyonacea, family Melithaeidae) sometimes used for jewellery – but are of low value owing to their skeletal quality (Tsounis *et al*., 2010; Cooper *et al*., 2011) – are not considered in this document. Products made from Melithaeidae (sponge coral) as well as shell products that could be mistaken for (and may be marketed as) items made from Coralliidae, by virtue of their natural or dyed red colour (Cooper *et al.,* 2011) are also excluded.

[7] Large enough and with sufficient density and consistency to be suitable for cutting, carving and polishing into black coral jewellery.

2. BIOLOGY

2.1. Black corals (order Antipatharia)

Despite the importance of black corals in the culture and economy of many societies, very little is known about the basic biology and ecology of these organisms. To date, the limited information on the biology and ecology of antipatharians has not been reviewed comprehensively, and the majority of summaries on this group are found in taxonomic monographs that were published close to a century ago (Wagner *et al*., 2012 and references therein).

2.1.1. Taxonomy (Antipatharia)

In the eighteenth century the German scientist Peter Simon Pallas separated antipatharians from gorgonians, as proposed by Linnaeus in 1758. He based this decision on their spiny skeleton, which remains one of their most prominent and evident features.

Black corals are characterized by: 1) polyps with six unbranched tentacles that are non-retractile; 2) six primary mesenteries which extend throughout the polyp coelenteron; 3) skeletons that are primarily proteinaceous and covered with spines; and 4) the fact that they consist entirely of colonial species (Brugler *et al*., 2013).

In the 1970s, a major work of taxonomic revision began on Antipatharia corals, and is still in progress, with new species constantly being recorded and/or discovered in recent years in various areas and habitats (see Opresko and de Laila Loiola, 2008; Wagner and Opresko, 2015; Wagner and Shuler, 2017; Molodtsova, 2017; Molodtsova and Opresko, 2017; Opresko, 2017). For instance, the Hawaiian antipatharian coral previously identified as *Leiopathes glaberrima* (Esper, 1792) and *Leiopathes* sp. has been recently assigned the new name of *Leiopathes annosa* sp. nov. (Wagner and Opresko, 2015). Opresko (2015) has described seven new species in the territorial waters of New Zealand and adjacent regions.

In addition, modern exploration of deep abyssal plains is now shedding light on the enormous biodiversity of these organisms, especially on continental shoals and seamounts. The classification of Antipatharia has been historically complicated by the existence of species established only on the basis of dry, incomplete museum material, and by the absence of clearly, universally recognized taxonomic characters defining the hierarchy of the order.

One of the most important criteria for taxonomic research within the group Antipatharia is the morphology of the skeleton, which requires the use of scanning electron microscopy to reveal the subtlest details. The latest revisionary works of Opresko (2001, 2002, 2003, 2004, 2005, 2006) lead to the current classification including about 235 species, 40 genera, 7 subfamilies and 7 families, approximately a quarter of which have been described in the past two decades (Daly *et al.,* 2007). Higher figures were reported by Brugler *et al*. (2013): 41 genera and 247 species. These figures could be higher considering the recently discovered species (44 genera, 265 recognized valid species; see Molodtsova and Opresko, 2018).

At the end of the report, in the Appendix, Table A lists all species currently included in the antipatharians (Molodtsova and Opresko, 2018).

2.1.2. Distribution (Antipatharia)

A majority of antipatharian species inhabit tropical and subtropical regions in deeper to abyssal water environments, but they can also be found in subtidal and polar settings (Wagner *et al*. 2012; Brugler *et al*., 2013). Indeed, black corals are found over a wide depth gradient, ranging from waters as shallow as 2–4 m for wire corals in the tropical Pacific (Parrish and Baco, 2007; Bo, 2008; Brugler *et al*., 2013), down to depths of 8 900 m for *Schizopathes affinis* from the Pacific Kurile–Kamchatka and Aleutian Trenches (Brugler *et al*., 2013 and references therein). Despite this broad bathymetric range, over 75 percent of described antipatharian species are restricted to depths below 50 m (Cairns, 2007; Yesson *et al*., 2017). Black corals are restricted to marine ecosystems. They have not been found in areas with brackish waters, although some species inhabit areas with decreased salinities such as the fjords of New Zealand (Wagner *et al*., 2012).

Studies on the biogeographical distributions of antipatharian species are very scarce, because a large proportion of black coral species are only known from their type locality and consequently have very limited known ranges (Opresko, 2001, 2002, 2003, 2004, 2005a, 2006; Molodtsova, 2005). Lists of the antipatharian fauna have been made for several regions around the globe including the waters surrounding East Africa and the Mergui Archipelago, Maldives and Laccadive Islands, Diego Garcia and the northern Indian Ocean, the Gulf of Manaar, Japan, the Republic of Korea, China, Indonesia, the Moluccan Islands, Antarctica, the Hawaiian Islands, the Aleutian Islands, Alaska, the Gulf of Mexico, the Caribbean, Jamaica, Brazil, Madeira, the Canary Islands, the Bay of Biscay, and the Mediterranean (Wagner *et al*., 2012 and references therein).

An attempt to increase knowledge on the number of antipatharian species was undertaken by compiling a list of black corals below 3 000 m (from abyssal and hadal zones) from the Indian and Atlantic Oceans (Terrana *et al*., 2017) (Family Schizopathidae: genera *Abyssopathes, Bathypathes* and *Schizopathes*) while the black corals inhabiting abyssal depths often appeared to be widespread or cosmopolitan species (Molodtsova, 2014). Recently, black coral specimens assigned to seven species including four genera (*Antipathes, Stichopathes, Tanacetipathes* and *Distichopathes*) were discovered around Bermuda (Wagner and Shuler, 2017). Of these, three species (*Stichopathes* sp*., S. pourtalesi,* and *D. filix*), one genus (*Distichopathes*) and one family (Aphanipathidae) were reported from Bermudan waters for the first time (Wagner and Shuler, 2017).

New black coral records for the western Coral Sea adjacent to northeast Australia were reported by Horowitz *et al*., (2018), including the first record for the family Cladopathidae. The first report of black corals (Antipatharia) in shallow waters off the continental coast of Chile extends their geographical range in shallow waters of the upper continental shelf of South America (Gorny *et al*., 2018).

A new antipatharian species, *Heteropathes opreski*, from the northern border of the Oceanographer Fracture Zone (northeastern Atlantic) is described by de Matos *et al*. (2014a), greatly expanding the known distribution of this genus, as it was not previously reported to occur in the northeastern Atlantic. Similarly, the first record of *Antipathella subpinnata* for the Azores archipelago, described by de Matos *et al*. (2014b), extends the species' westernmost boundary of distribution in the northeastern Atlantic.

Antipatharians are generally found in areas with hard substrates, with the exception of species within the genus *Schizopathes*, which have a hook-like hold on soft surfaces (Wagner *et al*., 2012 and references therein). Additionally, black corals require low light and strong currents. Under favourable conditions, some black coral species form dense aggregations to the point of becoming ecologically dominant.

Yesson *et al*. (2017) showed that antipatharian distribution is strongly affected by temperature, with warmer temperatures being preferred (> 3 °C). The importance of seabed temperature and habitat complexity has been further confirmed by a study performed in the American waters of the Gulf of Mexico, which undertook a habitat suitability model for the antipatharian *L. glaberrima* (Etnoyer *et al*., 2017). Higher values for slope and curvature appear as preferred habitat, coupled with food availability. Antipatharians have indeed been documented to dwell in several geomorphologies of the continental margin including: submarine canyons, rocky terraces and outcrops arising from soft substrates (Opresko and Sánchez, 2005; Bo *et al*., 2012b; Bo *et al*., 2014; Bo *et al*., 2015b; Deidun *et al*., 2015; Ingrassia *et al*., 2015; Cau *et al*., 2017c; Massi *et al*., 2018).

In tropical and subtropical regions such as the Indonesian Archipelago, Hawaii, Palau or the Caribbean Sea, these corals are particularly abundant in shallow-water coral reefs where they create multiple, specific communities dwelling across various types of rocky habitat (Bo *et al*. 2012a; Wagner *et al*., 2012; Wagner, 2015). Shallow-water antipatharians are usually found in dim light conditions, with a more frequent occurrence in caves or crevices, or in temperate areas where light levels are substantially reduced by peculiar environmental factors (e.g. New Zealand fjords; Kregting and Gibbs, 2006).

Antipathes spp. preferably settle in depressions, cracks or other rugged features along steep ledges, with few colonies found on smooth basaltic substratum (Grigg, 1965). The lower depth limit of distribution coincides with the top of the thermocline in the Hawaiian Islands (approximately 100 m; Lumsden *et al*., 2007). Some other black coral species occur in shallow waters, underneath ledges and in caves (*Cirrhipathes anguina* can occur at depths of 4 m). Depth appears to correlate with the distribution of various coral taxa, whereas substratum and environmental conditions (low, sedimentation) seem to influence the patchiness of precious corals (Grigg, 1976).

Unlike other cnidarians, antipatharian tissues have no structural protection against abrasive forces, and muscular systems are poorly developed, so that tentacles can only contract slightly but not retract into a groove like other anthozoans. This physiological feature does not allow these organisms to dwell in habitats characterized by high rates of sediment resuspension, which can be detrimental to the soft tissues of antipatharians (Wagner *et al*., 2012 and references therein).

Nowadays, six species of black corals are recognized as being present in the Mediterranean Sea: *Antipathes dichotoma, Antipathes fragilis, Parantipathes larix, Leiopathes glaberrima, Antipathella subpinnata* and *Antipathella wollastoni*. Information is lacking on the Mediterranean populations of two species: *A. wollastoni* and *A. fragilis*, which are very rare or even taxonomically doubtful. The other four species are widely distributed in the western Mediterranean basin, including the Ionian Sea and, at least *L. glaberrima*, also in the Levantine area. In terms of abundance, *A. subpinnata* is the most common black coral species in the basin (Bo *et al*., 2011). These species are found on flat or gently sloping rocky hard grounds that are subject to moderate current conditions and variable silting. When aggregating into large forests, this species is inhabited by a wide variety of sessile and vagile organisms, for shelter or food (Otero *et al*., 2017).

2.1.3. Reproduction (Antipatharia)

Despite the importance of black corals in the culture and economy of many societies, very little is known about the basic biology of these organisms (Wagner *et al.*, 2012)

With the exception of a few studies on shallow-water species (< 50 m), most information on the sexual reproduction of antipatharians is derived from scarce notes on the anatomy of reproductive tissues accompanying taxonomic descriptions (Wagner *et al*., 2011c). Even though most of this information is derived from the examination of a few specimens, published data on sexual reproduction exist for at least 56 nominal species, representing > 20 percent of the described antipatharian fauna (Table 2.3 in Wagner *et al.*, 2012). Recently, additional studies have been published for the species *Dendrobathypathes grandis* (Lauretta and Penchaszadeh 2017; Lauretta *et al*., 2018), and *A. wollastoni* (Rakka *et al*., 2017).

On the basis of the published data, some common features of the antipatharian reproductive system can be listed: 1) of the six primary mesenteries, only the two in the transverse plane bear the filaments and gametes; 2) individual polyps are strictly gonochoric, polyps with only oocytes or spermatocysts have been found; 3) entire colonies are either female or male, with the exception of *Stichopathes* *saccula,* which has mixed colonies with both male and female polyps (Pax *et al*., 1987); and, 4) there is no evidence for internal fertilization within the Antipatharia.

Very little is known about the larval biology and reproductive seasonality. To date, only larvae of the species *Antipathella fiordensis* have been studied in laboratory cultures (Miller, 1996). The ciliated planulae, negatively buoyant and weak swimmers, were 200 µm in length (Miller, 1996). The larvae were likely non-feeding and stayed alive for a maximum of 10 days, but none settled (Miller, 1996). The few studies on the reproductive seasonality of black corals have all been conducted in shallow waters of less than 70 m (see Wagner *et al*. 2012 and references therein). Grigg (1976) performed histological observations on *Antipathes griggi* specimens collected in Hawaii in both July and March. The female sampled in July differed from those collected in March insofar as it contained both mature and immature oocytes, underlining evidence of an annual reproductive cycle culminating in the summer months (Grigg, 1976). Larvae of *A. griggi* and *A. grandis* were both negatively phototaxic, thus explaining their settlement behaviour at depths of low light intensity, generally below zones of active growth of shallow-water reef corals (Grigg, 2010).

Goenaga (1977) studied the sexual reproduction of two *Stichopathes* spp. in Puerto Rico and concluded that reproduction occurred almost all year round, although there were slight temporal differences in the appearance of mature gametes between species. Parker *et al*. (1997) reported that *A. fiordensis* has an annual gametogenic cycle, with spawning coinciding in the month of March when the warmest temperatures are registered in New Zealand. Bo (2008) monitored polyp fecundities of *Cirrhipathes* sp. in Indonesia and inferred spawning by the disappearance of gametes which occurred multiple times through the year. *A. subpinnata* from the Mediterranean Sea showed no fertile colonies in September–November when water temperatures were low (14 °C), and fertile colonies were observed in August when temperatures were higher (16 °C) (Gaino and Scoccia, 2010). Several tropical Indo-Pacific antipatharians reproduced in the summer during a period of approximately two months (Schmidt and Zissler, 1979). In summary, the reproductive cycle of various shallow-water (< 70 m) antipatharian species is seasonal and correlated with seasonal temperature fluctuations, with peak maturity occurring when water temperatures are higher. Whether deepwater black corals, such as *L. glaberrima*, also have a seasonal reproductive cycle is currently unknown and should be examined by future studies (Etnoyer *et al*., 2017).

In *A. wollastoni*, the reproductive season coincided with an increase in seawater temperature; spawning, however, as inferred from the disappearance of gametes, likely happened after the sea surface temperature peak of the year (September). Polyp fecundity ranged from 1 to 309 oocytes/polyp. A decrease in polyp fecundity was detected in samples at the higher pre-spawning maturity stage, indicating possible repetitive spawning or oocyte absorption. Intra-colonial comparisons revealed a longer reproductive cycle in the medial colony section, and a gradient of increased oocyte size towards the apical section, possibly due to intra-colonial differences in energy allocation between reproduction and other biological processes, or as a strategy against predation on gametes/larvae. Colony height was positively correlated with polyp fecundity indicating that the reproductive output increases with colony size (Rakka *et al*., 2017).

Black corals are known as gonochoric organisms, meaning that separate male and female colonies can be found – although hermaphroditic species have been documented. Observations conducted in the field have always reported the absence of clear external morphological differences between the sexes. This seems to be the general rule, both in fertile and infertile colonies. Gametes are situated in relation to the primary transverse mesenteries in all species; in some cases, these mesenteries can reach into the cavities of lateral tentacles. Also, there is apparently no evidence of internal fertilization in antipatharians, suggesting that fertilization and larval development likely occur externally in the water column, and not internally within polyps (Wagner *et al*., 2011).

Dendrobathypathes grandis, a gonochoric species, is peculiar for the presence of gigantic oocytes, which can reach 1 500 µm in diameter – a volume more than twenty times the biggest oocyte reported, and over 800 times more when compared with the common oocyte size of the group (Lauretta and Penchaszadeh, 2017). The gametes of one of the seven females studied contain a funnel-shaped cluster of specialized cells that facilitate the transport of nutritive substances, known as the trophonema. This structure has been found within all orders of anthozoans, but this is first time the presence of a trophonema has been reported within the order Antipatharia (Lauretta *et al*., 2018).

Asexual reproduction is of vast importance to cnidarians and antipatharians are no exception to this trend. Modes of asexual reproduction that have been reported for black corals include: 1) budding of new polyps; 2) breakage of fragments which can subsequently reattach to the substrate; and 3) production of asexual larvae in aquaria under stressful conditions (Wagner *et al*., 2012 and references therein). The latter two modes of asexual reproduction result in the formation of new colonies, whereas the former leads to new polyps on existing colonies (Wagner *et al*., 2012 and references therein).

The artificial transplantation of black coral fragments has been attempted for *Antipathes* sp. in Cuba (Gonzalez *et al*., 1997), *Stichopathes* cf. *maldivensis* and *Cirrhipathes* cf. *anguina* in Indonesia (Bo *et al*., 2009c), and *A. griggi* in Hawaii (Montgomery, 2002), with mean survival rates of 93 percent, 80 percent, 68 percent and 45 percent respectively. Collectively, these high survival rates indicate that asexual reproduction via fragmentation can be successful under favourable conditions, but the likelihood of reattachment in the field is so low that asexual reproduction via colony fragmentation is predicted to be rare.

Molecular studies on the population genetic structure of *A. fiordensis* in New Zealand suggest that asexual reproduction is both common and geographically widespread within this species, because genetic clones have been reported to occur frequently and over geographic distances of up to 60 km (Miller, 1997; 1998).

2.1.4. Recruitment (Antipatharia)

Early research by Grigg (1976) provided a preliminary list of the habitat requirements of both species of black coral, including: a firm and rugose substratum for settlement and attachment, and strong bottom currents for the active transport of detritus and micro-planktonic food particles, and to sweep the bottom clear of sediments.

2.1.5. Growth and mortality (Antipatharia)

The order Antipatharia comprises some of the longest-living marine organisms (Roark *et al*., 2009).

Their skeleton is cylindrical, with a central canal surrounded by a series of concentric layers of deposited organic material. These micro-layers are responsible for the growth rings visible in transversal sections, actually used to infer colony age and growth rate. Antipathin, the component of antipatharian skeleton is a combination of chitin (a polysaccharide that also makes up the cuticle of insects and crustaceans) and a non-fibrous scleroprotein (Lartaud *et al*., 2016). The absence of carbonatic elements (e.g. sclerites) makes the skeleton extremely flexible and able to bend with the current flow, which is a crucial environmental element that enhances antipatharian distribution.

Literature data concerning growth rates focus mainly on radial growth rates recorded with radiometric studies using radioactive carbon (^{14}C). Isotopic studies on organic and carbonate coral skeletons in fact allow for paleoclimatic reconstructions of superficial and deepwater temperatures in the recent past (Raimundo *et al*., 2013).

Several methodologies have been used to estimate growth rates and longevities of antipatharians, including: 1) time-series measurements of colony fragments in aquaria; 2) time-series measurements of tagged colonies in the field; 3) measurements of colonies on artificial structures of known age; 4) growth ring counts; 5) analysis of size-frequency distributions; and 6) radioisotope dating techniques.

Studies that have tracked black coral growth report either changes in colony height over time (i.e. vertical growth rate) or changes in the diameter of the main stem over time (i.e. radial growth rate). The latter is particularly useful when colony height is difficult to estimate owing to the flexibility of branches (Olsen and Wood, 1980); however, it also requires more precise measurements because radial growth is more limited than vertical growth (see Appendix Table B).

In general, the studies report that the formation of rings is roughly annual (Wagner *et al*., 2012 and references therein), although Noome and Kristensen (1976), counted about 2 000 growth rings in a *Stichopathes gracilis* colony settled on an artificial structure over the course of a six-year project, concluding that growth rings are formed daily by this species.

The fastest growing antipatharians dwell in shallow tropical waters (10–36 m), with height growth rates of several centimetres per year (Warner, 2005), occasionally showing some of the highest growth rates ever recorded for colonial cnidarians. The fastest growing antipatharians appear in shallow waters (10–36 m), tropical wire corals, with vertical growth rates ranging between 3 and 7 cm/year for *Stichopathes* spp. from Puerto Rico (Goenaga, 1977) up to 159 cm/year for *Stichopathes* cf. *maldivensis* from Indonesia (Bo *et al*., 2009c). Longevity has not been reported for other *Sticophates* species. Individual colonies have been monitored for 2–3 years with no signs of senescence, indicating that lifespans exceed these time periods (Goenaga, 1977; Bo, 2008).

On the other hand, the slowest growing antipatharians dwell in deepwater habitats and belong to the family Leiopathidae; the specimens dwelling in Hawaiian waters were found to be the oldest colonial organisms discovered to date (i.e. 4 265 years old) and among the oldest animals on the planet (Roark *et al*., 2009). Specimens from the genus *Leiopathes* have been documented as the oldest skeletal-accreting marine organisms, with the slowest growth rates observed in species that have the greatest life spans (Carreiro-Silva *et al*., 2013).

In the Gulf of Mexico, the average estimated age of *L. glaberrima* colonies was 143 years, with an average range of 0–909 years. Among all measured colonies (n = 357), the most abundant size class (n = 191, or 53.5 percent) was 15–30 cm high, which corresponded to an age range of 50–100 years. The next most abundant size class (n = 68, or 19 percent) was < 15 cm high, which corresponded to an age of less than 50 years. The 30–45 cm size class (n = 55, or 15.4 percent) correlated with an age of 101–250 years. The size class of 45−60 cm (n = 22, or 6 percent) corresponded to an age of 250–400 years. The size class of 60−74 cm (n = 12, or 3.3 percent) corresponded to ages of 401–500 years. Colonies that were 74 cm and taller (n = 9, or 2.5 percent) correlated to ages > 500 years. Large colonies were present in both the northwestern and eastern parts of the Gulf of Mexico. The largest *L. glaberrima* coral colonies were over 1 m tall, with an age range estimated to be a minimum of 565 and maximum of 2 000 years old, depending on the growth rate (Etnoyer *et al*., 2017).

Other colonies belonging to the species *L. glaberrima* were documented to dwell in a presumably pristine environments in the Mediterranean (Bo *et al*., 2015a; Cau *et al*., 2017b), showing some colonies with considerable sizes in basal diameter (> 7 cm). Mediterranean specimens of about 4 cm in diameter have been dated as 2 000 years old. Comparable ages were already reported for *Leiopathes* species from deeper oceanic regions (Roark *et al*., 2009; Prouty *et al*., 2011; Corriero-Silva *et al*., 2013), but the Sardinian record also suggested a millennial stability of a deep-sea biocoenosis in a semi-enclosed, heavily exploited basin. Considering results of radiocarbon dating and the population size structure, it is probable that in the garden of the Carloforte shoal a major proportion of the colonies are hundreds of years old, with few millennial specimens (Bo *et al*., 2015).

A total of 155 colonies of *L. glaberrima* had an average height of 70 ± 3.4 cm and an average width of 69 ± 4.4 cm with a regular isometric growth. Three very large specimens (with heights and widths exceeding 200 cm) were recorded, characterized by a thin, fan-like profile. The basal diameter of the stem, on average 1.0 ± 0.1 cm, was linearly correlated with the height of the colony even though there are few large specimens with an exceptionally large diameter (maximum of 6.8 cm in a colony 200 cm high over a surface area of 4 m^2; Bo *et al*., 2015).

Longevities of smaller magnitude than *Leiopathes* species have been reported for *Stauropathes arctica* in the Newfoundland and Labrador region (33–66 years; Sherwood and Edinger, 2009), *Plumapathes pennacea* from Jamaica (35 years; Oakley, 1988), *Antipathes* sp. from the Red Sea (81 years; Risk *et al.*, 2009), and *Antipathes dendrochristos* from California (140 years; Love *et al.*, 2007).

Slower growth rates are evident for the genus *Antipathes*. Grigg (1964) observed growth of fragments of the Hawaiian species *A. griggi* (as *A. grandis*) kept in aquaria, and reported vertical extensions of 1.2 cm/ year. Studies using a different methodology (time-series measurements of tagged colonies in the field) then showed faster vertical growth rates ranging between 4.0 and 11.6 cm/year for *A. griggi*, and 2.92–6.12 cm/year for its sympatric congener *A. grandis* (Grigg, 1974, 1976; Montgomery, 2006). Assuming constant growth rates throughout the lifespan of corals, Grigg (1974) estimated longevities of 40 and 100 years for the *A. griggi* and *A. grandis* species, respectively. These ages appear higher than those obtained for *A. griggi* using radiocarbon dating techniques (12–32 years; Roark *et al*., 2006).

In summary, the available information suggests that all antipatharians are generally slow-growing and long-lived organisms, with longevities varying from decades to millennia (see Appendix Table B).

The ecological factors considered most important for fishery management are rates of growth, recruitment and natural mortality, along with reproductive behaviour and seasonality. In spite of their importance, no data on mortality of antipatharians are present in literature. The only available data are those recorded by Grigg (1976) for *A. griggi* and *A. grandis* **(**M=0.07)**.**

2.1.6. Population structure and density (Antipatharia)

In areas with favourable conditions, some antipatharians can reach high population densities (see Appendix Table C) and build monotypic aggregations that extend over large areas. Furthermore, black coral beds represent important reservoirs of biodiversity because they typically host distinct communities, including a myriad of organisms that are restricted to such habitats (Wagner *et al.*, 2012 and references therein). The highest densities among antipatharians have been recorded for several species of wire corals (*Stichopathes* spp. and *Cirrhipathes* spp.). In particular, *Stichopathes spiessi* populations can reach densities of up to 20 colonies/m^2 on deepwater (550–1 150 m) seamounts in the eastern North Pacific (Genin *et al*., 1986; Opresko and Genin, 1990). Dense aggregations of wire corals are also common in many deepwater (> 150 m) environments around Hawaii, although quantitative population densities have not yet been reported (Chave and Malahoff, 1998; Parrish *et al*., 2002). High population densities of *Stichopathes* spp. are also widespread throughout several shallower (< 50 m) locations in the Caribbean, with values up to 7.32 colonies/m^2 (Sanchez *et al*., 1998). In the Mediterranean, *A. subpinnata* forms massive aggregations of up to 5.2 colonies/m^2 at depths ranging from 70 to 100 m (Bo, 2008; Bo *et al*., 2008, 2009b).

In the Mediterranean, *L. glaberrima* showed a typical heterogeneous distribution, with colonies aggregated in small patches, particularly along the bench terraces. The colonies usually displayed a vertical development; some of those observed also inclined, but were never over-hanging (Bo *et al*., 2015). *L. glaberrima* showed an average abundance of 1.4 ± 0.8 colonies/m^2 (ranging from

0.3 to 2.5 colonies/m^2). Peaks of about eight colonies/m^2 were observed in some frames. Considering the total number of counted colonies in the frames, it is possible to estimate the occurrence of over 2 600 black coral colonies along the entire recorded video transects.

In slightly shallower waters (5–70 m), dense black coral populations exist in several locations around the world including in Indonesia, New Zealand, Palau, Hawaii and the Caribbean, with densities of up to 2.5 colonies/m^2 (see Appendix Table C). At the other end of the spectrum, several black coral species are not seen in aggregations (Grigg, 1988; Chave and Jones, 1991; Chave and Malahoff, 1998). Although population densities are typically not reported for antipatharians in this latter category, several of these occur in deeper water (> 300 m) and include several species in the family Leiopathidae and Schizopathidae (Grigg, 1988; Chave and Jones, 1991; Chave and Malahoff, 1998). Taken as a whole, the available information suggests that population densities are highly variable among antipatharian corals, but that at least some black coral species can reach particularly high densities to the point of becoming ecologically dominant.

Despite the fact that densities of antipatharians are highly variable, in areas with favourable habitat conditions they can form dense mono-specific forests that can extend over vast areas, becoming ecologically dominant in the hosting environment. Such aggregations are considered marine forests that are composed of animals rather than plants, enhancing the three-dimensional development of the habitat and consequently its complexity (Rossi 2013; Bo *et al*., 2014; Bo *et al*., 2015b). As such, they show the same ecological features of their terrestrial counterpart by providing favourable habitat within their canopy for a variety of associated organisms, at all trophic levels (Cerrano *et al*. 2010; Bianchelli *et al*., 2013; Cau *et al*., 2017b).

With regard to populations structures, a small number of studies exist for the most important commercial species that investigate populations to describe the impact of fisheries.

Grigg (2001) described the change in population structure in Hawaii after several decades of harvesting, comparing age frequency distributions of black coral (*A. griggi*) in the Maui Bed in 1975 and 1998. The age frequency distributions of sample populations in 1975 and 1998 exhibit a remarkably similarity. Except for the first two years, which are underrepresented in both surveys, year-class abundance exhibits a pattern of more or less steady decline from the third year to the oldest colonies in the population. The underrepresentation of young colonies (age 0–2 years) is likely a product of the difficulty of seeing young colonies of less than 5–10 cm in height, because larvae frequently settle in the shaded cracks and interstices of the reef. Although there is some small fluctuation between age classes, the overall population structure is relatively stable (Grigg, 2001). The only important difference between the age structure of the 1975 and the 1998 sample populations is the reduced number of large colonies of over 19 years of age: 8.6 percent in 1998 versus 10.8 percent in 1975; a decrease of 2.2 percent. This shift to a younger age frequency is clearly a result of harvesting. Nevertheless, the lack of any substantial difference in the age structure of colonies younger than 19 years old, which represents the recommended size limit of 1.2 m, indicates excellent compliance by the divers with the management guideline over the past two dozen years (Grigg, 2001). Perhaps the most remarkable result of the study is the degree to which the population has renewed itself since 1975. The recruitment over this time period has been nearly continuous, and the fishery itself can therefore be considered sustainable. In fact, 97 percent of the bed in 1998 was entirely the result of recruitment since 1975 (Grigg, 2010b). Unfortunately, since that time, annual harvest has increased, and in 2001 an alien species, *Carijoa riisei*, was discovered invading the deeper zones of the black coral bed off Maui at depths of 80–100 m; it has since been overgrowing portions of the substratum as well as many adult colonies of both species of black coral (Grigg, 2010b and references therein).

In Mexico, Padilla and Lara (2003) studied both exploited and unexploited banks of black corals off the Mexican coasts (State of Quintana Roo). Significant differences were found among banks in terms of their densities and population structure. Near the intensely harvested area of Cozumel Island the populations reveal serious deterioration. Black coral abundance is low; the average density is 0.060 colonies/m^2, with a range of 0.005–0.140 colonies/m^2. Black coral populations along the southern coast of Quintana Roo are patchily spread out, forming coral banks in some areas where density is high (0.200–0.450 colonies/m^2), but in other areas black coral is practically absent. Four important black coral banks were found in this region where colonies of more than 1 m high are common. The size structure observed corresponded to populations in good condition with organisms in a broad size range (0.15–1.90 m); demographic analysis revealed a slow population growth. Surveys made in Chinchorro, not yet commercially exploited, showed black coral populations in good condition. The colony size of *P. pennacea* differs along the Mexican Caribbean reefs, showing that in Banco Chinchorro it is possible to find colonies in a wide range of sizes, although bigger colonies are more frequent on the bank. As concerns *Antipathes caribbeana*, it occurs in Banco Chinchorro in a broad size range, from large colonies down to small recruits; densities were estimated as 0.050–1.300 colonies/m^2 (Padilla and Lara, 2003).

Similarly, Gress and Andradi-Brown (2018) studied two species of interest to the jewellery industry (*P. pennacea* and *A. caribbeana*), the harvesting of which began in the early 1960s around Cozumel Island (Mexico) and stopped in 1995. The population structure was compared between 1998 (a few years after harvesting ended) and 2016. *P. pennacea* in 2016 are substantially lower than in 1998. However, the 2016 *P. pennacea* population has shifted to be dominated by larger colonies, suggesting disproportionate juvenile mortality or recruitment failure. On the contrary, no change in population density or colony size of *A. caribbeana* was detected between 1998 and 2016 (Gress and Andradi- Brown, 2018). Despite harvesting occurring for almost 70 years in the Mexican Caribbean, no information on reproduction, recruitment and other dynamics of the targeted species is available (Gress and Andradi-Brown, 2018).

2.1.7. Genetics and genetic stock identification (Antipatharia)

Antipatharians have been included in several phylogenetic studies; however, sample sizes are small and taxonomic coverage minimal (Brugler *et al*., 2013 and references therein; Terrana *et al*., 2017).

For instance, to quantify genetic variation in the black coral mitogenome, Brugler *et al*. (2013) analysed the DNA sequences of 26 of 41 genera, representing all families and subfamilies. They constructed the first multilocus phylogenies of the Antipatharia. Reconstructions revealed that species in the genus *Stichopathes* are split across two families, *Sibopathes macrospina* groups among North Atlantic *Parantipathes* (suggesting the actinopharynx and mesenteries were secondarily lost), and that three families are polyphyletic. These and other results provide novel, independent insights into the evolutionary history of antipatharians and support the placement of species into higher-level groupings based on microscopic skeletal features rather than gross colony morphology (Brugler *et al*., 2013).

Recently, genetic analyses based on the internal transcribed spacer (ITS1) of the ribosomal DNA have been used to assess the phylogenetic relationships between 25 species (4 new to science) of the shallow-water corals off the southwestern coast of Madagascar. Sequences of the members from the Myriopathidae family did not discriminate the different species; more surprisingly, some whip corals *Cirrhipathes* were included in the latter clade while other *Cirrhipathes* species were grouped in three other clades (Terrana *et al*., 2017).

However, the full list of these studies is beyond the scope of the present report. On the contrary, genetic studies of populations are very limited in spite of how important these types of data are to assess population status. A key component of the resilience of coral populations lies in their inherent genetic diversity or capacity to cope with environmental change – but also in processes such as larval dispersal, which can facilitate population maintenance and recovery from disturbance, as well as the colonisation of new habitats (Miller *et al*., 2010).

Miller (1997) studied the populations of *Antipathes fiordensis* in New Zealand by examining the genetic variation and population subdivision both within and among fiords. Allozyme electrophoresis at 10 polymorphic loci revealed an unusual population genetic structure with significant genetic variation between sites, suggesting larval dispersal is restricted even at distances of 10–15 km.

Asexual reproduction was apparent in populations of *A. fiordensis*, as evidenced by genotypic diversity ratios < 1, as well as significant departures from random mating associated with a combination of heterozygote deficits and excesses at the majority of sites. The atypical genetic structure observed represents a population that has not yet reached equilibrium due to a combination of the effects of recent colonisation, asexual reproduction and the potential longevity of individual coral genotypes (Miller, 1997).

Miller (1998) examined the fine-scale (< 50 m) pattern of relatedness between black coral colonies at three sites to infer dispersal distance. The results suggested dispersal of black coral larvae is highly philopatric and that larvae settle very close to parent colonies. The study suggests that gene flow is restricted and patch size in this species may be relatively small, with evident implications for its conservation.

Thoma *et al*. (2009) showed that *Bathypathes* Brook, 1889 Type A and *Parantipathes* Brook, 1889 Type B both collected along the New England and Corner Seamounts (northwest Atlantic), may be connected by gene flow over an area of 1 599 and 1 461 km, respectively. However, these results do not preclude the possibility that cryptic variation and endemism not revealed by mitochondrial DNA may become evident, should more variable markers be developed (Thoma *et al*., 2009).

Miller *et al*. (2010) assessed connectivity among deep-sea coral populations on seamounts and slopes in the Australian and New Zealand region. They found evidence of genetic subdivision across the ocean for the antipatharians *Antipathes robillardi* and *Stichopathes variabilis*.

These results provide further support for the prediction that ocean expanses of 500–1 000 km are effective barriers to ongoing gene flow for deep-sea coral species and that communities of benthic organisms on two of the ridges (Lord Howe and Norfolk) are dissimilar to one another, at least in part, because of topographically controlled current patterns that promote connectivity within ridges rather than between them.

Consistent with the genetic differentiation found by Miller *et al*. (2010) are observations of antipatharian larvae that suggest they may be negatively buoyant and only weak swimmers and thus unlikely to disperse far (Miller *et al.*, 2010 and references therein).

On the contrary, no evidence of genetic subdivision between sites within regions for *Stichopathes filiformis* was found, suggesting sufficient gene flow occurs in order to maintain genetic homogeneity at scales of tens to hundreds of kilometres (Miller *et al.*, 2010).

One aspect of this study which may have contributed to the lack of genetic structure observed within and between regions is sample size. Coupled with relatively high levels of haplotype diversity, this may have reduced our power to detect structure, if it exists. This is an unavoidable consequence of the generally limited sampling in deep-sea habitats, as well as the seemingly patchy distribution of effort. Nonetheless, the results form a baseline database for future studies as more samples and regions are gradually explored. Ultimately, however, a more complete understanding of the genetic structure and dispersal of coral larvae among seamounts will require a targeted programme that ensures adequate sampling replication across a variety of spatial scales (Miller *et al.,* 2010).

Recognizing that some seamount regions and coral populations may or may not be effectively isolated will be a key component of successful management planning, based on marine protected area (MPA) networks both within and beyond national jurisdictions (Miller *et al*., 2010).

Brugler *et al.* (2013) quantified divergence in the mitochondrial DNA at the intraspecific level for 100+ colonies of *Antipathes griggi,* collected from eight sites across five Hawaiian Islands (Hawaii, Maui, Oahu, Kauai and Niihau), with 566 km separating the two most distant sampling sites (great circle distance). Two haplotypes were found, but it is not clear if they represent two different species (*A. griggii* and *A.* cf. *griggi*) or intraspecific variation. Except for three locations characterized by small sampling, all localities included colonies of both haplotypes. By considering *A.* cf. *griggi* a different species, no sequence divergence was observed at the intraspecific level for *A. griggi* for any of the five mt gene regions examined, supporting the general inability of anthozoan mitochondrial DNA to provide population-level variation (Brugler *et al*., 2013).

Ruiz-Ramos *et al*. (2015) investigated multiple colour morphs of *L. glaberrima* that grow sympatrically in the Gulf of Mexico. Morphological, mitochondrial and nuclear ribosomal markers supported the hypothesis that colour morphs constituted a single biological species and that colonies, regardless of colour, were somewhat genetically differentiated east and west of the Mississippi Canyon. Gene flow was disrupted between and within two nearby hard-ground sites (distance = 36.4 km) where two sympatric micro-satellite lineages – which might constitute cryptic species – were recovered. Lineage one was outbred and found in all sampled locations across 765.6 km in the northern Gulf of Mexico. Lineage two was inbred, reproducing predominantly by fragmentation, and restricted to sites around Viosca Knoll. It was concluded that *L. glaberrima* is phenotypically plastic with a mixed reproductive strategy in the northern Gulf of Mexico. Such strategy might enable this long-lived species to balance local recruitment with occasional long-distance dispersal to colonize new sites in an environment where habitat is limited (Ruiz-Ramos *et al*., 2015).

Recently, through the combination of a genetic and physical models applied to the millennial coral *L. glaberrima* dwelling in the Gulf of Mexico, Cardona *et al.* (2016) investigated the scale of their dispersal via planktonic larvae. Overall, results showed limited spatial dispersal capabilities, emphasizing how, in case of perturbations, any recovery would depend on local larvae produced by surviving individuals (Cardona *et al*., 2016).

2.1.8. Threats and environmental issues (Antipatharia)

Black coral communities, both in the shallow and deep waters, are under increasing threat from a range of direct and indirect anthropogenic impacts. In particular, black corals are considered vulnerable because they display life-history characteristics often associated with susceptibility to local extirpation (i.e. they are long-lived and slow-growing), in addition to the threats to their persistence associated with fishing, pollution (including nutrient pollution), climate change and ocean acidification (Miller *et al*., 2010 and references therein).

Apart from the direct harvesting of a few species, primary threats are damage by destructive fishing practices, and damage by entanglement in lost longlines and netting.

For the few species that live on soft sediments, the main threat is mechanical disturbance from fishing activities, particularly bottom trawling gear, beam trawls and dredges. Bottom trawling, together with other fishing methods, physically impacts the sea floor and has consequences for the benthic communities that can lead to changes in the trophic structure and function of these benthic communities. The damage to these communities might have dramatic consequences for the sustainability of their populations, considering the combination of the destructive effect of bottom trawling and dredges, combined with their slow growth rates, reduced larval dispersal and the patchy and highly episodic nature of recruitment events (Otero *et al*., 2017).

In the Mediterranean, recent ROV (remotely operated vehicle) and manned submersible surveys provide evidence that fishing impact is a major concern for hard-bottom communities (Bo *et al*., 2013, 2014, 2015; Orejas *et al*., 2009; Angiolillo *et al*., 2015, 2016; Deidun *et al*., 2015; Cau *et al*., 2015, 2017; Consoli *et al.,* 2018). In particular, longlining and benthic gillnets used by recreational and professional fisheries are very damaging fishing practices in the Mediterranean, since they are generally practised in deeper waters than trawling, very shallow areas or in untrawlable zones that are good habitats for different anthozoan species (e.g. black corals, cold water corals, etc.).

Bycatch of black corals is also recorded with the use of professional trammel nets, gillnets and longlines (Mytilenou *et al*., 2014; Otero *et al*., 2017 and references therein).

Ghost fishing by abandoned or discarded small-scale gears is another issue of potential importance in the Mediterranean basin. Anchor lines, ropes connecting lobster pots, gill nets placed across gorgonian populations and lost monofilament lines may cause tissue abrasion or the detachment of whole coral colonies (Bavestrello *et al*., 1997; Tsounis *et al*., 2012). Ghost gear has been showed to directly impact benthic organisms, primarily gorgonians (*Paramuricea clavata, Eunicella cavolini, Callogorgia verticillata, C. rubrum*, etc.), black corals (*A. subpinnata, A. dichotoma and L. glaberrima*) and sponges in the Tyrrhenian Sea, Campania, Sicily and Sardinia (Bavestrello *et al*., 1997; Tsounis *et al.,* 2012; Bo *et al*., 2013, 2014; Angiolillo *et al*., 2015; Cau *et al*., 2015, 2017).

Recent studies have also shown that the deep rocky environments of the Tyrrhenian Sea, with a relatively high abundance of debris (mainly lines and nets, plastic items), have a heavy impact (covering and abrasive action) on gorgonians, *C. rubrum*, antipatharians (*A. subpinnata, A. dichotoma* and *L. glaberrima*), and other invertebrates (Bo *et al*., 2014; Angiolillo *et al.*, 2015; Cau *et al*., 2015, 2017).

The effect of anthropogenic alterations on the shoreline and deeper areas is less documented for antipatharians. The impact of pollution (crude oil and chemical dispersants) has been recently described for a population of *L. glaberrima* in the Gulf of Mexico (Ruiz-Ramon *et al*., 2017).

Finally, for the Mediterranean it has been proposed that sea-floor drilling activities as part of oil exploration or mining, and the deployment of submarine cables and pipelines, all threaten the integrity of deep benthic communities such as those where *A. subpinnata, A. dichotoma, P. larix or L. glaberrima* might occur (Otero *et al*., 2017).

In a recent assessment of the conservation status of Mediterranean anthozoa, five species of Antipatharia were included in the Mediterranean IUCN Red List in the following categories: one Data-Deficient (*A. wollastoni*), three Near-Threatened (*A. dichotoma, A. subpinnata,* and *P. larix*), and one Endangered (*L. glaberrima*) (Otero *et al*., 2017). The same species have also been included in the Red Book of Corals of Croatia as Critically Endangered (CR) for *A. subpinnata*, Endangered (EN) for *L. glaberrima* and Vulnerable (VU) for *A. dichotoma*. In Croatia, apart from the destruction of colonies by fishing gear the main threat is diving (collection by recreational divers) (CITES document AC29 Doc.22). In the Italian Red List (Bo *et al*., 2014), *L. glaberrima* is listed as EN (over 1/3 of populations have disappeared); *A. subpinnata, A. dichotoma* and *P. larix* are listed as LC (Least Concern) because no evident signs of decline have been documented in the populations.

2.2. White, pink and red corals (family Coralliidae)

2.2.1. Taxonomy (Coralliidae)

The family Coralliidae comprises 43 valid species, which are listed in Annex 1, Table 1, with the indication of both the traditional (van Ofwegen, 2004) and the new classification (Tu *et al*., 2016), to facilitate the reader accustomed to the past nomenclature (possibly still used in trade records). The traditional division of the family Coralliidae in two genera (*Corallium* and *Paracorallium*), has also been reconsidered (Ardila *et al*., 2012; Figueroa and Baco, 2014; Tu *et al.,* 2015a; Tu *et al.,* 2016; Uda *et al*., 2013). By means of molecular phylogenetic analysis, Tu *et al.* (2015a) distinguished 7, 18 and 18 species within the three genera of Coralliidae (*Corallium, Hemicorallium* and *Pleurocorallium*).

Twelve species have only been described and formally named in recent years (Nonaka *et al.*, 2012; Simpson and Watling 2011; Tu *et al*., 2016; Tu *et al*., 2015b; Tu *et al*., 2012). However, several species that have not yet been described are still reported to exist within the Coralliidae. Nearly half of the coralloid specimens at the Smithsonian National Museum of Natural History remain unidentified (Figueroa and Baco, 2014). In particular, recent molecular data confirm the possible occurrence of cryptic species: for instance, the cosmopolitan *Hemicorallium laauense* could be a species complex containing a cryptic species *Hemicorallium* sp.8 (Tu *et al.*, 2015a). Other species, such as *Hemicorallium imperiale* and *Hemicorallium ducale*, might also be species complexes containing cryptic species (Tu *et al.,* 2015a). In general, species within *Hemicorallium* cannot be delimited by mitochondrial markers, but most of them can be distinguished by morphometric measurements coupled with multifactorial analyses (Tu *et al*., 2016).

As far as the species of commercial importance are concerned, according to recent molecular phylogenetic analyses both Mediterranean and Japanese red coral are now classified as *Corallium* (i.e. *Corallium rubrum* and *Corallium japonicum*). The genus *Paracorallium* is indeed not considered a valid taxon anymore, given that its species are all nested within *Corallium* (Ardila *et al*., 2012).

Similarly, based on genetic data, the boundaries between three commercial species (*Pleurocorallium carusrubrum,*[8] *P. elatius* and *P. konojoi*) were ambiguous; these three species were therefore grouped together as the '*P. elatius* species-complex' (Tu *et al*., 2015a). Apart from genetic results, intraspecific morphological variability is also recognized (Nonaka *et al*., 2006; Nonaka and Muzik, 2010). For this reason, the classification of both species needs to be re-examined from both morphological and genetic points of view.

By means of ROV surveys, an unidentified species belonging to the family Coralliidae showing similar morphologic features as precious corals was collected at depths of 1 420–1 620 m from the waters around the Ogasawara Islands (Hasegawa *et al*., 2010) and observed around Koko Seamount (Emperor Seamounts) at a depth of 425 m (Fisheries Agency of Japan, 2008; see Annex 2). Again, taxonomic studies are needed in order to properly identify this deep-sea dwelling species.

2.2.2. Distribution (Coralliidae)

Species in the family Coralliidae inhabit tropical, subtropical and temperate oceans (Bruckner, 2016).

Several studies have revealed that the diversity hotspots of coralliid species are located in the west and central Pacific, including the seas surrounding New Caledonia, Taiwan Province of China, Japan and the Hawaiian Archipelago (Bayer 1956; Tu *et al*., 2012; Nonaka *et al*., 2012). By contrast, species diversity appears lower in the Atlantic. A few species are known from the Indian Ocean, eastern Pacific Ocean, the Gulf of Alaska, and off the coast of California. The depth range of this family extends from about 5 to 2 400 m (Bayer 1956; Tsounis *et al*., 2010 and references therein).

Nevertheless, only two areas have historically large populations and are commercially exploited: the Mediterranean Sea and the northern Pacific Ocean.

The following paragraphs describe the distribution of the main Coralliidae species of commercial importance.

[8] See Footnote 1.

2.2.2.1. Mediterranean Sea and Atlantic Ocean

Corallium rubrum is endemic to the Mediterranean and neighbouring Atlantic coasts, at depths ranging from 5 to more than 1 000 metres. Its distribution within the Mediterranean basin is mostly centred around the central and western basin with smaller populations in deeper waters in the eastern basin and off the coast of Africa around the Canary Islands, southern Portugal and around Cabo Verde (Chintiroglou *et al*., 1989; Marchetti, 1965; Weinberg, 1976; Zibrowius *et al*., 1984; Boavida *et al*., 2016; Cattaneo-Vietti *et al*., 2016; Otero *et al*., 2017; Ghanem *et al.,* 2018) (see Annex 1, Figure 1). Nonetheless, a very recent paper by Çinar *et al*. (2018) suggests that the distributional range of *C. rubrum* within the Mediterranean Sea could reach Anamur, in the Levantine Sea (Turkey), as the eastern point of its distribution.

Knowledge on the bathymetric limits of Mediterranean red coral distribution has been greatly expanded: from 350 to 1 016 metres depth in the Central Mediterranean (Straits of Sicily and Malta waters) (Costantini *et al*., 2010; Freiwald *et al*., 2009; Taviani *et al*., 2010; Knittweis *et al*., 2016; see details in Annex 1).

Several studies (see the following sections) agree on the occurrence of different typologies of *C. rubrum* populations:

- shallow-water populations (depth range 10–50 m), dwelling in caves, crevices, on overhangs and protected interstices. Many of them are reported to be overexploited, as a result of the pressure exerted by SCUBA diving since the 1950s, which enables the picking of red coral in areas inaccessible to dredges. According to Recommendation GFCM/35/2011/2 Paragraph 4, shallow-water populations are now fully protected from exploitation, to allow for their recovery.
- deepwater populations (depth range of 50–130 m), typically dwell on open surfaces. Commercial harvesting is now focused on these populations. In the past, they were harvested by the use of dredging gears (now forbidden), but nowadays divers can reach depths of 100–130 m.
- deepest-water populations (depth range 130–1 000 m) are poorly known. No legal harvesting is undertaken on these populations. The lack of knowledge on the distribution and demography of these population represents a major gap; however, the only available information indicates that these populations are very sparse with low densities; they therefore do not representing a profitable resource for exploitation, which in any case is also limited by the maximum SCUBA depth of 100–130 m.

In the Mediterranean, *C. rubrum* inhabits subtidal rocky substrates and is one of the dominant and important components of Mediterranean “coralligenous” species assemblages (Ballesteros, 2006), coexisting with other gorgonians, large sponges and other benthic invertebrates. The coralligenous biocoenosis that hosts most of red coral populations is classified as a ‘priority habitat for conservation’ – in other words a habitat whose conservation is required because of its vulnerability, heritage value, rarity, aesthetic and economic values (Relini and Giaccone, 2009). More generally the coralligenous outcrop habitats represent: 1) a hotspot for species diversity, considered a benthic habitat of high conservation interest (Ballesteros, 2006; UNEP/MAP, 2009), and 2) areas of special concern, as they are increasingly enduring a range of anthropogenic disturbances. In particular, gorgonian corals play a very important role in the coralligenous biocoenosis as ecosystem engineers by providing structural complexity and biodiversity. *C. rubrum* is also found outside the coralligenous biocoenosis, in semi-dark caves and upper enclaves (facies with *C. rubrum*).

2.2.2.2. Pacific Ocean

Within the Pacific context, commercially valuable species of Coralliidae are distributed across two depth zones, 50–400 m and 1 000–1 500 m (Bruckner, 2016) in the waters around Hawaii, Japan, the Philippines and Taiwan Province of China (Annex 2, Table 1).

P. konojoi (white coral) and *P. elatius* (pink coral) occur from 50 to 320 m, in southern Japan to the north South China Sea and Viet Nam. One of the largest *P. elatius* specimens was harvested off Okinawa in 2006; it measured 1.1 m in height and 1.7 m in width and weighed 67 kg (Tsounis *et al*. 2010; Iwasaki and Suzuki, 2010).

C. japonicum (Japanese red coral) inhabits waters from 75 to 300 m on rocky bottoms from southern Japan to the north South China Sea and the Philippines. Nonaka and Muzik (2009) report that these species all live in deeper habitats in the southern sea areas of the Ryukyu Archipelago (Amami, Okinawa, Ishigaki) as compared to northern areas (off Kagoshima). Pink corals, on the other hand, grow in colonies that are larger in size in southern compared to northern sea areas (Annex 2).

Two further species, possibly collected by Taiwanese fishermen (see following paragraphs and Annex 2) have very limited distribution: *P. carusrubrum, which* is currently only found off Pengjia Islet, the top of a seamount off northeastern Taiwan Province of China, at depths of 120–180 m; while *Hemicorallium sulcatum* (miss coral) is known exclusively along the Boso Peninsula in the Chiba Prefecture, Japan and Taiwan Province of China at depths of 180–550 m (Tu *et al*., 2012).

Pleurocorallium secundum (Midway coral) is found on flat exposed substrata throughout the Hawaiian archipelago at depths from 160–600 m (Parrish *et al.*, 2017), with the largest known populations found in Makapu'u bed off Oahu, and smaller populations identified in at least 16 other locations (Bruckner, 2016). This species is reported to have been collected off Lanyu Island (Taiwan Province of China) (Huang and Ou, 2010; Fisheries Agency, Council of Agriculture, Executive Yuan, Taiwan Province of China) however further investigation is required since no scientific identification has been provided and this is the only reported occurrence of the species in Chinese waters. This raises potential doubts over the legitimacy of its inclusion in CITES Appendix III by China in 2008 (See Annex 2 and IWMC 2009).[9]

Hemicorallium regale (garnet coral, possibly confused in landings with *H. laauense* – see following sections) is known to co-exist with *P. secundum* in the Hawaiian archipelago and United States Pacific Islands, at depths of 365–719 m (possibly up to 1 815 m). It prefers encrusted, uneven rocky bottom habitats, with no shelf areas (< 400 m depth), off populated islands, and substrata periodically covered with sand and silt (see Tsounis *et al*., 2010 and references therein).

Corallium sp. nov. (Midway deep-sea coral) is found on the Emperor Seamounts and Midway at depths of 900–1 500 m.

Despite the wide distribution area of Pacific precious corals, there is little information available to date on habitats and ecological gradients driving the distribution of the species that are currently commercially exploited. Most of these data are limited to pinpoint surveys conducted in Japanese waters (e.g. Okinawa and Ogasawara Islands and Kochi Prefecture), that do not offer a comprehensive picture of the distribution of precious corals, and are even further from providing a clear picture of resource status. Surveys concerning the distribution of precious corals (including density and biomass) are essential for the management of the species, so it is desirable that each sea area is investigated more thoroughly (see Annex 2).

[9] Letter from IWMC to CITES Secretariat (2019).

2.2.3. Reproduction (Coralliidae)

2.2.3.1. Mediterranean Sea and Atlantic Ocean

Many aspects of the reproductive biology of *C. rubrum* have been studied, but there are still many knowledge gaps and uncertainties that prevent scientists from fully understanding all the factors that drive species reproduction. To date, except for the historical work of Lacaze-Duthiers (Lacaze-Duthiers, 1864), few reports on red coral reproduction have been published for shallow-water colonies (Santangelo *et al*., 2003; Torrents *et al*., 2005; Tsounis, 2005; Tsounis *et al*., 2006a, b; Vighi, 1970, 1972; Weinberg, 1979). Since 2013, new knowledge on the fecundity of deep colonies has been made available for Italy and Spain (Porcu *et al*., 2017; Priori *et al*., 2013; Santangelo *et al*., 2015).

Recently, a genetic study based on single nucleotide polymorphisms (SNPs) demonstrated the occurrence of an XX/XY genetic sex-determination (GSD system) in *C. rubrum*: this was is the first record for non-bilaterian species (Pratlong *et al*., 2017).

C. rubrum has a limited capacity for asexual reproduction. The sexual status of the population appears completely gonochoric at both the colony and the polyp level (Porcu *et al*., 2017; Santangelo *et al*., 2003; Tsounis *et al*., 2006; Vighi, 1970, 1972). Red coral is an iteroparous species that undergoes internal fertilization and broods larvae internally (planulator). Gonadal development follows an annual cycle with a synchronized release in summer (Porcu *et al*., 2017; Santangelo *et al*., 2003; Tsounis *et al*., 2006a; Vighi, 1970). The embryonic period lasts about 30 days (Lacaze-Duthiers, 1864; Vighi, 1970). Colonies do not fuse together (in nature) and each adult colony therefore is likely to have originated from a singular planula (Stiller *et al*., 1984; Weinberg, 1979). Larvae exist in the water column for a period of a few hours to days before settling near the parent's colonies. It is therefore very likely that single populations are genetically isolated.

Martínez-Quintana *et al*. (2015) studied the pelagic larval duration (PLD), buoyancy and larval vertical motility behaviour. PLD ranged from 16 days (95 percent survival) to 42 days (5 percent survival). Larvae exhibited negative buoyancy with a free-fall speed decreasing linearly with age, at a velocity varying from -0.09 ± 0.026 cm/s on day 1 to -0.05 ± 0.026 cm/s on day 10. *C. rubrum* larvae maintained active swimming behaviour for 82 percent of the time. This larval motility behaviour, combined with the extended PLD, confers on *C. rubrum* larvae an unsuspectedly high dispersive potential in open waters.

Data on fecundity is summarized in Annex 1, Table 2. Fertile female polyps of shallow water colonies produced 1–4 mature oocytes per year, while each fertile male polyp produced 6 ± 3.5 spermiaries on average (Santangelo *et al*., 2003). A range of 1–11 sexual products is registered in the Sardinian seas per year (Porcu *et al*., 2017). Female fecundity and fertility are constant over time, but decrease after March, despite the fact that oocytes do not turn into planulae during this period (Santangelo *et al*., 2003). Female fecundity ranges of 0.05–3 mature oocytes per year have been found for deepwater colonies in the Tuscany Archipelago (Priori *et al*., 2013), while average fecundity values of 0.87 and 2.09 ± 0.65 of female sexual products per polyp have been recorded for deep colonies caught in Spain and Sardinia respectively (Porcu *et al*., 2017; Santangelo *et al*., 2015).

Size has a positive effect on reproductive potential. Reproductive output increases exponentially with colony size. Generally, large colonies contain more oocytes and sperm sacs and may produce a hundred or more planulae, as compared to the small ones (some tens of planulae) (Santangelo *et al*., 2003; Torrents *et al*., 2005; Tsounis *et al*., 2006a). The actual age of first reproduction is probably 7–10 years, corresponding to colonies about 24 mm in height, 3.6 mm in basal diameter and 0.6 g in wet weight (Torrents *et al*., 2005). Female colonies reach fertility at a minimum age of approximately 10 years (corresponding to a diameter of 2 mm), although reproductive parameters (percentage of fertile colonies and fertility) are significantly lower in small than in medium and large colonies (Torrents *et al*., 2005). Male colonies, on the other hand, may develop gametes much earlier. The youngest fertile male colony of the sample with a basal diameter of 1.2 mm is estimated to be no older than six years (Gallmetzer *et al*., 2010).

Priori *et al*. (2015) quantified the effects of the partial predation exerted by the gastropod *Pseudosimnia carnea* on the reproductive features of *C. rubrum*: colony fecundity was reduced by 81 percent with a consequent reduction in population reproductive output and thus limited resilience to intense commercial harvesting.

2.2.3.2. Pacific Ocean

While many aspects of Mediterranean red coral have been investigated in recent decades, very little is known about basic life history of most species of Pacific precious corals.

Kishinouye (1904) examined eggs and spermatids of Japanese red coral, collected in March and in September. The eggs and spermatids in March were larger in number and bigger in size than in September, leading to the conclusion that the reproduction season for the red coral was spring.

Nonaka *et al*. (2015) investigated three species (*C. japonicum*, *P. elatius*, and *P. konojoi*) from the Ryukyu Archipelago in Japan; they all shared a sexual reproduction strategy described as gonochoristic broadcast, spawning during summer, mostly from May to August. The gonads are formed in the siphonozooids, with the fertilisation external and the oocytes and sperm discharged into the sea. Planula larvae have not been observed in these species (Nonaka *et al*., 2015; Sekida *et al*., 2016). In Mediterranean red corals (*C. rubrum*), the gonads are formed in the autozooids. Fertilization takes place in the autozooids of female colonies, after which planula larvae develop and are released into the sea.

Sekida *et al*. (2016) developed a method of using a liquid chelating agent to decalcify the Japanese red coral *C. japonicum* axis and sclerites, rendering the coenenchyme semi-transparent and enabling the observation of the gonads. Using this method to measure the size of the gonads of Pacific red corals – collected monthly from 2006 to 2014 from the waters off Kochi – revealed that the gonads are formed during the period from March to May. Also, since the number of colonies with gonads was seen to decrease sharply from July onwards, they are thought to release their eggs and sperm all at once in July. In consideration of the useful findings documented in this study, since March 2012, the government of Kochi Prefecture has prohibited coral fishery from June and July in order to protect precious corals during the spawning season (Annex 2).

The reproduction season of *C. japonicum*, *P. elatius*, and *P. konojoi* around Okinawa is estimated to be from May to August (Nonaka *et al*., 2015). On the other hand, while the exact location is unknown, Kishinouye (1904) reports that the eggs and seminal vesicles of mature red corals collected in March were larger and more numerous than of those collected in September. Such heterogeneity calls for the close examination of each sea area, since it indicates that the timing of the reproduction season may show different patterns according to the geographical area. In addition, effective resource management requires data concerning the age at which corals attain sexual maturity; this is currently lacking, in addition to date on the relation between the size of a colony and its level of maturity.

In the Hawaiian archipelago, a few studies have been undertaken on the most accessible coral beds (Makapu'u population). *P. secundum* and *H. regale* are described as gonochoristic (Waller and Baco, 2007), and are estimated to reach reproductive maturity at 12–13 years, as determined from growth ring studies (Grigg, 1993). However, this method, based on the assumption that rings were deposited annually, was contested by subsequent radiometric studies (Roark, 2006 and references therein), which indicate much lower growth rates than those inferred from growth ring size relationships for Hawaiian *P. secundum* (Grigg, 1976, 2002). These appear to underestimate the age of larger individuals by at least a factor of two (Roark, 2006).

2.2.4. Recruitment (Coralliidae)

Recruitment is one of the main processes determining both population structure and dynamics (Caley *et al.,* 1996). The rate of recruitment can vary greatly in relation to space, time and habitat. It could be different if it occurs in natural or artificial substrates. There are few studies on the recruitment process in *C. rubrum* populations (Annex 1, Table 3).

Saturation of space and interference competition appears to be a major controlling factor in red coral recruitment (Garrabou *et al.,* 2001; Santangelo *et al.,* 1988). This could explain why recruitment rates were higher after harvesting events (Linares *et al.,* 2000; Santangelo *et al.,* 1997).

Similar studies are not known for the Pacific precious corals.

However, within a research project focused on breeding techniques, the settlement process was studied using tiles made of vinyl chloride or ceramics, placed in a sea area inhabited by precious corals for over a year. Unfortunately, no precious coral larvae settlement was observed, but large numbers of hydrozoan corals were found to have settled on vinyl chloride tiles (Annex 2).

2.2.5. Growth and mortality (Coralliidae)

2.2.5.1. <u>Mediterranean Sea and Atlantic Ocean</u>

C. rubrum is long-lived, but exactly how "long" its lifespan can be is the subject of controversy and research. Until now, three different methods have been applied to determine the age of coral colonies:

- the petrographic method (Abbiati *et al*., 1992; Garcia-Rodriguez and Massò, 1986a; Santangelo *et al*., 1993);
- the organic matrix staining of thin sections from the base of colonies (Marshal *et al*., 2004), which allows one to read annual growth rings;
- direct measurements of newly settled colonies of known age, a non-destructive approach to the study of colony growth rate, based on artificial or semi-natural substrates on which new settled colonies can be followed during their growth for a determined time interval (Bramanti *et al*., 2005; Cerrano *et al*., 1999; Garrabou and Harmelin, 2002).

Unfortunately, the dark bands highlighted with the first method (petrographic method) were not annual, so colony age was underestimated. A study comparing the petrographic method of growth ring counting to that of organic growth rings in *C. rubrum* from the Mediterranean, showed that the petrographic method significantly underestimated, by as many as 10 years, the known age samples (20 years old; Marschal *et al*., 2004). The counting of growth rings from the staining of the organic matrix underestimated the known age by 3–4 years, and resulted in growth rates ranging from 140 to 750 µm/year (mean of 340 ± 150 µm/year) (Marschal *et al*., 2004),

Only the "organic matrix staining method" allowed growth ring readings that were proved to be annual by calcein labelling in vivo (Caley *et al*., 1996). In general, colonies exhibit low growth rate that varies between locations, depths, and habitats (Abbiati *et al*., 1992; Bramanti *et al*., 2005; Cerrano *et al*., 1999; Garcia-Rodriguez and Massò 1986a; Garrabou and Harmelin, 2002) (see Annex 1, Table 4).

Galli *et al.* (2016) applied a mechanistic numerical model to describe the growth of a *C. rubrum* colony (polyp numbers, polyp and gametes biomass, skeletal inorganic and organic matter) as a function of food availability and seawater temperature. They found that large colonies are more likely to undergo negative growth in shallow waters, where they experience both high temperatures and food shortage at the same time during late summer and are, thus, less likely to be found near the upper distribution limits of *C. rubrum*. In deeper waters, larger colonies may have less constraints as a result of relatively stable temperature and food input. If this temperature effect on size distribution is superimposed on the harvesting effect, the occurrence of large colonies could be further limited to deeper waters as mean sea temperatures increase. Increasing trends of harmful heatwave events would also increase this risk (Galli *et al.,* 2016).

After settlement, the growth rate of shallow water colonies is about 1 mm/year for base diameter, and 10 mm/year for height (Cattaneo-Vietti and Bavestrello 1994). However, after 4–5 years, growth virtually stops and become negligible (Bavestrello *et al.,* 2010). The growth rate of deepwater colonies seems to be narrower (from 0.21 to 0.26 mm) (Benedetti *et al.,* 2016; Priori *et al.,* 2013). Similar value of growth rate has been registered (0.23 mm/year) in Atlantic deep red corals dwelling at 60–100 m depth in southern Portugal (Boavida *et al.,* 2016).

Natural mortality of *C. rubrum* arises from competition over space with sponges and other sessile biota, dislodgement from the substrate due to the action of boring species (Harmelin *et al.,* 1984) or seismic (volcanic) movement (Di Geronimo *et al.,* 1994), predation by the small gastropod *Pseudosimnia carnea* and the crustacean *Balssia gasti* (Abbiati *et al.,* 1992), as well as sedimentation increase and ocean acidification.

Phenomena of mass mortality events have been observed in shallow-water populations since the late 1990s, including several mass-mortality events linked to elevated temperature anomalies (Bramanti *et al*., 2005). A mass mortality event occurred in the northwestern Mediterranean, from Tuscany (Italy) to Marseille (France), in summer 1999. The phenomenon, in which about 80 percent of the colonies were affected, was attributed to a fungal and protozoan disease, and linked to temperature anomalies (Cerrano *et al*., 2000; Garrabou *et al.,* 2001; Perez *et al.,* 2000; Romano *et al*., 2000). In the same year (late summer 1999) some shallow-water red coral populations were affected by mass mortality associated with anomalous temperature increases in the eastern Ligurian Sea (Calafuria, Italy; Bramanti *et al*., 2005), as well as the western Ligurian Sea.

Mortality in *C. rubrum* had a different impact, depending on the colony size. Large colonies are more resilient to natural stressors. On the contrary, small colonies prevalently suffer from higher virulence, showing higher whole-colony mortality rates. However, further studies are urgently required to provide basic information regarding red coral population dynamics as a basis for the hypothesis on the actual recovery capability of affected populations.

Recently, *C. rubrum* deepwater mass mortality events have been described occurring in the Gulf of Naples (Italy). It has been suggested that the mortality is due to: 1) formation of local down-welling currents inducing an unusual drop in the thermocline; 2) sudden warm water emissions (sulphur springs) in an area characterized by important volcanic activities; or 3) local landslides generating turbidity currents along the steep slopes (Bavestrello *et al*., 2014).

Many species (mainly sponges, crustaceans, brachiopods, molluscs and echinoderms) have been documented as living on or in strict association with red coral colonies (Calcinai *et al.,* 2010; Crocetta and Spanu, 2008). Some of these species, especially the parasitic ones, may increase the red coral mortality rate and therefore profoundly affect its population structure (Corriero *et al.,* 1997). Apart from being the main causes of natural mortality, sponges have the ability to damage colonies and thus reduce the commercial value of red coral (Calcinai *et al.,* 2010).

2.2.5.2. Pacific Ocean

Precious corals in the family Coralliidae will form large bushes when undisturbed for long periods, and all species exhibit slow growth. They are long-lived organisms with low growth rates. *P. elatius* is one of the largest species in the family, achieving a maximum size of 1.1 m in height and 1.7 m in width and a weight of 67 kg (Tsounis *et al.*, 2010; Iwasaki and Suzuki, 2010). *P. konojoi* is one of the smallest commercially harvested coral in this family, with a maximum height of only about 30 cm, while *P. secundum* and *C. japonicum* have intermediate heights (maximum 80 cm and 100 cm respectively; see Bruckner, 2016 and references therein).

With respect to Pacific coralliids, to date, three methods have been used to estimate the growth rate of colonies in *P. secundum, C. japonicum*, *P. elatius* and *P. konojoi*: high-precision analysis of the colours of the growth rings in a transverse section of the axis using a high-resolution digital microscope (Luan *et al*., 2013); analysis of the two-dimensional distribution of infrared absorption spectra using high-luminance infrared synchrotron radiation microbeams (Iwasaki *et al*., 2014); and age determination using the naturally occurring radionuclide lead-210 (210Pb) (Hasegawa and Yamada, 2010) or ^{14}C dating method (Roark *et al*., 2006).

Annex 2, Table 2 shows the results of these methods. There is different growth depending on coral species, habitat and environmental condition.

Diametric and linear growth rates (DGR and LGR) of three species (*C. japonicum*, *P. elatius*, *P. konojoi*) were determined based on annual growth rings in unstained slabs. LGRs of *C. japonicum* were 2.22–5.82 mm/year on average, and 2.76 mm/year for *P. elatius*, with a descending pattern from the tip to the base. *P. konojoi* did not show the same pattern, and it was much faster (7.15 mm/year) than the other three species studied (Luan *et al*., 2013), *P. secundum* was estimated to increase 9 mm in height per year using a similar petrographic method (Grigg, 1976).

As concerns the DGRs, *C. japonicum* showed the slowest growth rate (0.2–0.27 mm/year) compared to *P. elatius* (0.30 mm/year) and *P. konojoi* (0.44 mm/year) (Luan *et al*., 2013). Using the infrared radiation method, the DGR was calculated for *C. japonicum* (0.22 mm/year), *P. elatius* (0.294 mm/year) and *P. konojoi* (0.218 mm/year) (see Iwasaki *et al*., 2014). More accurate radiometric studies have demonstrated similar growth rates for *P. secundum* (0.34 mm/year; Roark *et al*., 2006), *P. elatius* (0.3 mm/year) and *P. konojoi* (0.26–0.66 mm/year; Hasegawa and Yamada, 2010; Yamada unpublished).

The DGR calculated for *C. rubrum* are quite variable, depending on location and depth, but also on the method used (see previous paragraph). In general, it seems that the DGRs of the Pacific precious corals are slower than the Mediterranean counterpart, while their LGRs can be faster.

In Japan and Taiwan Province of China, scientific research studies on mortality of precious coral resources are long overdue; particularly for those data (e.g. biomass, recruitment, mortality) necessary to construct a population dynamics model to estimate future resources and landings is almost non-existent, with only growth rate as the uniquely defined parametre (Annex 2).

2.2.6. Population structure and density (Coralliidae)

Data on population structure are the most useful data in monitoring population status, and in identifying changing proportions of mature/immature colonies, which is more functional as a basis for management decisions that need to ensure minimum recruitment, especially for sessile animals that require a certain density to ensure fertilization success. Drastic changes induced in pristine shallow populations when larger/older colonies have been selectively removed led to shifts towards higher densities and smaller size. One possible cause of such shifts could be the inhibitory effect exerted by larger/older colonies on larval settlement and settler growth (Bramanti *et al.,* 2009; Angiolillo *et al.,* 2016).

On the contrary, numbers of colonies per unit area are unlikely to provide an indication of the population status or trends. This is because these measures differ depending on how they are assessed (colony density measured over the entire suitable habitat is much less than the density of small patches occupied by the coral within this habitat), and the life-stage of the population. Furthermore, shifts in the size structure of populations due to fishing pressure can be directly compared, while density and abundance cannot.

2.2.6.1. Mediterranean Sea and Atlantic Ocean

While the populations living in the shallower portion of the species' distribution range have been the focus of several previous studies, the knowledge of the deepwater (50–130 m) commercial ones is improving in recent years.

In general, at shallower locations (10–50 m depth) the population structure is significantly different from those at deeper locations, which is also the result of being exposed to heavy harvesting pressure.

Populations studied along the Costa Brava (Spain) were mainly composed of colonies smaller than 8 cm, with a mean basal diameter of about 5 mm (Tsounis *et al*., 2007; Bramanti *et al.,* 2014), below the smallest commercial size (7–8 mm). Therefore, the present-day shallow-water red coral populations appear to be greatly compromised, considering that in the 1960s colonies as large as a man's palm could still be found at 35 m depth (Tsounis *et al*., 2006). A similar situation was observed along the Tuscany coast (Santangelo *et al*., 2007).

A study (Linares *et al*., 2010a) documented that the demographic structure of red coral populations from three of the oldest Mediterranean MPAs (all in France: Scandola Nature Reserve, Cèrbere-Banyuls Nature Reserve and Carry-le Rouet Marine Protected Zone) has changed with time and it is now significantly different from unprotected populations (Linares *et al,*. 2010a). Within the MPAs the size values are higher than those reported for most of the shallow populations and deep-dwelling populations. When analysing the mean basal diameter and colony height for the 30 largest colonies at each location, differences between harvested and non-harvested populations were found. Regarding the basal diameter, the values found for the three MPAs (ranging from 9.8 to 18.6 mm) showed a high mean diameter (18.6 mm) than harvested populations (range 5.4–13.7 mm; Linares *et al*., 2010a). Similarly, the mean colony height of the largest colonies found for the three MPAs (ranging from 117.1 to 130.5 mm) were higher than the values reported for harvested populations (from 38.5 to 98.3 mm, Linares *et al*., 2010a). Differences in the observed size distributions are more closely related to the structure at the beginning of the reserve than to the number of years of protection. Despite these positive effects, colony sizes did not reach characteristic values of presumably pristine populations estimated from museum specimens (Garrabou and Harmelin 2002),[10] most likely as a result of other impacts such as poaching and diving, which do not allow their total recovery (Linares *et al*., 2003). The percentages of colonies with a basal diameter greater than 7 mm or colony height greater than 100 mm has been proposed as a useful descriptor for evaluating the conservation status of each population.

According to scientific surveys in Spain (GFCM, 2017) and Croatia (AC29 Doc.22 and its annexes), intensive harvesting of red coral shallow beds has resulted in a decline of almost 75–90 percent of the populations. In Costa Brava (Spain), where 80 percent of the harvesting concentrates at less than 50 m of depth, the latest scientific studies showed that the average size of the basal diameter of colonies varied from 5 to 8 mm and their height ranged from 3 to 6 cm, leading to the adoption of stricter management measures: halving the fishing effort in 2017 and imposing a ten-year moratorium on harvesting in the internal waters of Catalunya from 2018 (GFCM, 2017). According to the GFCM Recommendations (see also the section on Management for details), the populations from 0 to 50 m

[10] Seventy-five colonies, belonging to a private collection collected in 1962 in shallow-water habitats (25–35 m depth) on the northern face of the Cap de Creus (northeast Spain) were considered as a proxy of presumably pristine populations. Mean maximum basal diameter for these "pristine" non-harvested populations ranged from 1.16±0.28 to 1.60±0.39 cm, while mean maximum height ranged from 11.45±2.46 to 11.81±1.9 cm. In harvested populations the mean values of basal diameter and height were lower, ranging from 0.56±0.09 cm to 0.71±0.13 cm and 5.99±1.07 cm to 9.72±1.7 cm, respectively.

of depth should have already been protected from harvesting, but the national management measures have not been updated in all countries to fully protect shallow populations (GFCM, 2017) or have used existing derogations to allow harvesting at less than 50 m of depth (Annex 1).

In deepwater locations, the percentage of large colonies exceeding the legal minimum size of 7 mm of basal diameter is larger, but varies between studies and locations. For instance, 46–79 percent were large enough to be legally harvested in Cap de Creus, while only 9–20 percent of the shallow water colonies met the 7 mm legal basal diameter to be collected (Rossi *et al*., 2018). The branching pattern was also better developed in deeper colonies, as up to 16 percent of the colonies showed fourth-order branches, compared to less than 1 percent of the shallow-water colonies (of which 96 percent consisted of only one single branch) (Rossi *et al*., 2008).

C. rubrum populations from the western Mediterranean (north and central Tyrrhenian Sea) from 50 to 130 m depth, despite being historically and recently subjected to red coral commercial exploitation, contained a high percentage (38–46 percent on average) of harvestable colonies (> 7 mm basal diameter) corresponding to about 30 years and a maximum life span of 93 years for the oldest specimen in the study, with some colonies with fifth-order branches (Priori et al. 2013; Angiolillo *et al*., 2016).

In deepwater commercial banks from the central-western coast of Sardinia (western Mediterranean), large colonies (> 10 mm basal diameter) represented a big portion of the population (> 38 percent of the total), suggesting that harvesting effort did not affect the maximum size of colonies yet (Follesa *et al*., 2013). However, the population structures of *C. rubrum* appeared statistically different off the northern coast of the island, at depths of 84–93 m, in populations that in the past were exposed to considerable human impact. This area is characterized by a near absolute dominance of small colonies, with only 10 percent of large colonies (> 10 cm in height). Finally, an intermediate structure has been found on the southwestern coast at depths of 85–107 m; the colonies found do not exceed 10 cm in height, with only 17 percent of all colonies belonging to the height class of 10–15 cm (Cannas *et al*., 2011).

In a recent survey in Tunisian waters (Jaziri *et al*., 2017), from 44 to 83 m depth, basal diameters were comparable to those of Sardinian colonies harvested below 80 m depth (Cannas *et al.,* 2011). Red coral colonies above 50 m depth in Bizerte and Tabarka/Sidi Mechrig have the lowest basal diameter (6.2 ± 1.6 and 6.57 ± 0.9 mm, respectively) whereas at more than 50 m depth, including in Esquerquis benches, the red coral colonies have a largest basal diameter with an average of 8.83±3.2 mm. For all Tunisian northern coasts, fishermen consider that: 1) red coral is not exploited (virgin) in 20 percent of the sites, overexploited in 10 percent of the sites and that the exploitation is optimal in 70 percent of the sites; 2) more than 80 percent of harvested colonies have mean or bigger sizes; and 3) the density of colonies is medium-to-high in about 60 percent of exploited zones (Gaamour in GFCM, 2017). However, no scientific criteria were provided on when/how the exploitation was considered ‘optimal’.

The density of red coral population varies from place to place, according to depth and exploitation (Rossi *et al*., 2008; Tsounis *et al*., 2006a). Schematically, we can distinguish two different spatial situations: 1) coastal populations, occurring up to 50 m depth, characterized by high density (up to 1 000 colony/m^2) and small colony size (up to 5 cm high); 2) deeper populations, extending up to 200 m depth and more, characterized by low density and high colony size (Annex 1, Table 5 to Table 8). At this depth range, colonies are characterized by more extensive branching patterns (Santangelo *et al*., 2007) forming small aggregates on individual banks and hard ground areas, where colonies are concentrated on the exposed surface facing high-current areas (Cannas *et al*., 2010; Rossi *et al*., 2008). The situation in the 200–1 000 depth range is far less known; at present sporadic point records have been described at the maximum depths in the Sicilian Channel and off Maltese waters (Freiwald *et al*., 2009; Costantini *et al*., 2010; Taviani *et al*., 2010; Knittweis *et al*., 2016).

Nevertheless, ROV surveys showed that populations dwelling from 80 to 170 m depth can exhibit a continuous range of population density (from 2 to 75 colonies per 0.25 m^2) (Cau *et al.,* 2016; Cau *et al.,* 2015a; Cau *et al.,* 2017; Cau *et al.,* 2015b). Young populations composed of small and dense colonies dominated along rocky vertical walls, whereas mature populations characterized by large and sparsely distributed colonies were found only in horizontal beds not covered by sediment. An inverse relationship between maximum population density and mean colony height was found, suggesting that self-thinning processes may shape population structure. It has been suggested that, in the long term, shallow protected populations should resemble present deep populations, with sparsely distributed large colonies (Cau *et al.,* 2016).

The recent discovery of an exceptional red coral population from a previously unexplored shallow underwater cave in Corsica (France), characterized by the coexistence of a relatively high abundance of small (< 3 cm) colonies with large colonies (> 10 cm in height) older than 100 years, challenge current assumptions on red coral populations. Prior to intense exploitation, red coral may have lived in relatively high-density populations with a large proportion of centuries-old colonies, even at very shallow depths (Garrabou *et al.,* 2017).

2.2.6.2. Pacific Ocean

There are numerous gaps in the knowledge of the density of precious coral species in the Pacific Ocean.

Deep coral research in Hawaii highlighted that in 1970, approximately four years after the discovery of *P. secundum*, the Makapu'u Bed off Oahu contained the largest known population of *Corallium* in the United States Pacific, with colonies occurring over an area of about 3.6 km^2. The density and size structure of this population was first characterized in 1971, following three years (1966–1969) of non-selective fishing (1 800 kg of *P. secundum* were removed using a coral tangle dredge). The area was resurveyed in 1983 and 1985, following six years of manned submersible harvest (total harvest = 6 200 kg). This coral was found to have density of 0.02 colonies/m^2 in 1971 and 0.022 ± 0.01 colonies/m^2 in 1985. Additional surveys in 2001 found that the bed was larger than previously estimated (4.3 km^2) and corals occurred at a higher density (0.3 colonies/m^2). There was a slight shift in the age-frequency distribution. A positive result, from a management standpoint, was an increase in the percent of older year classes of *P. secundum* in 2001 compared to surveys in 1971 and 1983 and 1985 (Grigg, 2002). Year class groups of 20–25, 25–30, 30–35, 35–40, and 40–45 years all increased in relative abundance over the same year classes in 1983, showing substantial recovery from the harvest years in the 1970s. Even though all year class groups in 2001 had not fully recruited (e.g. year classes 45–50 and 50–55 were still under-represented), the medium-sized colonies in 2001 were relatively more abundant and present in greater density compared to 1971, making the overall biomass in the bed 45 percent greater in 2001 than 1971. In terms of biomass, recovery in 2001 exceeded 100 percent. This result is due to the greater percent of older and larger colonies in the population's age–frequency distribution of (Grigg, 2002).

In the 1970s, the act of harvesting colonies may have triggered the release of larvae, or, alternatively, and perhaps more likely, the harvest (removal) of large coral colonies from the population may have caused their natural predators to disperse, thereby lessening mortality especially to young colonies. In other words, *P. secundum* may have increased in abundance due to increased survival of recruits in the 1970s as a result of reduced predatory pressure (Grigg, 2002).

The total number of colonies in characterized beds in the United States of America is low including an estimated 2 500 legally sized colonies of *H. laauense* (*H. regale*) at Cross Seamount and up to 7 000 legally sized at Keahole Point Bed (Bruckner, 2009). Available data suggest individual species have small effective population sizes and a restricted distribution within a larger geographic range.

Less information is available from the Asian Pacific area. Iwasaki *et al*. (2012) investigated and compared the age structure of both harvested and unharvested patches of *C. japonicum* around Amami Islands (Kagoshima Prefecture, Japan). In the harvested population, the estimated modal ages are 10–20 years, while unharvested population extends widely from 20–70 years, with a main mode from 20–40 years, and a small but distinct secondary mode from 50 to 60 years. According to the authors, the difference in the modes of non-harvested and harvested populations suggests that harvested populations reach a harvestable size after at least a 10–20 year period of biological rest.

From 2009 to 2012, 20 sites off Kochi, Kagoshima and Okinawa were investigated by means of ROV to document the distribution and density of *C. japonicum*, *P. elatius* and *P. konojoi*; the reported density was 0.45 colonies per 100 m^2, 0.19 colonies per 100 m^2, and 0.07 colonies per 100 m^2 respectively, and the average of the three species combined was 0.74 colonies per 100 m^2 (Annex 2).

No data on abundance, density or size structure are available for Coralliidae in the waters of Taiwan Province of China.

2.2.7. Genetics and genetic stock identification (Coralliidae)

2.2.7.1. <u>Mediterranean Sea and Atlantic Ocean</u>

The bulk of available genetic knowledge refers to the more accessible shallow-water populations of *C. rubrum* (from 15 to 60 m of depth) (Abbiati *et al*., 1992, 1993, 1997; Aurelle *et al*., 2011; Calderon *et al*., 2006; Casu *et al*., 2008; Costantini *et al.*, 2003, 2007a,b, 2011; Del Gaudio *et al.,* 2004; Ledoux *et al*., 2010a,b). These studies found evidence of breeding isolation and population sub-structuring suggesting that larval dispersal may not be able to ensure sufficient gene flow to preserve the species' genetic homogeneity. Several studies confirmed the occurrence of genetic differentiation at spatial scales of ten of metres (that is the effective larval dispersal range may be restricted to < 10 m). The strong genetic differentiation between nearby samples implies that the recovery of overexploited populations must be mainly due to self-recruitment. Moreover, a pattern of Isolation by distance was described in Ledoux *et al*. (2010b), i.e. the more distant, the more different populations are. All studies revealed, using differentially powerful markers, significant deviations from Hardy-Weinberg equilibrium due to elevated heterozygote deficiencies, consistent with the occurrence of inbreeding (mating between consanguineous colonies).

For deepwater *C. rubrum* populations, the number of genetic studies is limited, but has been growing in recent years. Costantini *et al*. (2011) analysed colonies along a depth gradient (from 20 to 70 m) which showed strong patterns of genetic structuring among the samples both within and between two study sites (Catalan and Ligurian Sea), with a pattern of reduction in genetic variability with depth. A threshold in connectivity was observed among the samples collected across depths of 40–50 m, supporting the hypothesis that separations between shallow- and deepwater red coral populations occur. This finding could have major implications for management strategies and the conservation of commercially exploited deep red coral populations.

In Sardinian populations (range from 80 m to 120 m of depth) high levels of genetic differentiation were measured, over different spatial scales from hundreds to less than 1 km (Cannas *et al.*, 2010; Cannas *et al.,* 2011; Cannas *et al.,* 2015; Cannas *et al.,* 2016). Results indicate the existence of strong genetic differentiation among populations over the different depths (banks from 30/60 m vs. banks from 80/120 m), underlying possible restrictions to gene flow along the depth gradient (Cannas *et al.,* 2016). The genetic results indicated that for precious coral populations in Sardinia the 'deep reef refugia hypothesis', which envisages the capacity for deep corals to act as seed banks for the shallower impaired (over-harvested) populations, is not supported (Cannas *et al.,* 2016).

The highest genetic diversity recorded in Sardinia for all areas and depths with respect to other Mediterranean areas indicates that strict local management has been effective, since harvesting has not yet led to a substantial erosion of the genetic pool. Possible causes for the high levels of diversity observed in Sardinia are in relation to hydrological conditions, its geographical position and its proximity to some putative glacial refugia (Cannas *et al.,* 2016).

Strongly differentiated red coral populations were described in the Tyrrhenian with respect to the Algero-Proveṇal Basin (Cannas *et al.,* 2015). From a management perspective, the high genetic diversity and demographic stability recorded indicate that Sardinian deep red coral populations are stable and still sustainably exploited. Nevertheless, the significant spatial structuring at the scale of less than 10 km indicates that they are sustained largely by local recruitment, excluding the possibility that they can help in recovering shallower banks (Cannas *et al.,* 2015).

The genetic spatial structuring of deepwater *C. rubrum* colonies found in the 58/118 m depth range within the Tyrrhenian Sea confirmed the occurrence of significant genetic differentiation at both small and large spatial scales (Costantini *et al.,* 2013).

Costantini and Abbiati (2016) analysed the genetic variability of *C. rubrum* populations dwelling at depths of 55–120 m, from the Ligurian to the Ionian Sea, along about 1 500 km of Italian coastline. The estimated gene flow seems to indicate that each basin contains the major sources of its own populations, and small proportions of gene flow from other populations. Ligurian and Tyrrhenian populations showed lower genetic diversity compared to the populations from the Sardinian Sea (Cannas *et al.,* 2015; Cannas *et al.,* 2016), suggesting that they may have smaller population size and possibly have been overexploited.

Recently, (Jaziri *et al.,* 2017) provided the very first knowledge on genetic and biological features of commercial populations from the southwestern Mediterranean Sea (Tunisia, North African coast; depth range 44–83 m). The main results indicate that Tunisian populations have a weak genetic structuring but significant differentiation between coastal and offshore locations. Moreover, harvesting seems to not have altered the genetic structure of red coral populations yet.

Concerning the deepest red coral populations, a few colonies (a total of 12 fragments from five sites from Malta and Linosa shelves) from the lower limit of red coral depth range in the Mediterranean Sea (up to 819 m of depth) were genetically characterized by Costantini (Costantini *et al.,* 2010). The authors found differences between shallow and deepwater samples, but the small sample size of the deepwater collections does not allow the authors to make final conclusions on their degree of isolation (Costantini *et al.,* 2010).

Finally, an interesting experiment was recently performed combining transplant and genetic analyses (Ledoux *et al.,* 2015). Two populations dwelling in contrasting temperature regimes and depths (20 m and 40 m) were reciprocally transplanted. The analyses reveal partially contrasting PEIs (population-by-environment interactions) between shallow and mesophotic populations separated by approximately 100 metres, suggesting that red coral populations may potentially be locally adapted to their environment with differential phenotypic buffering capacities against thermal stress (Ledoux *et al.,* 2015). The study questions the relevance of the deep refugia hypothesis and highlights the conservation value of marginal populations as a putative reservoir of adaptive genetic polymorphism.

The adaptive response of *C. rubrum* to thermal stress has been recently studied (Haguenauer *et al.*, 2013) on three populations from different depths (5 m, 20 m and 40 m). In particular, the 5 m population seems to have a higher thermal tolerance: they showed higher gene expression response (induction of HSP70 after heat shock), which might be explained by local adaptation or acclimatization. The authors suggest that, facing climate change, these shallow populations may be an irreplaceable source for evolutionary rescue and re-colonization potential, and thus worthy of special attention and protection (Haguenauer *et al.,* 2013).

New candidate markers potentially implicated in the response to thermal stress, which could be good candidates for the study of thermal adaptation for the red coral have been proposed by Pratlong *et al.* (2015).

Overall the genetic studies performed within the Mediterranean Sea so far highlight a strong genetic heterogeneity even at very small spatial scales, and suggest that management of red coral has to be planned on a local basis. Individual harvesting plans for each bank should also to be considered; that is, each commercial stock should be characterized from the genetic point of view to identify population boundaries and define management units.

On a larger spatial scale, mitochondrial data suggest that Atlantic *C. rubrum* is genetically distinct from its Mediterranean counterpart. Atlantic red corals may have introgressed the Mediterranean ones after migration via the Algeria current (Boavida *et al.,* 2016).

2.2.7.2. Pacific Ocean

Genetic studies have been conducted on Pacific Coralliidae species, mostly focused on phylogenetic aspects (Ardila *et al*., 2012; Uda *et al*., 2011, 2013; Tu *et al*., 2012, 2015a). On the contrary, to date genetic studies on populations have been performed on two species.

A preliminary study, based on three microsatellite loci, examined the population genetic structure of widely-distributed populations of *H. laauense* within and among eight Hawaiian seamounts (Baco and Shank, 2005). Gene analyses revealed relatively high levels of genetic diversity as well as low yet significant levels of population differentiation among several of the seamounts and islands (predominantly among population comparisons with any site and the Kauai and Makapu'u coral beds). Heterozygote deficiency was found in every population with at least one locus. The low heterozygosity throughout the Hawaiian Archipelago raises the concern that this species may be suffering from inbreeding depression. Further investigation of precious coral population demographic structure is needed to assess the risk of habitat loss and fisheries activities on these seamounts (Baco and Shank, 2005).

P. secundum populations from 11 sites in the Hawaiian Archipelago collected between 1998 and 2004 have been analysed and compared with the population genetic structure and dispersal capabilities of *H. laauense*. Microsatellite studies on *P. secundum* populations show little heterozygote deficiency and are separated into three distinct regions. On the contrary *H. laauense* had significant heterozygote deficiency in most populations; two populations appear to be significantly isolated from others and may be separate stocks. The results suggest recruitment is from local sources with only occasional long-distance dispersal events. In addition to having fisheries management implications for these corals, the results of these studies also have implications for the management and protection of seamount fauna (Baco-Taylor 2006, unpublished data).

Next-generation sequencing (NGS) is currently being used to obtain SNPs in order to analyse the genetic diversity of coral populations in each sea area and the gene flow between them (Annex 2).

2.2.8. Threats and environmental issues (Coralliidae)

2.2.8.1. Mediterranean Sea and Atlantic Ocean

The Mediterranean red coral is considered a vulnerable resource in the Mediterranean Sea because it is a long-lived species, with slow growth, low fecundity and limited dispersal capabilities and consequently strong genetic differentiation of neighbouring populations at spatial scales of tens of metres.

Because of its high economic value, *C. rubrum* has been exploited since ancient times. The impact of this direct fishing is driving significant shifts in the size structure of shallow *C. rubrum* populations, with an increasing exploitation of red corals living below 100 m (Otero *et al.,* 2017). Recent Mediterranean studies have echoed the vulnerability of *C. rubrum* (e.g. Bavestrello *et al.,* 2014a; Garrabou *et al*., 2009; Harmelin, 2004). When it comes to observed local depletions, and an estimated decline that almost certainly exceeds 30 percent over the past 30 years (one generation), *C. rubrum* has been listed as ‘Endangered’ by IUCN specialists in the recent assessment of Mediterranean anthozoa. As threats are ongoing, its future decline may well exceed 50 percent unless the threats are addressed (Otero *et al*., 2017). *C. rubrum* has been also included in the IUCN Italian Red List as ‘Endangered’ (Bo *et al*., 2014a), and the Red Book of Corals of Croatia as ‘Critically Endangered’ (CITES document AC29 Doc.22). However, doubts can be raised on the data used for the Italian and Mediterranean assessments, especially when they refer to the decline in yields over the past years (“decrease in catch landings of 60 percent during the last thirty years”; Otero *et al*., 2017), because precise figures are not available on this aspect and the FAO data (they possibly rely on) are known to have severe shortcomings (see following paragraphs on landings). The Croatian assessment is still unpublished (CITES document AC29 Doc.22), and hence precise details on the source of data used for the assessment are lacking.

Recent studies also showed that deep rocky environments in the Mediterranean with high relative abundance of debris (mainly lines and nets, plastic items), have a heavy impact (covering and abrasion action) on *C. rubrum*, gorgonians, antipatharians and other invertebrates (Bo *et al*., 2014a; Angiolillo *et al*., 2015; Cau *et al.,* 2015, 2017).

Fishing impacts are worsened by natural stressors and climate change, especially in shallow water where mass mortality events have also been documented since the late 1990s, attributed to a fungal and protozoan disease and linked to temperature anomalies (Bramanti *et al.,* 2005; Cerrano *et al.*, 1999; Garrabou *et al.,* 2009*,* 2001; Perez *et al.,* 2000; Romano *et al.*, 2000). In particular, the Mediterranean red coral is expected to be particularly susceptible to acidification effects linked to climate change owing to the elevated solubility of its Mg-calcite skeleton. Experimental studies have shown, for the first time, evidence of detrimental ocean acidification effects on this valuable coral species (Bramanti *et al.,* 2013; Cerrano *et al.,* 2013).

Besides harvesting, other types of anthropogenic disturbances threaten *C. rubrum* populations: pollution, tourism, recreational diving, incidental takes, and habitat degradation (Boavida *et al.,* 2016; Cau *et al.,* 2015a*,* 2017; Coma *et al.,* 2004; Garrabou *et al.,* 1998; Lastras *et al.,* 2016; Linares *et al.,* 2003*,* 2010b).

The most destructive impact affecting Mediterranean coralligenous communities is the action of trawling gear (Cinelli and Tunesi, 2009). In the past, special trawling used to collect the precious red coral by the so-called "Italian Bar" or "Saint Andrew's Cross”, which proved to be highly destructive, causing degradation of large areas of coralligenous up to a depth of about 200 metres. Negative impacts on coralligenous communities can also be caused by traditional artisanal fishing (longlines, trammel nets, pots for catching lobsters), recreational fishing, diving activities and anchoring. At present, fishing on coralligenous formations is prohibited, although there are also reports of illegal fishing (Cattaneo-Vietti *et al.,* 2017). As most of the builder species are long-lived, have low recruitment and complex demographic patterns, destruction of the coralligenous structure is critical as their recovery will probably take several decades or even centuries (Ballesteros, 2006). For instance, after 20–30 years of protection within French and Spanish MPAs, red coral colony sizes did not reach the values of pristine populations (Garrabou and Harmelin 2002; Linares *et al.*, 2010a; Tsounis *et al.,* 2006b) suggesting that full recovery will require decades of effective protection (Linares *et al.,* 2010a; Torrents and Garrabou, 2011; Tsounis *et al.,* 2006b). Given the longevity of red coral (more than 100 years) (Garrabou and Harmelin 2002; Marschal *et al.,* 2004; Tsounis *et al.,* 2007), it seems reasonable to speculate that a few decades of protection may not be sufficient to reach pristine population sizes.

2.2.8.2. Pacific Ocean

For most commercially valuable Pacific corals, unsustainable harvesting is the primary threat, as it has triggered large shifts in population dynamics and abundance.

In March 2017, the Japanese Ministry of the Environment included red, pink and white corals on the Red List of Threatened Species under the classification “Near Threatened”. While these species’ risk of extinction is not significant at present, the reason for this classification was increased pressure on fishery as a result of dwindling fishing resources, greater numbers of permitted fishing vessels, and poaching by foreign boats. Since red coral is not threatened with extinction but a depletion of resources is feared, the Fisheries Agency publication “Basic Data on Rare Wild Aquatic Life in Japan” classifies it under Category 3: Diminishing Species (species that are clearly decreasing in number) (Fujioka, 1996; in Annex 2). On the contrary, precious corals and black corals have still not been included on the Red Lists of Taiwan Province of China, the Republic of Korea, the Philippines, India and Sri Lanka (Annex 2).

Moreover, the use of non-selective fishing gear is another issue. Dredges are the most widely used gear for harvest, whereby the coral is entangled in nets and pulled to the surface. This practice is a destructive and wasteful process that breaks and dislodges coral, with high percentages of colonies detached and lost during collection, much of which dies as a consequence (Bruckner, 2016). Dredging operations dislodge and remove non-target sessile invertebrates, including undersized precious corals of low value that are subsequently discarded. Dredges also destabilize the bottom, reducing the availability of hard substrates for the future settlement of larvae (Bruckner, 2016).

However, damage to Coralliidae populations due to unintended impacts from fisheries are thought to be minimal in the United States Pacific Islands (Parrish *et al.,* 2009).

3. FISHERY & CATCH DATA, KNOWLEDGE OF POPULATION STATUS

3.1. Black corals (order Antipatharia)

The harvesting of black coral is known to occur/have occurred in several areas: in the Indo-Pacific, the Caribbean, Latin America, and very sporadically in the Mediterranean Sea. In addition to targeted commercial harvesting, black corals are also inadvertently caught in bottom trawls (New Zealand, Australia, Canada; Wagner *et al*., 2012 and references therein).

The earliest known fisheries for black coral were in the Red Sea. One of the largest fisheries was observed out of Jeddah, Saudi Arabia, where corals were harvested 100 miles off the coastline (Bruckner, 2016). In Indonesia black coral harvesting has been known about since the eighteenth century at least.

In the Pacific, black coral (*Antipathes ternatensis* and *Cirrhipathes* spp.) has also been extensively collected off the Vietnamese coast and Sumatra Malaysia region, while *Antipathes arborea* was historically harvested off Sri Lanka. Southeast Asia and the islands of the South Pacific continue to be an important source of black coral for international markets, however very limited information is available on the status of the industry or the amount of harvest (Bruckner, 2016). In the 1980s, an estimated 60–100 tonnes of black coral were harvested from the Philippines each year, with most of the remainder (about 8 percent of all world harvest) collected from South Pacific countries, principally Tonga, Fiji and Papua New Guinea. Across the Pacific Islands, as part of the PROCFISH project (http://coastfish.spc.int/en/projects/procfish), black coral stands were commonly noted on dives across hundreds of sites across 22 Island States and Territories of the Pacific as part of deeper water sea cucumber assessments (20–45 m, anecdotal report; K. Friedman, personal communication). In 1977, the trade of precious and semi-precious corals was prohibited in the Philippines, and stocks were required to be sold within three years (Tsounis *et al.*, 2010).

In Indonesia, and Palau, the amounts were generally small and the quantities landed were rarely registered (Wells, 1983; Grigg, 1993). In Indonesia a small-scale fishery still exists, consisting of diving using an air compressor (Annex 2).

Commercial black coral harvesting was recorded from the 1960s to the 1980s in Ecuador, where black corals were intensively collected around the Galapagos (Ecuador); these species have completely disappeared from several sites (Martinez and Robinson, 1983).

In the Caribbean, the main species harvested were *P. pennacea*, *A. caribbeana A. dichotoma*, *Leiopathes glaberrima*, and *Cirrhipathes lutkeni*, primarily for use in the tourist jewellery trade and not for export. This is because more of Caribbean fisheries were short-lived and collection was sporadic as the resource was patchy and shallow-water populations were easily overexploited. By the early 1980s, a local depletion of populations was registered in St. Lucia, Barbados, the Netherlands Antilles, Bahamas, and British Virgin Islands (Bruckner, 2016 and references therein). In the 1960s large populations of black coral (*A. caribbeana*) were first discovered in Cuba. Then, in the mid-1980s the fishery expanded in the Cazones Gulf, with 1 355 kg of black coral harvested from 1987 to 1993. Black coral was also extracted from Pinar del Rio, with a maximum of 301 kg taken in 1992 (Guitart *et al*., 1997).

In Mexico, black coral is an important resource for jewellery and handicraft, providing economic support for authorized fishermen, craftsmen and merchants. The harvesting of black corals started in the late 1960s in the Mexican Caribbean Sea (State of Quintana Roo), targeting mainly two species (*P. pennacea* and *A. caribbeana*). At first collection was accomplished by snorkelling to depths of about 20 m. The SCUBA made access to deeper banks at depth around 80 m, using a saw to cut the colonies. The fishery expanded rapidly, some 1 000–1 500 kg were caught in the 1980–1990s. For over 30 years the harvesting concentrated around Cozumel Island; however, because of the inadequate management and the consequent overexploitation of the banks, the fishery was closed there in 1997 (Padila, 2001; Padila and Lara, 2003). New collecting sites were searched and exploited. Today, harvesting of black corals occurs along the Mexican coastline, with the only exclusion of MPAs (Padila, 2001). The strategy of constant exploration of new banks followed by exploitation until depletion represents a serious problem to this fishery and a risk for black coral as a resource in Quintana Roo (Padila and Lara, 2003).

In the Mediterranean Sea the only known black coral fishery started in 1984 in Maltese waters, undertaken by a government-owned company. The fishery relied on SCUBA divers using helium-based breathing gas mixtures, an ROV, and a manned submersible. Some 250 kg of black coral were harvested from 500 to 600 m between 1984 and 1987 (Deidun *et al*., 2010).

As a result of the overexploitation of black coral populations in several regions, some governments have banned harvesting of antipatharians (Wagner *et al*., 2012 and references therein).

The black coral fishery in the United States of America (Hawaii) has enjoyed a more profitable and continuous history. In Hawaii, commercial harvesting of black coral started in 1958, and has occurred in three locations (Auau Channel off Maui, southwestern coast of Hawaii, and southern coast of Kauai) at 30–90 m depth by SCUBA divers; it has continued continuously for over 60 years (Grigg, 2001). The Hawaiian black coral fishery was the first fishery that was managed on the basis of an extensive fishery research programme, which was conducted at the University of Hawaii in the early 1970s.

Highly variable catch records have been reported (Grigg, 2010b), but this is primarily a product of incomplete reporting by the Hawaii Division of Aquatic Resources, especially during the years 1981–1997. In spite of this problem, the landings do not appear to have exceeded estimates of OY (optimal yield 5 000 kg/year for the Maui Bed). In this regard, it is important to clarify that harvest of black coral has always been selective, favouring large colonies and discouraging the collection of undersize colonies (Grigg, 2010b).

Between 1963 and 1970, divers harvested over 23 000 kg from two locations, most of which was *A. dichotoma* (90 percent) with lesser amounts of *A. grandis* (10 percent) and *A. ulex* (1 percent). During the 1970s and 1980s the demand for black coral was greatly reduced, and the market shifted from *Antipathes* to *Corallium* and *Gerardia* (Grigg, 1993, 2001), although landings still quite remained high. Grigg (2010) reported that approximately 8 000 kg/year were harvested from the Maui Bed and 4 000 kg/year from the Kauai bed over this period. Since 1980, all black coral harvested in the Hawaiian Islands has been taken from the Au'Au Channel Bed (total size of about 1.7 km^2) with a lesser harvest from the smaller (0.4 km^2) Kauai Bed. In the 1990s and early 2000s landings of black coral in Hawaii increased. From 1981 to 1990 the state of Hawaii reported that landings of black coral amounted to 6 200 kg, with an annual take of 72–1 977 kg. Over the next 7 years (1992–1998) it increased to over 9 000 kg and continued to increase exponentially from 1999 to 2005, comprising 58 percent of the total harvest since 1985 (Grigg, 2010b and references therein).

In the United States of America, quotas were amended based on recommendations of a downward adjustment of maximum sustainable yield (MSY), given the results of biological surveys in 2002–2004, which illustrated a downward shift in the age structure of colonies as well as a reduction in biomass of about 25 percent, and a decline in both recruitment and the abundance of legal-sized black coral colonies (Grigg, 2004). This decline was driven by the combination of two main factors: the continued spread of the invasive coral (Kahng and Grigg, 2005) and a large increase (25–50 percent) in the demand for black coral since 1998, leading to increased harvest pressure.

In the Mediterranean Sea, some black corals are targeted by illegal, unreported and unregulated (IUU) fishing activities (e.g. Giusti *et al*., 2015; Maldonado *et al*., 2013; Deidun *et al*., 2015). However, there is no information on the extent of these practices at the country level, and more efforts are needed to assess and enforce, if necessary, measures to prevent, deter and eliminate the fishing of these protected species (Otero *et al*., 2017).

In the Pacific area the illegal collection of black corals, whose export is controlled as a species listed in CITES Appendix II, is often reported by news sources. Since illegal operations are proof of demand, cases of violations need to be reviewed not only from the point of view of species protection and resource conservation, but also for the implementation of CITES. Several cases reported in the Philippines and India are discussed in Annex 2.

Black coral harvests are reported to occur without any control or any management in many tropical islands and particularly in Madagascar, where the illegal trade is continually expanding. Since 2011, an illegal traffic of black corals has been occurring in the main cities of the southern and coastal regions of Ambovombe and Tolagnaro. In 2014 and 2015, hundreds of kilograms of black coral let and a lot of diving material were seized by the authorities in the Anosy and Androy regions (Terrana *et al*., 2017; Todinanahary *et al*., 2016).

According to all the people questioned, poachers come mainly from China and have ample equipment and means at their disposal, including vehicles and diving materials. It is estimated that a collection of 4–5 months could lead to the production of more than 5 000 kg of black coral. In Madagascar, black corals are so lucrative that local people have named them the "rosewoods of the sea". Fishermen claim that the value of unworked black coral sold from the villages ranges from MGA 10 000 to MGA 15 000 /kg from SCUBA divers/fishermen to local collectors and MGA 15 000 to MGA 20 000 from local collectors to national collectors.[11] The price per kg from the capital Antananarivo for international exportation, mainly headed to Asia, ranges from MGA 500 000 to MGA 700 000. A SCUBA-diver harvesting black corals can earn MGA 750 000 /day, his monthly salary reaching more or less MGA 12 million. This is higher than the salary of a government official even when all costs related to the collection of corals are deducted. This income is a thousand times higher than the monthly revenue received by a traditional fisherman in Madagascar (Todinanahary *et al*., 2016).

3.2. White, pink and red corals (family Coralliidae)

Among the different species in the family Coralliidae, harvesting is known for nine (ten considering that part of *P. elatius* catches in Taiwan Province of China are probably *P. carusrubrum*; (see previous section on systematics). However, three species distributed in the coastal waters of Hawaii and around Midway Atoll, are no longer collected commercially, although they have been fished in the past. The remaining six species are harvested at present in the Mediterranean and Atlantic Ocean (*Corallium rubrum*), and in the Pacific Ocean (*Pleurocorallium secundum, P. konojoi, P. elatius, Hemicorallium sulcatum,* and *C. japonicum)*. Information related to the harvesting of these precious corals, in the past and in the present, is described in the following paragraphs.

3.2.1. Corallium rubrum fisheries

3.2.1.1. History

The history of the exploitation of precious corals can be described as the history of the Mediterranean red coral exploitation (*C. rubrum*), the precious coral par excellence. In late Palaeolithic, Mediterranean red coral was collected in the form of fragments and large branches that washed up on the shore after storms (Tescione, 1973). Intentional precious coral exploitation has been dated to 5 000 years ago, when iron hooks were used to harvest red coral (Grigg, 1984). These tools were first used between the first and fourth century B.C. by Greek Roman divers up until the first century A.D. Greek sponge divers were capable of free diving to a depth of over 80 m with iron hooks called

[11] MGA (Malagasy Ariary) 1 = USD 0.0003 (OANDA, 1 April 2019)

“kouralio” and other tools to dislodge the corals. During the Neolithic period, boats armed with nets used to drag for the coral became the principal tools for exploitation. After that, dredges of a similar style, but varying mainly in their size, as well as the number of nets attached, and the weight and shape of materials used for ballasting, gradually spread throughout the region. In the tenth century, precious coral fishing became more efficient when Arabic fishermen developed a wooden dredge known as the “ingegno”, or “Saint Andrew’s Cross”. This consisted of a wooden cross with nets attached that was dragged along the bottom, entangling red coral in the Mediterranean Sea. These early dredges were used by Spanish, Sardinian, and Catalonian fishers until replaced with a larger and heavier metal dredge, the “*barra italiana*” during the industrial age (Galasso, 2000). The *barra italiana* was made of a heavy iron bar (> 1 tonne) with chains and nets attached along its length. Although the *barra italiana* could be dragged in deeper water and was considerably larger, it still was not very efficient: it is estimated that only about 40 percent of the coral broken off the substrate was entangled and retrieved (GFCM, 1984).

The Mediterranean coral fishery has fluctuated between periods of prosperity and decline, on the basis of the discovery of new beds, commercial demand, political regime changes, wars, restrictions on fishing, and various political struggles for supremacy over fishing grounds (Bruckner, 2016). Those countries with some history of red coral harvesting in the Mediterranean Sea number less than a dozen.

The introduction of the *barra italiana* allowed the fishery to switch from a fishery practised only by individual divers to industrial-scale operations, i.e. fleets of boats armed with multiple dredges used simultaneously. With the advent of industrial fisheries, all European coral fishermen moved to the area off the coastlines of Africa, which became the main fishery and processing centre. For example, in the twelfth and thirteenth century most of the exploited coral banks were located in Tunisia and Algeria, with fishermen from Catalonia, Genoa and Marseille (Tescione, 1973), while in the sixteenth and seventeenth centuries, most coral fishing boats from Genoa and Naples harvested coral off the North African coast. By the late seventeenth century, the majority of the coral fishermen were based out of Lisbon and Marseille, while Livorno and Torre del Greco emerged in the eighteenth century as the major production centres (Tescione, 1973; GFCM, 1989). In the mid- to late 1800s, the Italian fishery dominated with a period of prosperity, replacing the African centre. In a short time, effort in the Italian fishery increased with thousands of small sail and rowing boats, tens of thousands of workers and dozens of processing centres. In 1862, there were 347 registered coral fishing boats. By 1865 this had increased to 1 200 vessels, with 24 Italian factories processing coral, employing around 17 000 fishermen and jewellers (Tescione, 1965, 1973). In the 1880s, the discovery of huge dead red coral deposits off Sciacca triggered a true red coral rush (Rajola, 2012). About 18 000 tonnes of sub-fossil, dead coral were harvested, depleting the banks in about 30 years (Liverino, 1998), using a new gear, *la codata* (the tail). This tool consisted of a 200-metre-long rope with attached bundles of old nets at regular intervals of 1.5 m, in which coral branches remained entangled (Gangemi, 2011, 2014). In this period, the yield reached an average of 2.6 tonnes/boat/year. The best harvests occurred in 1880 and 1881 when about 4 492 tonnes and 2 630 tonnes were collected, respectively (Cattaneo-Vietti *et al*., 2017). At its peak, over 2 000 boats worked these banks, levels of harvest quadrupled, and the number of processing factories increased from 40 to over 80. Intense fishing quickly depleted these grounds, and they were abandoned around 1915, but the glut of low-quality coral simultaneously lowered prices and reduced fishing in other areas (Tsounis *et al*., 2010 and references therein). An excessive abundance of coral gave rise to a crisis, caused by market saturation and depreciation of a large part of the production. In just ten years, the excess of supply made its dramatic effects felt, and many manufacturers were forced to close (Ascione, 2010). Considerable stockpiles of “Sciacca coral” are still held in Italy (Ascione, 2010). Red coral fishing stopped definitively between 1914 and 1918. Simultaneously, imports of Japanese pink and red coral started to reduce the demand for Mediterranean red coral (Tescione, 1973; Ascione, 2010). Later, demand for

Mediterranean coral increased again in the 1930s, followed by another decline during the early 1940s. Shortly after, SCUBA diving emerged as a new, highly profitable method for the harvest of red coral.

From the first use of SCUBA diving for harvesting red coral, divers have been forced to move to deeper and deeper depths in response to the depletion of shallower areas; in 1956 divers worked at 30–35 m; in 1958 at 40–45 m; and by 1964 at depths of 72 m. Inevitably a growing number of accidents were recorded as the result of the spreading 'coral fever' among the divers (Liverino, 1984). Others similarly documented that by the late 1950s divers in France and Italy already had to descend to depths of 80 m, and at times to even more than 100 m, to find coral (Galasso, 2000). In 1974, helium-based mixed-gas diving techniques started to spread among coral divers, permitting them to work at 120 m for 20 minutes without the dangers of nitrogen narcosis (Liverino, 1984).

In any event, coral dredges continued to be used as the major tool to collect red coral in the Mediterranean, as they could be used to drag for coral in deeper areas that could not be exploited by divers. Small boats were replaced by larger motorized vessels capable of dragging immense (7 metres long, 800 kg) metal dredges at much greater depths (180 m) in the 1970s (GFCM, 1989). Both sampling methods (divers and dredges) continued to expand in the Mediterranean. In 1982, there were 150 factories, employing 4 000 workers and 1 600 fishermen (Tsounis *et al*., 2010). Following the documentation on the damage caused by dredges to coralligenous habitat in the Mediterranean, provided by different research projects, Mediterranean countries progressively banned their use. Algeria was the first country to do so in 1977, followed by Morocco, Spain, Tunisia and Croatia in the mid-1980s. By 1994, the *barra italiana* and other types of coral dredges were completely prohibited by European Union.

3.2.1.2. C. rubrum *landing (1978–1993)*

A large part of the data reported below are based on Liverino's data (Liverino, 1998); they are unofficial, because for a long time in Italy (and probably elsewhere) any administration or recognized organization (Coast Guard, Unions, Customs, Fisheries Department, etc.) was supposed to collect information systematically and/or elaborate statistical figures on this important resource (Liverino, 1998). According to the author, these data are often incomplete and disorganized; they were gathered thanks to the collaboration of fishermen, buyers and all the people involved in the harvesting of red coral, based on their memories and personal annotations. These figures are thus approximate, but in any case they are reported here because of their importance in reconstructing the evolution of the *C. rubrum* fishery in the Mediterranean region. In fact, apart from the production (yield in tonnes) for each country, Liverino also provided the number of boats (using dredging gears) and divers (using the pick) exploiting red coral in the different countries and years, information that is not available in other datasets (e.g. the FAO global capture database, see below).

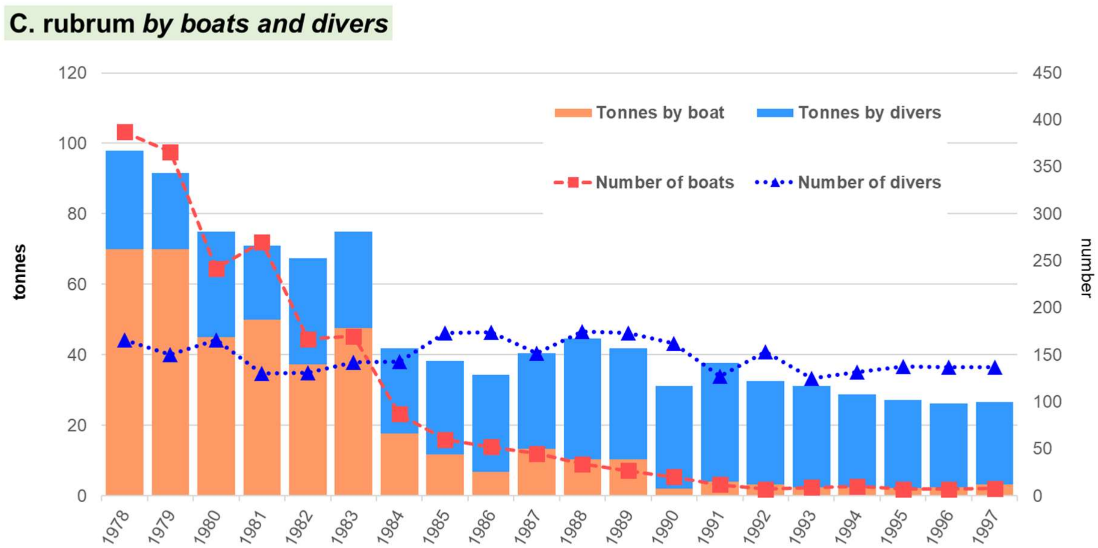

Figure 1 Data on C. rubrum *production for the period 1978–1997. Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino, 1998).*

In the period 1978–1997, the production dropped from 98 to 26.7 tonnes (Figure 1). This decrease in landings was mainly due to the change in harvesting technique (from destructive dredging to selective diving) that has greatly reduced the fishing effort exerted on the species (Pani, 2010). In fact, in 1978 there were 387 boats using dragging gears (the cross and/or ingegno), but only eight were still actively harvesting red coral in 1997. In the same period, the amount of red coral collected by boats shifted from 70 to 3.2 tonnes. On the other hand, the number of divers and the amount taken by them remained largely stable in those years, with small fluctuations (1978: 28 tonnes collected by 28 divers; 1997: 23.5 tonnes by 137 divers; see Figure 1).

Considering the immense ecological damage that dredging inflicts on coral habitats (Thrush and Dayton, 2002), coral dredging was banned in European Union waters in 1994, through European Union (EU) Regulation 1626/94. Actually, dredging was phased out even earlier in some countries (see section on gears and country profiles for further details). A small number of boats continued to work, sometimes illegally, for some more years after the ban (Figure 2), even though this type of fishery proved to be more and more unsustainable in light of the decline in abundance and its high economic overheads (GFCM, 1989).

The vast majority of these vessels were Italian working mainly in Italy, Tunisia, and Spain (Figure 2 and Figure 3). A relatively small number of vessels worked in France (with a peak of 21 in 1978, and the last vessel in 1991), and in Morocco (from 1987 to 1991, with a maximum of four boats in 1987–1988). Lower numbers were recorded in Greece, Albania and sporadically in former Yugoslavia (Figure 3 and Figure 4). Since the mid-1980s the ban of all dragging gears was imposed by law in some countries, but a small number of boats kept on using it for a while (Liverino, 1998), after which it finally disappeared ‘officially' throughout the Mediterranean.

The large fluctuations in numbers (of boats and divers) and the amount of coral harvested reflect the fortuitous discovery of new fishing grounds and/or the entry into force of local regulations in the different countries (with the opening or closing of certain areas, the banning of dredging gears, limits on the number of fishermen, boats and harvesting months, etc.).

In the period 1990–1997, 2 194 divers and 220 boats collected a total of 241.4 tonnes of *C. rubrum*; the vast majority of it was sold to India and Japan (Liverino, 1998).

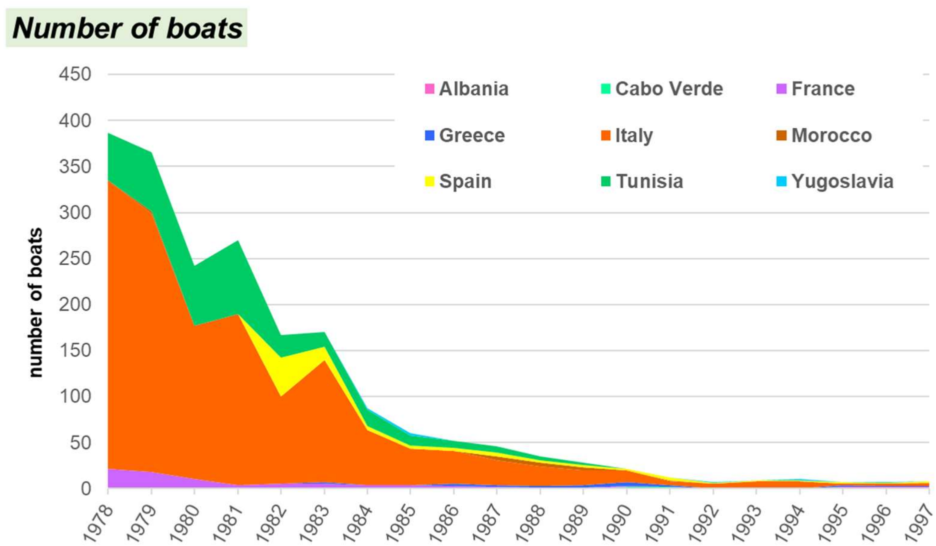

Figure 2 Number of boats using dredging gears in the different countries in the period 1978–1997 (Source: Liverino, 1998).

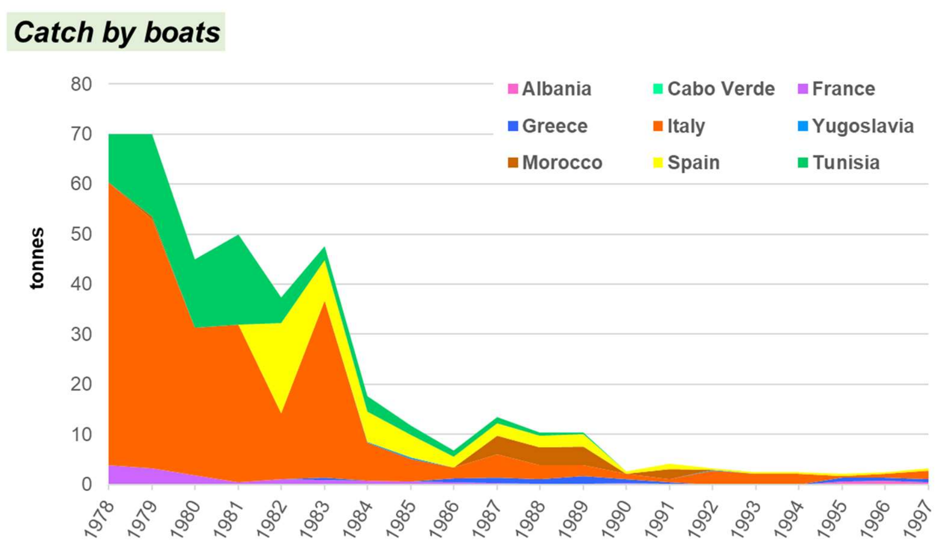

Figure 3 Red coral yields (in tonnes) by boats using dragging gears in the different countries in the period 1978–1997 (Source: Liverino, 1998).

Initially, divers worked principally in Italy, France, Spain and Tunisia (Figure 4 and Figure 5); thereafter exploitation expanded to other countries such as Morocco, Algeria, Greece, and former Yugoslavia (especially on the Dalmatian coast and in the islands of Viz and Lastovo, now part of Croatia). In many cases, the same divers moved to the different countries to exploit new areas once the coral was hard to find at home. Especially in countries with no tradition of deep diving, fishing was carried out over precise periods by joint ventures employing foreign divers (GFCM, 1989). In particular, Italian (and French) divers worked on the North African coasts for local shipowners where they also contributed to training local divers (Liverino, 1998).

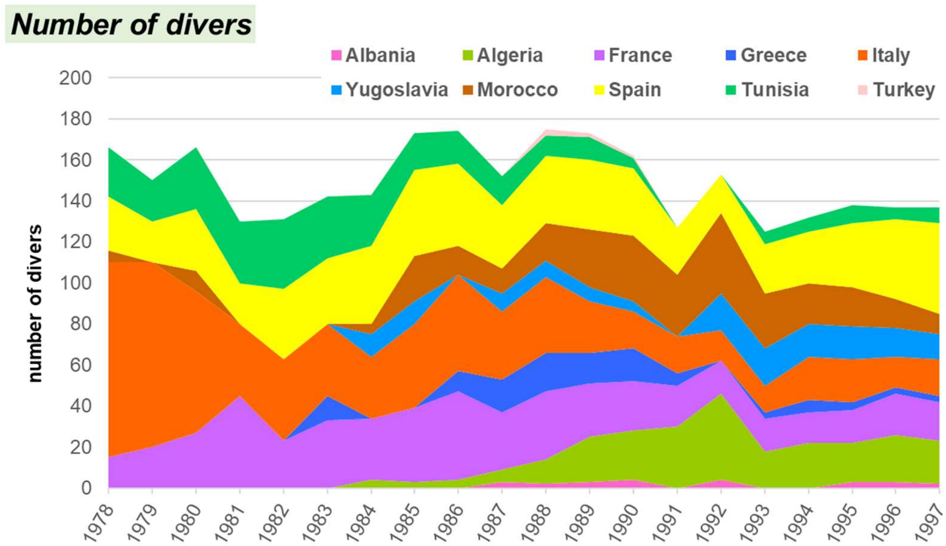

Figure 4 Number of divers employed in the different countries; manual collection of coral with the pick in the period 1978–1997 (Source: Liverino, 1998).

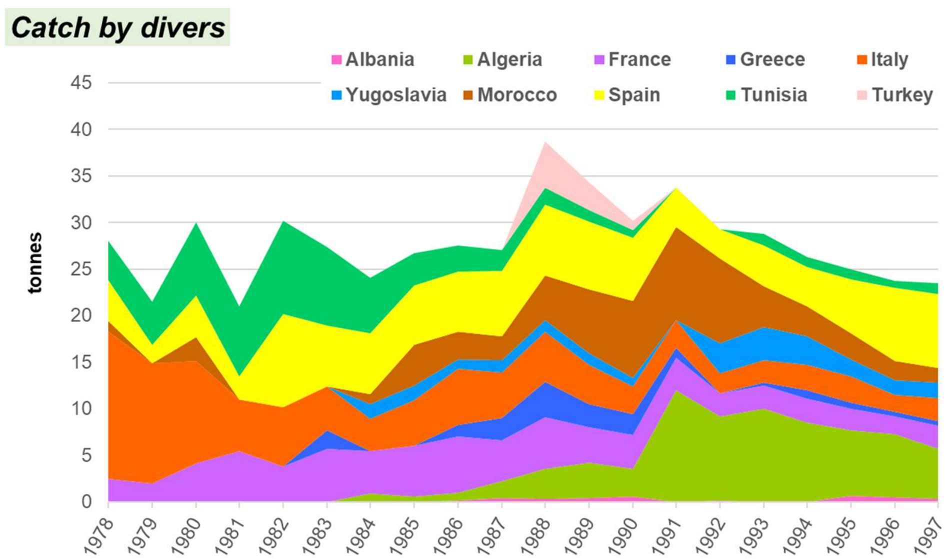

Figure 5 Red coral yields (tonnes) by divers (manual collection with the pick) in the different countries in the period 1978–1997 (Source: Liverino, 1998).

3.2.1.3. C. rubrum *current fishing methods*

Since the mid-1980s coral dredging has been banned in many countries (e.g. 1977 in Algeria; 1985 in former Yugoslavia and Tunisia; 1989 in Sardinia (Italy); and in 1994 in European Union waters, in compliance with EU Regulation 1626/94).

Trawling gears are extremely destructive for coral populations and their habitats even on rough sea floors, as recently demonstrated by ROV surveys, which provided evidence of the devastating effect of illegal and continuous use of the 'ingegno' gear for red coral fishing in the Sicily Channel (Cattaneo-Vietti *et al*., 2017). The trawling gears harvest only 45 percent of the total coral biomass, lose about 9 percent as fragments, while 46 percent of the colonies are left *in situ*, many of which are left entangled by net shreds. These data highlight the potential efficiency of the 'ingegno' gear in removing colonies over time and the long-term impact they have on the surviving colonies (Cattaneo-Vietti *et al*., 2017).

The recent data support the notion that this fishing technique, by perpetrating its impact over time on a limited area, is able to completely erase a red coral population, irreversibly changing the structure and the composition of the whole benthic community and its habitat (Cattaneo-Vietti *et al*., 2017).

Nowadays, SCUBA diving using manual picks is the only harvesting method legally allowed in the *C. rubrum* fishery (see also the management section).

Divers use a pick to break the chosen branches, leaving the rest untouched. SCUBA harvesting inflicts little direct damage on non-target species in the same habitat. Responsible divers cut the red coral base instead of extracting the whole colony; leaving the base in place leaves a chance that this colony might regrow, as has been sporadically observed (Rossi *et al*., 2008). Studies on catches confiscated from poachers confirmed that up to 60–70 percent were entire colonies with the substratum still attached to their base (Linares *et al*., 2003; Tsounis *et al.,* 2010b; Hereu *et al*., 2002). However, as regards the selectivity of this fishing method, it is entirely theoretical. In the 1980s divers reportedly made a "clean sweep" of an entire precious coral population at one site, and poachers harvested young corals in shallow waters (GFCM, 1989).

Diving at significant depths is a very dangerous activity and it requires a very long decompression, after which the divers usually go into a hyperbaric chamber for 6–9 hours (Cicogna, 2000). Furthermore, the considerable time, pressure and difficulty of working underwater and at those depths is quite incapacitating, so that red coral divers may not be able to consistently perform a precise size selection or partial harvest of corals (Tsounis *et al*., 2010b).

Today, robotic extraction is not permitted for harvesting Mediterranean red coral in the GFCM competence area; it is only used for scientific purposes as part of a specific project (Annex 1, GFCM 2018).

In recent years ROVs have proven to be very useful in the Mediterranean for habitat mapping, studying biocoenosis and quantifying the distribution, structure, abundance and status of benthic biocoenosis, especially in areas that could not be sampled using traditional methods (SCUBA, trawl) owing to the depth or roughness of the terrain. ROV surveys have also been applied to red coral studies, to describe their occurrence, spatial distribution, and population structure (Rossi *et al*., 2008; Angiolillo *et al*., 2009; Taviani *et al*., 2010; Bo *et al*., 2011). Similarly, ROV videos enabled the gathering of data on the spatial distribution and structure of commercial banks (from 80 to 130 m) around Sardinia seas (Pedoni *et al*., 2009; Cannas *et al*., 2010; Cannas *et al*., 2011; Follesa *et al*., 2013; Cau *et al*., 2015a; Cau *et al*., 2016).

In recent years ROVs have been increasingly employed to scout for potential beds in Sardinia (Italy) and France (GFCM, 2017). Basic ROVs consisted of a motorized, real-time video camera controlled from the boat via a cable that also transmits the video signal to a topside monitor and recorder. ROVs could also be equipped with a robotic arm that permits remote-controlled harvesting, although this option considerably raises the purchase price.

Remote harvesting was considered problematic when compared to manual methods. Currents, nets, and the topography of coral habitats make it difficult to handle the tethered machines, and without a dedicated technician a minor malfunction may easily render an ROV unusable for an entire expedition (Tsounis *et al*., 2010b).

Furthermore, many different problems may arise from the use of ROV for harvesting red coral. Considering the peculiarities of these machines (not limited by the physical constraints of divers, and hence capable of diving deeper and for longer than humans) the number of ROV licenses, its operational time/day, season length, depth limits etc. should be carefully defined before the gear is employed on a large scale, in order to avoid the risk that its unregulated (mis)use will lead to a sudden and unsustainable increase in the amount of coral harvested (A. Cau, personal communication; Tsounis *et al*., 2013, 2010; Bruckner, 2016).

Last but not least, based on experimental use in the Pacific, ROVs may damage precious corals if not used with care because currents, nets and the topography of coral habitats make the handling of these tethered machines difficult (WPCouncil, 2007); as a consequence, potentially long-lasting damage to the ecosystems (the coralligenous communities) also needs to be considered (Cau, in GFCM, 2010).

Considering the high risk associated with the legalisation of a new harvesting methodology (robotic harvesting) without comprehensive knowledge of the fishing effort and the sustainability of the gear, the use of ROVs has been limited in the GFCM competence area since 2011, in line with Recommendation GFCM/35/2011/2. ROVs were initially authorized for observation and prospection reasons, providing that ROV models were not equipped with manipulator arms or any other device allowing the cutting and harvesting of red coral. The use of ROVs was then limited to scientific experimental projects both for observation and harvesting during a limited period not extending beyond 2015, in order to evaluate the impact and advisability of using ROV for the direct harvesting of red coral.

According to data from a recent programme of onboard scientific observers undertaken in Sardinia (Italy) in the 2012–2015 period, the use of ROV in commercial vessels for the prospecting of coral beds was linked to a high increase in yield per unit (i.e. the number of colonies collected per dive) when compared to dives performed without ROVs. Thanks to ROV prospecting, the time for searching the right spot to collect corals during a dive was drastically reduced, whereas the time for collecting corals considerably increased (Cannas in GFCM, 2017).

Today, according to Recommendations GFCM/40/2016/7 and GFCM/41/2017/5, the use of ROVs for prospecting in the GFCM area of application is no longer allowed and it has been further limited to observation for scientific purposes in the context of research programmes and scientific experimental campaigns led by scientific institutions (until 31 December 2020), with a strict prohibition on the commercialization of *C. rubrum* colonies collected as part of research programmes.

3.2.1.4. C. rubrum *current capture data in the FAO database (up to 2016)*

Harvest statistics for *C. rubrum* have been available in the FAO global capture database since 1963 for Spain and since 1974 for Tunisia. The coverage extended to other countries in the late 1970s. Unlike the other catch statistics included in the database, which are usually submitted by national official sources, data on red coral have been provided consistently since the mid-1980s by a major red coral import–export and jewellery production wholesaler (Garibaldi in GFCM, 2010 and 2011). However, the data available in the FAO database present clear shortcomings (i.e. possible conflict of interest for an industry data provider; data may refer in some cases to trade information rather than to actual annual harvest) and therefore should be considered cautiously. The constant provision by the same source may provide useful information for trend analysis (Garibaldi in GFCM, 2010 and 2011).

Data from the FAO database (Figure 6) show a decline in total catch from 98 tonnes in 1978 to 20.5 tonnes in 1998 and then a progressive increase up to 60.92 tonnes in 2016.[12] With the exclusion of catch data from Atlantic areas by Cabo Verde (1990–1991), Morocco (1988–2016) and Spain (1996–2006), all other catches are from the Mediterranean Sea (Figure 7).

12 FAO 2018, Fishery and Aquaculture Statistics, Global capture production 1950–2016 (FishstatJ). In: FAO Fisheries and Aquaculture Department [online], Rome, Updated 2018, www.fao.org/fishery/statistics/software/fishstatj/en

The countries with the largest catches in 2016 were Croatia (18.5 tonnes), France (11.1 tonnes), Italy (9.8 tonnes) and Tunisia (8.5 tonnes).

It should be noted that FAO aimed to progressively incorporate as much data as possible from official national sources into its database, rather than those provided by the industry (Garibaldi in GFCM, 2011). However, to discontinue the provision of data from the industry, the number of countries reporting red coral data should have increased significantly; currently only a few countries officially submit their annual red coral harvest to FAO.

IUU activities and the black-market trade of red coral are known to be common in the Mediterranean, although it is difficult to quantify their extent (Santangelo *et al*., 2009; Tsounis *et al*., 2006; Dounas *et al*., 2010). For some countries, data reported to FAO by the industry sources are higher than those available from scientific or other national sources. This may be also due to the fact that the data from the industry also cover quantities actually traded, which could come from IUU activities and therefore not reported in national data (the discrepancies between FAO and national data are discussed in the following sections by country).

Poaching has been confirmed on the Costa Brava (Spain) (GFCM 2017), Italy (Santagelo *et al*., 2009; 2014; Cattaneo-Vietti *et al*., 2017), Greece (Dounas *et al*., 2010), France (Linares *et al*., 2010) and Croatia (CITES document AC29 Doc 22 and its annexes) and is probably common throughout the Mediterranean, especially on the North African coast (Tsounis *et al.*, 2013 and references therein). In some cases, it seems that poachers sell their harvest through licensed divers (Tsounis *et al*., 2009). The enforcement of the new GFCM regulations is therefore critical to limit illegal harvests.

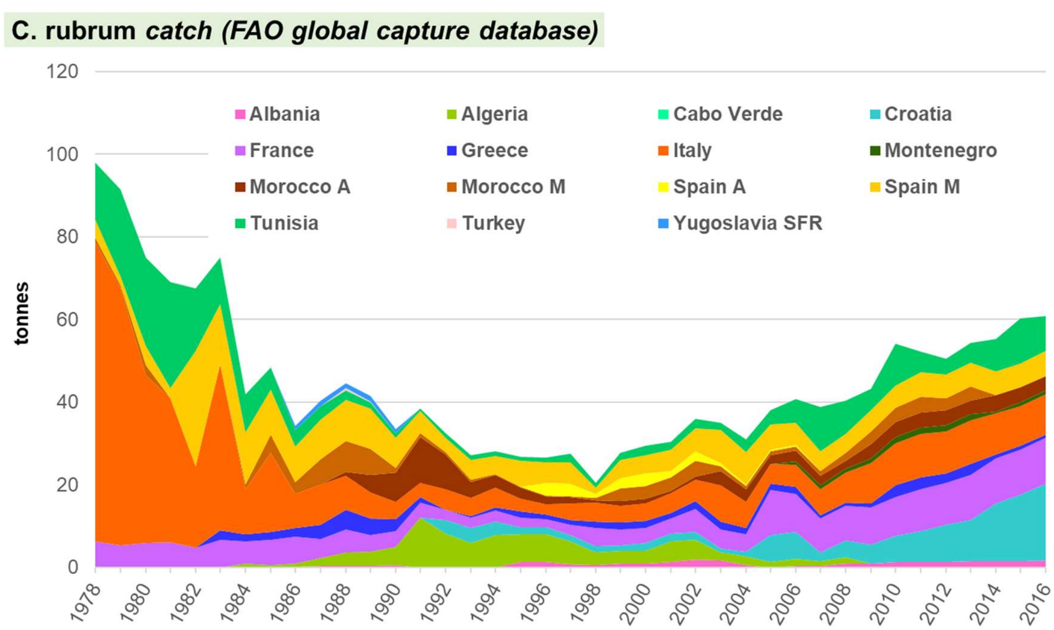

Figure 6 Data in the FAO global capture database for C. rubrum *(Source: FAO, 2018).*

In 2016, apart from small amounts from the Atlantic coast off Morocco (3.42 tonnes), all red corals were harvested in the Mediterranean Sea (57.5 tonnes) (Figure 7).

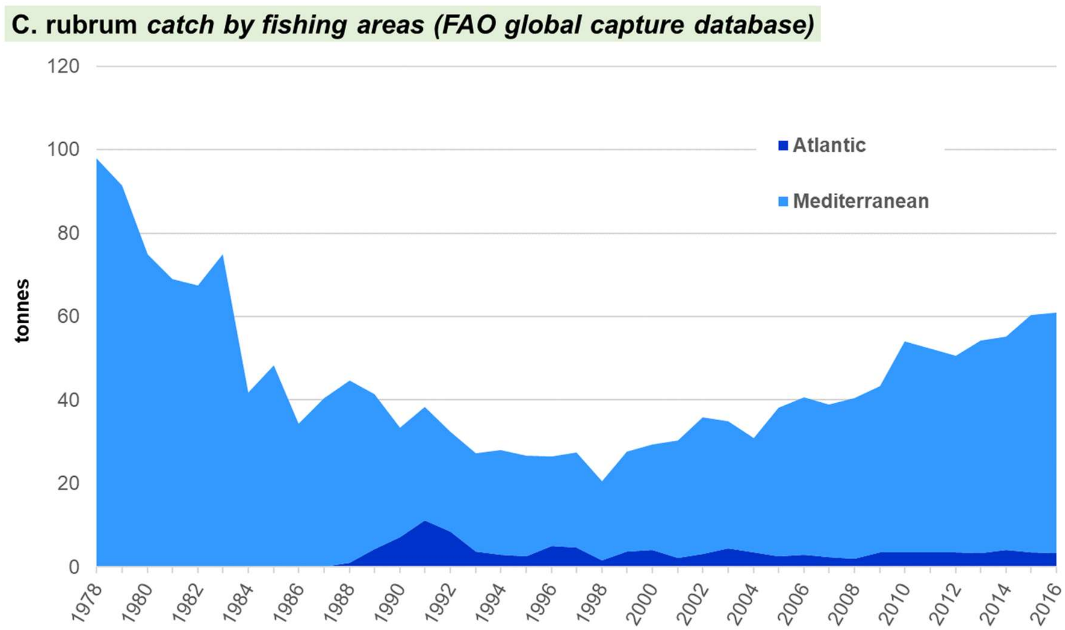

Figure 7 Data in the FAO global capture database for C. rubrum *by FAO major fishing area (Source: FAO, 2018).*

Since the inception of SCUBA fishing, landings reported by individual countries have continued to show sharp peaks and troughs, which are suggestive of the discovery of large aggregations of coral in a particular area, followed by the rapid overexploitation of these populations. Individual peaks in landings during a single year reflect the pulse fishing mode associated with SCUBA harvest, whereby individual beds are selectively cleared of large colonies, and then a new area is targeted. SCUBA fishing was originally concentrated in shallow waters, extracting corals from areas that were largely inaccessible to dredges. Over the last two decades, SCUBA fishing has been progressively moving into deeper areas, in response to a depletion of corals in shallow waters.

According to a recent publication (Tsounis *et al*., 2013) the yield has been able to remain stable or even increase in the last decade thanks to: 1) the harvesting of ever-smaller corals after depleting larger size classes; 2) the harvesting in ever-deeper waters; and 3) the use of ROV technology for scouting (legal) and harvesting (illegal). Furthermore, anecdotal reports linking the fear of the listing in CITES and the incoming GFCM Regional Management Plan (RMP) probably pushed coral divers to collect as much as possible before potentially stricter trade and harvesting regulations are implemented (Tsounis *et al*., 2013). On the other hand, the possibility of introducing bureaucratic elements into the trade has been perceived negatively by both Mediterranean divers and manufacturers given the high probability that it will lead to significant reductions of red coral in selling and processing (Stampacchia, 2010).

3.2.1.5. <u>C. rubrum *current fishing data: official figures*</u>

Unfortunately, official data on *C. rubrum* production (provided by state agencies, ministries, fisheries departments etc.) are only regularly available for a few countries; information available in the following sections was retrieved from published papers, reports and proceedings, and integrated with the data directly transmitted by members to GFCM in the present and past years. Wherever possible they were compared to the FAO data. It is worth highlighting that, since 2013 the compulsory measures provided in the recent GFCM decisions request contracting parties and cooperating non-contracting parties (CPCs) exploiting Mediterranean red coral to submit an array of compulsory and optional information on harvesting activities to the GFCM Secretariat.

Compulsory information includes: area of exploitation (geographical subarea (GSA) and Statistical Grid), area name, name of landing port, annual catch of red coral, average diameter (mm) of the colonies harvested; percentage in weight of undersized (i.e. < 7 mm) colonies; effort expressed as annual number of fishing days or of dives; range of harvesting depth.

Trend analysis per country by both yield/year and yield/dive indicate that total landings of red coral are decreasing in some countries and stable in others (Annex 1, Figure 2). Information related to catches, the size structure and the biology of the species is generally not sufficient to provide an overall assessment of the status of red coral populations. In general, important compulsory data, such as the average colony diameter and the percentage of undersized colonies, has not yet been made available, and an improvement in the quality of data is expected as a result of the entry into force of Recommendation GFCM/41/2017/5.

3.2.1.6. C. rubrum country profiles

Hereinafter, data are discussed by country, in alphabetic order, and shown in Figure 8 to Figure 40, comparing data from different sources wherever possible (Liverino, 1998; FAO, 2018; GFCM, 2018; and national data). If not specified otherwise the FAO figures for the period 1978–1997 are totally consistent with those from the Liverino's book (Liverino, 1998). In fact, FAO data are from the same source (Garibaldi in GFCM 2010, 2011); small discrepancies are probably the result of revisions.

3.2.1.6.1. Albania

In Albania, the number of boats and divers involved has always been very limited. Since 1995 the fishing of corals and sponges is prohibited (Article 22 of Law No. 7908 of 1995), but Liverino (Liverino, 1998) and FAO still report small amounts of coral harvesting (Figure 8 and Figure 9).

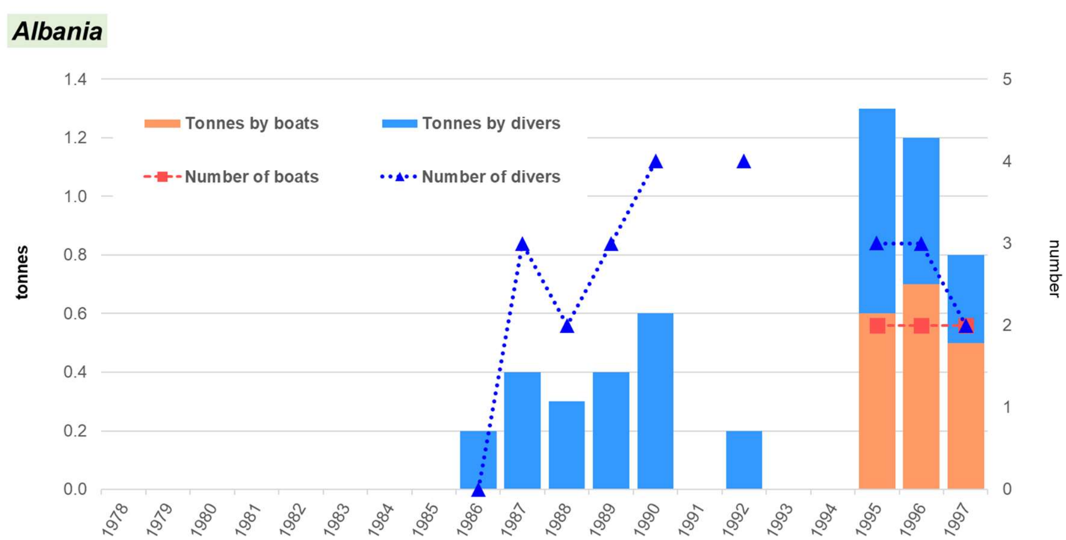

Figure 8 Data on C. rubrum *production for the period 1978–1997 in Albania. Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino, 1998).*

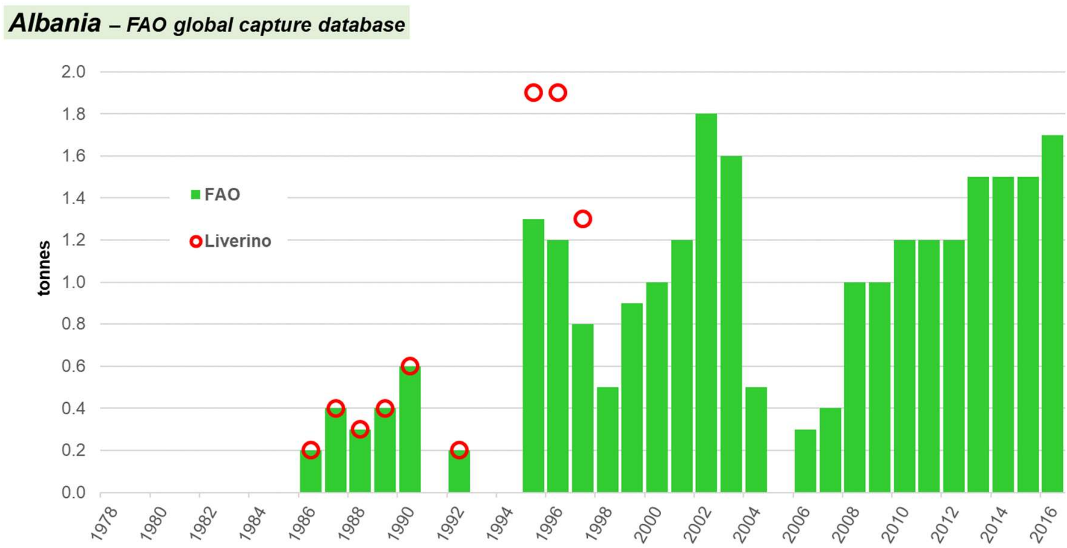

Figure 9 Data in the FAO global capture database for C. rubrum *in Albania (green bars); the white dots correspond to the figures given by Liverino (Source: FAO, 2018; Liverino, 1998).*

3.2.1.6.2. Algeria

In Algeria, after a period of stasis following independence from France (1962) red coral fishing was resumed (1975–1977) totalling some 10 tonnes. During these years, the 'Office Algérien des Pèches (OAP)' was in charge of the exploitation and transformation of red coral resources (GFCM, 2014). Coral fishing was banned from 1977 to at least 1982 pending the chance to regulate the fishery (Akrour, 1989). In 1982, the fisheries activities restarted again thanks to a public company (Ex-Enapêches) in conjunction with private companies. From 1978 the Saint Andrew's cross was prohibited (GFCM, 1989). In 1980, and up to 1986, one vessel equipped for SCUBA diving was operating in the El Kala region alone (GFCM, 1989). According to Liverino (Liverino, 1998) Algeria reopened coral harvesting to divers in 1984 (Figure 10).

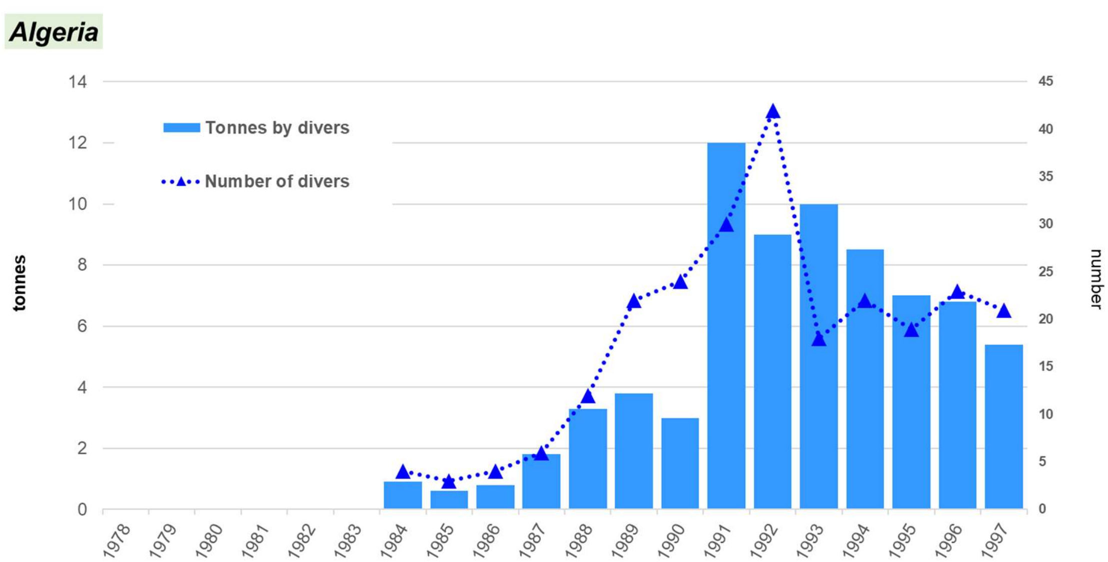

Figure 10 Data on C. rubrum *production for the period 1978–1997 in Algeria. Number divers (using picks) and yields (tonnes) are illustrated (Source: Liverino, 1998).*

In 1987, two private shipowners were granted the permit of collecting coral in the El-Kala area, the following year more authorizations were issued for Algerian fishermen collaborating with Italian divers and investors (GFCM 2014; Akrour 1989).

At that time, the ministerial circular N°639/88/SPM du 19 Octobre 1988 defined the operational rules for the exploitation.

Since 1995 (Décret exécutif n° 95-323 du 21 Octobre 1995 réglementant l'exploitation des ressources corallifères), red coral harvesting was allowed only if in possession of a personal permit, strictly linked to a specific area. Each area should be harvested for not more than five consecutive years, and closed for a minimum period of 15 years following harvesting.

The Official data provided by Algeria to the GFCM Secretariat in 2014 for the red coral exploitation in Algeria (Figure 11) indicate that:

- the domestic production of the coral has been characterized by a gradual increase from 1987 to 1991; in that year, the production reached a maximum of 7 864 kg (an increase of about four times the actual production in 1987 (1 765 kg);
- from 1992 to 1996 the production remained relatively constant with an average of about 7 000 kg/year;
- the highest production was recorded in 1997 with a national collection of 13 259 kg;
- from 1987 to 2000, a total of 86 300 kg was collected, with an average annual production of more than 4 500 kg/year.

The fleet and the number of SCUBA divers, meanwhile, experienced a continuous increase from 1 boat (5 divers) in 1987 to 43 boats (46 divers) in 1997. This fleet, which used foreign divers, dropped to 20 divers in 1998 as a consequence of the entry into force of the provision banning foreign divers. This followed a grace period of three years granted to the beneficiaries of concessions to hire Algerian staff (GFCM, 2014).

Overall, this production has resulted in a turnover of nearly DZD 1 billion.[13] It led to the creation of nearly 250 direct jobs and almost 500 in supporting activities and coral processing before commercialization. To these 750 jobs, it is necessary to add, indirectly, the 5 000 jewellers whose product is the raw material of their activities (GFCM, 2014).

The comparison of catch data for Algeria from different sources points out the largest differences in the data are in 1991 and 1997. On the overall, considering the years 1987–2000, the catches total 86.3 tonnes (official data) or 74.27 tonnes (FAO data; Figure 11).

Since 2001 the harvest of red coral has been completely prohibited again, as formally implemented by an executive decree (Décret exécutif N° 01-56 15 February 2001). However, the FAO global capture database also reports data for the period 2001–2008, when fishing is officially prohibited in the country; these numbers (provided to FAO by a coral wholesaler) may refer to corals worked in that period by Torre del Greco manufacturers.

[13] DZD (Algerian Dinars) 1 = USD 0.0083 (OANDA, 1 April 2019)

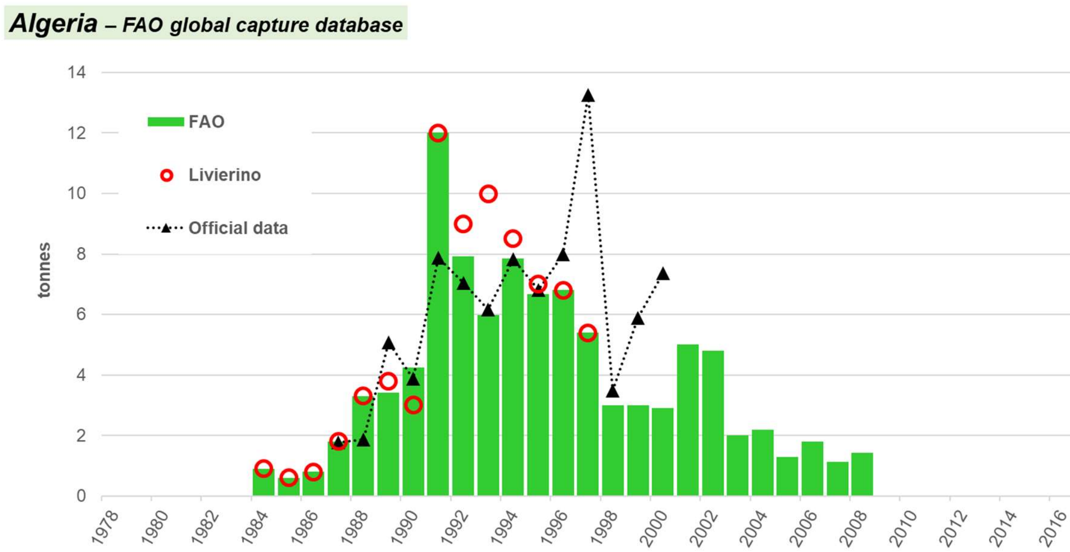

Figure 11 Data in the FAO global capture database for C. rubrum *in Algeria (green bars); the white dots correspond to figures given by Liverino (Liverino 1998) while the black triangles are the official data transmitted by Algeria to the GFCM Secretariat in 2014 (Source: FAO 2018; GFCM 2014).*

In 2017, during the GFCM Workshop held in Tunisia, the imminent re-opening of harvesting was announced, and the new "Management of the red coral fishery in Algeria" was presented. It was recalled that the closure of Algerian fisheries was a temporary measure to avoid the illegal trade of dive licenses and to carry out an assessment of the status of the fishery, so that adequate legislation and monitoring, control and surveillance (MCS) rules could be developed before reconsidering opening the fishery again.

Specific rules for the conditions and modalities of coral harvesting are outlined in Executive Decree n°15-231 of 26 August 2015 and by other legislative instruments, considering GFCM recommendations, specifically: size, the depth, the designation of landing ports, the prohibition of using ROV and the use of hammer as the only permitted gear. Algerian experts underlined that the management of red coral fisheries in their coastal area was based on rotating periods of 5 years of exploitation and 20 years of total closure, and that they obtained very positive results with this approach (GFCM, 2017).

3.2.1.6.3. Cabo Verde

In 1990–1991 a very small amount of coral (0.14 t) was collected by 2 boats, owned by Italians, in Cabo Verde (Eastern Atlantic) and exported, presumably, to China (Liverino, 1998) (Figure 12). These banks were first discovered in 1870, but only later did the Italian boats (1982) reached the Cabo Verde Islands and collect red corals (14 kg); these banks were soon deserted because of the poor quality of the coral.

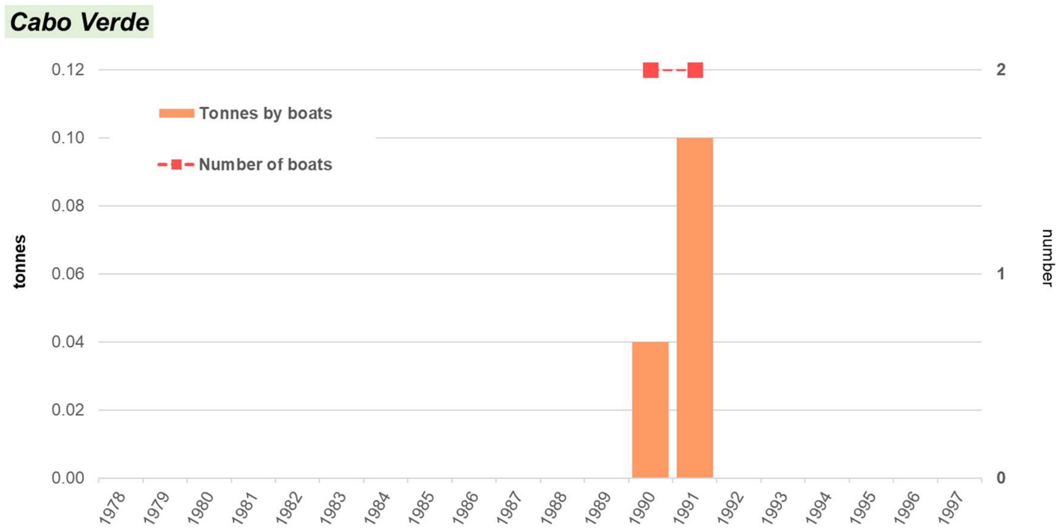

Figure 12 Data on C. rubrum *production for the period 1978–1997 in the Cabo Verde Islands (Eastern Atlantic). Number boats (using dredges) and yields (tonnes) are illustrated (Source: Liverino 1998).*

3.2.1.6.4. Croatia (before 1992 Yugoslavia SFR)

In the waters of Yugoslavia (SFR, Socialist Federal Republic of) limited resources were known to occur and these mainly along the coasts of Dalmatia and in the Islands of Viz and Lastovo (Croatia), from 45 m to 120 m, and at shallower depths in caves. Historically, only one village traditionally carried out red coral harvesting using the cross, but in 1988 this fishery had already been abandoned. In 1988, a limited number of divers harvested *C. rubrum* with average annual yields of 30 kg/diver (GFCM, 1989).

The ban of dredges could date back to 1985 when the former Yugoslavia seems to have limited the harvesting of red coral to SCUBA divers (Article 51 and 53 Regulation of 19 June 1985).

According to Liverino (Liverino, 1998) the exploitation of the Croatian coasts started in 1984 and continued into the 1990s (Figure 13).

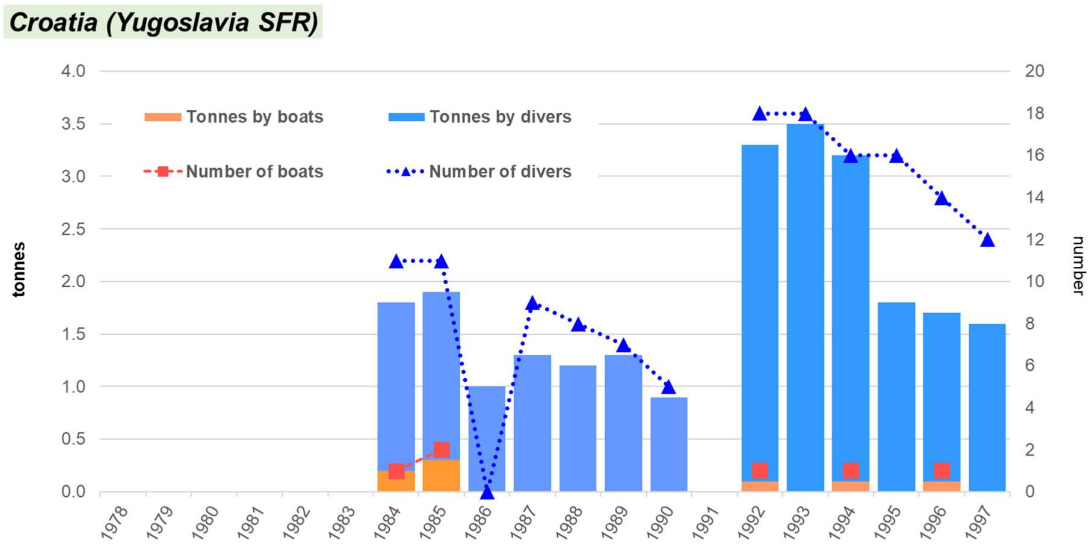

Figure 13 Data on C. rubrum *production for the period 1978–1997 in the former Yugoslavia. Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino, 1998). Since 1992 the data refers to Croatia, after the breakup of Yugoslavia SFR.*

According to the FAO data (Figure 14), there has been an increase in catches since 2005 and an exponential growth since 2010, driving Croatia to be in 2016 the country with the highest records (18.5 tonnes; GFCM 2018).

However, the information provided by Croatia for the period 2013–2016 within the GFCM Red Coral data collection is totally different. In general, the figures show a slightly decreasing trend in total amount of coral harvested from 1.1 tonnes in 2013 to 0.722 in 2016 (GFCM, 2018).

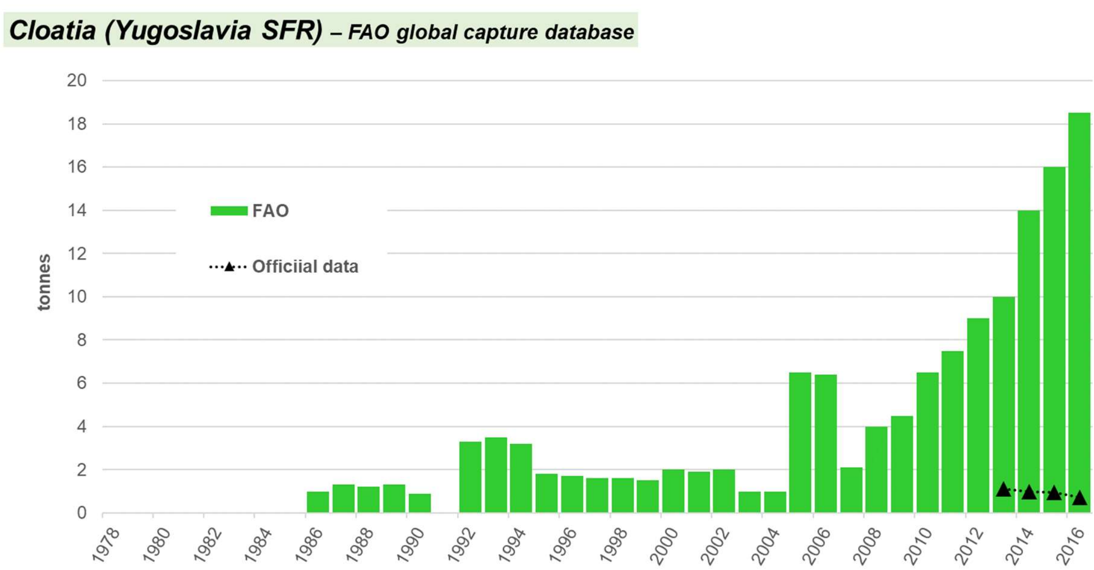

Figure 14 Data in the FAO global capture database for C. rubrum *in Croatia (before 1991 are from Yugoslavia SFR), the black triangles correspond to the official data transmitted by Croatia to the GFCM Secretariat (Source: FAO, 2018; GFCM, 2018).*

3.2.1.6.5. France

In France *C. rubrum* fishery is traditionally practised in two main areas: France 'continentale' (Côte d'Azur and Marseille) and Corsica (Figures 15–17). While on the mainland divers were exclusively collecting red coral, in Corsica divers were harvesting *C. rubrum* along with a certain number of boats (using the cross).

According to the information provided by the 'Direction des Affaires maritimes de Marseille' during the First Consultation on red coral resources organized by FAO in 1983, the number of authorized divers was 29 in 1968 (10 in the mainland and 19 in Corsica); the number peaked to 55 in 1979 (35 of which were working on the continental coasts), and later decreased to 27 in 1983 (Annex E in GFCM, 1984). There were 17 boats using the cross in Corsica in 1968, which decreased to 6 in 1983 (GFCM, 1984).

In France, the official catch statistics for red coral are probably inaccurate, however. In the 1980s the production of divers was estimated at about 5 tonnes/year and that of vessels using the Saint Andrew's cross at about 1 tonnes/year (GFCM, 1984). Uncontrolled fishing by amateur divers was also reported to occur (GFCM, 1984).

The Liverino figures are sometimes discordant from those previously reported, but the general trend of catches is confirmed (Figures 15–17).

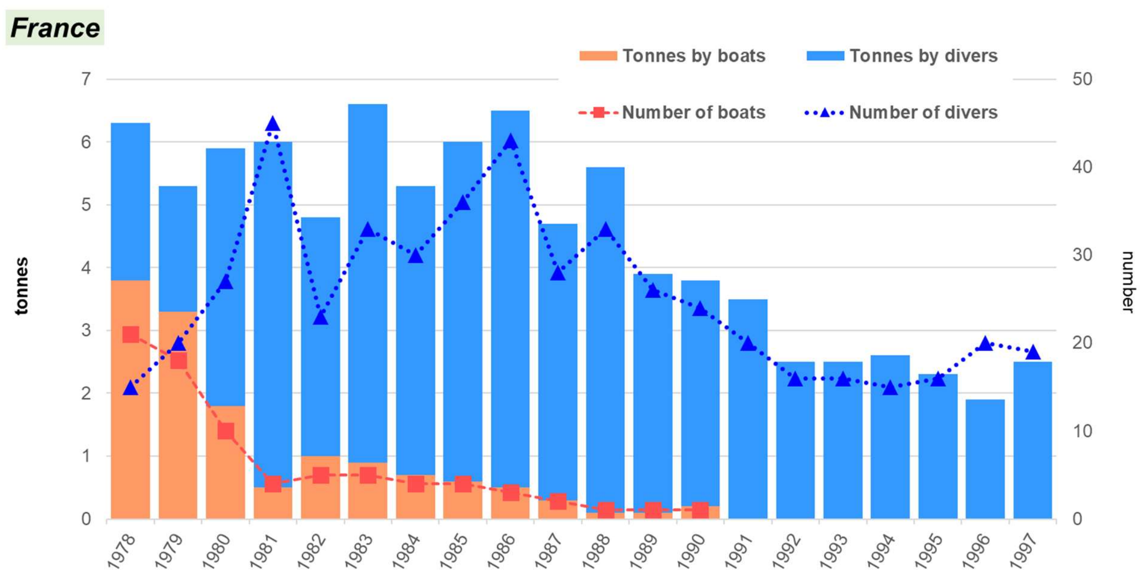

Figure 15 Data on C. rubrum *production for the period 1978–1997 in France (western Mediterranean). Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino, 1998).*

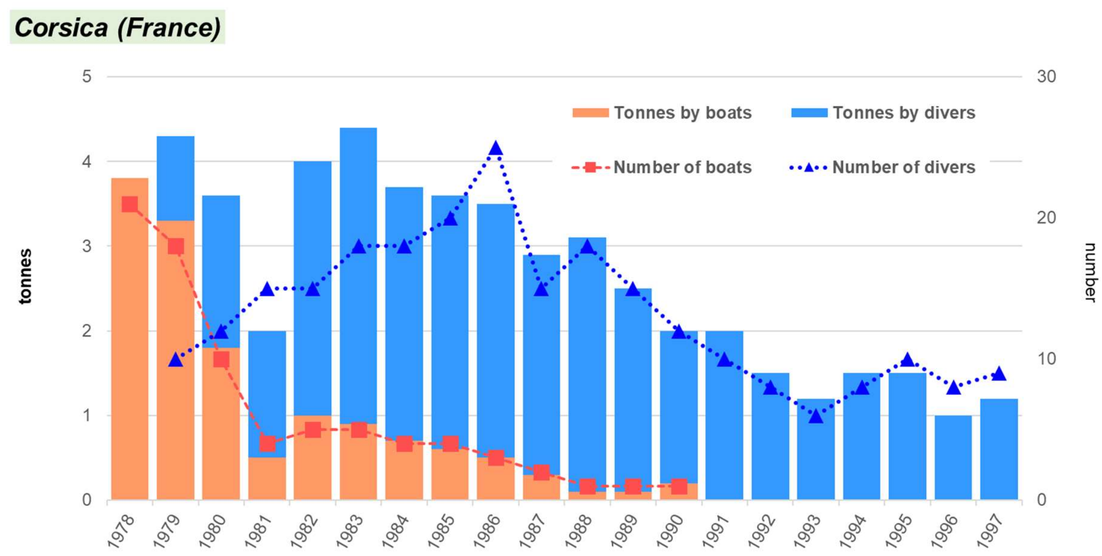

Figure 16 Data on C. rubrum *production for the period 1978–1997 in Corsica (France). Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino, 1998).*

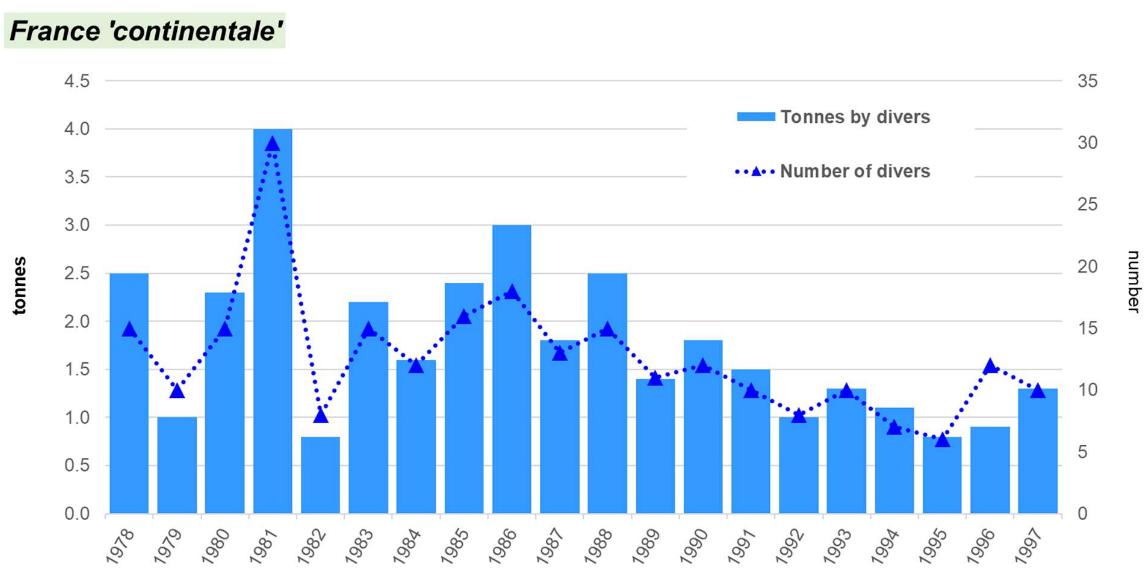

Figure 17 Data on C. rubrum *production for the period 1978–1997 in France 'continentale' (Côte d'Azur and Marseille, western Mediterranean). Number of divers (using picks) and yields (tonnes) are illustrated (Source: Liverino, 1998).*

According to FAO data (Figure 18), there has been an increase in catches since 2005, driving France to be in 2016 the country with the second highest records in the Mediterranean (11.1 tonnes; GFCM 2018).

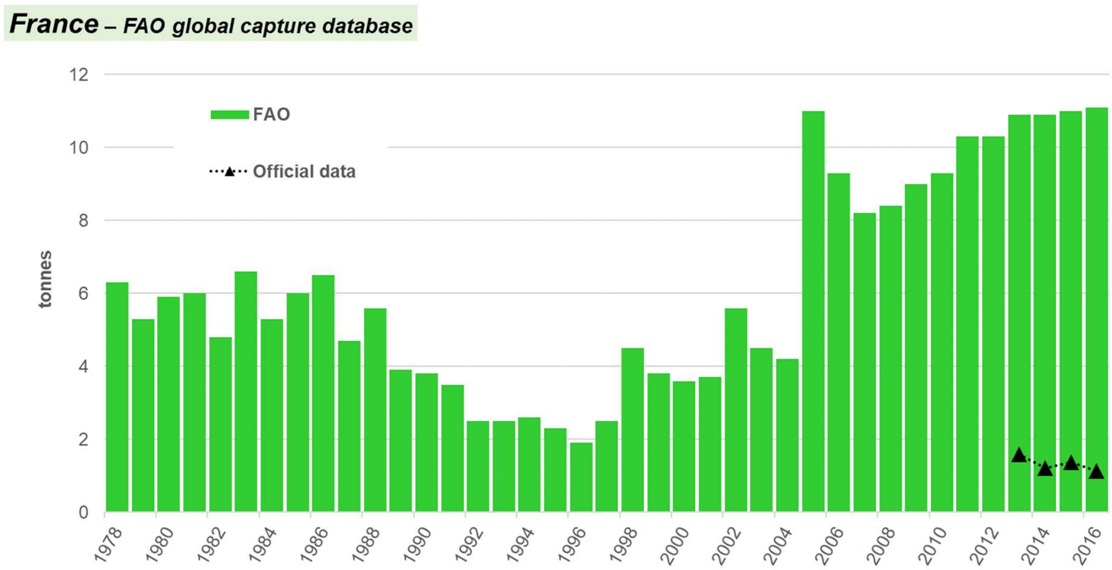

Figure 18 Data in the FAO global capture database for C. rubrum *in France; the black triangles correspond to the official data transmitted by France to the GFCM Secretariat (Source: FAO, 2018; GFCM, 2018).*

However, official data on red coral harvesting in 2013–2016 by France (Ministère de l'Ecologie, du Développement Durable et de l'Energie – Direction des Pêches Martitimes et de l'Aquaculture) within the GFCM Red Coral data collection system, are totally different.

In general, the figures show a slightly decreasing trend in total amount of coral harvested from 1.51 t in 2013 to 1.1 in 2016. In general, the amount of *C. rubrum* collected is higher in Corsica than in the PACA (Provence-Alpes-Côte d'Azur) region. A small number of ROVs were used in both areas since 2015 (GFCM 2018).

3.2.1.6.6. Greece

For a long period, *C. rubrum* was commercially (though unofficially) exploited in Greek waters (Dounas *et al*., 2010). Only in 1984, Liverino provided the first data from the Greek waters (Figure 19). The Ministry of Agriculture established relevant legislation for the first time in 1987 (Greek Law 1740/1987). According to this legislation the harvesting, processing, and trade of the red coral were to be allowed only after the purchase of a special license. Five years later, scientific surveys confirmed its presence in large populations in certain areas of the Greek seas and at depths ranging from 50 to 110 m (Chintiroglou *et al*., 1989). Initially, according to the Greek national legislation there were three harvesting methods allowed: manual harvesting by SCUBA divers using a pickaxe, and two dragging gears; that is the 'ingegno' and the 'Columbus' egg' (an egg-shaped piece of marble to crack corals and detach them from the seabed; later they are collected by nets). The first method is the only one still in use, while the latter two gears are now illegal; the use of the St Andrews' cross was always forbidden officially (Dounas *et al*., 2010).

Once the legislative framework was in place in 1994, a rotating national harvesting system was set up; it comprises five large fishing geographical zones. In the period 1995–1997 only Zone I (North Aegean) was open to exploitation (Dounas *et al*., 2010).

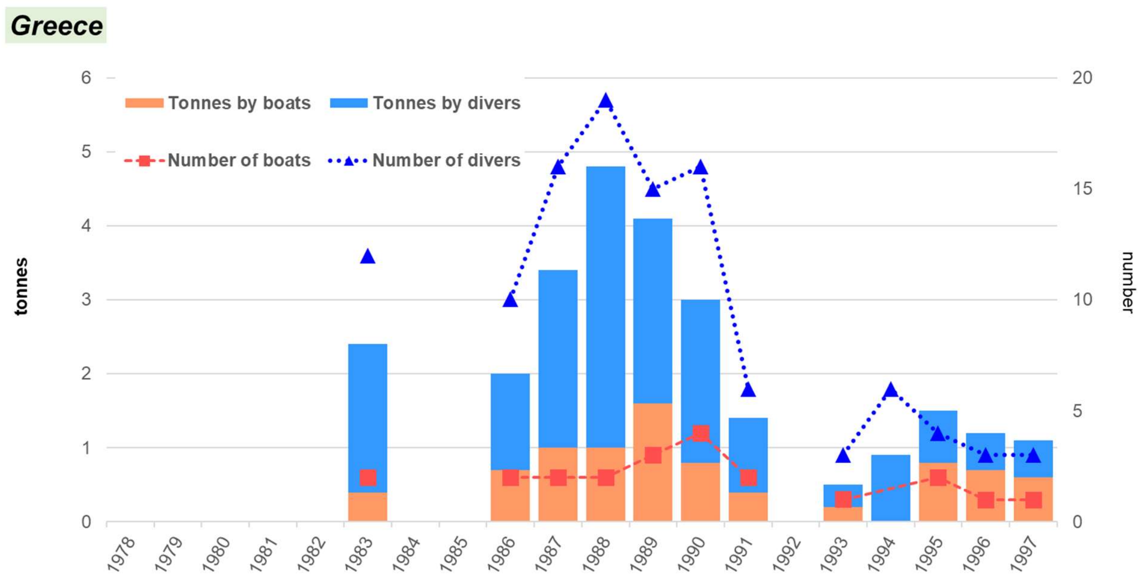

Figure 19 Data on C. rubrum *production for the period 1978–1997 in Greece. Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino, 1998).*

According to the data on red coral landings provided by the General Fisheries Directory of the Greek Ministry in Agriculture, and presented during the International Workshop on Red Coral Science, during the period 1995 to 2005 landings amounted to 17.05 (FAO data) – or, according to the official records, to 22.9 tonnes (from 1995 to 1997 Zone I (north Aegean) = 15.2 tonnes; from 1998 to 2000 Zone IV (Cretan and Libyan Sea) = 1.2 tonnes; from 2001 to 2005 Zone V (Ionian Sea) = 6.5 tonnes) (Figure 20). According to the official data there is a significant decrease of shallow-water stocks (up to 60 m) at the end of the harvesting period in each zone. Zone III (central Aegean) was open to exploitation from 2006 to 2008, finally while Zone II (south Aegean) is opened in 2009 (official data not available) (Dounas *et al*., 2010).

Total landings from the north Aegean Sea were twice as high as those recorded from the Cretan and Ionian Seas taken together, while landings from the coasts of Crete were the lowest.

Depletion of red corals in the shallower areas, where harvesting was allowed, has forced professional divers to harvest colonies at greater depths (up to 125 m) by means of advanced mixed gas diving equipment, as today this is the only effective legal fishing method (Dounas *et al*., 2010). Another major issue, related to the depletion of red coral stocks in Greek waters, is the intrinsic difficulty experienced by both local and central authorities in tackling the illegal fishing of this renewable biological resource (Dounas *et al.,* 2010). There is growing concern in the red coral fisheries sector that fishing by unauthorized vessels, illegally using destructive dragging gears (e.g. Saint Andrews' crosses), has increased considerably. A possible reason for this practice could be the high cost of purchasing and maintaining the legal, though often highly sophisticated and expensive, harvesting equipment (e.g. mixed gas apparatus) (Dounas *et al*., 2010).

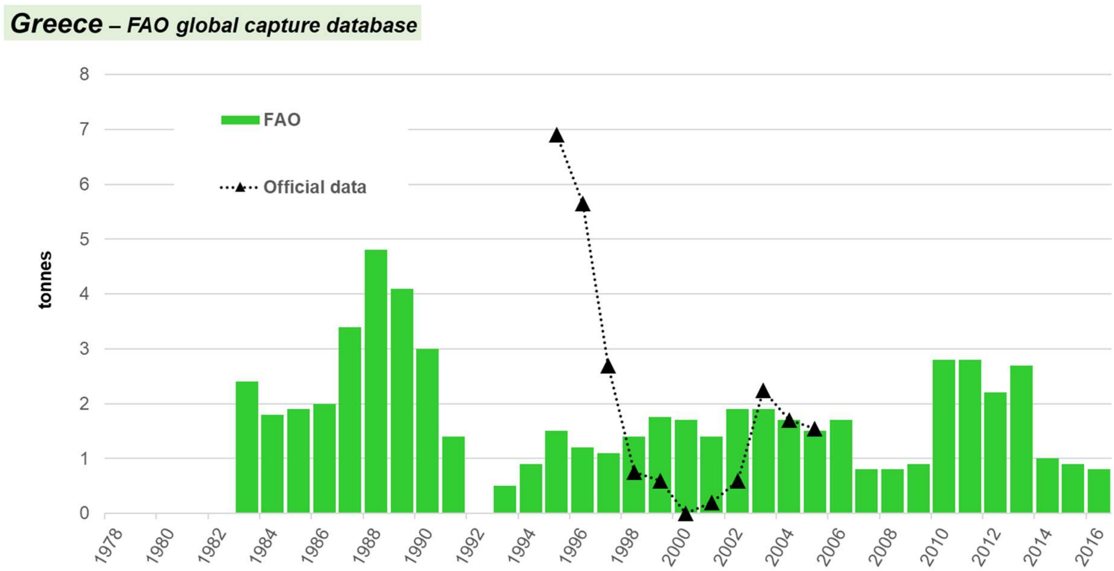

Figure 20 Data in the FAO global capture database for C. rubrum *in Greece; the black triangles correspond to the official data provided by the General Fisheries Directory of the Greek Ministry in Agriculture (Source: FAO, 2018; Dounas et al., 2010).*

According to the information officially transmitted from Greece to GFCM, no harvesting of *C. rubrum* has occurred in the country in the period 2013–2016 (GFCM, 2018). However, the FAO global capture database also reports data for this period (Figure 20).

3.2.1.6.7. Italy

Red coral harvesting has been practised along the Italian coasts since ancient times; in the period 1978–1997 red coral was actively collected in Sardinia, Sicily as well as in other areas both along the western (Toscana, Lazio, Calabria and Campania) and eastern coasts (Puglia) (Figure 21–24) (Liverino, 1998).

The 1979 to 1983 period saw a particularly favourable climate for coral fishing due to the rediscovery of the Skerki Banks in the Sicily Channel. The landing data show that, with respect to a total harvest in the entire Mediterranean of about 70–100 tonnes/year, the Italian production reached an average of 40 tonnes/year. More than half of all coral fished in the Mediterranean basin in the last few years of the twentieth century came from Italian banks, although it is difficult to distinguish catches that occurred on the Italian continental platform and those obtained by Italian SCUBA-divers along the North African coasts (Cattaneo-Vietti *et al*. 2016).

In the period 1978–1997, the coralline and SCUBA- diver yields from the Tuscany Archipelago and Gulf of Naples reached, on average, 1.8 tonnes/year: only 6.9 percent of the amount collected in the Sardinia waters in the same period (Liverino, 1998).

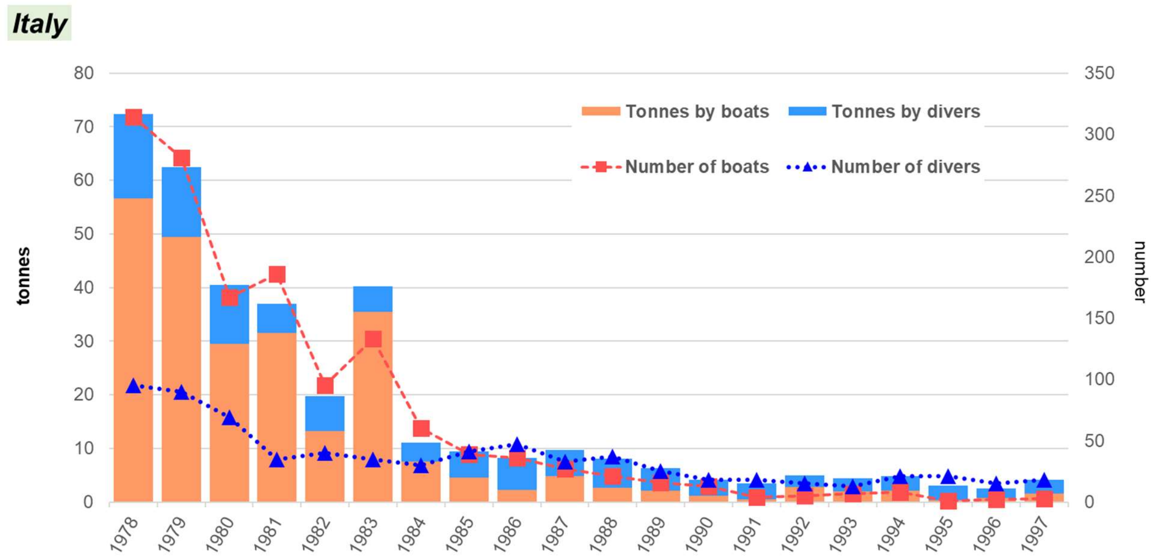

Figure 21 Data on C. rubrum *production for the period 1978–1997 in Italy. Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino 1998).*

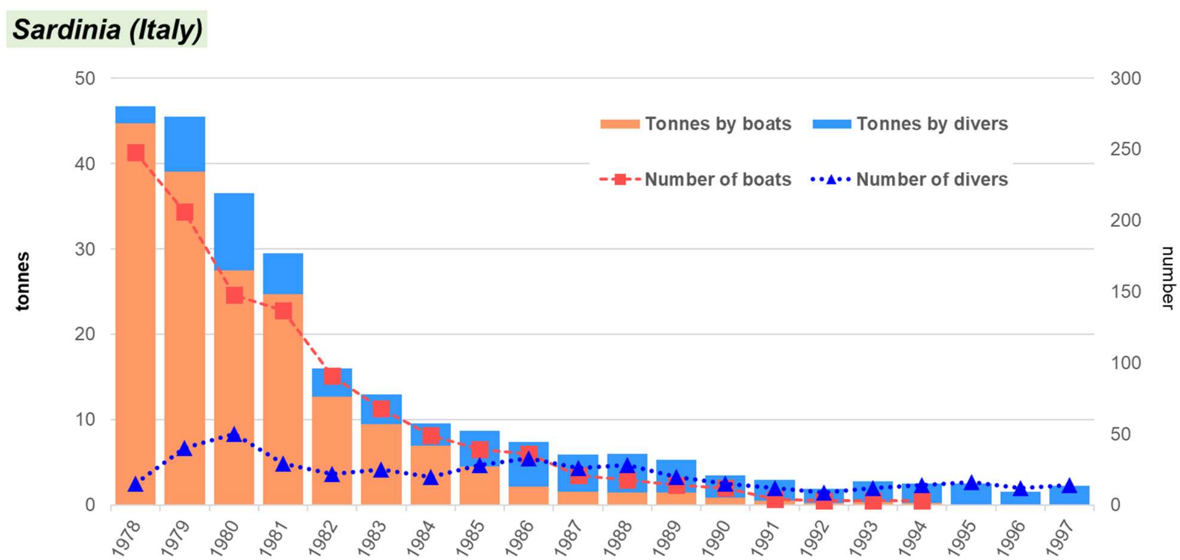

Figure 22 Data on C. rubrum *production for the period 1978–1997 in Sardinia (Italy). Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino 1998).*

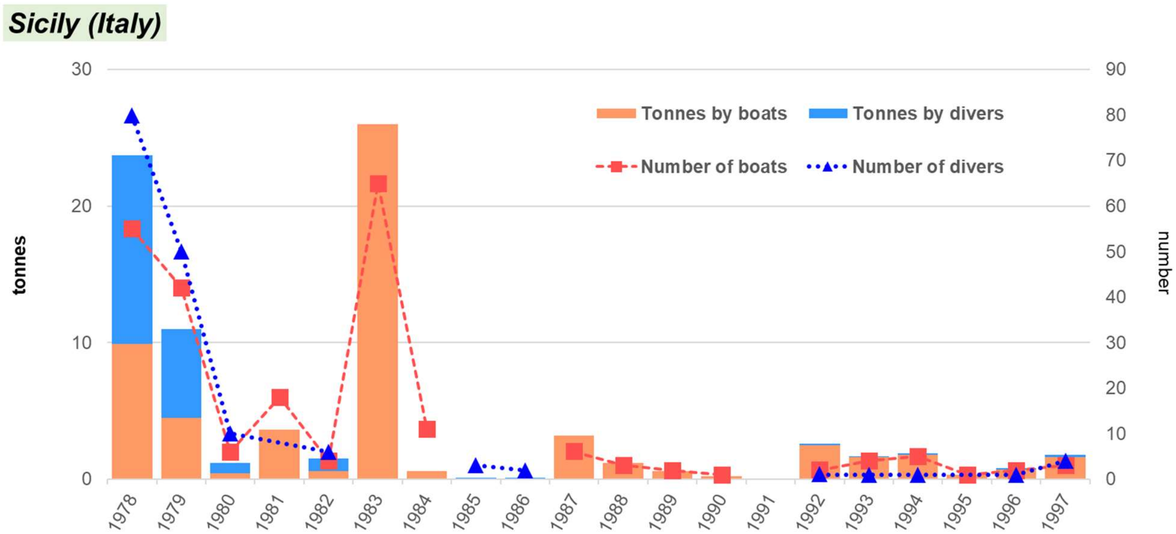

Figure 23 Data on C. rubrum *production for the period 1978–1997 in Sicily (Italy). Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino 1998).*

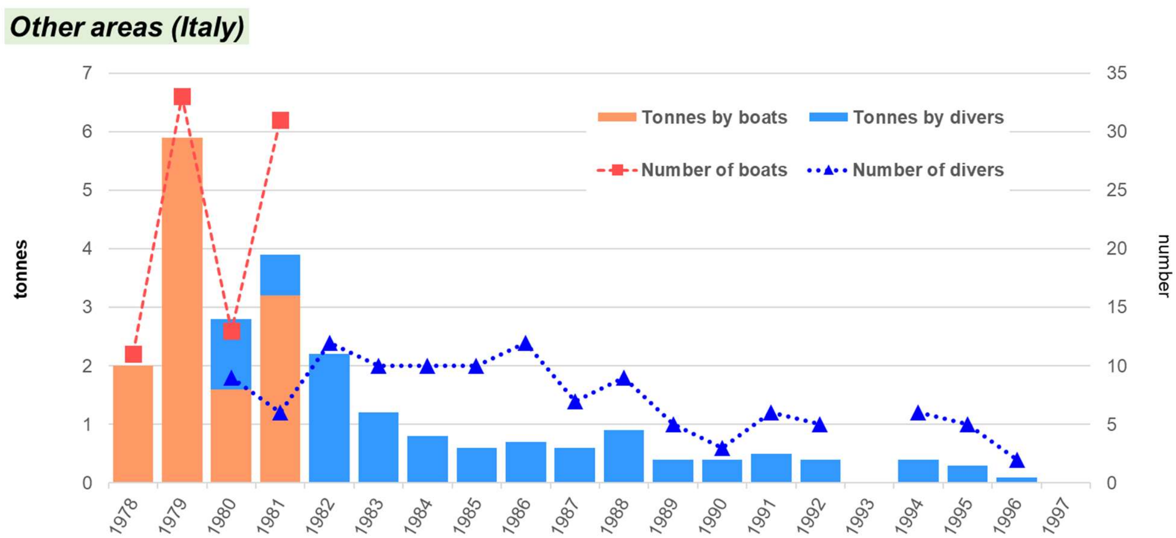

Figure 24 Data on C. rubrum *production for the period 1978–1997 in Italy (excluding Sardinia and Sicily). Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino 1998).*

With regards to Sardinia, in the period 1978–1997 about 70 percent of the Italian red coral was collected there, as compared to only 23 percent coming from Sicily. In 1979, Liverino reports that the collection of an outstanding colony (4 kg of weight, among the biggest ever recorded for *C. rubrum*) in western Sardinian waters (off Oristano) (Liverino, 1998). In the same year the first Sardinian local regulation (Regional Law LR n 59 July 5, 1979) entered into force, aiming to protect the resource from an excessive fishing pressure; it banned the 'ingegno', imposing a fine of 7 millions of Italian lire and limited activity to 5 months. The lobbying activity of manufacturers and fishermen succeeded in modifying the law slightly; the following year two small 'crosses' (3 m in length) were permitted in each boat, the fee was reduced to 2 million Lire and the season extended to 7 months. In

1989, a new law (LR 23/1989) imposing stricter regulations (e.g. the ban of all dragging gears), led to a progressive reduction in to the amount of coral harvested (Figure 22).

In Sicily, two accidental discoveries contributed to increasing yields: in 1978 the large shoal of 'Skerki' in the Sicilian Channel (between Sicily and Tunisia) and in 1983 the 'Terrible Bank' off Pantelleria (three miles east of the historical banks of Sciacca). In particular, the Skerki bank was regarded as a coral 'el Dorado'. About 80 divers of different nationalities (French, Spanish and Italian) along with 20 boats collected 10 tonnes in only the first year of exploitation (1978). The following year the number of boats (13), divers (50), and yield (6.5 tonnes) declined (Figure 23). All the coral was landed in Trapani (Sicily) and worked there. Greeks and French divers (1972–73) probably exploited this bank years earlier (Liverino, 1998). In March 1983, a single 'corallina' boat collected 3.5 tonnes near the 'Pantelleria' coral. Two months later there were 50 Italian boats working there. However, 'Pantelleria' red coral was almost all dead ('decaduto'): with empty branches and small in size, it was all collected in a few months (Figure 23).

Cattaneo-Vietti *et al*. (2016) provide a detailed overview of all recent and historical knowledge on landings, fishing and geographic distribution of the red coral banks along the Italian coasts from 1861 to 2014.

Today, fishing is limited to Sardinia and the Sicily Channel, while the banks off Liguria, Tuscany Archipelago, Latium, Gulfs of Naples and Salerno, Calabria, and Ionian Sea, which once produced a significant amount of red coral, fishing has allegedly been abandoned because the yields have become negligible (Cattaneo-Vietti *et al*., 2016). According to the GFCM Mediterranean and Black Sea production database, in recent years (2007–2014) over 76 percent of harvesting has occurred in Sardinia. Official Italian data exist for Sardinia only. In 1996 the Autonomous Region of Sardinia started collecting information on catches through the logbooks compulsorily filled out by divers at the end of each harvesting season. Figure 25 illustrates the annual amount of red coral in Sardinia (official data by RAS (Regione Autonoma della Sardegna)) compared to the figures available in the FAO dataset. In the period 1996–2010, total catches amounted to some 37.6 tonnes (official data from RAS), less than half of that reported in the FAO datasets (88.5 tonnes). The main differences from the two sources are in 2003 and since 2006. In particular, in 2007, the Regional Government decided to forbid red coral harvesting, applying the precautionary principle to better preserve the biological resource assessing the need of deepening scientific studies on stock condition in Sardinian coastal waters, while data are available in the FAO database. Moreover, in 2006 and in 2012–2017 the limitation of the harvesting period and a decrease in the number of authorized (active) divers led to a marked decrease in the annual coral harvest in Sardinia. In 2017 the number of fishermen actively harvesting *C. rubrum* was only 14, for a total of about 780 kg of red coral (Source: RAS Regione Autonoma della Sardegna, 2018).

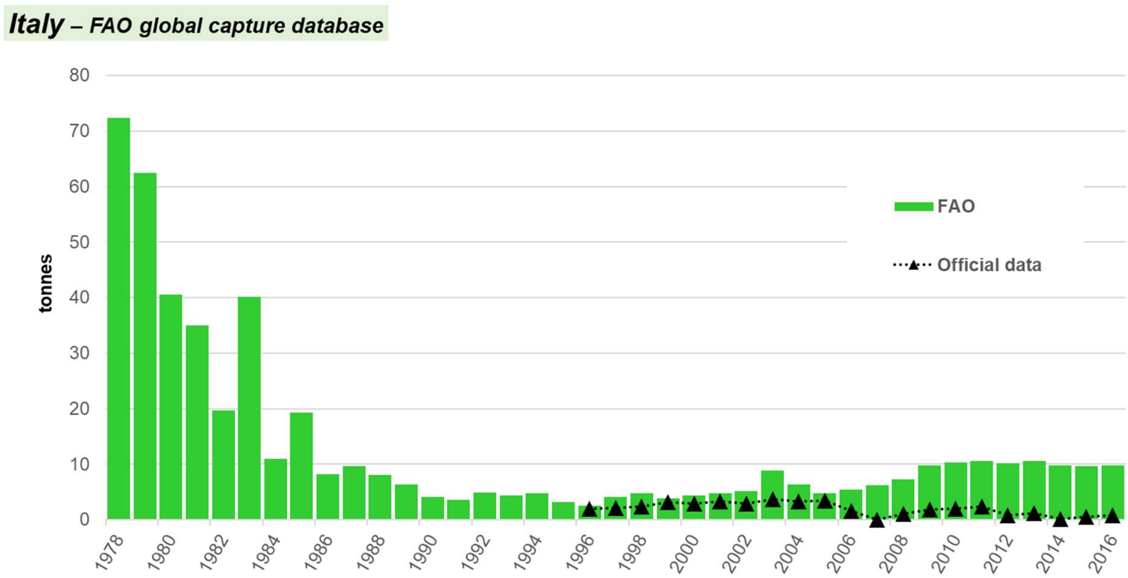

Figure 25 Data in FAO global capture database for C. rubrum *in Italy; the black traiangles correspond to the official data provided by RAS (Source: FAO 2018; Regione Autonoma della Sardegna 2018).*

3.2.1.6.8. Malta

According to the data presented during the Second GFCM Technical Consultation on Red Coral in the Mediterranean (GFCM 1989), very small amounts of coral (0.8 tonnes) were collected in Maltese waters in the period 1985–1987. Only 1–2 boats and one diver were operating in the country (GFCM, 1989).

3.2.1.6.9. Montenegro

The FAO global capture database records data for the period from 2006 to 2016 (Figure 26) even if *C. rubrum* is protected under the Decree on putting the protection on certain plant and animal species (Official Gazette RM no. 76/06) issued by the Institute for Nature Protection 12.12.2006. Moreover, since 2015, it is explicitly forbidden to catch *C. rubrum* in the fishing sea of Montenegro (Article 18 of the Law on Marine Fisheries and Mariculture (Official Gazette of Montenegro N.56/09; LEX-FAOC151759)). Therefore, the FAO data again provide unofficial and unconfirmed records.

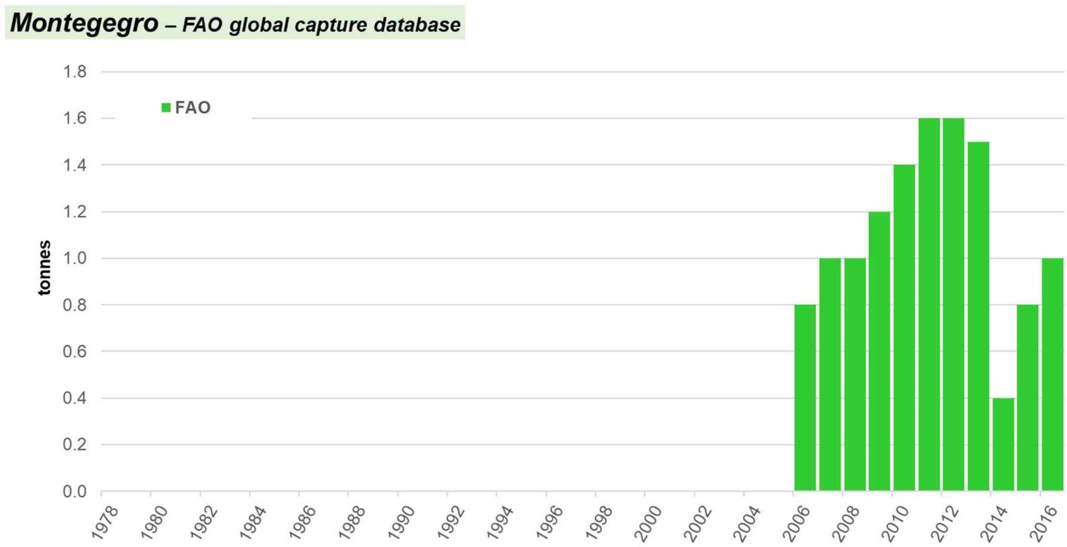

Figure 26 Data in FAO global capture database for C. rubrum *in Montenegro (Source: FAO 2018).*

3.2.1.6.10. Morocco

As reported in the GFCM report (GFCM 1984, 1989) Morocco prohibited coral fishing as a general rule, allowing it occasionally on a discretionary basis.

Until the 1980s, aside from any traditional fishery for red coral that occurred in Morocco, the only harvesting being conducted was by Italians that had fishing contracts with the Government of Morocco; they benefited from a limited number, dependent on the fishing licenses issued within the framework of chartering of their ship. For instance, in a five-month period (June to November 1980) a single Italian company collected a total of 667 kg of red coral in the bank of Tofino (Berraho in GFCM 1984).

The exploitation of coral began in the 1980s by foreigners in the zone of Al Hoceima (Mediterranean coast). During this period 2–10 boats, often of foreign provenance (corailleurs) exploited red coral (Dridi, 2009). In the 1970s foreign boats illicitly undertook coral fishing in the same zone, at the level of the Mediterranean. In fact, at the end of the 1970s Italian divers found traces of the passage of Saint Andrews cross at the bottom (El Alloussi and Zoubai in GFCM 2010).

The first gear used in the exploitation of red coral was the Saint Andrew's cross; however, during the 1980s this technique was prohibited and replaced by SCUBA diving (Dridi *et al*., 2010).

The zone of Al Hoceima was the most prosperous zone for red coral exploitation (8.6 tonnes in 1985), which was completely exported raw. In three to four years production decreased (Dridi 2009). At the end of the 1980s the fishing of coral become unprofitable in Al Hoceima (El Alloussi and Zoubai GFCM, 2010).

In terms of demographic structure the Mediterranean area was no longer interesting anymore because the collected colonies had significantly reduced in size. Consequently, their commercial value decreased, which encouraged the 13 units exploiting the coral to change their zone of activity from M'diq towards the Atlantic zone (Dridi, 2009).

In 1988, a large bank of coral was discovered in the Atlantic waters of Morocco; it was exploited starting from the following year (Figure 28) (Liverino, 1998). In 1990, the harvesting of coral was concentrated around Asilah (between Cape Spartel and Larache); after 13 years of exploitation, the zone was closed to exploitation (Dridi *et al.,* 2010).

The Liverino figures (Liverino 1998) sometimes differ with those provided by Dridi (Dridi, 2009), but the general trend of catches is confirmed (Figure 27, Figure 28).

During the 1990s, the application of new conditions for coral fishing enabled a reduction of the number of fishing boat to 10 units (and only of Moroccan nationality). In this period, coral landings continued to fall, with the number of trips in 1991–1992 ranging from 9 to 23 trips/month. The catch per trip shows a variability ranging between 2 and 6 kg. In 1992, out of the 146 trips carried out, 80 percent were in the area of Tofinio with a catch of 3.4 kg/trip, and 20 percent of the trips were in the area of Topo with a harvest of 2.1 kg/trip. The average harvest was 3 kg/trip (Dridi, 2009).

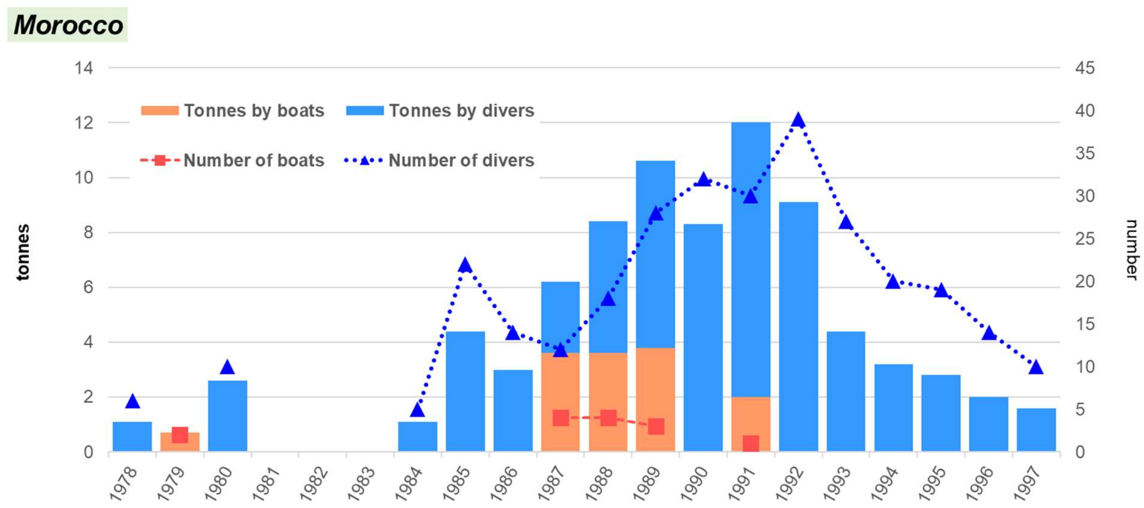

Figure 27 Data on C. rubrum *production for the period 1978–1997 in Morocco (Mediterranean and Atlantic waters). Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino, 1998).*

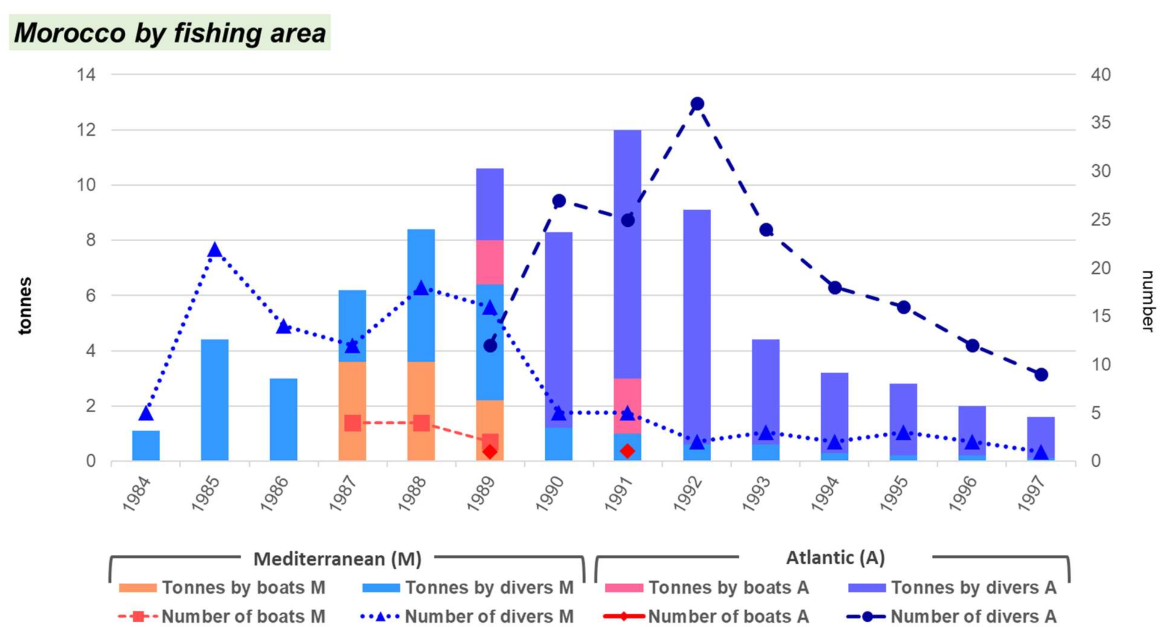

Figure 28 Data on C. rubrum *production for the period 1978–1997 in Morocco (Mediterranean and Atlantic waters separately). Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino, 1998).*

In the 1980s harvesting was exclusively undertaken in the Mediterranean, later the fishing was closed there from 1993 to 2003 (Dridi, 2009), while in the same period it was open in the Atlantic. Since 2004 the Mediterranean (Al Hoceima) was opened again for coral fishing (Dridi *et al*., 2010). In 2010 Mediterranean banks were protected and the fishing units moved again to the Atlantic.

Nowadays the Atlantic and the Mediterranean Moroccan coast have very important red coral resources (Dridi *et al*. 2010). In the Atlantic, Asilah region the fished coral is located on rocky bottom in a depth extending between 40 and 120 m. The depth of exploitation is increased from 40 metres at the beginning of the 1990s to 95 metres in 2001. During the last years of fishing, 57 percent of the production is fished at depths of 65–85 m (El Alloussi and Zoubai in GFCM 2010).

In the Mediterranean, the Topo region (15 miles west of Al Hoceima and one mile off the coast), the fishing area is in the range 35–85 metres. In 2001, 83 percent of the production is fished at depths of 55–75 m (62 percent 65–75 m and 22 percent 55–65 m) (El Alloussi and Zoubai in GFCM, 2010).

In the Mediterranean, the Tofinio region (approximately 24 Miles east of Al Hoceima), the coral is on the sides of rocks beyond 100 metres of depth. In 2001, 87 percent of the production is fished at depths of 95–115 m (66 percent at 95–105 m and 20 percent at 105–115 m) (El Alloussi and Zoubai in GFCM, 2010).

Generally, the exploitation of red coral in Morocco, both in the Mediterranean and in the Atlantic, shows a typical trend: the production reaches a maximum after the opening of the zone of activity and starts declining progressively (Dridi *et al*. 2010).

According to the information presented by the Ministry of Agriculture and Fisheries, Fisheries Marine Department and the National Institute of Fisheries Research (INRH) during the GFCM First Transversal Workshop on red coral, the amount of declared catches in period 1980–2009 totalled about 135.3 tonnes (73 tonnes in the Atlantic and 62.3 in the Mediterranean) (El Alloussi and Zoubai in GFCM 2010). In particular, 16.8 tonnes were harvested in the Al Hoceima area in first 10 years of exploitation; 66.5 tonnes were collected in Asilah in following 12 years; 45.5 tonnes in Al Hoceima in the next 7 years, and finally 6.5 tonnes in last years in the Atlantic waters (Asilah) (El Alloussi and Zoubai in GFCM 2010).

For the same period the figures for Morocco in FAO database are quite close to the official recorded landings: a total of 126.35 tonnes (67.45 in the Atlantic and 58.9 in the Mediterranean) (Figure 29).

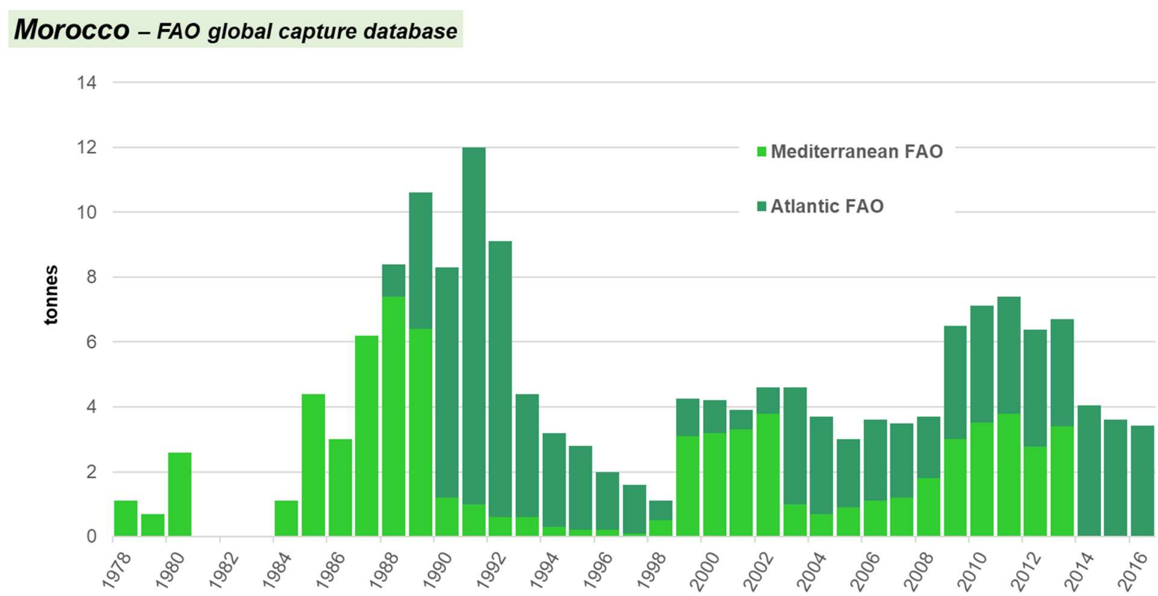

Figure 29 Data in FAO global capture database for C. rubrum *in Morocco (Source: FAO, 2018).*

However, in general, the recorded landings for a given year are lower than the amount of coral exported (Figure 30). The differences in the levels of production and exports can be explained by the addition of quantities already harvested in past years (Dridi *et al*., 2010).

Italy imported 77 percent and Switzerland 11 percent of coral product between 1980–2009 (El Alloussi and Zoubai in GFCM, 2010) but from 2010 India absorbs about 35 percent of corals exports from Morocco (GFCM, 2017).

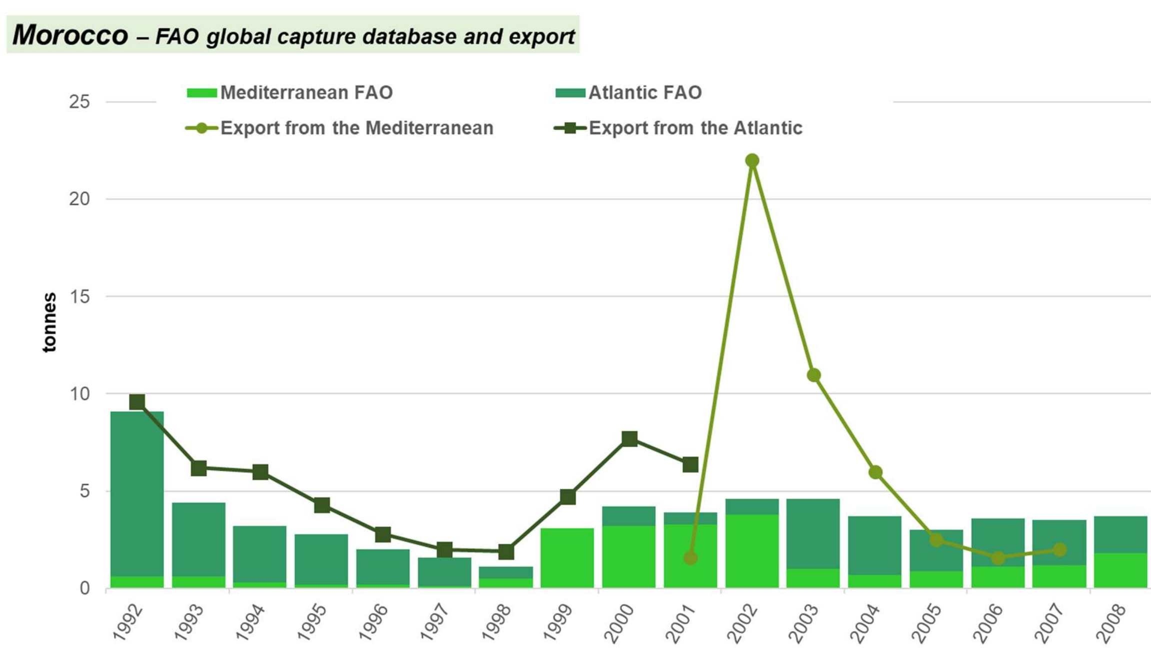

Figure 30 Comparison of production data and export data for Morocco (Source: FAO, 2018; export data from Change Office in Dridi et al.*, 2010).*

3.2.1.6.11. Spain

In the 1980s in Spain red coral was harvested in four areas: the Catalan Sea, the Alboran Sea, the Columbretes Islands and the Archipelago of the Balearics (GFCM 1984). The first was alarmingly overexploited, the second was exploited in an irrational manner, the third was over-exploited by illegal fishermen (both Spanish and foreign), and the fourth was approaching over-exploitation (GFCM, 1984).

In Spain, official statistics on coral fishing have been collected since 1957, but the data are not reliable (GFCM, 1984). It is estimated that the average amount annually fished by divers since this date is about 1 360 kg/year (8 000 kg in 1983) (GFCM 1984).

Data of catches illustrated in Figure 31 are from Liverino's book (Liverino, 1998) and refers to the total production from the Costa Brava (Catalan Sea), Mallorca and the Alboran Island.

For Spain, a major event was the discovery (on 15 September 1981) of a large coral bank off Almeria and Melilla (Alboran Island) by Italian divers; they collected in the first three days about 1 tonne; later the same year 6–7 boats, spread one every 2 miles, started working there and harvested up to 0.1 tonne each day (Liverino, 1998). The following year, despite the Spanish government confiscating some Italian boats, they evaded surveillance and kept on harvesting. In 1982, a total of 21 tonnes were collected both by boats and divers (Italian and French) (Figure 31). In 1984, four Almerian boats were allowed to use the barra italiana in the protected area of the Isla de Alborán, this type of gear was used in the Almeria banks from 1984 to 1986. The quota was fixed at 700 kg/boat/year, and these high yields led to the emergence of local processing industries in Adra and Almeria. However, impact studies on biocoenosis, made by the Spanish Institute of Oceanography in 1986, found high rates of mortality among the coral-associated fauna leading to the ban of this gear while licenses were still granted to divers and underwater devices.

In 1987 a submarine was used for the first time in Spain (Alboran Island) for harvesting corals; according to Liverino (1998) it collected about 3 tonnes in the period 1987–1993 (Figure 32).

Because the area around the Isla of Alboran is characterized by the existence of strong currents and adverse weather conditions the harvesting by SCUBA divers was difficult and often impossible, halting their work for several months, except for the summer months. The average yield was, in the period 1990–1995, of 174.2 kg/diver/year, quite below the maximum legal quota of 400 kg/diver/year. The highest peaks of harvesting were located in the period between April and November. With a total of 113 days worked during 1990–1995, the average activity for the year is 27 days/year.

Intrusion of foreign poachers in the territorial waters of Spain around the Isla de Alboran, lead to the initiative of proposing the listing of *C. rubrum* in the CITES Appendix II (CITES CoP6, Ottawa, Canada 1987). The proposal was rejected but led to several meetings organized by GFCM to improve the management of the species in the Mediterranean Sea.

With regard to the Catalan Sea and Balearics Archipelago, data on catches are shown in Figure 33. Spanish divers started exploiting their Atlantic coral banks only later (since 1996, Figure 34).

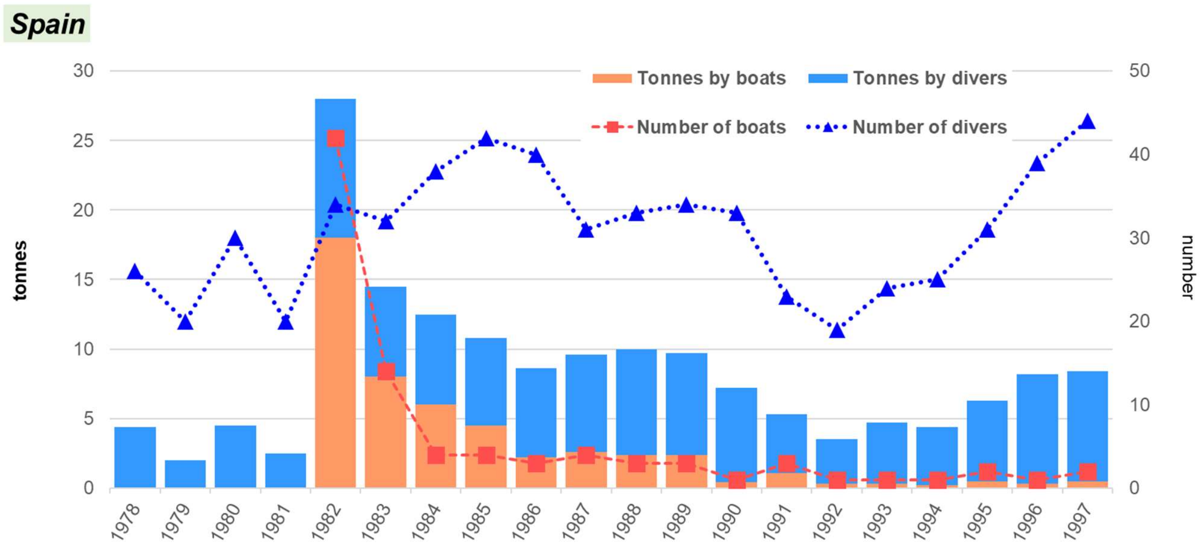

Figure 31 Data on C. rubrum *production for the period 1978–1997 in Spain (Mediterranean and Atlantic waters). Number of boats (using dredging gears and including a submarine), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source Liverino, 1998).*

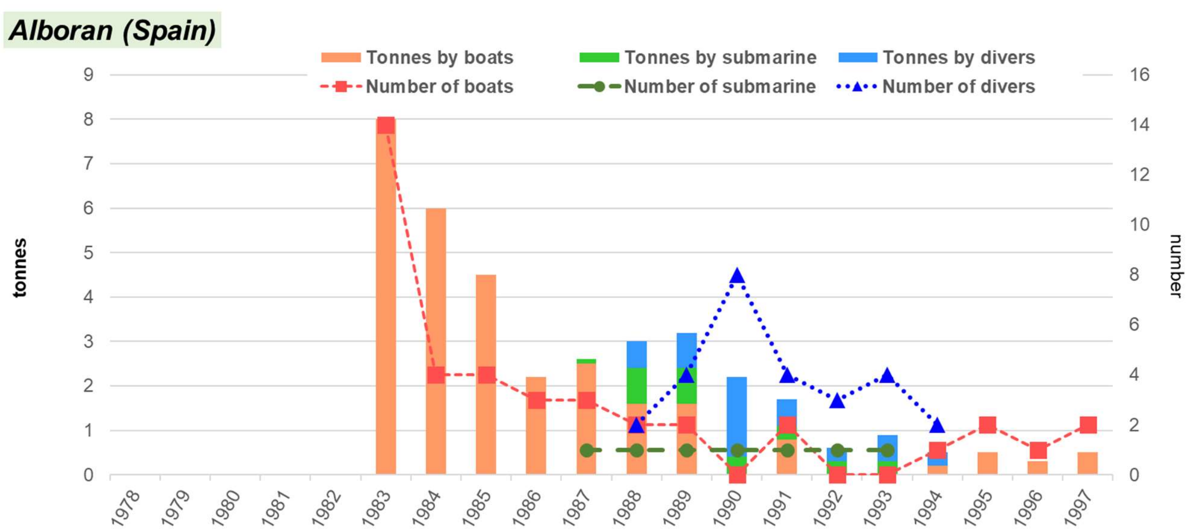

Figure 32 Data on C. rubrum *production for the period 1978–1997 in Spain (Alboran Islands, western Mediterranean). Number of boats, divers (using picks), and yields (tonnes) by the different categories are illustrated. The quantities taken by the submarine are given separately from those of boats using dredging gears (Source: Liverino 1998).*

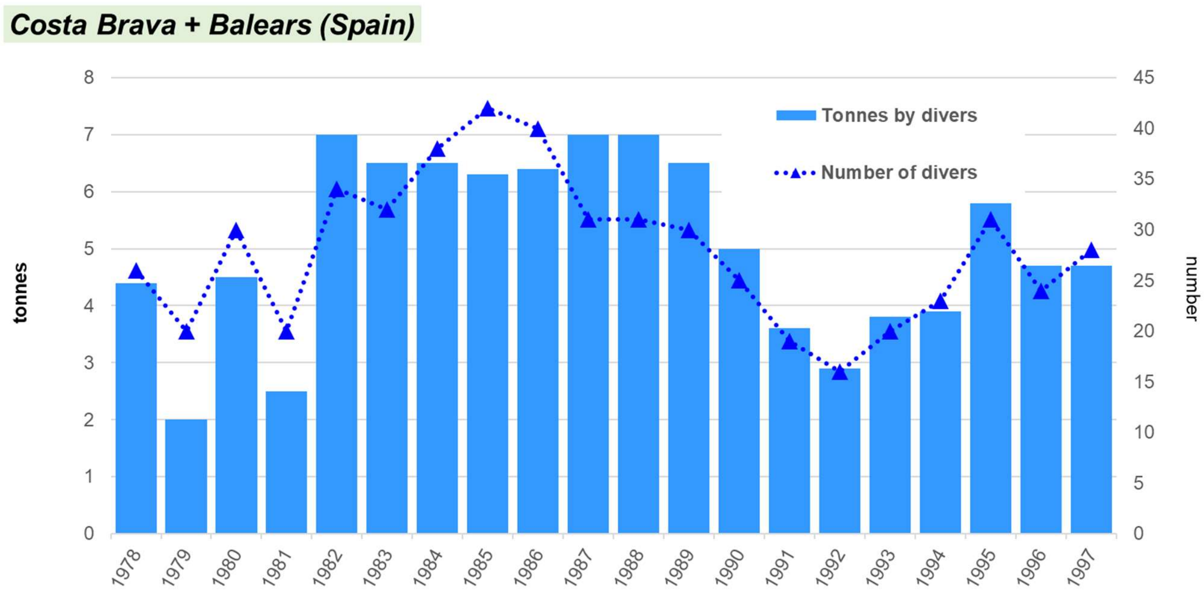

Figure 33 Data on C. rubrum *production for the period 1978–1997 in Spain (Costa Brava and Balearic Islands, western Mediterranean). Number of divers (using picks), and yields (tonnes) are illustrated (Source: Liverino, 1998). In the period 1978–1982 data are exclusively from the Costa Brava.*

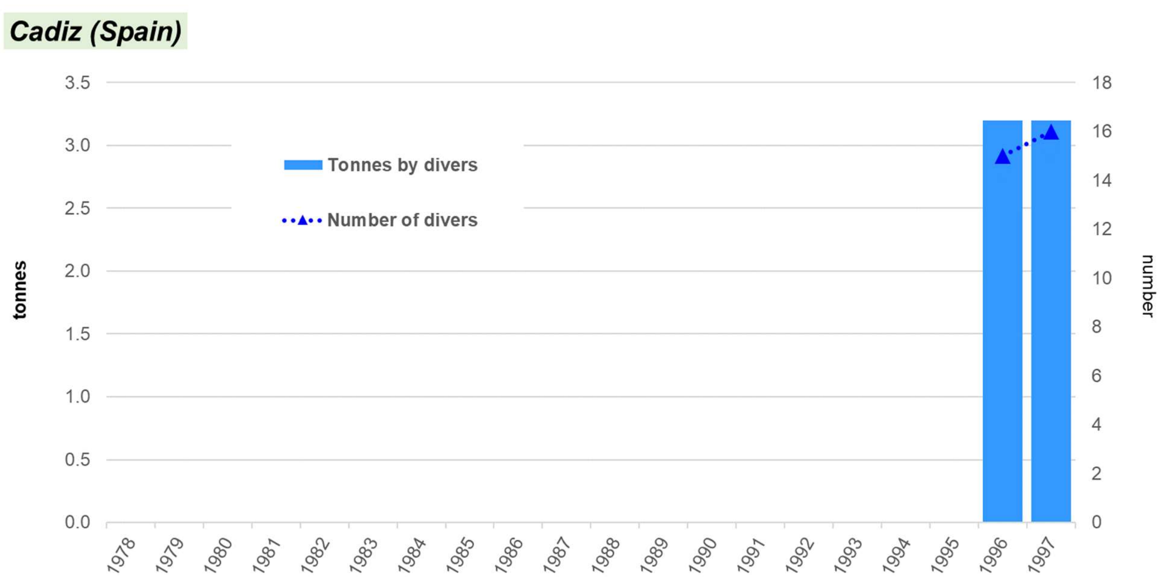

Figure 34 Data on C. rubrum *production for the period 1978–1997 in Spain (Cadiz, eastern Atlantic). Number of divers (using picks), and yields (tonnes) are illustrated (Source: Liverino, 1998).*

Recent data on landings (kg), effort (number of dives), depth range, and average diameter (mm) provided by the Secretaria General De Pesca to GFCM are shown in Figures 35. They are generally much lower than the FAO data.

On overall, the amount of coral harvested per year has increased, as well as the kg/dive, while the diameter of colonies decreased. The highest increments in catches have been documented in the Balearic Islands (Mallorca and Menorca), where the figures almost doubled in 2015 (GFCM 2017).

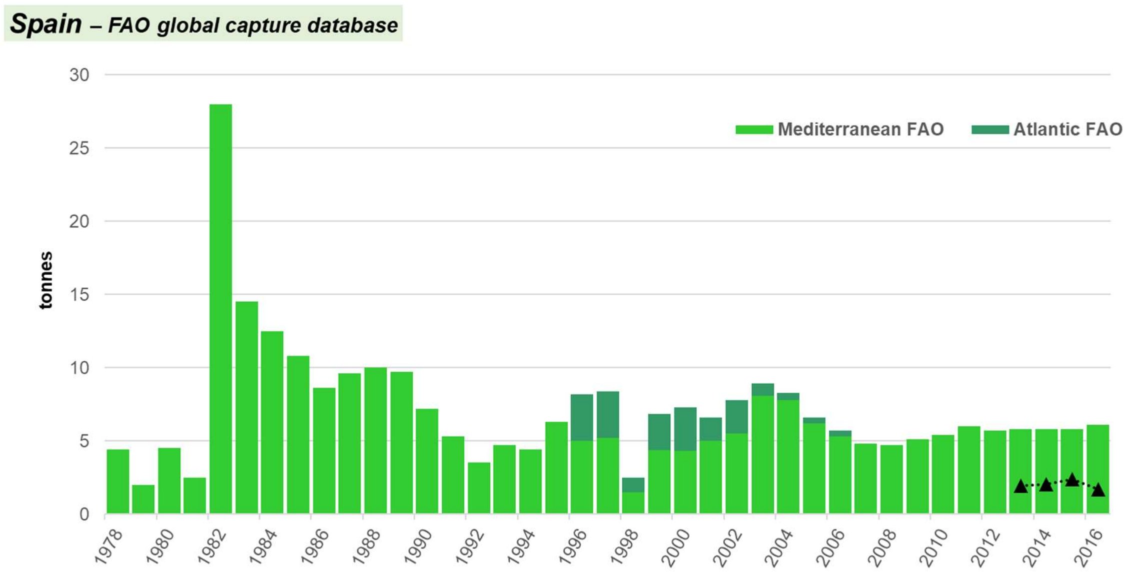

Figure 35 Data in FAO global capture database for C. rubrum *in Spain; the black triangles correspond to the official data provided by Spain to GFCM (source FAO 2018; GFCM 2017).*

3.2.1.6.12. Tunisia

During the period from 1832 to 1956 (the year of the independence of Tunisia from France) the exclusive right of exploitation of Tunisian banks rested with France (Mustapha in Annex N, GFCM 1989). The harvesting of red coral occurred from Porto Farina to Calle (Algeria). According to Liverino (Liverino, 1998) since 1978, the discovery of a large bank off La Galite (Tunisia), intensively exploited in the following years, attracted a large number of boats; the landings decreased quite rapidly in the subsequent years (Figure 36).

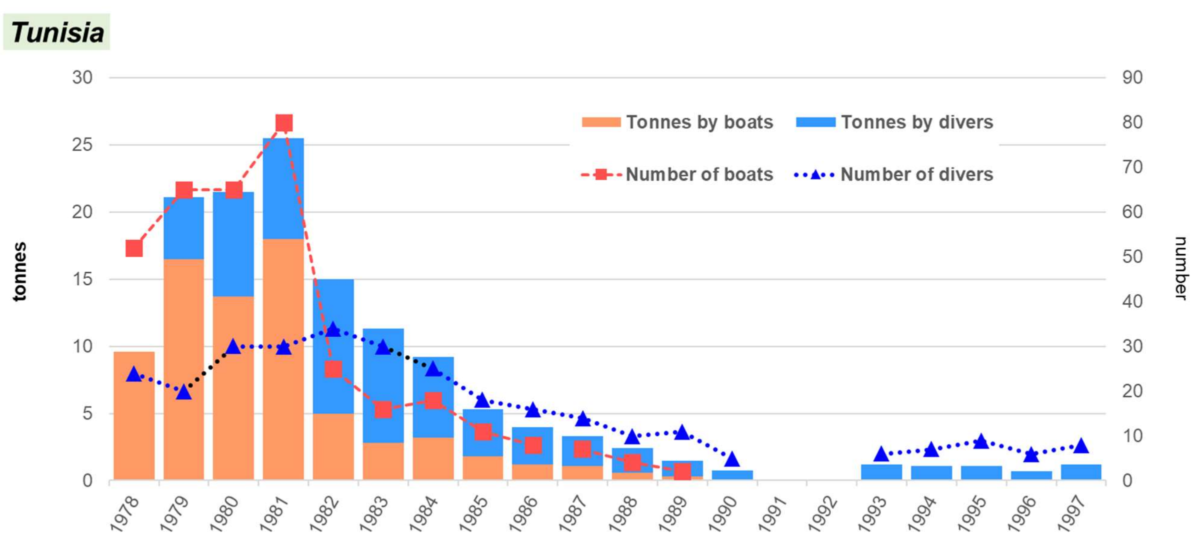

Figure 36 Data on C. rubrum *production for the period 1978–1997 in Tunisia. Number of boats (using dredging gears), divers (using picks), and yields (tonnes) by the two categories are illustrated (Source: Liverino 1998).*

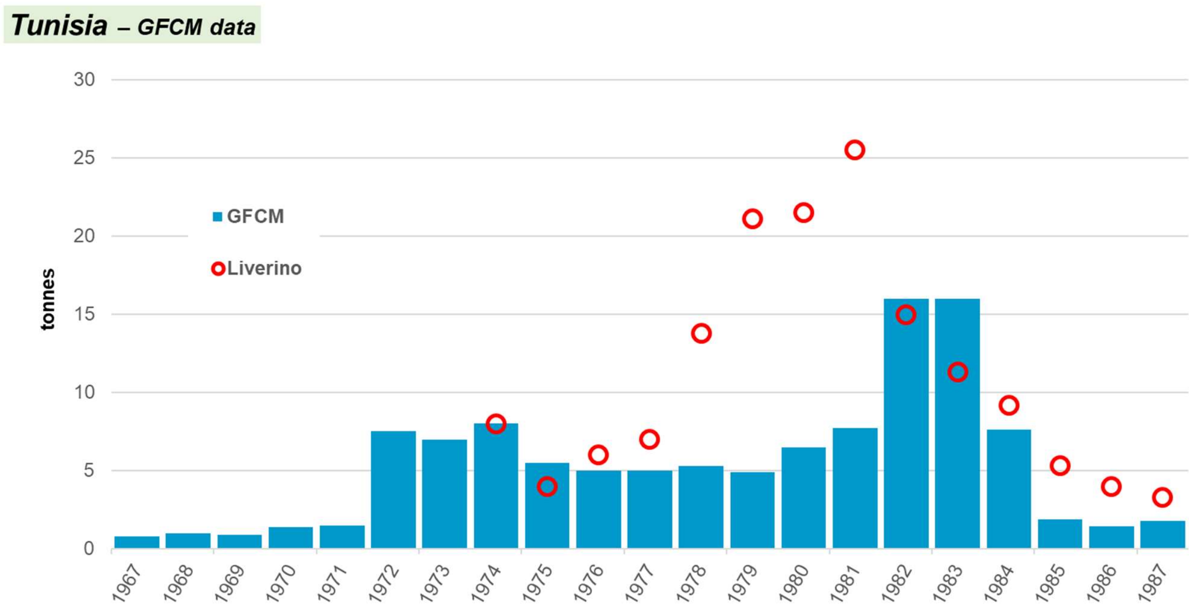

Figure 37 Data on C. rubrum *production for the period 1967–1987 in Tunisia (Source: Table I by Mustapha in GFCM 1989; Liverino 1998).*

According to the information presented in the Second GFCM technical consultation on red coral (Mustapha in GFCM 1989), the Tunisian national yield started to increase in 1972 (7.5 tonnes) and then decreased again in 1975 (5.5 tonnes). The catches increased again during the 1980s (the peak was reached in 1981 according to Liverino or in 1983 if the GFCM 1989 data are considered) (Figure 37).

In the 1980s the fishing zones were concentrated from Bizerte to the border with Algeria. The local yields in the two areas of Bizerte and Tabarka experienced a similar evolution: a rapid increase from 1979 to 1983, followed by a decrease in production (Mustapha in GFCM 1989) (Figure 38). In Bizerte, the maximum was reached in 1983 (333.33 kg/license), while the minimum (70.59 kg/license) was recorded in 1987, despite the falling number of authorized divers (Mustapha in GFCM, 1989). The above-mentioned figures suggest the occurrence of an overexploitation of Tunisian banks (up to 70 m of depth) by SCUBA divers (Mustapha in GFCM, 1989).

In 1982, with the decree of the 26 February 1982, Tunisia decided to not release new permits for the use of the Saint Andrew's cross, so that it could be used only by those who already had the license. Red coral fishery using the cross was allowed in the maritime zone of Tabarka, La Galite and Cap Negro, and only below 100 m of depth. Initially, the cross could not be used from 1 April to 15 September of each year. Starting from the 1 April 1985 this type of fishing was forbidden. Moreover, harvesting was prohibited in the areas of Bizerte, from Cap Blanc to Cap Zebit as well as the fishing off "Cani" islands above 50 m of depth.

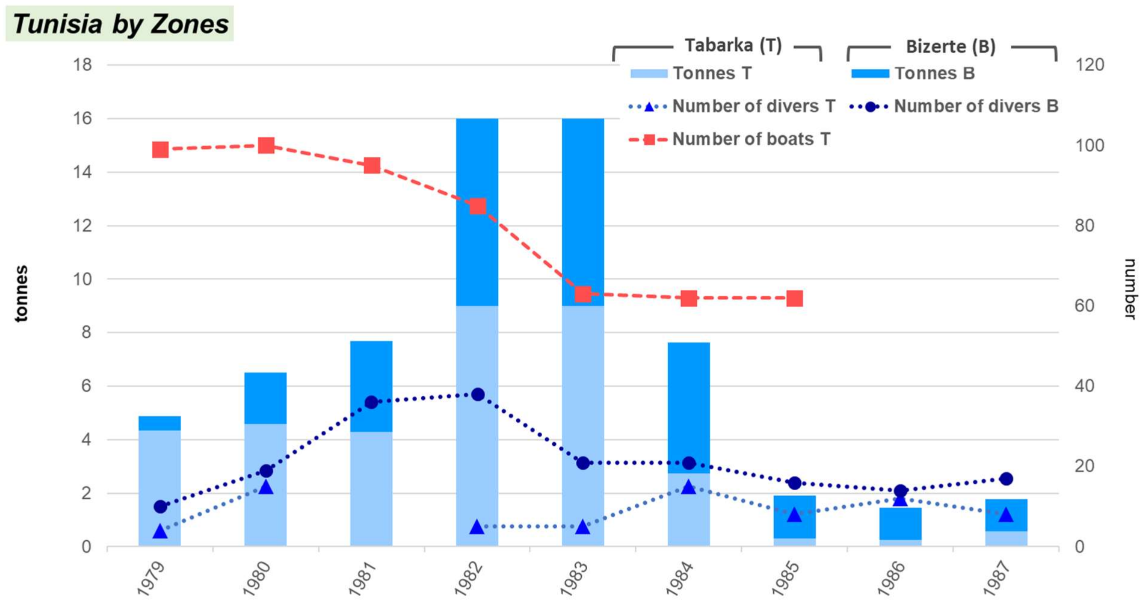

Figure 38 Data on C. rubrum *production for the period 1979–1987 in Tunisia by zones (Bizerte (B) and Tabarka (T)) (Source: Table II and IV by Mustapha in GFCM 1989).*

According to the information provided by Tunisia (Direction Générale de la Pêche et de l'Aquaculture) to GFCM, nowadays, the exploitation of red coral involves about 25 boats with on board 2–4 divers each mainly in the zones of Bizerte and Tabarka.

The official data on red coral catches in the last decade are almost coincident with those reported in the FAO database (Figure 39).

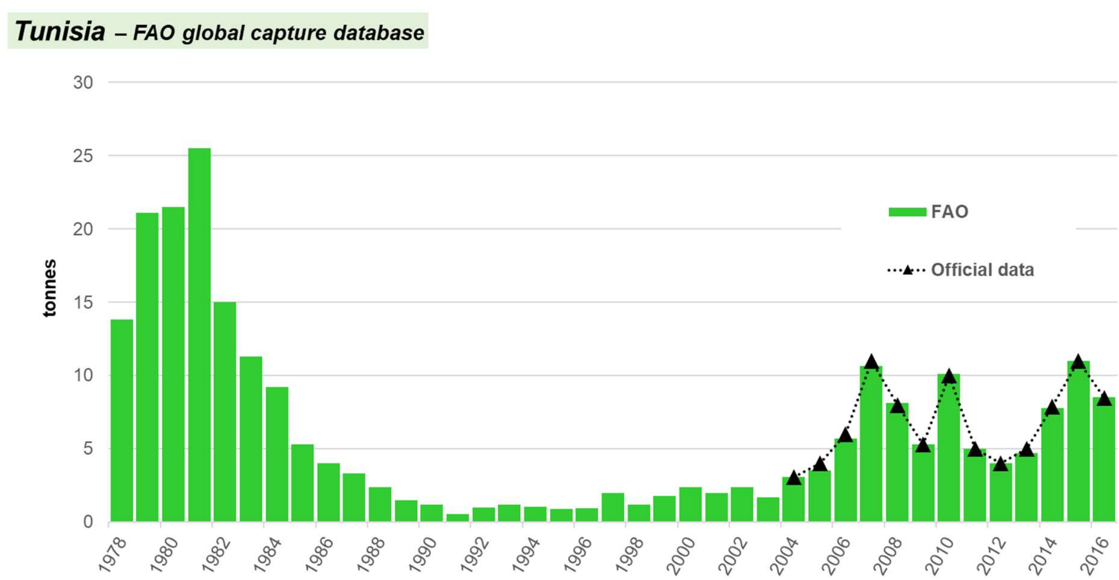

Figure 39 Data in FAO global capture database for C. rubrum *in Tunisia; the black triangles correspond to the official data provided by Tunisia to GFCM (Source: FAO 2018; GFCM 2014, 2017).*

3.2.1.6.13. Turkey

In 1988, it was possible to confirm for the first time the real existence of a few divers working in Turkish waters (Figure 40). The species was collected in the Turkish Mediterranean Sea until 1990, below 40 metres, in banks and marine caves mostly between Anamur and Kas (Öztürk, 2010).

Since 1990, the harvesting of corals is prohibited according to the Turkish legislation governing fisheries (Turkish Fishery Law 1380). The main reason of this prohibition was the destruction of many benthic species wrought by coral divers (Öztürk, 2010). Nowadays, it is prohibited to catch aquatic products by diving, except for sponges (Fisheries Regulation n 22223 10/03/1995, Article 17).

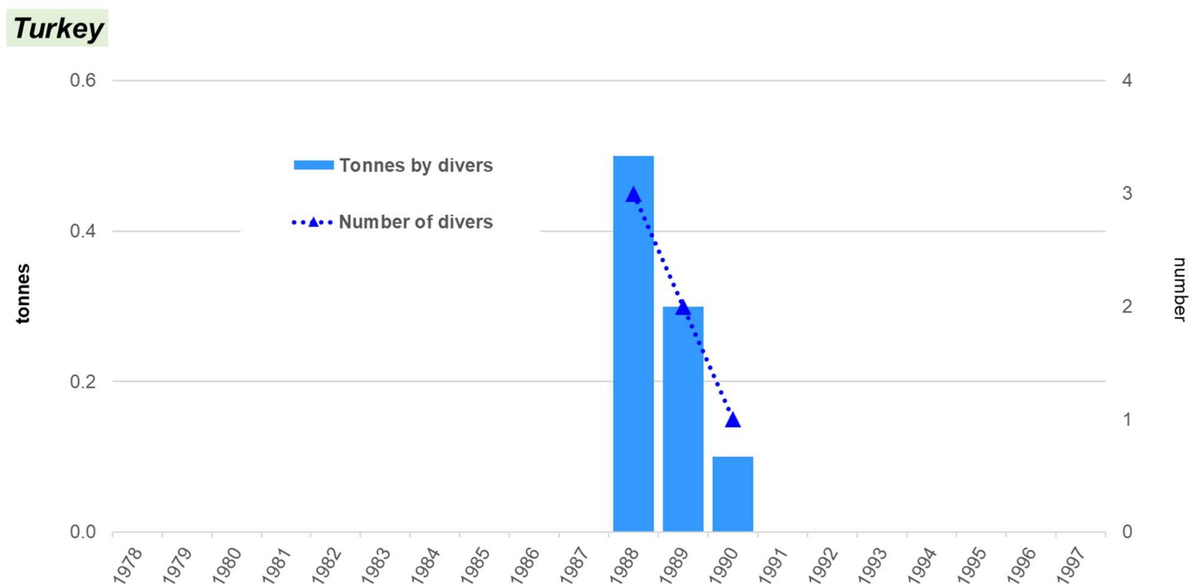

Figure 40 Data on C. rubrum *production for the period 1978–1997 in Turkey. Number of divers (using picks) and yields (tonnes) are illustrated (Source: Liverino 1998).*

3.2.2. Pacific Coralliidae fisheries

3.2.2.1. History of the fishery

The harvesting of precious corals is relatively young in the Pacific area, if compared to the historical tradition of the *C. rubrum* fishery in the Mediterranean. Three countries have been primarily involved: Japan (since the beginning of the nineteenth century), Taiwan Province of China (a century later), and the United States of America (since 1966). Five species have been harvested in different areas, periods and amounts: *Corallium japonicum*, *Pleurocorallium elatius* and *P. konojoi* by Japanese and Taiwanese vessels since the beginning of the fishery; and *Corallium* sp. nov. and *P. secundum* mainly in international waters starting from the mid-1960s to 1990s. The latter species, along with smaller amounts of *Hemicorallium laauense (H. regale*), have also been taken within Hawaiian waters.

Data derived from the FAO global capture database in the period 1965–2016 can be seen in Figure 41 (FAO, 2018). FAO started collecting data in 1950, however the data are incomplete until 1983 for *C. japonicum*, *P. elatius* and *P. konojoi*, started in 1965 for *P. secundum* and later in 1975 for *Corallium* sp. nov., immediately after their discovery.

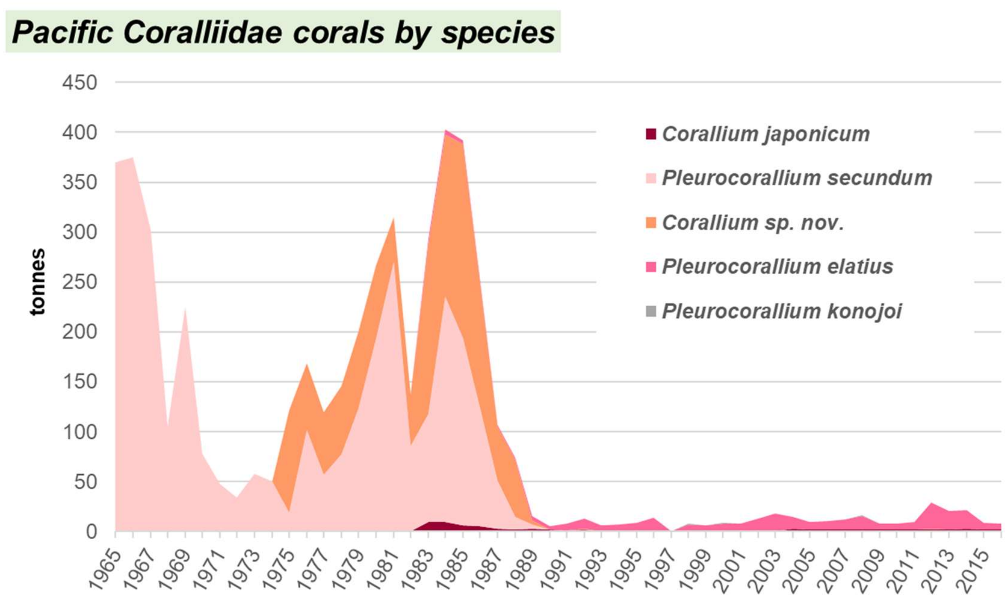

Figure 41 Production data for the Pacific Coralliidae corals in the period 1965–2016 (Source: FAO 2018).

3.2.2.1.1. Japan

The first record of precious coral harvests in Japan was in 1812, when a fisherman found a precious coral entangled in his net off Muroto, Kochi Prefecture (Suzuki, 1999). For the next few decades, as a consequence of a ban on coral dredging under the Tokugawa Shogunate law, coral fishing was very limited. In 1868, during the Meiji Reform, the Tokugawa feudal system was abolished and along with it the Tosa Clan laws banning coral fisheries.[14] By 1871, precious coral fisheries began operating in the Muroto region of Tosa Bay (Kosuge, 1993). The fisheries quickly expanded into western and southern Japan (Okinawa and the Bonin (Ogasawara) Islands), with about 100 boats gathering coral from 100 to 400 m depth (Kitahara, 1904). At first, rectangular nets attached to bamboo sticks were dragged across the bottom, but by the end of the nineteenth century this was modified by Konojo Ebisuya to include additional netting at the rear of the dredge,[15] thereby increasing the efficiency of harvesting (Kosuge, 1993).

Production statistics are available from 1897 to 1925 for Kochi (Annex 2, Figure 11). However, the yearly fluctuations in that period do not necessarily reflect the quantities of resources. In those days, when attractive fishing grounds were discovered in other prefectures, fishing boats moved away from Kochi and production there decreased (Ogi, 2010a). Annual production from the waters off Kochi peaked at 45.3 tonnes in 1902. After the production of 29.6 tonnes in 1909, the quantity declined rapidly to an annual average of 5.8 tonnes (1911–1922; Annex 2, Figure 11).

According to the records, when coral fishing began in the waters off Kagoshima (around 1902–1903), the percentage of live coral in the landings was as high as 81.6 percent for red coral, 67.1 percent for pink coral, and 86.3 percent for white coral (Ogi, 2010a). In a fishing survey conducted near the Ogasawara Islands in 1935, live coral made up 67.0 percent of the 2 844 g red coral sample and 65.5 percent of the 917 g white coral sample. For fishing grounds in southwestern Kochi prefecture (Hata District), there are records for around 1919, roughly 40 years after coral fishing began there in

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[14] This episode is anecdotal and not supported by firm evidence.

1871. These records indicate that live coral made up 20 percent of the catch at that time, a significant drop from earlier years (Ogi, 2010b).

During the Second World War, fishing effort dropped by about 80 percent (Grigg, 1971a; Kosuge, 2010), with the golden era for capture production of Pacific *Corallium* being from 1954 to 1980 (Table 2).

*Table 2 Changes in the fishing ground of the Japanese precious coral (*Corallium *and* Pleurocorallium*) (Source: Liverino, 1983)*

YEAR	SPECIES	LOCATION
1950/51	*C. japonicum* (Aka) *P. elatius*, (Momo)	Hachijo Island (240 km off Tokyo)
1952/53	*C. japonicum, P. elatius*	Amami Island
1952/53	*P. elatius, P. konojoi* (White)	Amami and Goto Islands
1960	*P. elatius*	Okinawa
1961/63	*C. japonicum, P. elatius*	Sumisu and Hachijo Islands
1965	*P. elatius, P. secundum* (Boké)	South China sea
1965	*C.* sp. nov. (Midway coral)	Midway

Specifically, from 1950 to 1955, the harvesting was concentrated on the islands of Shikoku and Kyushu. Hachijo and the islands Amami and Goto were noted as particularly rich in *P. elatius, P. konojoi* and *C. japonicum* (Liverino, 1983). Subsequently, after discovering banks at the Ryukyu Islands south of Kagoshima (Okinawa, Amami) and those souths of Tokyo (Ogasawara, Hachijo, Sumisu), the fleet started making ever-more-distant fishing trips. Small vessels (5 tonnes) were used for the inshore fishery, while long-range larger boats (150–180 tonnes), manned by a 25- to 30-man crew, undertook two- to five-month trips. The boats could employ up to 18 mini dredges simultaneously. The dredges consisted of a round 10 kg stone that had five nets attached to it and could harvest up to 80 kg/day (Liverino, 1983). The fishery reached a peak of 150 t in 1966 but had finally depleted the stocks, and yield remained low for the next 5 year (Grigg, 1994).

During these trips, fishermen expanding their search also into international waters. In 1964, they discovered a very large bed of a new species, the pink coral *P. secundum* in 400 m depth at the Milwaukee Banks, Mellish Bank and surrounding seamounts in the Emperor Seamount chain north of Midway Island, near the Hawaiian Archipelago. However, only about 10 percent of the entire area (the so-called Midway Grounds) lies within the United States of America EEZ (Grigg, 1993).

After this discovery, most of the world's *Corallium* landings came from that bed in huge amounts (about 1 000 tonnes from 1965 to 1967 only) (FAO, 2018).[15] Then a previously undescribed deepwater species (deepwater Midway coral, *Corallium* sp. nov.) was discovered at depths of 900–1 500 m (Grigg, 1993). Harvesting by Japanese vessels of deepwater Midway coral peaked at 100 t in 1975 and finished with less than 1 t in 1986 (Figure 42).

[15] FAO 2018, Fishery and Aquaculture Statistics, Global capture production 1950–2016 (FishstatJ). In: FAO Fisheries and Aquaculture Department [online], Rome, Updated 2018, www.fao.org/fishery/statistics/software/fishstatj/en

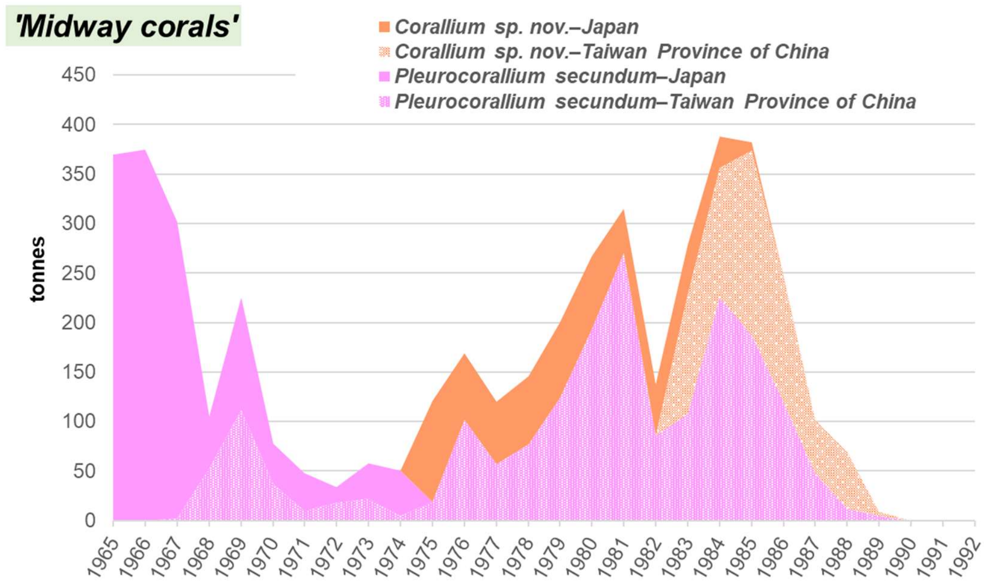

Figure 42 Production data for the 'Midway corals' in the period 1965–1991 (source FAO 2018 modified according to Chang et al., 2013).[16]

The cause of the decline of the Emperor Seamount fisheries has been widely debated. Several reports from Japan and Taiwan Province of China indicate the precipitous declines are due solely to a decrease in effort but not the depletion of the resource (Iwasaki and Suzuki 2010; Chang *et al*., 2013). In the years after, coral fishing has not been resumed in those areas largely because: 1) these corals are less valuable than other species; 2) there are reports of remaining stockpiles; and 3) the costs of fuel and operations associated with offshore fisheries exceeds the value of the coral (Chang *et al*., 2013). Although few biological data exist from these areas, submersible surveys conducted by Japan in 2008 on Emperor Seamounts included areas targeted by coral draggers in the 1960s–1980s. These surveys identified isolated live colonies and dead and broken colonies, but they failed to identify a single large patch of *Corallium* (Fisheries Agency of Japan, 2008 in Bruckner, 2016). It suggests that the resource was overexploited beyond limits of recovery, and even in the absence of fishing for more than 20 years populations have not rebounded (Bruckner, 2016). Japanese fishermen also abandoned coral beds around the Emperor Seamounts in the late 1980s when they became depleted, and many inshore fisheries began to focus more on food fishes due to the higher value, with coral harvest occurring secondarily (Chen 2012; Chang *et al*., 2013).

Even if Japan continued to harvest *Corallium* during the 1980s and 1990s, many of the historic fishing grounds were depleted and landings remained at historic low levels. Areas that were no longer productive included: 1) the Penghu Islands near Taiwan Province of China; 2) continental slope areas near the southern tip of Taiwan Province of China; 3) Oza Bank near Miyako Island; 4) the Danjo Islands near Kyushu; 5) Japan Banks of Tosu; and 6) Smith (Sumisu) Bank, Torishima Island and Sofu Gan located between Japan main land and the Bonin (Ogasawara) Islands (Grigg, 1971).

The largest quantity traded in Kochi was 8.2 tonnes in 1978, but this was followed by a period of continuous decline, reaching a low of 0.4 tonnes in 1993 (Annex 2, Figure 9). The peak of the bid market quantity in 1978 likely reflects the booming coral fishery around Midway Atoll. When the

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[16] Japanese deep-sea coral catches (*C.* sp. nov.) since 1965 up to 1974 (when deep-sea coral has not yet been discovered) have been considered erroneously mislabelled angel skin coral catch (*P. secundum*).

coral fishing in Midway Atoll slowed down, the quantity traded in the market also decreased subsequently.

Nakanishi (2010) points out that the small quantities traded in the first half of the 1990s were due to fishermen giving up coral fishing in favour of fishing for alfonsino (*Beryx decadactylus*). Few fishermen specialised in coral fishing at the time; many of them were also engaged in other types of fishing. Their turning to alfonsino fishing in pursuit of higher profits is thought to have caused the decrease in precious coral trade (Chang *et al*., 2013; Nakanishi, 2010). As the value of precious corals rebounded, harvest has resumed and continues to the present day.

3.2.2.1.2. Taiwan Province of China

In 1923, occasional discoveries of fishing equipment that had hooked coral by Japanese longliners gave birth to the coral fishery in Taiwan Province of China (Huang *et al*., 2010). Industrial coral harvesting started in 1924 in offshore regions of Taiwan Province of China and south of Japan (Chang *et al*., 2013). The coral fishery in Taiwan Province of China attained wealth during the Japanese Colonial Period between 1930 and 1940, and during Taiwan Province of China's Early Restoration between 1960 and 1990, with up to 180 vessels fishing at one time for coral and accounting for over 80 percent of total worldwide production (Huang and Ou, 2010). The industry came to a halt only during the Second World War (from 1941 to 1945) and from 1946 to 1963; it declined from 1991 to 2005 (Huang and Ou, 2010).

Since coral fishing in Taiwan Province of China began in 1924, annual production has peaked three times, once in the late 1960s and twice in the 1980s (Figure 1 in Huang and Ou, 2010; Chang *et al*., 2013). Production exceeded 100 tonnes in the first peak, and 200 tonnes in the second and third. However, most of this was collected near Midway Atoll so these figures do not represent the catch from Taiwan Province of China coastal waters; coral fishing grounds in domestic waters are traditionally relatively small and therefore contain limited corals (Huang and Ou, 2010). Huge hauls from Midway caused coral prices to fall and fishing boats eventually withdrew from the area. Even in recent times, a substantial amount of the stockpiles of low-commercial-value Midway corals harvested in the 1980s remain (Chang *et al*., 2013). Since then, annual production has remained below 20 tonnes.

Fishermen from Taiwan Province of China have mainly used non-selective tools, similar to the gear used by the Japanese, and they continue to use this tool also today. Their coral tangle net dredges are comprised of a cobblestone weighing approximately 30 kg tied with steel wire in a crisscross fashion, with four to five nets attached to the steel wire around the cobblestone. Individual boats may deploy several dozens of these dredges simultaneously, depending on the number of winches and the size of the boat (Bruckner, 2016). Precious corals harvested by Taiwanese fishermen included four species, *C. japonicum*, *P. konojoi*, *P. secundum* and *P. elatius*. After the discovery of the *P. secundum* bed near the Emperor Seamount in 1965, about 200 vessels from Taiwan Province of China and Japan made up to seven different trips per year. The number of Taiwanese fishing vessels was greater than that of Japanese vessels, and they continued to target these grounds for a longer duration (until 1992). Landings from these beds reported by Taiwan Province of China were comparable to those of Japan between 1968 and 1970, but they then dropped off until 1976 when they climbed to values that were two to three times greater than that reported by Japan with peaks in landings of *P. secundum* in 1969 (112 tonnes), 1976 (102 tonnes), 1981 (270 tonnes) and 1984 (225 tonnes), and even higher amounts of *Corallium* sp. nov. from 1983 to 1989 (680 t), most of which was harvested off Midway Island.

During the coral boom in the Midway Islands, the larger fishery of Taiwan Province of China was partially attributed to government fishery enhancement programmes designed to promote and modernize the precious coral fishery industry. At the end of the project, a regulation was implemented to reduce the number of vessels. The combined impact of the new regulation and a substantial drop in the value of *Corallium* could be the cause of a rapid decline in landings and effort during 1982, which later contributed to a rapid collapse of the Midway coral fishing grounds and its abandonment at the beginning of 1989. Despite everything, the Taiwan Province of China fishery continues to be active.

Although only three coral fishing boats were licensed in 2007, an investigation carried out by Taiwan Province of China Fisheries Agency identified 96 unregulated vessels equipped with coral fishing gear that were illegally fishing and not reporting their catch (Huang and Ou, 2010). To address IUU fishing, the government issued new management standards for Taiwan Province of China's precious coral fisheries.

Regarding the harvesting practised in the past in Viet Nam (using a manned submersible south of Ho Chi Minh in 2008, probably currently inoperative), and the Philippines by Japanese and Taiwanese boats, few details of these activities were found (Annex 2).

3.2.2.1.3. Hawaii

In Hawaii, *Corallium* harvesting was initiated in 1965, following the discovery of a large bed (Makapu'u) of *P. secundum* approximately 9 km off Oahu in the Molokai Channel (Grigg, 2010b). Using tangle net dredges, approximately 1 800 kg of *Corallium* was harvested from this bed between 1966 and 1969. Years later, landings of approximately 6 400 kg of coral were collected using a submersible (1973–1978), but as a result of the high cost and low quality of landings, the operation was discontinued. A *Corallium* fishery was revived briefly in 1988 (500 kg were dredged) and later between 1999 and 2001, using a one-person submersible. A total of 1 216 kg *P. secundum* was collected from the Makapu'u Bed, along with 61 kg of *H. laauense* (*H. regale*) from exploratory areas off Kailua, Kona. No harvest of *Corallium* has occurred since 2002 (Grigg 2010b).

Poaching by foreign vessels has been a problem in the past because, during the 1980s coral vessels from Japan and Taiwan Province of China continuously violated the EEZ near the Hancock Seamounts. In 1980, about 20 coral draggers from Taiwan Province of China reportedly poached about 100 tonnes of *Corallium* from seamounts within the EEZ north of Gardner Pinnacles and Laysan Island (Grigg, 1994). However, it appears that since the 1980s poaching within the EEZ by foreign coral fishing has been negligible, in part due to the general fishing activity in the area. Fishing has now been terminated with the declaration of the area as a national monument, so it is not clear if current enforcement will prevent the reoccurrence of poaching (Tsounis *et al.,* 2010 and references therein).

3.2.2.2. <u>The Pacific Coralliidae current fishery</u>

3.2.2.2.1. Japan

The methods currently used for coral harvesting consist of coral nets and ROVs (Annex 2). Coral nets have the same structure as the traditional fishing tools used in the past, the main difference is found in the higher stone weight that has risen from 10 kg in the past to the current weight of 30 kg. In Kochi today, several coarse-mesh nets of just over 1 m in length are attached to either a stone weight (Annex 2, Figure 3) or a chain weight (Annex 2, Figure 4). When stone weights are used, three stones are tied to one rope, with five nets attached to each stone. In the case of a chain weight, four or five nets are attached at regular intervals to a horizontal bar. These are dragged along the sea floor by tidal movements, and corals are entangled in the nets. This method does not allow selective fishing of precious corals according to either species or size depletion (Annex 2).

Some experiments carried out to investigate the real impact of these tools on the sea floor underlined the presence of a real impact. According to Iwasaki (unpublished data), although the impact of a single operation is localized and limited, when multiple fishing boats drag nets repeatedly over a small area, the impact and time needed for recovery are expected to be considerable. As far as ROVs are concerned, unless restrictions are imposed there is a danger that small, immature colonies and those living in places inaccessible to drag-nets may be fished to depletion (Annex 2).

Currently, precious coral harvesting exists in the following Japanese sea areas: off Hahajima in the Ogasawara Islands (Tokyo-to); the Kii Peninsula, Wakayama (Wakayama Prefecture); Cape Ashizuri (Kochi Prefecture); Cape Muroto (Kochi Prefecture); Uwajima (Ehime Prefecture); the Goto Islands (Nagasaki Prefecture); the Uji Islands, Mishima, Tanegashima, Yakushima, Toshima, the Amami Islands (Kagoshima Prefecture); and the Ryukyu Islands (Okinawa Prefecture). Operations are conducted using ROVs in Kagoshima and Okinawa Prefectures and by coral drag-nets in the other areas. The number of vessels that are licensed in these areas is over 500 (Annex 2)

3.2.2.2.2. Taiwan Province of China

According to the "Regulations Governing Fishing Vessels that Also Engage in Coral Harvesting", coral fishing is permitted in 5 sea areas – northeast of Keelung, east of Yilan County, south of Orchid Island, south of the Hengchun Peninsula, and west of Kaohsiung (Huang and Ou 2010, Figure 1) – and there are 60 licensed vessels.

The five permitted fishing grounds in Taiwanese coastal waters cover a total area of 7,811 km^2. This is an average of 120 km^2 per fishing boat, which makes for relatively dense fishing operations. In addition, 60 percent of the permitted vessels operate in one sea area. It is feared that, over time, this situation will further deplete coral resources in the permitted fishing grounds (Huang and Ou, 2010).

The same coral fishing nets are used in Taiwan Province of China as in Japan, but the scale of operations is larger in Taiwan Province of China. In Kochi, Japan, 1 fishing boat of up to 20 tonnes operates with 1 net, while in Taiwan Province of China boats of 20–29 tonnes (around 4–6 fishermen) operate using 6–12 nets (Chen, 2012, 2014; Hu, 2003; Chen, 2015).

3.2.2.2.3. Hawaii

The precious coral fisheries in Hawaii is considered moribund, with the exception of the relatively shallow black corals (*Antipathes griggi*) (CITES annex AC29 Doc.22). However, detailed information on the management, along with areas, gears allowed etc. in provided in paragraph 5.2.1.3.

3.2.2.3. <u>Pacific Coralliidae fishery – landing data</u>

It is difficult to identify the current coral production in the Pacific because data in some areas are collected by national authorities but not disclosed, or else because the systematic collection has started only recently, after the implementation of new management systems (see following paragraphs).

Since 1950, FAO compiles the global capture database, in which annual production by country, species and FAO major fishing area is recorded according to the data officially submitted by countries and additional sources. Statistics for precious corals have been available since 1954. However, there are some discrepancies between data in the FAO database and those from other sources.

In particular it is not clear if FAO production data for precious corals include both live colonies and dead colonies, given that fishermen in Taiwan Province of China and Japan report their harvests under these categories. This could obscure the true impact of harvesting activities that varies, depending on whether colonies were live or already naturally dead when harvested (Shiraishi, 2018).

Bearing in mind all these limitations, FAO statistics are presented over the following paragraphs, considering that they can help somehow to illustrate production and its changes in recent years.[17]

3.2.2.3.1. Japan (FAO data)

It is impossible to accurately identify Japan's total coral production and its fluctuations because data on coral landings in most regions (Wakayama, Ehime, Kochi, Nagasaki, Kagoshima and Okinawa prefectures) are not disclosed, while those of other regions are incomplete. For instance, the data from Tokyo include reef-building corals (Annex 2).

It is unclear if the FAO global capture database includes all catches from undisclosed areas in Japan. Moreover, discrepancies are evident, especially when Japanese FAO data are compared with those from other sources (Annex 2, Figure 9). For example, comparing the FAO data for 1983–1991 with Grigg's (1993), some years are roughly in agreement, while others differ by as much as about five tonnes (1984; Annex 2, Figure 8). In addition, the FAO reports production of only 0.1 tonnes for Japan for 1997, while the quantity traded at bid markets in Kochi prefecture that year amounted to as much as 1.4 tonnes (Annex 2, Figure 8 and Figure 9). Nonetheless, for 2003–2006, the total quantities traded at bid markets throughout Japan (Kochi, Kagoshima, and Okinawa prefectures) reported by the Kochi Department of Fisheries (2007) differ by only 0.2–0.9 tonnes from FAO data, and the sources report similar yearly fluctuations (Annex 2, Figure 8)

The data from the Fishery Agency of Japan and FAO indicate that the annual production peaked in 1979 (14.9 tonnes) when Midway Atoll fishing ground was found and a significant amount of precious coral was fished there. After that, the fisheries in the area diminished due to the fall of prices. In fact, the catches sharply declined to 3.8 tonnes about 10 years later. It then increased and decreased repeatedly, recovering to 5.3 tonnes in 2010. In recent years, production has remained at around 5 tonnes (Figure 43; Annex 2, Figure 8, Figure 10).

Currently, during the seven years from 2010 to 2016, annual production has not changed significantly: the average annual production in Japan is 5.15 metric tonnes, and Japan accounts for 7.31 percent of the global total (Annex 2, Table 6).

17 FAO 2018, Fishery and Aquaculture Statistics, Global capture production 1950–2016 (FishstatJ). In: FAO Fisheries and Aquaculture Department [online], Rome, Updated 2018, www.fao.org/fishery/statistics/software/fishstatj/en

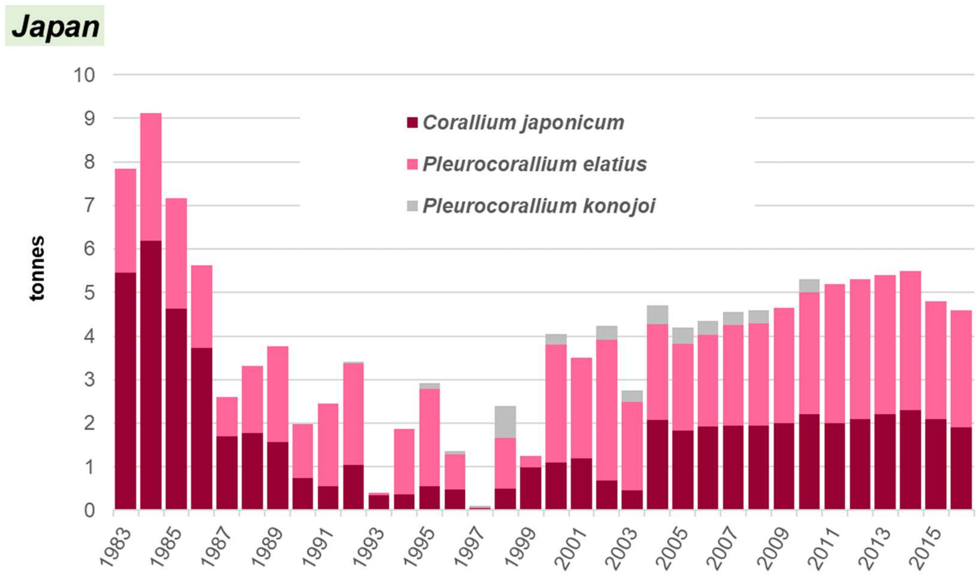

Figure 43 Production data for the Japanese precious corals in the period 1983–2016 (Source: FAO 2018).

3.2.2.3.2. Japan (trade data)

Kochi Prefecture is the birthplace of coral fishing and the principal production area in Japan, so its statistics for precious corals are relatively well maintained. Bid markets are held several times a year, and almost all the precious corals landed in the prefecture are brought there by the fishermen and sold to processors and distributors. Unlike fresh fish, precious corals can be preserved after they are landed. This means sellers can hold on to the corals when market prices are low; there could therefore be some difference in the timing of when corals are landed and when they are traded at a bid market. In addition, off-market distribution has been increasing in recent years. Therefore, the quantity traded at bid markets could not be supposed to be equal to the quantity landed in the same year.

The average annual quantity traded in recent years was 3.6 tonnes (2010–2011), which amounts to only 20 percent of the average annual production of 17.5 tonnes in the peak period (1899–1920) (Annex 2). From 1989 to 2011, the percentage of live coral (red coral, red coral twigs, pink coral, and white coral) in the quantities traded at Kochi bid markets averaged 11.1 percent, while dead corals were 88.9 percent. In particular, dead red coral (*C. japonicum*) averaged 68.1 percent of the annual quantities traded (Annex 2, Figure 12).

From 2010 to 2011 the quantity of live coral traded at bid markets increased for *C. japonicum* (red coral, red coral twigs), and *P. konojoi* (white coral) (Annex 2, Figure 13–Figure 16) in coincidence with a jump in the number of vessels receiving permits for coral fishing, encouraged by soaring prices (Annex 2, Figure 17) due to the large number of boats carefully searching within the sea areas permitted for coral fishing, the discovery rate of live colonies increased, causing the quantities traded at bid markets to rise.

In particular, average bid market prices rose sharply from 2010 after gradually increasing from 1990 (Sakita, 2016). This was due to increased demand from wealthy Chinese as their economy boomed (Machida *et al.*, 2014) (The Nikkei, 2010). The price of red coral, which is preferred in China, soared: in 2015 the unit price of raw coral reached an historic high (585g, JPY 28 890 000, JPY 49 385 /g),[18] with one piece being traded for the highest price ever recorded (3 326g, JPY 91 200 000) (Miyazaki,

[18] JPY (Japanese Yen) 1 = USD 0.009 (OANDA, 1 April 2019)

2015; Four Seasons of Jewelry, 2015). Since then, the average unit price has followed a downward trend but continues to remain high up to 2017 (Annex 2).

3.2.2.3.3. Taiwan Province of China (FAO data)

According to the FAO global capture database, Taiwan Province of China landed overwhelming quantities in the 1970s and 1980s, with 9 out of those 20 years surpassing 100 tonnes. They correspond to the huge amounts of 'Midway coral' fished in international waters and in part in national American waters (see previous paragraph and Figure 41). After that, the territory's production declined. Regarding the other corals (pink, red and white) no official reliable catch record exists for these three species before1983 (Chang *et al*., 2013). Large amounts of pink coral have been reported for the period 1989–2008, with three main peaks (1996: 12 tonnes; 2003: 15 tonnes; 2008: 11.5 tonnes; FAO 2018). Since the implementation of the Management Regulation in January 2009, during the six years from 2010 to 2016, the production did not change significantly being stable at 2–3 tonnes a year (Figure 44; Annex 2, Figure 10).

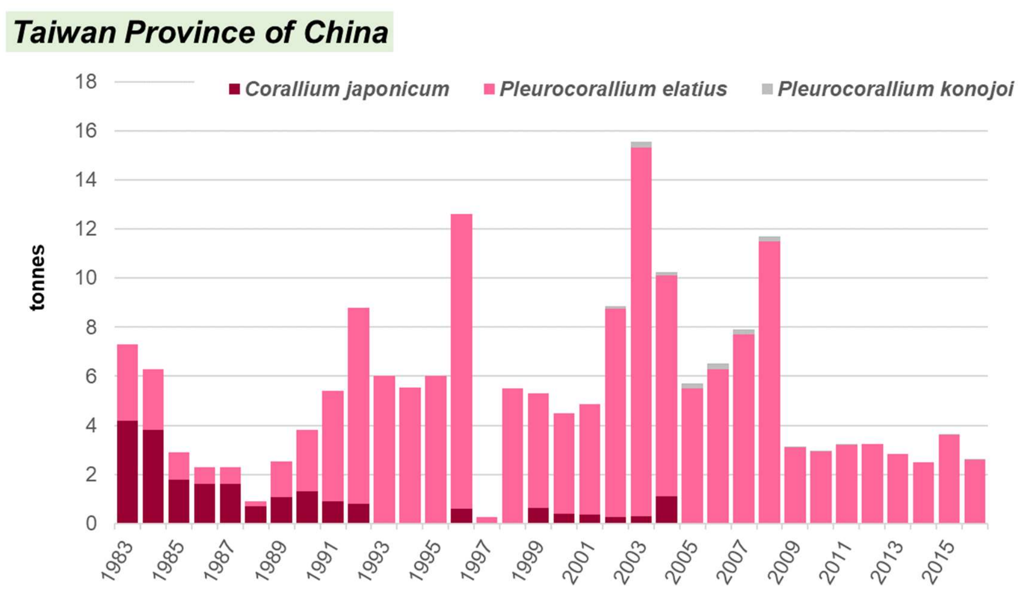

Figure 44 Production data for Taiwanese precious corals in the period 1983–2016 (Source: FAO 2018).

In the period 2009–2012, after the regulations governing coral fishing were tightened and landings were recorded in logbook and inspected, average annual production was 2.9–3.2 tonnes (Chen 2015). Pink coral (*P. elatius*) made up the greater part of the catch (63–86 percent), followed by miss coral (*H. sulcatum*, 1–9 percent) and red coral (*C. japonicum* 3–4 percent) (Chen, 2012; 2015). In 2009, production of live coral was 109 kg, but this fell to 40 kg the following year, and then followed a gradual upward trend (Chen 2015). The percentage of live coral in the catch was as low as 2–5 percent. Pink coral (*P. elatius*) accounted for 60–90 percent of the live coral, followed by red coral (*C. japonicum*) at 6–21 percent (Chen 2015).

Landings of pink coral (*P. elatius*), and especially red coral (*C. japonicum*), varied in different sea areas: in sea area E, while over 11 kg of red coral were landed in 2009, there were no landings in subsequent years (Chen, 2012; 2015). Huang and Ou (2010) point out that the small proportions of live coral indicate deteriorating resources that are ill-adapted for further exploitation. They therefore call for an immediate reduction in the number of permitted vessels and operating days.

3.2.2.3.4. Taiwan Province of China (trade data)

Initially, the price for Midway corals was approximately USD 190 /kg (raw materials) in Taiwan Province of China. The spotty and low-quality corals and excessive catches during the boom years of the early 1980s caused the prices to decline. In the late 1980s, the price declined to the list price of approximately USD 50 /kg with a 10 percent discount in actual trading because buyers were seldom satisfied with the list price. Although the price for stockpile Midway corals (materials) has recently risen to approximately USD 250 /kg, no vessels are interested in fishing for Midway corals now because of: 1) the low quality of the corals; 2) the low price compared to the price of over USD 10 000 /kg for other precious corals; 3) the higher harvesting costs (especially fuel costs) now than in the 1980s; and 4) the limited number of licenses for coral fishing (Chen *et al*., 2013).

Considering the corals taken within Taiwanese water in recent years, data are available from the first four open auctions in Suao since the implementation of the new regulations (2009). Some 279 kg of *P. elatius* (USD 900–2 300 /kg) and 3 kg of *C. japonicum* (USD 1 170 /kg) were traded (Huang and Ou, 2010). Considering an average trading price of about USD 1 500 /kg, and the total catches from January to October it means about USD 8 300 /vessel/month. The net profit after depreciation and overheads such as labour costs, fuel, provisions and maintenance is very little, and in some cases not even achieved (Huang and Ou 2010).

3.2.2.3.5. China (FAO data)

Records are available in the FAO database for China from 2011 despite the fact that precious corals are designated as protected species and fishery is therefore prohibited in Chinese waters (see previous paragraphs). It is unknown whether the statistics are faulty, whether certain coral fishing is legally allowed or whether FAO statistics includes also IUU data. (Annex 2) (Figure 45).

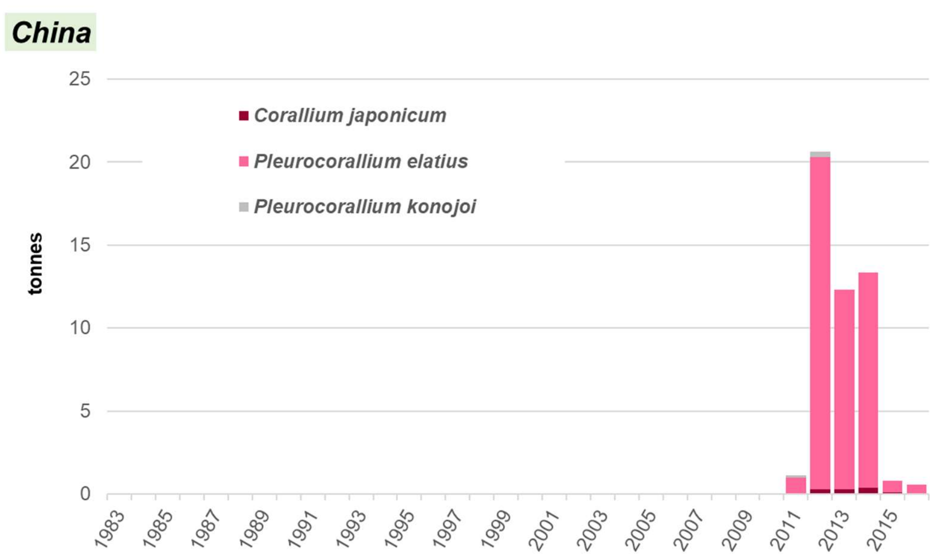

Figure 45 Production data for Chinese precious corals in the period 1983–2016 (Source: FAO 2018).

In fact, poaching started to be documented since 2014 around Ogasawara Islands and was mostly conducted by Chinese boats, as the request of precious coral exponentially increased in line with the thriving Chinese economy (Machida *et al*., 2014; Japan Coast Guard, 2015). By late 2014 the Japanese and Chinese governments agreed to a definitive and consistent crackdown, employing stronger measures such as strict penalties for offenders, in order to stamp out illegal coral fishing by Chinese boats. As a result, no more poaching incidents were reported near the Ogasawara Islands after late January 2015 (Annex 2). Japanese Fisheries Agencies also guided research project focused on the estimation of impacts deriving from those illegal activities within Japanese competence area, documenting that precious corals were still present even in impacted areas.

Similarly, in Taiwan Province of China more and more observations of or arrests for illegal poaching have been reported. Such fishing activities have mainly occurred in the waters around the Diaoyu Islands and Miyako Islands and have more recently extended into waters off Taiwan Province of China (Chang, 2015 and references therein).

The resulting harvests have been sold at low prices (e.g. three boxes of poached materials valued at USD 3.7 million were sold for USD 0.08 million in 2013), to Chinese processors mainly and, as rumoured, to a few Taiwanese processors (Chang*,* 2015 and references therein).

4. USE AND TRADE

4.1. On the use and value of precious corals

4.1.1. On the use and value of black corals (Antipatharia)

Black corals have long been used by humans for a variety of purposes ranging from jewellery to their presumed ability to fend off evil and ailments. In fact, the name of the order Antipatharia is derived from the Greek words ‘anti’ and ‘pathos’ and literally means against evil or disease (Wagner *et al.,* 2012). Antipatharians have a skeleton made of a hornlike protein called gorgonin that is typically dark brown to black; its consistency resembles hard wood. Black coral can also be polished to an onyx-like lustre. Because of its hornlike protein skeleton, it can also be bent and moulded when heated (Bruckner, 2016).

As with *Corallium*, mystical powers and medicinal properties have been attributed to black coral. The skeleton of the larger black coral species has been used for jewellery and religious articles from at least the time of the ancient Greeks. In Indonesia, the Red Sea, North Africa and China, black coral bracelets are used because they are said to increase virility; coral powder is used to cure rheumatism, eye diseases, as well as an aphrodisiac and to relieve pain, fever, stop bleeding and soften hard masses (Wagner *et al*., 2012). In many tropical nations, especially Hawaii and the Cayman Islands, high-end jewellery and carvings of black coral are commonplace. In Hawaiian culture, the high cultural importance of black coral is still apparent today, as it is the official state gemstone of the State of Hawaii (United States of America) (Bruckner, 2016).

4.1.2. On the use and value of white, pink and red corals (family Coralliidae)

Red coral has been a precious commodity since prehistoric times. It has been found in archaeological digs of prehistoric graves of both Mediterranean marine people and European inland ones. Since the Palaeolithic, *Corallium rubrum* has been used for rituals, ornaments, talismans, medicines, aphrodisiacs and currency (De Simone, 2010). During the Classical Period, Rome, India, Persia, and China carried on an extensive international trade in coral and other luxury goods (De Simone, 2010; Bruckner, 2016 and references therein). In the Middle Ages coral products could be found in most Mediterranean countries and red coral became an important commodity along the great Eurasian caravan routes (Ascione, 2010; Bruckner, 2016 and references therein). Red coral has continued to play an important role in religious art, jewellery, sculptures, ornaments and other artefacts up to the modern age. Much of the jewellery up to the early 1800s consisted of uncarved beads, while later the Renaissance-inspired corals involved new designs that included allegoric and mythological compositions. With the introduction of Pacific *Corallium* in the early twentieth century, larger sculptures, inlays and statues were created thanks to the larger sizes and different colours of these species.

Newly emerging fisheries and the discovery of unusually large precious coral beds in the 1960s–1980s inundated markets with Pacific *Corallium*, and new production centres emerged. Coral artisans from Torre del Greco (Italy) continue to produce some of the most intricate and valuable designs, but international markets are flooded with pink and red coral jewellery, sculptures and other decorations from India, Taiwan Province of China and Japan, and the Native American tribes of Arizona; new markets have also emerged in China (Chang *et al*., 2015).

Nowadays, the skeletons of precious corals still provide the raw materials used in jewellery, sculptures, figurines, amulets, clothing, ornamentation and a variety of art objects, and are also reported to have medicinal and mystical powers (Bruckner, 2016).

Coralliidae corals have very hard, magnesium-rich calcitic skeletons. Their value varies depending on their colour, lustre and hardness, condition, and the size of the skeleton (Torntore, 2002). The colour tends to differ between species and depending on the locality where it is collected. Coralliidae corals have the widest spectrum of colours, ranging from pure white to shades of pink, salmon, and orange, and a very dark blood red. These colours are the result of a specific carotenoid pigment called canthaxanthin (Cvejic *et al*., 2007). Individual colonies often lack a uniform coloration throughout the skeleton, and the presence of striations of different colours may lower the value.

Corallium skeletons, composed primarily of calcium carbonate (82–87 percent) and magnesium carbonate (7 percent), can be easily broken but they are hard enough to be polished. The value varies depending on the condition of the skeleton, that is the state of the specimen when it was collected, and the extent of bio-erosion. Japanese fishermen distinguish four conditions corresponding to decreasing values: colonies harvested when attached and living (*seiki*), dead but still attached (*ichi-kare*), dead and toppled (*ni-kare, ochiki*), and long-dead unattached colonies (*san-kare, ochiki*) (Bruckner, 2016; Cooper *et al*., 2011; Iwasaki, 2010).

4.2. Trade of black corals (order Antipatharia)

With regard to the order Antipatharia, according to the CITES trade statistics (derived from the CITES Trade Database, UNEP World Conservation Monitoring Centre, Cambridge, UK)[19] total global imports of Antipatharia between 1981 and 2016 were about 10 million pieces (13 percent raw corals, 82 percent carvings, 5 percent unspecified) and 78.6 tonnes (85 percent raw corals, 14 percent carvings, 1 percent unspecified). The main purposes of the transactions are trade, scientific medical and personal, while the main sources are specimens taken from the wild and confiscated specimens.[20]

The pieces of black corals traded were higher in the 1981–1998 period, then gradually decreased, and are very low up to 2016 (Figure 46). The main importers of black corals in term of pieces were the United States of America (73.94 percent), Japan (15.47 percent), Cuba (5.09 percent), the Dominican Republic (1.83 percent), and France (1.81 percent). The top exporters of black corals in term of pieces were Taiwan Province of China (90.38 percent), the Philippines (4.22 percent), Dominican Republic (3.38 percent) Cuba (0.44 percent), and China, Hong Kong SAR (0.26 percent). When the identification of the genus or the species is provided, international imports/exports have been reported to involve at least 30 genera and 31 species. *Cirrhipathes anguina* accounts for 34.65 percent of the pieces traded, followed by *Antipathes densa* (14.26 percent), *Myriopathes japonica* (6.93 percent), and *Antipathes grandis* (5.49 percent). However, there are many pieces traded as *Antipatharia* spp. (23.19 percent) or *Antipathes* spp. (14.61 percent). This is because in most cases identification to genus or species is possible only if an entire colony is available. Therefore, worked products often cannot be identified beyond the order (Cooper *et al.,* 2011). In recent years. *Antipathes ceylonensis* is the species most commonly reported in statistics (59.23 percent). Actually, it refers to a transaction in 2013 of 4 800 raw corals imported into the United States of America from China, Hong Kong SAR (origin China; CITES, 2018a).

Cirrhipathes is considered to be of inferior quality and value with respect to *Antipathes*; it is often bleached and marketed as “gold” or “golden” coral (Cooper *et al*. 2011).

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[19] The data have been extracted on 2nd of October 2018. The figures are based on the net trade tables for numbers (pieces) and amounts (kg) of precious corals traded as well as importers and exporters, and on the gross trade tables for source, term, purpose, origin. All amount have been transformed to kg. When no indication of unit was available the records have been discarded and not included in the analyses.

[20] The terms are according to the ‘A guide to using the CITES Trade Database Version 8’.

The amount of black coral traded peaked in specific years (Figure 47). The main importers of black corals in term of the amount were the Republic of Korea (50.89 percent), Japan (23.55 percent), United States of America (20.09 percent), Taiwan Province of China (3.11 percent), and China, Hong Kong SAR (0.71 percent). The top exporters of black corals in term of the amount were China (52.49 percent), Taiwan Province of China (35.68 percent), the Philippines (7.22 percent), United States of America (3.3 percent), and Indonesia (1.30 percent). About 6.55 percent of the amount of black coral are recorded as introductions from the sea (ZZ). *Antipathes dichotoma* accounts for 34.57 percent of the amount traded while *A. grandis* for 25.25 percent of the amount. Over 26 percent of black corals in terms of the amount were traded as *Antipatharia* spp., 8.24 percent as *Antipathes* spp., and 4.40 percent as *Leiopathes* spp..

The main purposes of the transactions are trade, scientific, medical and personal, while the main sources are specimens taken from the wild and confiscated specimens. However, since the taxonomic classification of most items circulating on the market is unknown, future research is awaited. Small items that have been cut and polished have lost the morphological characteristics that distinguish them, so identification is impossible in most cases. A variety of black materials may be confused with modified and polished antipatharian corals – including plastics, casein-based plastics, bovid horn keratin, and mangrove roots (Espinoza *et al*., 2012). For this reason, non-destructive methods of analysis, such as Fourier-transform infrared spectroscopy (FTIR) and X-ray fluorescence spectrometry (XRF) are being tested to identify items made of black corals (Espinoza *et al*., 2012).

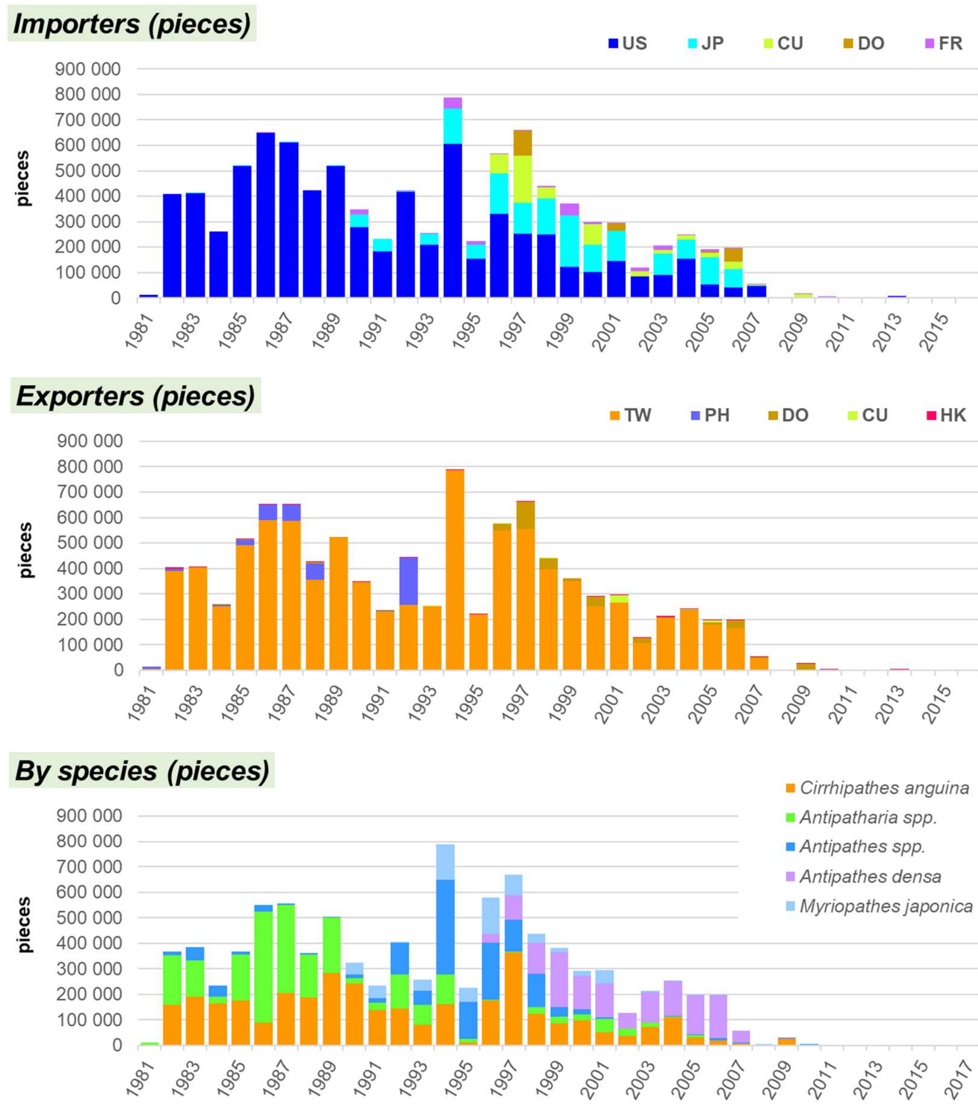

Figure 46 Antipatharia net trade data (pieces): five top importers, exporters and species (Source: CITES, 2018a).

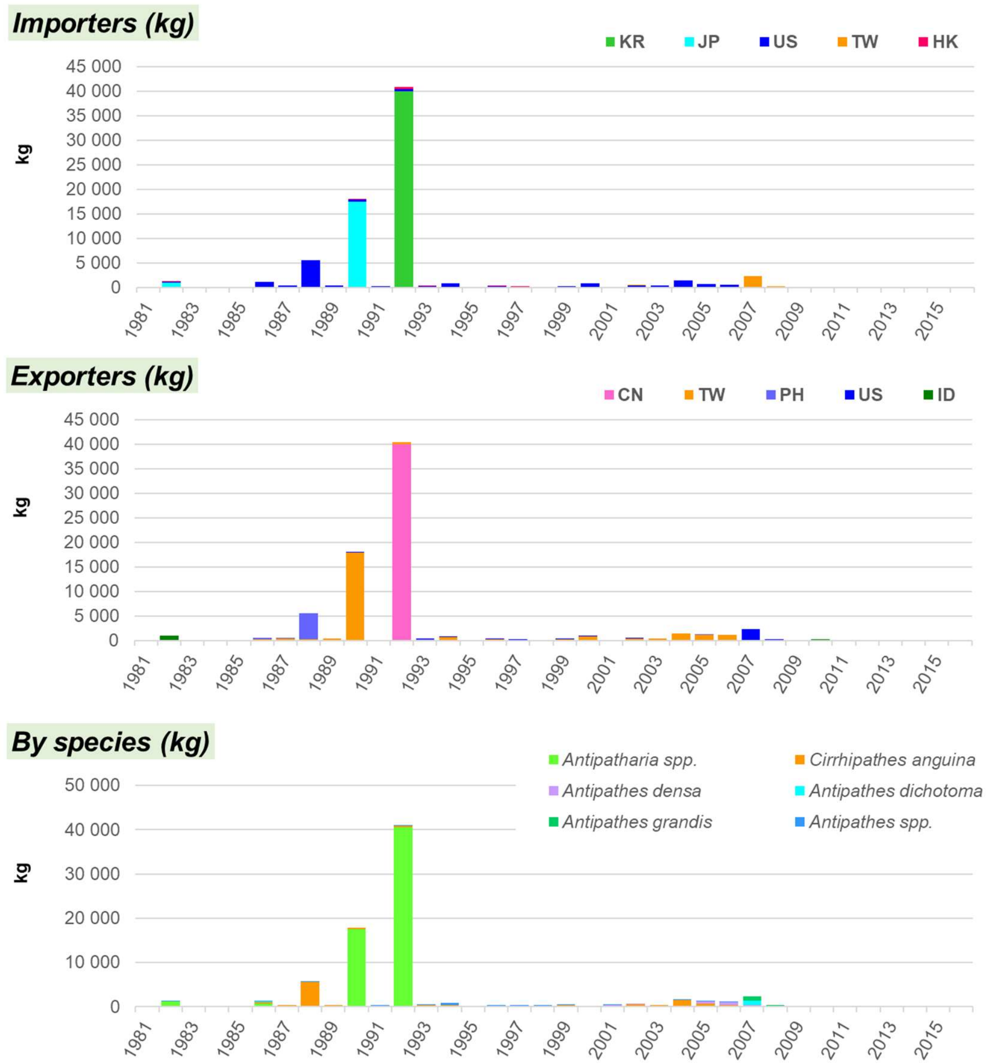

Figure 47 Antipatharia net trade data (kg): five top importers, exporters and species (Source: CITES, 2018a).

4.2.1. Mediterranean Sea

Historical records for black corals do not suggest a significant trade from the Mediterranean Sea, either from a fisheries point of view (GFCM) or a trade perspective (CITES) (Oceana, 2013). Mediterranean black corals are not a profitable species in general, as they have relatively low value when compared to other precious corals such as *C. rubrum* (Oceana, 2013). Moreover, more than 75 percent of the biomass is wasted during the sculpting phase, due to the soft skeleton which is not very malleable to mechanical manipulation. In addition, the extraction of the raw material is difficult because black corals are colonized by encrusting epibionts (Deidun *et al.,* 2010). In Malta, antipatharian species were targeted, albeit on a smaller scale (250 kg) between 1984 and 1987 (Deidun *et al.,* 2010), but the trade does not appear in the CITES database. Black coral jewellery from Maltese stocks was mainly sold to the German market. It is not clear if the fishery exploited one or several species, the landing were recorded at that time as *Antipathes* spp., although there is a high probability that it was *Leiopathes glaberrima*. Black coral populations are no longer exploited in Maltese waters (Deidun *et al.,* 2010).

4.2.2. Pacific area

Black corals are utilised and in circulation on domestic markets in various Asian and Pacific Island countries. In the period from 1985 to 1997, Taiwan Province of China, the Philippines, China, China, Hong Kong SAR and Thailand were the principal exporters among Asian nations, and Japan and the Republic of Korea were importers (Annex 2). In recent years, black coral trade is known to occur in Japan, Taiwan Province of China, the Philippines, and Indonesia. However, in 2018 Japanese black coral retailers stated that they were commercializing stock imported from Hawaii before all antipatharians were listed in CITES Annex II. Nowadays, surveys are needed to assess the actual situation concerning the fishery, processing and trade in the different countries (Annex 2).

4.2.2.1. Hawaii

The black coral harvest industry is currently moribund in Hawaii, with the exception of the relatively shallow species *Antipathes griggi*. There are only a few commercial harvesters who harvest a small number every year, and these harvests are monitored closely by the state of Hawaii (CITES, document AC29 Doc.22 and its annexes).

4.3. Trade of white, pink and red corals (family Coralliidae)

Species of Coralliidae are traded as whole, dried colonies and unworked branches and branch fragments, as well as beads and manufactured jewellery (Cooper *et al.,* 2011). A few species within the family Coralliidae are/were most likely encountered in international trade: four non-CITES species (*C. rubrum, Hemicorallium regale (H. laauense),*[21] *Corallium* sp. nov., *Hemicorallium sulcatum*) and four CITES-listed species (Appendix III) (*Pleurocorallium elatius, Pleurocorallium secundum, Pleurocorallium konojoi, and Corallium japonicum*).

Records of products in trade for the four CITES species is available since their listing in Appendix III of CITES (2008), while data on trades of the three non-CITES species is more difficult to obtain.

4.3.1. CITES species

With regard to the family Coralliidae, according to the CITES trade statistics (derived from the CITES Trade Database, UNEP World Conservation Monitoring Centre, Cambridge, UK; extracted in October 2018) total global imports between 2008 and 2016 were about 325 000 pieces (32 percent raw corals, 66 percent carvings, 2 percent jewellery) and 67.3 tonnes (77 percent raw corals, 22.8 percent carvings, 0.2 percent derivates) (Figure 48 and Figure 49). The main purpose of the transaction was trade, while the main sources were pre-convention specimens, specimens taken from the wild and confiscated specimens.

[21] Problems about *in situ* identifications have been reported between *H. laauense* and *H. regale* and the fact that the name *H. laauense* was used in place of *H. regale* to report findings in publications. Subsequent discussions determined that the name *H. regale* should be used to refer to what appears to be red coral for consistency until modified by a published taxonomic analysis (Parrish *et al*. 2009).

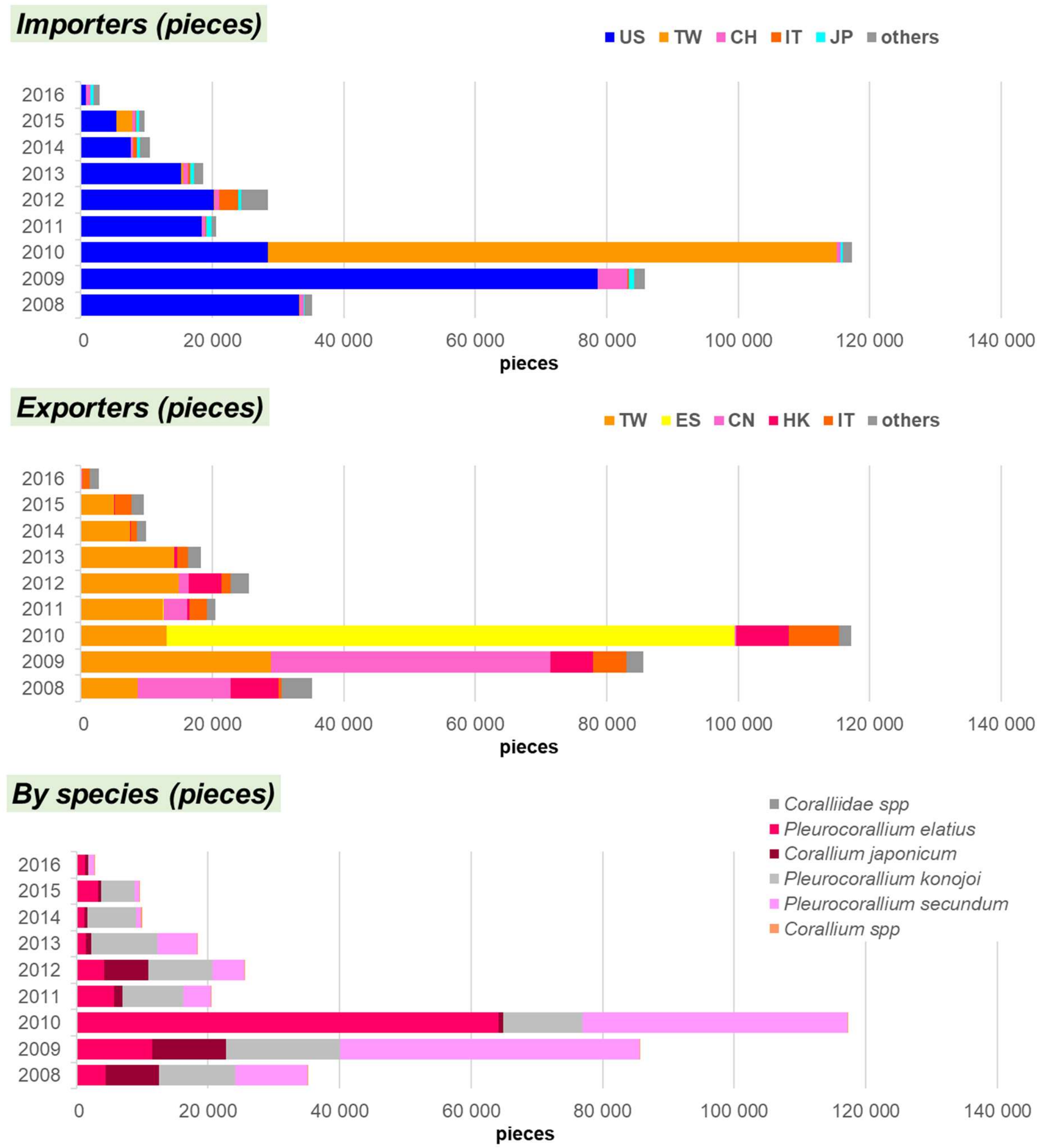

Figure 48 Coralliidae net trade data (pieces): top importers, exporters and species (Source: CITES 2018).

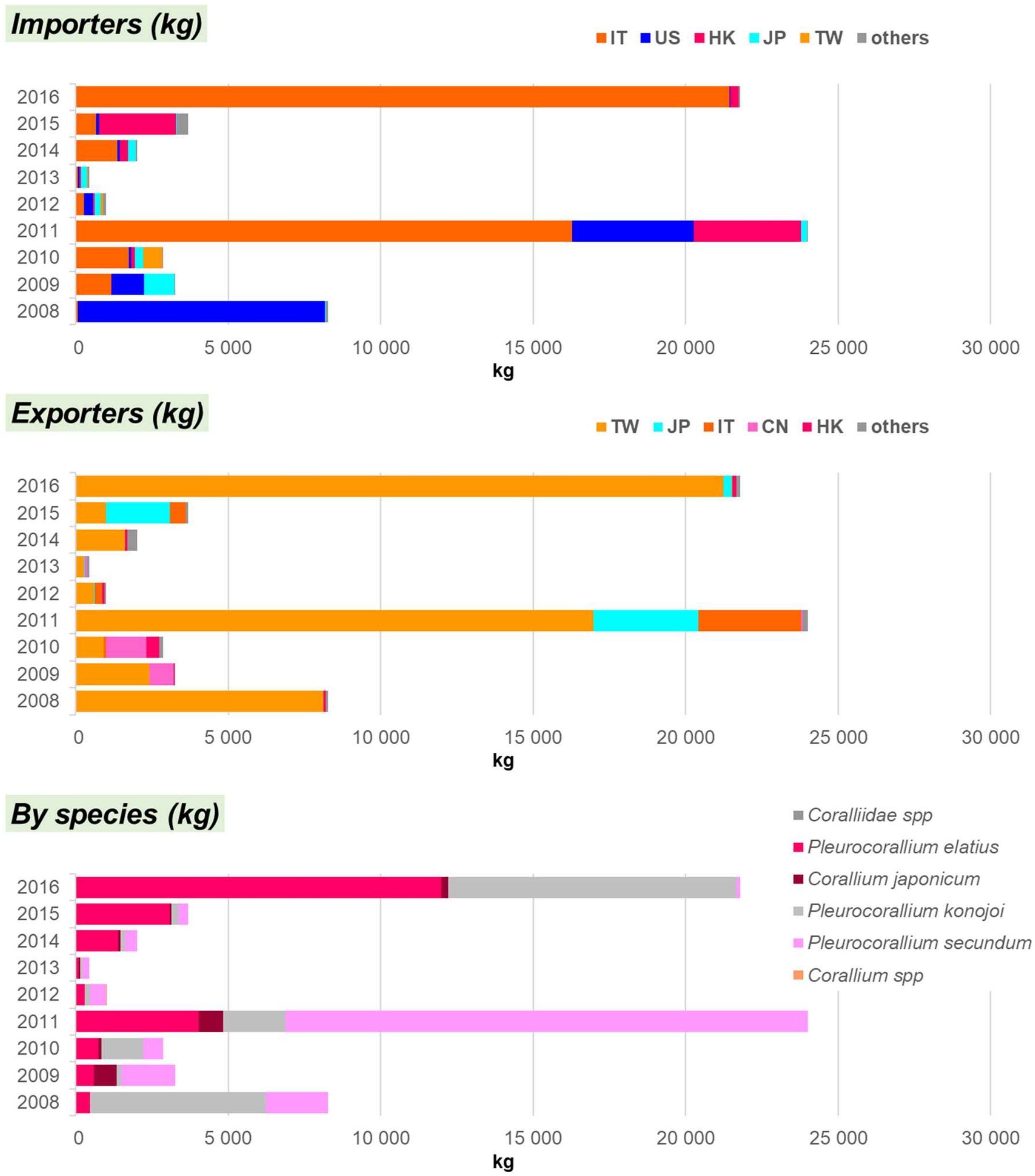

Figure 49 Coralliidae net trade data (kg): top importers, exporters and species (Source: CITES 2018).

The pieces of Coralliidae traded peaked in 2010, then gradually decreased, and up to 2016 are very low (Figure 48). The main importers of Coralliidae in term of pieces were the United States of America (63.26 percent), Taiwan Province of China (27.15 percent), China (2.08 percent), Japan (1.36 percent), and Italy (1.32 percent). The top exporters of Coralliidae in term of pieces were Taiwan Province of China (35.17 percent), Spain (27.31 percent), China (16.49 percent), China, Hong Kong SAR (9.43 percent), and Italy (7.98 percent).

Most of the pieces traded were *P. secundum* (35.15 percent), then in decreasing order *P. elatius* (29.9 percent), *P. konojoi* (25.42 percent), and *C. japonicum* (9.38 percent). It is worth highlighting that a new species recently discovered by Tu *et al.* (2012) has been reported to be circulating on the market as pink coral (Jeng, 2015). Therefore *P. elatius* commercialized in Taiwan Province of China may contain *P. carusrubrum* (Annex 2).

As concerns the amount of Coralliidae traded, they peaked in specific years (Figure 49). The main importers of Coralliidae in term of the amount were Italy (63.97 percent), the United States of America (20.51 percent), China, Hong Kong SAR (10.11 percent), Japan (3.18 percent), and Taiwan Province of China (1.08 percent). The top exporters of Coralliidae in term of the amount were Taiwan Province of China (72.43 percent), Japan (12.39 percent), Italy (7.92 percent), China (3.77 percent), and China, Hong Kong SAR (2.25 percent). Most of the amount traded were *P. elatius* (33.72 percent), then in decreasing order: *P. secundum* (4.26 percent), *P. konojoi* (3.55 percent), and *C. japonicum* (0.39 percent).

According to the CITES Trade Database, there were 211 trade records of seizures or confiscations of *Corallium* corals between 2008–2015, representing a total of approximately 870 kg and 67 400 pieces.

The majority of cases (95 percent) were reported by the United States of America. Raw corals accounted for 90 percent by weight and 76 percent by number of pieces. By weight, *C. japonicum* was the most frequently confiscated species, accounting for 83 percent between 2008 and 2015 and *P. secundum* was responsible for 60 percent of seizures by number of pieces. Of the raw coral confiscations with the country of origin were recorded, 89 percent originated from Japan (Shiraishi, 2018).

In general, raw and whole dried specimens of Coralliidae are relatively easily identified to the species level using an electron microscope (Torntore, 2009); where worked specimens are concerned, it may only be possible to identify them to the level of the family Coralliidae (Bruckner and Roberts, 2009).

The above figures contrast with those reported by Shiraishi (2018), who analysed a shorter period (2011–2015) and merged data from two sources (CITES trade database and CITES trade data from Taiwan Province of China obtained by the Bureau of Foreign Trade Ministry of Economic Affairs). The most striking differences are in the amount of corals traded in the period 2011–2015: about 220 tonnes versus about 31 tonnes, according to Shiraishi (2018) and CITES (2018), respectively. Regarding the number of pieces traded, they are almost double in Shiraishi (2018) with respect to CITES (2018) for the same 2011–2015 period: about 145 000 pieces as compared to about 84 000 pieces. Huge amounts of raw corals are reported to have been traded in 2011 (145 tonnes) and 2012 (almost 40 tonnes), after which imports declined considerably (Shiraishi, 2018). Similarly, for carvings the number of pieces imported reached more than 18 000 pieces in 2012, after which the imports gradually decline (Shiraishi, 2018).

These discrepancies are largely due to the lack of trade data for Taiwan Province of China in the CITES trade database, which were included in the analysis by Shiraishi (2018), in addition to some differences in extracting and analysing data from the CITES trade database in Shiraishi (2018) and in the present report.

Detailed figures for the four species of Coralliidae listed in CITES Appendix III are described in the following paragraphs.

4.3.1.1. Corallium japonicum *(CITES trade database)*

According to the CITES trade database (CITES, 2018a), a total of about 2 110 kg of *C. japonicum* was traded in the 2008–2016 period, with three main peaks in 2009, 2011, and 2016 (Figure 50). *C. japonicum* was imported into China, Hong Kong SAR (48.63 percent, 2011 and 2016) and the United States of America (40.31 percent, 2009), and exported mainly from China (in 2009) and Japan (in 2011, 2016), principally as raw corals.

The *C. japonicum* figures reported by Shiraishi (2018) are astonishingly different: 130 tonnes and 28 500 pieces traded in the 2011–2015 period, with Taiwan Province of China as the main importer (99 percent in weight and 88 percent in pieces). Trades reached a peak of 114 tonnes in 2011, after which they gradually decreased (Shiraishi, 2018). The large quantity of imports in 2011 was largely due to imports in Taiwan Province of China. However, this total is anomalous and, according to the same author, may have been misreported (Shiraishi, 2018).

According to the CITES trade database (CITES 2018), a total of about 30 470 pieces of *C. japonicum* were traded in the 2008–2016 period, with two peaks in 2009 and 2012 (Figure 50). About 84 percent of *C. japonicum* pieces were imported into the United States of America (raw corals and carvings), Switzerland (4.58 percent, raw corals and carvings), Japan (3.28 percent, carvings and jewellery), the United Kingdom of Great Britain and Northern Ireland (3.01 percent, carvings), Italy (1.42 percent raw corals, carvings and derivates). They were exported mainly from Taiwan Province of China (37.34 percent, mainly raw corals), China, Hong Kong SAR (22.84 percent, raw corals and carvings), China (15.97 percent, carvings), Thailand (9.73 percent, raw corals), and Italy (4.73 percent, raw corals and carvings).

In the Shiraishi's analysis, *C. japonicum* was the most traded raw coral species among the four CITES-listed corals (it is the least traded in the CITES database), while imports of carvings were scarce compared to other species: only 81 kg and 1 100 pieces (Shiraishi, 2018).

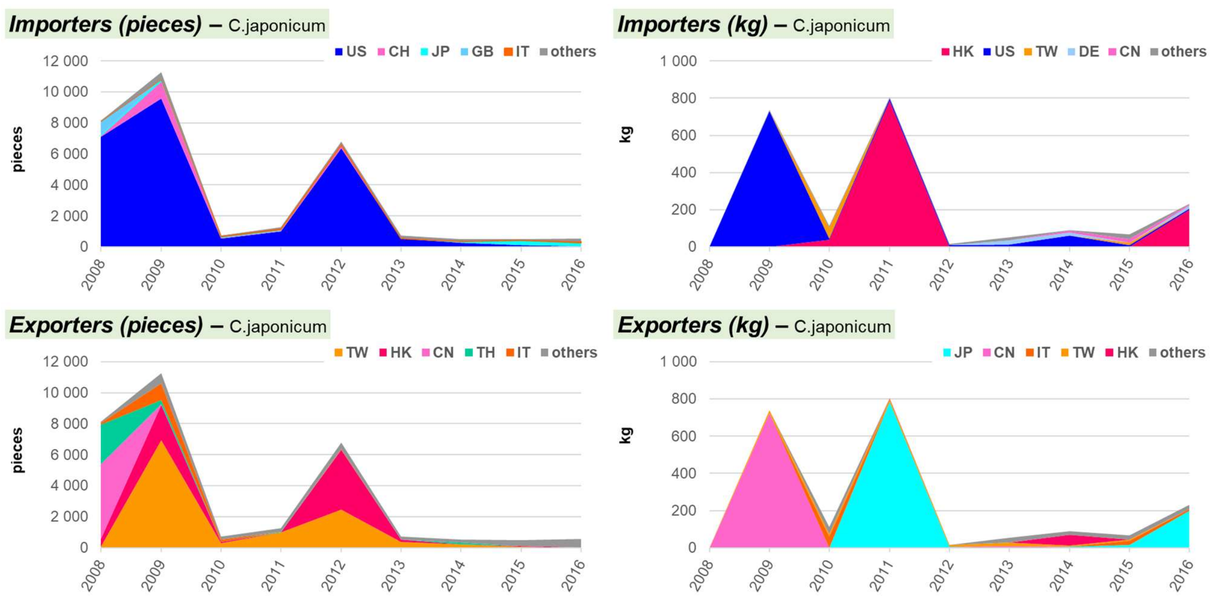

Figure 50 C. japonicum *net trade data: top importers and exporters (Source: CITES 2018).*

4.3.1.2. Pleurocorallium elatius *(CITES trade database)*

A total of about 22.7 tonnes of *P. elatius* were traded in the 2008–2016 period, according to the CITES trade database (CITES 2018), with two peaks in 2011 and in 2016 (Figure 51). *P. elatius* were imported into Italy (70.57 percent, raw corals), China, Hong Kong SAR (13.61 percent, mainly raw corals), the United States of America (11.49 percent, mainly carvings), Taiwan Province of China (2.85 percent, raw corals), Japan (0.62 percent, carvings). They were exported from Taiwan Province of China (73.43 percent, raw corals), Japan (11.48 percent, raw corals), Italy (8.80 percent, carvings), China, Hong Kong SAR (3 percent, raw corals and carvings) and China (1.04 percent, raw corals).

A total of about 97 100 pieces of *P. elatius* were traded in the 2008–2016 period, according to the CITES trade database (CITES 2018), with a peak in 2010 (Figure 51). *P. elatius* were imported into Taiwan Province of China (65.54 percent, carvings), the United States of America (21.84 percent, raw corals and carvings), Switzerland (5.51 percent, carvings, jewellery and raw corals), Italy (2.64 percent, carvings and raw corals) and China, Hong Kong SAR (1.02 percent, carvings). They were exported from Spain (63 percent, carvings), Italy (10.94 percent, carvings and raw corals), Taiwan Province of China (10.71 percent, carvings and raw corals), China (4.38 percent, raw corals), and China, Hong Kong SAR (4.13 percent, carvings and raw corals).

The *P. elatius* figures reported by Shiraishi (2018) are astonishingly different: 60 tons and 40 000 pieces traded in the period 2011–2015, with Taiwan Province of China being the main importer (84 percent in weight and 82 percent in pieces). However, Italy overtook Taiwan Province of China the leading position in 2014. Trades reached a peak of 26.5 tons in 2012, after which they gradually decreased (Shiraishi, 2018).

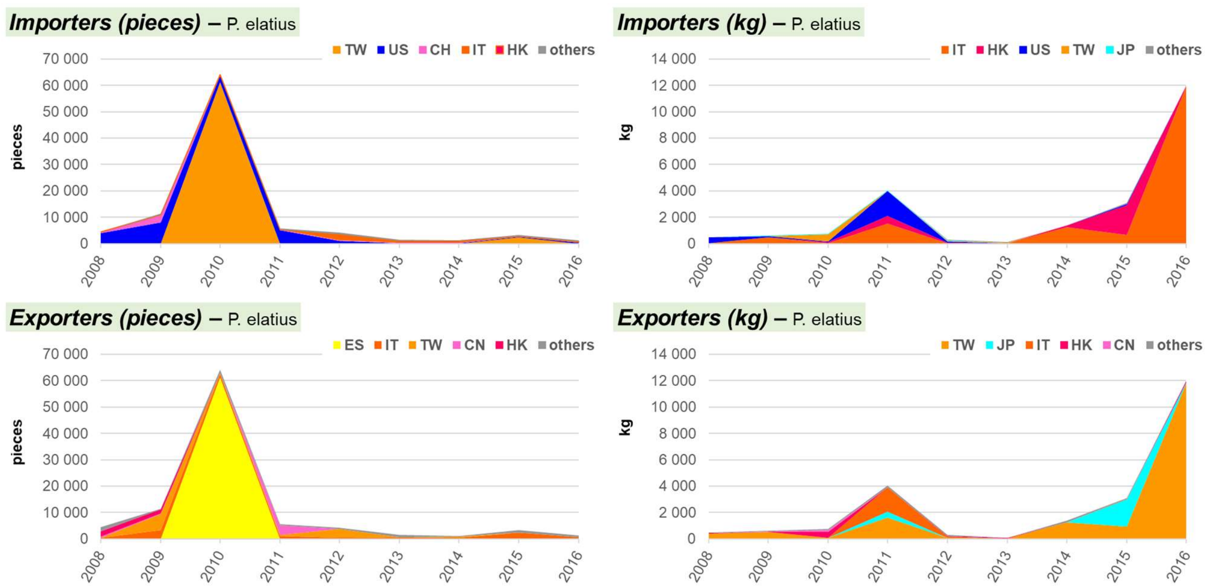

Figure 51 P. elatius *net trade data: top importers and exporters (Source: CITES 2018).*

4.3.1.3. <u>Pleurocorallium konojoi *(CITES trade database)*</u>

A total of about 19.3 tonnes of *P. konojoi* were traded in the 2008–2016 period, according to the CITES trade database (CITES, 2018a), with three peaks in 2008, 2011 and in 2016 (Figure 52). *P. konojoi* were imported into Italy (57.17 percent, raw corals), the United States of America (30.68 percent, carvings and raw corals), China, Hong Kong SAR (9.08 percent, raw corals), Japan (2.24 percent, carvings), and China (0.58 percent, raw corals). They were exported from Taiwan Province of China (84 percent, carvings and raw corals), and Japan (8.49 percent, raw corals), China (6.09 percent, raw corals), Italy (1.16 percent, raw corals).

A total of about 82 500 pieces of *P. konojoi* were traded in the 2008–2016 period, according to the CITES trade database (CITES, 2018a) (Figure 52). *P. konojoi* were imported into the United States of America (97.3 percent, carvings, jewellery and raw corals), Japan (2.04 percent, carvings and raw corals), China (0.3 percent, carvings). They were exported from Taiwan Province of China (65.54 percent, carvings, jewellery and raw corals), China, Hong Kong SAR (9.85 percent, carvings and raw corals), China (3.17 percent, carvings and raw corals), Italy (2.3 percent, carvings and raw corals).

The *P. konojoi* figures reported by Shiraishi (2018) are quite different: only 4.4 tonnes and 40 000 pieces traded in the 2011–2015 period, with Taiwan Province of China and China, Hong Kong

SAR being the main importers (52 percent and 46 by weight percent, respectively. Trades reached a peak in 2011, after which they declined dramatically (Shiraishi, 2018).

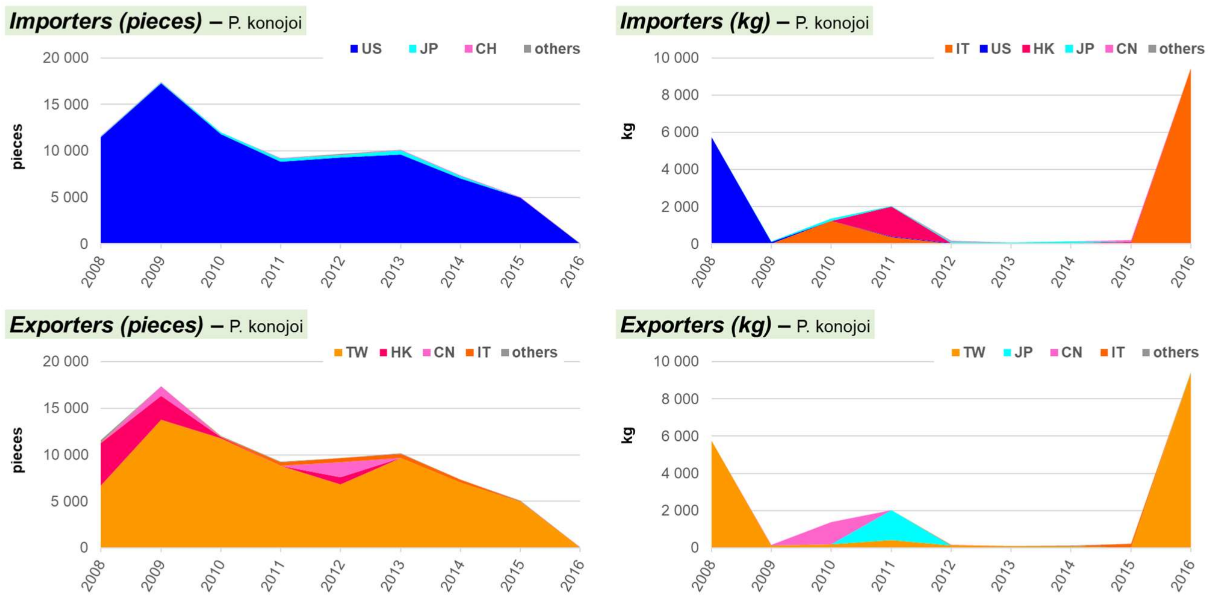

Figure 52 P. konojoi *net trade data: top importers and exporters (Source: CITES 2018).*

4.3.1.4. Pleurocorallium secundum *(CITES trade database)*

A total of about 23.2 tonnes of *P. secundum* were traded in the 2008–2016 period, according to the CITES trade database (CITES, 2018a), with a peak in 2011 (Figure 53). *P. secundum* were imported into Italy (68.92 percent, raw corals and carvings), the United States of America (19.04 percent, raw corals and carvings), Japan (6.67 percent, raw corals and carvings), China, Hong Kong SAR (4.04 percent, raw corals and carvings), and China (0.6 percent, raw corals). They were exported from Taiwan Province of China (86.9 percent, raw corals and carvings), Italy (8.18 percent, carvings and raw corals), Japan (2.73 percent, raw corals).

A total of about 1 141 000 pieces of *P. secundum* were traded in the 2008–2016 period, according to the CITES trade database (CITES, 2018a) (Figure 53). *P. secundum* were imported into the United States of America (70.99 percent, raw corals and carvings), Taiwan Province of China (22.47 percent, carvings), China (1.94 percent, carvings), Italy (1.17 percent, carvings), and Japan (0.77 percent, carvings, jewellery and raw corals). They were exported from China (44.11 percent, raw corals and carvings), Spain (22.20 percent, carvings), Taiwan Province of China (12.03 percent, raw corals and carvings), Italy (8.55 percent, raw corals and carvings) and China, Hong Kong SAR (7.48 percent, raw corals and carvings).

The *P. secundum* figures reported by Shiraishi (2018) are not very different in weight (approximately 25 tonnes) and pieces 34 000 traded in the 2011–2015 period. Italy was the leading importer in weight (68 percent), while the United States of America and Taiwan Province of China were in terms of pieces (Shiraishi, 2018).

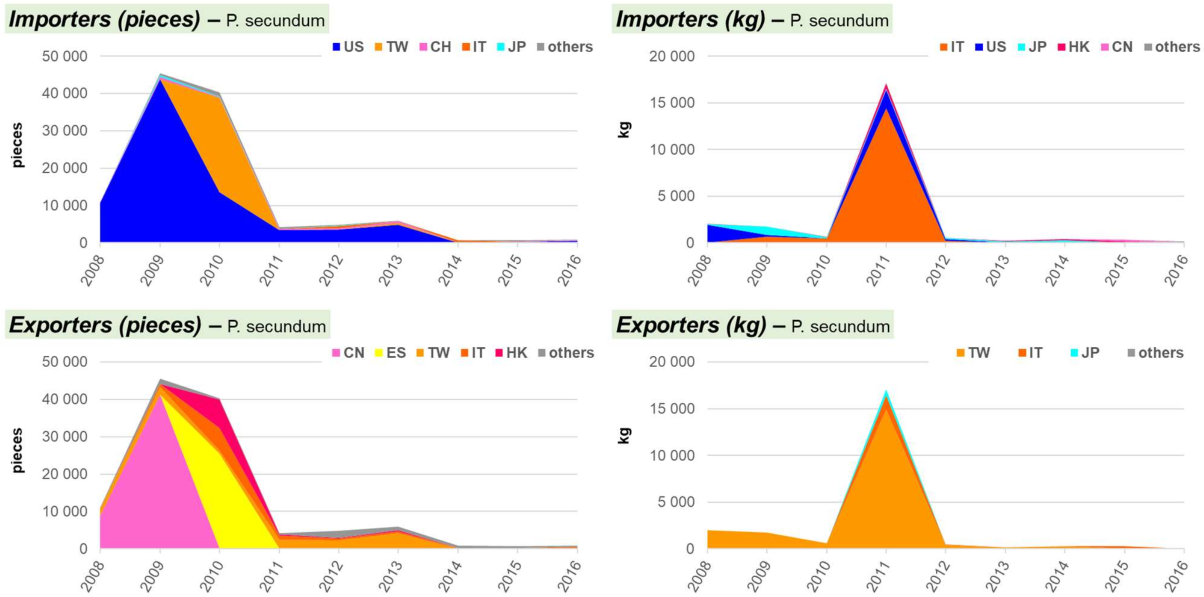

Figure 53 P. secundum *net trade data: top importers and exporters (Source: CITES 2018)*

Additional information on legal import and export of precious corals is available for Japan and Taiwan Province of China (Annex 2).

4.3.1.5. CITES species (Japan)

In Japan, data on trade at the species level are not available. The Japanese statistics include all coral species together, and it is difficult to distinguish processed and manufactured items because they are reported under the same code with semi-precious corals, bone, tortoise-shell, horn, antlers, mother-of-pearl and other animal carving materials. However, judging from the price, most of the imports are estimated to be stony coral, rather than precious corals. Incongruences in the statistics are said to exist in the country, with discrepancies between landings and exported quantities of raw corals (Annex 2).

After the Second World War, commercial trading resumed in 1947. According to the Trade Statistics of Japan, published by the Japanese Ministry of Finance, Japan began exporting raw coral to the United States of America and Italy the following year, and to India in 1949. Exports to Italy were dominant in both quantity and value until around 1980 (Annex 2, Figure 20 and Figure 21). From 1979 to 1981, the period when the largest post-war raw-coral imports were recorded, the average stock was 79.4 tonnes;[22] the corals were kept in Japan for processing or used to meet domestic demand (Annex 2, Figure 25). In the 1980s, export quantities to Taiwan Province of China increased along with an increase in the nation's share of Japan's exports. There was a sharp increase in the quantity of exports to the Republic of Korea in 1990 and to the United States of America in 2002. Assuming all the raw coral exports to be precious corals, large exports were recorded in 2003–2006 and 2010–2015, exports are consistently 2-digit figures while landings are single figures. It is possible that in recent years already existing stock was exported but not to the extent that justify these figures (Annex 2, Figure 8 and Figure 20). From 2010 to 2015, the proportions exported to the three major countries

[22] The total stock can be thought of as precious coral landings plus imports of raw corals and processed items, minus exports of raw corals and processed items.

in terms of quantity were 68.0 percent to the United States of America, 16.3 percent to the Republic of Korea, and 12.3 percent to Taiwan Province of China, while the latter accounted for an overwhelming 88 percent of the total export value during that period (Jeng, 2015; Annex 2).

The export prices (averages for 1988–2015) for the Republic of Korea and the United States of America were low at JPY 795 /kg and JPY 1 527 /kg respectively,[23] indicating that large quantities of low-priced items were exported to those countries (Annex 2, Figure 22) . Meanwhile, export prices to Taiwan Province of China, China, Hong Kong SAR, and European countries were over JPY 190 000 /kg, and small quantities at unit prices exceeding JPY 1 million /kg were exported, particularly to Switzerland and France. As for imports of raw coral, imports from Okinawa, then under US-military occupation, accounted for the major share of both quantity and value in the mid-1950s and early 1970s, (Annex 2**,** Figure 23 and Figure 24). However, in the mid-1960s, when the catch in Okinawa decreased, raw coral from mainland Japan was exported to Okinawa (Annex 2**,** Figure 20). From the mid-1970s to the present, imports from Taiwan Province of China have become dominant (Annex 2).

Precious coral prices soared in the years 2010–2015. Japanese exports to China, Taiwan Province of China and China, Hong Kong SAR during this period consisted of more valuable, higher-priced items when compared to 2003–2006 (Annex 2**,** Table 7).

High demand (until 2016) from Chinese customers has stimulated increasing demand from processors and jewellers both in Taiwan Province of China and China. The preference for the colour red among ethnic Chinese people due to its associations with joy, happiness and luck have made the price of Aka coral (mostly from Japanese waters) five times higher now than it was in 2008, making it the most expensive coral. Consequently, a substantial proportion of Japanese raw materials have been bid on by exporters (mainly from Taiwan Province of China), with higher prices to support the demand. This in turn has resulted in a shortage of coral for many small Japanese local processors and has closed them down, with the number of local processors decreasing from around 50 to less than 10 (Chang, 2015).

According to a recent market survey in Tokyo in March 2017, the main product types were earrings, necklaces and rings and all of these shops offered red coral products (species mostly unknown). The price varied between JPY 3 000 (small earrings) to JPY 9 800 000 (*C. japonicum* ball) depending on the colour, size of the coral and/or whether the products had diamonds. Generally, products made from *C. rubrum* (non-CITES-listed species) were less expensive. For instance, while a *C. rubrum* necklace was JPY 45 000, a *C. japonicum* necklace was JPY 75 000 (Shiraishi, 2018).

According to the shop staff, the products for sale had originated from and been processed in Japan. They indicated the demand for corals had increased over the past 3–4 years in mainland China and they sold corals mainly to them (foreign visitors) because the popularity of coral products had declined and they do not match the clothes of modern Japanese people (Shiraishi, 2018).

According to statistics from Japanese Customs (Shiraishi, 2018), Japan exported unworked coral to 11 countries/territories from 2007 to 2016, however the data may include non-*Corallium* corals.

According to Japanese Customs, there were ten cases of “acknowledgement of abandonment” from 2008 to 2016 – this is a procedure whereby an individual who owns the products abandons them voluntarily to the Japanese government if Customs finds appropriate documentation is missing but determines that stricter measures (e.g. prosecution) are not needed. It was not possible to calculate a total weight/number of pieces involved in these ten cases as the units used were different. All of them were seized either due to a lack of export permits or certificates of origin when imported. These corals were exported from mainland China (five cases), Italy (two cases), the Republic of Korea, Taiwan Province of China and China, Hong Kong SAR (1 case each). They were composed of dead coral

[23] JPY (Japanese Yen) 1 = USD 0.009 (OANDA, 1 April 2019)

(four), accessories (four) and specimens (two). Half of them were sent by post and the rest was shipped by commercial cargo. The biggest case was 127 kg of *Corallium* corals found in 2010 (Shiraishi, 2018).

Similarly, in mainland China, according to media reports, there were at least 38 seizures between 2008 and 2016, representing a total of 156.7 kg and more than 1 500 products/materials. The weight of seized specimens was considerably higher in 2013 than earlier years, at around 57 kg and a peak of 861 pieces seized in 2015 (Shiraishi, 2018). In September 2016, Huanggang Port Customs seized 30 red coral products from tourists returning from Taiwan Province of China intending to process and sell them in mainland China (Shiraishi, 2018).

4.3.1.6. CITES species (Taiwan Province of China)

Precious corals collected in Taiwan Province of China are sold at public auction at bid markets in the Suao fishing port. First-hand processors purchase the raw materials then further process them for jewellers, delivering either finished products or semi-products that are processed further by second-hand processors. Traders who usually hold some shares of the fishing vessels also purchase the raw materials and sell them to various processors in Taiwan Province of China and other countries. Products have mainly been sold in the markets of Taiwan Province of China during the last five years and are increasingly purchased by Chinese tourists who then take the products back to mainland China. Another main route for such products is their direct delivery by various processors/jewellers to China (including China, Hong Kong SAR), mostly for exhibitions (which are conducted almost every two weeks nowadays at different locations in China), with a portion of the exhibited products then being sold on the Chinese market. Processors/jewellers are required to apply for C/O from the Fisheries Agency for such deliveries, in order to obtain CITES documents from the Bureau of Foreign Trade of Taiwan Province of China. There were also some small amounts of products officially exported to China. In addition to the above main routes, some raw materials also come from stockpiles or are imported from sources in the Mediterranean Sea. Some percentage of all such raw material is processed into semi-products for the United States of America, Japanese, Korean, and EU markets, or is sent to Viet Nam for the labour-intensive early-stages of processing to take advantage of the cheap price of labour there (Chang 2015; Annex 2, Figure 27). It is reported that corals imported into Taiwan Province of China from Japan are resold to countries including China, Tibet, and India via processors in Taiwan Province of China (The Nikkei, 2010). China is therefore the principal market for precious corals from Taiwan Province of China (Chang, 2015).

Precious coral prices rose steeply on the back of the booming Chinese economy, increasing from USD 900 /kg in 2009 to USD 7 500 /kg in 2014. The price rise began in July 2008 when Taiwan Province of China passed legislation opening the gate to visitors from mainland China. The number of Chinese visitors shot up from 329 000 in 2008 to 2 875 000 in 2013.

The increase in demand while supplies have remained unchanged or have even decreased has led to an increase in prices, and China, with its strong purchasing power, has increased the price ceiling for high-quality corals. High profits from both the tourist and luxury markets have fuelled the boom of processors (including precious coral companies, workshops and factories) and jewellers in Taiwan Province of China; the current coral sector is estimated to be at least two-to-three times larger than it was in the late 1970s to early 1980s, when the coral industry reached its last historical peak. As the number of processors grows, even as some are unable to make a profit due to increasing competition for customers, the demand for raw materials in Taiwan Province of China is likely to only increase further (Chang, 2015 and references therein).

There are concerns that, if all the corals brought to bid markets and continue to sell out at high prices, this will stimulate greater fishing effort, resulting in more pressure on resources (Annex 2).

It is difficult to analyse customs data, since Taiwan Province of China has four Customs codes related to coral – namely “coral and similar material”, “powder and waste of coral and similar material (including for Chinese medicine)”, “worked coral material” and “articles of coral” – which may include trade in non-CITES listed *Corallium* corals (Shiraishi, 2018).

Trade data between Japan and Taiwan Province of China suggest much unworked coral harvested in Japan is exported to Taiwan Province of China (Shiraishi, 2018) with a peak in 2013. The import/export price dropped a little in 2015 and 2016, but was still higher than prior to 2012 (Shiraishi, 2018).

Although previous studies and media reports claim that demand for *Corallium* corals has increased in mainland China and most of them are shipped from Taiwan Province of China after processing (Chang, 2015), Taiwan Province of China Customs data between 2007–2016 as well as Taiwan Province of China CITES trade data did not indicate exports to mainland China have substantially increased (Shiraishi, 2018).

Taiwan Province of China Customs data suggest that Taiwan Province of China exports coral as unworked material or semi-worked coral, and imports semi-worked coral from Viet Nam (Shiraishi, 2018).

4.3.1.7. CITES species (China)

According to a recent market survey, the species of *Corallium* coral offered for sale at the markets/stores in China were *C. japonicum*, *P. elatius* and *C. rubrum*, with a few *P. konojoi*. Shopkeepers were usually unsure about the origin of products. *C. japonicum* was to be the most expensive, ranging from CNY 100 /g to over CNY 10 000 /g.[24] The larger and darker corals were, the more expensive they became. The price also varied depending on their shape and the presence of gold, diamonds and/or elaborated designs. Most coral specialist shops displayed the permit required under the Wildlife Protection Law to sell *Corallium* products (Shiraishi, 2018). Most shopkeepers confirmed the price is rising and was expected to continue to do so in the future, with a possible increase of 20–30 percent a year (Shiraishi, 2018).

4.3.2. non-CITES species

As far as the four non-CITES species are concerned, two species were harvested and thus commercialized in limited periods of time. They are no longer collected commercially although they have been fished in the past:

- *H. regale* (*H. laauense*): exploitation occurred in the coastal waters of Hawaii in 1999–2001 (Grigg, 2010);
- Midway deep coral (*Corallium* sp. nov.): exploitation occurred in the Milwaukee Bank and the surrounding Emperor Seamounts, located in international waters near the Hawaiian Archipelago at depths of 900–1 500 m. The discovery of the bed was in 1974, the peak of landings in 1985, but virtually no coral was landed from these beds after 1989 (Bruckner, 2016).

A third species, miss corals (*H. sulcatum*) was collected in huge quantities in the waters of Taiwan Province of China in the 1980s and 1990s. It is still harvested, representing 1–9 percent of the average annual production for the period 2009–2012 in Taiwan Province of China. Close examination is required to investigate the possibility that miss coral is also being collected in Japanese waters.

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[24] CNY (Chinese Yuan) 1 = USD 0.15 (OANDA, 1 April 2019)

The fourth species, the Mediterranean red coral (*C. rubrum*) has been extensively harvested and traded since prehistoric times.

4.3.2.1. non-CITES species (raw coral C. rubrum*)*

Since the Roman Empire, Italy has been the centre of Mediterranean red coral harvesting, manufacturing, and trading. Artisan manufacturers in Torre del Greco (Italy) started in 1805 when the first workshop to process this precious material was granted the sole right to do so, as issued by King Ferdinand V de Bourbon. These artisan manufacturers now work with Mediterranean raw corals from France, Italy, Morocco, Spain and Tunisia, and in the past (when available) also from Albania, Algeria and Greece. The imported raw coral is not usually re-exported but worked on site.

However, based on historic data collected and compiled by private industries, red coral has been imported and used in expensive jewellery in Taiwan Province of China (Huang and Ou, 2010). This is also confirmed by the fact that in the years 1986–1988 substantial quantities of Mediterranean red coral were said to be exported to Taiwan Province of China and Japan for processing (GFCM, 1989). Even in very recent years relevant amounts of raw Mediterranean red coral have been exported from Italy to the United States of America (CITES, 2010) and more recently to India and China, which are two new important trade destinations.

The market price of raw red coral varies consistent with its quality, origin and economic situation. Red coral colonies are classified in five different categories: tips, third category colonies (mainly < 7 mm of trunk diameter), second category colonies (7–9 mm of trunk diameter), first category colonies (> 10 mm of trunk diameter, usually 12–14 mm) and special category colonies (> 14 mm in diameter and > 100 g in weight). The most profitable red coral colony should have the following characteristics: 12–15 cm in size, approximate weight of 100 g, 10–15 mm diameter of the trunk and at least 4 mm thickness at the tips (GFCM, 1984). The quality (and value) can vary depending on whether the specimen was harvested as a living or dead colony (or a dead broken branch). Coral that was harvested live is the most valuable because of its deep colour and high translucency, decreasing as a dead coral ages on the ocean bottom (Cooper *et al*., 2011). The quality of products made is also affected by damage from boring sponges of the family Clionidae, that create a series of holes in the skeleton (Cooper *et al*., 2011).

Even though most of the GFCM countries have mandatory sale notes, and register data on the buyer and seller of *C. rubrum* (Annex 1, Table 11), official data on trade are not easily available.

According to the Chairman of the Association of Coral Producers of Torre del Greco, until the 1970s all kinds and qualities of coral were easily marketed (GFCM, 1989). Later, in the mid-1980s a strong demand for dark red, high-quality coral was recorded, while clearer corals could hardly be sold. Certain types of corals, although available on the market at reasonable prices were no longer of interest to the processing industry – this was also reflected in a decrease of the overall quantity of coral processed at Torre del Greco in those years (GFCM, 1989). *C. rubrum* is currently still sold for relatively high prices: in Torre del Greco medium- and large-sized colonies are prevalently worked, with prices ranging from a minimum of EUR 200 /kg to a maximum of EUR 2 000 /kg (in cases of colonies of extremely high quality)[25].

[25] EUR 1 = USD 1.12 (OANDA, 1 April 2019)

4.3.2.2. non-CITES species (processed coral C. rubrum*)*

Essentially, methods of processing Mediterranean red coral today are the same as in the past. It is a highly labour-intensive process, done mainly by hand, with mechanization appearing only in some steps (Stampacchia and de Chiara, 2000; Torntore, 2009 and references therein).

In general, the main division exists between artisan–artists who make the semi-finished product and the companies that perform the remaining steps of the assembly and placement of the product on the market. The artisanal production in Torre del Greco is characterized by an extremely wide range of products, making it difficult to establish a standard price for worked corals; they can have a commercial value at the wholesale market that is extremely variable and related to the size, nature and quality of the product. The price is decided by calculating not only the value of the raw material but also, and above all, the costs for the processing – which are in this case extremely high – and of the skilled labour.

Nowadays, about 60 percent of the coral used in Torre del Greco is of Mediterranean origin and 40 percent of Asian origin (Cau *et al*., 2013). However, in recent years, partly due to the economic crisis as well as the inclusion in CITES Appendix III of Pacific corals, the amount of coral imported from those countries has considerably decreased.

Even before the world economic crises of 2007 and 2011, a significant dichotomization of demand between products of low price/quality and medium–high price/quality has occurred. The first type of product is usually made with Pacific corals (Stampacchia, 2010). Despite the more recent developments, therefore, the bulk of high-quality production (and revenues) remains strongly linked to a specific raw material – the Mediterranean red coral – and to the skills and relationships that are associated with it, which in turn have spread and become customary for other materials and areas (Stampacchia, 2010).

Concerning the number of people involved in processing in Torre del Greco, in 1982 there were some 150 factories employing 4 000 workers (GFCM, 1989); in 1999 the number of workers declined to 1 900 employees, including 1 300 internal workers and 600 workers at home (Stampacchia and de Chiara, 2000). In 2009, there were about 270 companies in Torre del Greco with over 2 000 employees (Stampacchia, 2010). Finally, in 2013 there were about 2 600 employees, indicating a substantial stability in the sector. In comparison, today the principal competitor (the coral industry of Taiwan Province of China) includes some 2 000–3 000 companies/workshops/factories and over 30 000 people who work with coral and depend on the coral sector in some way for their livelihood (Torntore, 2009).

5. MANAGEMENT AND CONSERVATION

5.1. Management of black corals (order Antipatharia)

Information on the management of black coral species worldwide is not well documented or easily accessible, with the most detailed management found in the United States of America. According to the information provided by the United States of America (CITES AC29 Doc.22A and its annexes) three species have been harvested commercially in the past 60 years in the country (*Antipathes grandis, Antipathes griggi,* and to a lesser extent *Myriopathes* cf. *ulex*). The fishery has historically been managed as a single stock, in large part due to difficulties in differentiating the targeted species (Wagner, 2015). Harvests have only occurred in state and federal waters surrounding the Hawaiian Islands, and these have been done in accordance with management plans.

In Hawaii, commercial black coral beds are located in state and federal waters. In state waters, harvesting within three miles of islands is regulated by the Department of Land and Natural Resources. State management involves a system of licensing and reporting requirements, as well as maximum sustainable yields (MSY), and minimum size limits (i.e. it is unlawful to take, destroy, or possess any black coral with a base diameter of less than three quarters of an inch from state waters;[26] in federal waters live black coral harvested from any precious corals permit area must have attained either a minimum stem diameter of 2.54 cm, or a minimum height of 122 cm. In federal waters (United States of America EEZ) precious coral exploitation has been managed since 1983 by the National Marine Fisheries Service (NMFS) of the National Oceanic and Atmospheric Administration (NOAA) through the Precious Coral Fisheries Management Plan (FMP) of the Western Pacific Regional Fisheries Management Council (WPFMC) (Tsounis *et al*., 2010). The FMP has been amended eight times since its inception. The FMP imposes permit requirements valid for specific locations, harvest quotas for beds, a minimum size, gear restrictions, area restrictions and fishing seasons.

Estimates of MSY and optimum yield (OY) are used in the fishery management plan for precious corals in the Hawaiian Archipelago for the definition of harvest quotas (WPRFMC, 2009).

Using the Beverton and Holt model (BH), black coral MSYs were estimated at 6 174 kg/year for the Auau Channel and 1 480 kg/year for the area around Kauai (Grigg, 1976). More recently, Grigg discovered a greater impact on the black coral resource from an invasive soft coral, *Carijoa* sp. (previously identified as *Carijoa riisei*), and based on that, coupled with harvesting impacts, estimated a reduced MSY of 3 750 kg/year for the Auau Channel (Grigg, 2004).

In fact, the harvest quotas are based on extrapolations from “rounded down MSY values” for ecological and economic reasons (OY). All the beds for which OY has been determined are called ‘established beds’. All other beds known to contain precious corals, but in which biological data are not available (‘conditional bed’), the OY is calculated on a pro-rata basis, using the area of the Conditional Bed relative to the area of established beds. Moreover, two additional categories were identified: black coral refugia beds (areas set aside for baseline studies), and exploratory permit areas (the unexplored portions of the EEZ).

Outside the United States of America, in the Turks and Caicos Islands (United Kingdom of Great Britain and Northern Ireland), licenses are required to collect black corals. However, no such license has been issued for a number of years (CITES AC29 Doc.22A). No commercial harvesting permits are issued under New Zealand legislation for black corals. However, permits to catch and handle are issued for research purposes only (CITES AC29 Doc.22A).

[26] http://dlnr.hawaii.gov/dar/fishing/fishing-regulations/marine-invertebrates/

Size limits have been applied to black corals in other countries as well (e.g. Cuba), but in absence of other controls they appear to have been ineffective, and black coral populations have been depleted. This is the case of *Antipathes caribbeana*, where a minimum size of 1.2 m height and 2.5 cm diameter was established in the 1990s. Nevertheless, black coral populations declined along the Pinar del Rio Province, in Matanzas Bay (northeast Cuba), Puerto de Sagua (northcentral Cuba) and Cazones Gulf (Alcolado *et al*., 2003; Bruckner, 2016).

In Mexico, the management is based on quotas (150 kg/month in Quintana Roo State), size limits (> 2 cm basal diameter) and spatial restrictions (Padilla, 2001).

5.2. Management of white, pink and red corals (family Coralliidae)

5.2.1. CITES species

5.2.1.1. CITES species (Japan)

Coral fishing in Japan is regulated by each prefecture, and permits are issued by the respective governors (Annex 2, Table 3). Coral fishing is currently prohibited by nine regional authorities (Chiba, Kanagawa, Shizuoka, Aichi, Mie, Tokushima, Miyazaki, Oita, and Kumamoto Prefectures) and permitted by seven regional authorities (Tokyo metropolitan area, Wakayama, Kochi, Ehime, Nagasaki, Kagoshima and Okinawa Prefectures). In other prefectures, there is no record of operation because precious corals do not occur.

In 2015, Fisheries Agency of Japan issued technical advice for the management of its domestic precious coral fishery based on the Local Autonomy Act. The advice requests governors of relevant prefectures to take action to enhance the appropriate management of precious coral fishery such as the following:

- limiting fishing effort to current levels;
- clearly designating fishing zones in order to conserve precious corals in non-fishing zones (while avoiding clear designation of non-authorized zones since such designation might provoke poaching activities);
- recording the movement of authorized fishing vessels through vessel monitoring system (VMS), global positioning system (GPS), etc. and keeping a record;
- ensuring catch reporting based on each prefecture’s fishery adjustment rules.

Each prefectural government regulates precious coral fisheries in accordance with this technical advice.

Based on the technical advice mentioned above, each prefecture’s Fishery Adjustment Committee regulates rules and instructions for their respective waters, not only in terms of licensing requirements and fishing methods but also catch-reporting obligations. In Ogasawara (Tokyo) in 2014, restrictions were voluntarily imposed by the local fishermen in addition to public regulations. For this reason, the regulations differ between regional governments (Annex 2, Table 3). In Kagoshima and Okinawa Prefectures, the non-selective fishing method of drag-netting is prohibited, but selective fishing using ROVs is allowed. In Kochi Prefecture, fishing regulations were tightened from 2012 in response to increasing global consensus for the protection of precious corals. While fishing in January and February had already been banned, research on spawning seasons (Sekida *et al*., 2016) prompted the local government to extend the ban to include the spawning season of June and July. Limits were also set on fishing zones, catch quotas (up to 0.75 tonnes of live coral a year), fishing hours and coral size. For instance, there is a requirement that undersized corals (< 3 cm high or 7 mm in diameter) are “released”, but these are unlikely to survive as they have been detached from the bottom (Bruckner, 2016). Although the number of permitted fishing boats increased, stricter limits were set on the fishing periods and areas so that the total "fishing effort" remained unchanged.

The number of fishing boats using coral nets has continued to increase on the back of soaring precious coral prices. Coral fishing was resumed off the Ogasawara Islands in 2012 after a break of ten years. In Wakayama prefecture, where a coral fishery had never existed, 50 vessels were permitted to conduct trial operations in 2014, and in 2015 fishery permission was granted to a company jointly managed by 12 fishery cooperatives. This led to 152 registered vessels, with a cap of 50 vessels operating per day. Currently, the number of permits is 359 in Kochi (2018), 24 in Tokyo (eight in operation, 2016), five in Nagasaki (one in operation, 2017), three in Ehime (2017), one in Wakayama (actually 123 boats, 2017), one in Kagoshima (ROV, 2017), and three in Okinawa (one in operation, ROV, 2017).

5.2.1.2. CITES species (Taiwan Province of China)

Coral fishing is managed according to the "Regulations Governing Fishing Vessels that Also Engage in Coral Harvesting", established in January 2009. Regulations have been tightened, restricting the number of fishing vessels, fishing zones, catch quotas, designated landing ports, and requirements of a VMS. The regulations require fishermen to keep a logbook of fishing operations and to have an observer on board the boat, as well as to centralize auction markets (Hunag and Ou, 2010). Fishermen who violate these regulations are penalized with confiscation of their fishing equipment and revocation of their license. Prior to that, Taiwan Province of China had already been reducing the number of fishing vessels in stages since 1979 – by not permitting more than the already existing 150 vessels in 1983, then setting the number at 56 in 2009 and 60 in 2010 (Chen, 2012). These measures appear to have been effective in regulating the amount and spatial distribution of effort (Bruckner, 2016).

5.2.1.3. CITES species (United States of America)

Pleurocorallium secundum has been harvested commercially in the past years in American waters between 350–500 m (CITES AC29 Doc.22A and its annexes). The main fishery began in 1965 when Japanese fishermen discovered a very large bed of pink coral (*P. secundum*) at 400 m in the Milwaukee Banks, Mellish Bank and surrounding seamounts in the Emperor Seamount chain north of Midway Island, near the Hawaiian Archipelago. This discovery led to an incipient fishery that was initially unregulated since very little was known about the ecology of pink corals (Grigg, 2010). Later, harvests were conducted in accordance with strict management plans.

WPFMC's Precious Corals FMP has regulated the harvest of pink corals since 1983 within the 3 to 200 miles United States of America EEZ. The FMP imposes permit requirements valid for specific locations, harvest quotas for precious coral beds (only live corals are included in the quota), a minimum size for pink coral (live pink coral harvested from any precious corals permit area must have attained a minimum height of 25.4 cm), gear restrictions, area restrictions, and fishing seasons. The initial plan created four categories of coral beds: established beds, conditional beds, exploratory areas and refugia.[27] MSY has been estimated using a Beverton and Holt model for *P. secundum*, for a single bank for which biological data were available (Grigg, 1976). For the Makapu'u Bed the age

[27] Established beds are ones for which appraisals of MSY are reasonably precise. Conditional beds are ones for which estimates of MSY have been calculated by comparing the size of the beds to that of Makapuu Bed and then multiplying the ratio by the yield from the Makapuu Bed. Refugia beds are areas set aside for baseline studies and possible reproductive reserves. Exploratory areas are the unexplored portions of the EEZ.

at MSY has been estimated at 34 years and the harvest quota at MSY has been identified at 1 185 kg/year (Grigg, 1976).

NMFS specifies annual catch limits (ACLs) for precious coral, and accountability measures (AMs) to correct or mitigate any overages of catch limits. The ACL and AM specifications support the long-term sustainability of fishery resources of the United States Pacific islands. At the end of each fishing year, WPFMC will review catches relative to each ACL. If NMFS and WPFMC determine that the three-year average catch for the fishery exceeds the specified ACL, NMFS and the Council will reduce the ACL for that fishery by the amount of the overage in the subsequent year (CITES AC29 Doc.22A and its annexes).

A single bed, the Makapu'u Bed off Oahu Hawaii, is defined as an established bed for pink coral; four beds are established as conditional beds: Keahole Point, Kaena Point, Brooks Bank, and 180 Fathom Bank; Westpac Bed is established as a refugium where the harvesting of coral was not permitted.

Nowadays, for Makapu'u bed, a one-year harvest quota using selective gear is set at 2 000 kg for *P. secundum*. The quotas for the conditional beds were directly related to their size, assuming the same optimal yield as Makapu'u. Because these beds are substantially smaller, the quotas for pink coral range from 17 to 222 kg. Given the absence of data on the size of the bed and the population structure of corals within the EEZ exploratory areas, quotas of 1 000 kg are established for all species of precious coral combined, with the intent of modifying these based on future harvest records and research.

Exploratory areas open to commercial coral harvesting include the main Hawaiian Islands, American Samoa, Marian Archipelago, the United States of America Pacific possessions, while coral harvest is prohibited within the Northwest Hawaiian Islands (NWHI) National Monument, now called the Papahānaumokuākea Marine National Monument. These prohibited areas have been expanded recently to include areas out to the 200 nautical mile EEZ – including areas of the Emperor Seamounts in American waters that were the focus of intensive harvests in the 1960s. Yield data from the period 1965 2008 indicate that pink corals in the Hawaii have been harvested in a sustainable way. Yields have been relatively stable and have never exceeded estimates of MSY. Also, the age frequency distributions and recruitment rates have remained relatively stable (Grigg, 2010b). This indicates that precious corals are renewable resources and can be fished sustainably using selective methods (Grigg, 2010).

5.2.2. non-CITES species

*5.2.2.1. non-CITES species (*C. rubrum*)*

Management of *Corallium rubrum* fisheries has included a variety of national measures within territorial waters, and international legal instruments.

5.2.2.1.1. National legislation

Several countries have enacted specific laws related to *C. rubrum* management in its distribution area (Mediterranean Sea and adjacent Atlantic waters), all of whom are GFCM Contracting Parties. Actually, only the banks of *C. rubrum* within the limit of 12 nautical miles from the coast (territorial waters) can be regulated by national laws. A recent review analysed the main aspects of the red coral management measures in place in GFCM countries using as its main sources the FAO/GFCM lex database as well as the GFCM Red Coral Data Collection Form, which is used by contracting parties and cooperating non-contracting parties (CPCs) to submit data requested under existing GFCM recommendations (GFCM, 2017).

The harvesting of red coral was prohibited by national law in five Mediterranean countries (Albania, Malta, Monaco, Montenegro and Slovenia), while it was temporarily closed in the Mediterranean waters of three countries (Algeria, Greece and Morocco) and regulated by national law in five others (Croatia, France, Italy, Spain and Tunisia). Table 11 and Table 12 in Annex 1 provide summaries of the information collected on different aspects of the national measures in place, based on data transmitted by CPCs to the GFCM Secretariat since 2013 (GFCM, 2017).

Red coral harvesting is regulated at the national level in Croatia and Tunisia while in France, Spain and Italy local/regional regulations are in place and sometimes vary according to the area.[28] Only a few countries provide information on exploited banks and on the number of commercial banks. In most countries, parts of the national red coral banks are protected by marine protected areas (MPAs) or marine reserves while, in few cases, MPAs are specifically established to protect red coral. Management systems based on alternate areas (also called the rotating system) are in place in four countries, with closure–opening intervals ranging from 5 to 34 years.[29]

Information on licenses, temporal closures, minimum legal size for landing, allowed depth and areas are also provided. As far as monitoring, control and surveillance (MCS) is concerned, logbooks are mandatory in all countries, although it is not clear how the size limit is controlled in most; on the other hand, MCS measures are not fully in place in many countries. There are no observers at landing sites or on board, except in Sardinia (Italy) where an experimental programme of on-board observers is implemented and there is a good level of collaboration between scientists and fishers. Fishers in Corsica (France) and Sardinia (Italy) have received authorizations for the use of ROV for prospecting purposes while, according to the data provided, sampling and research programmes are in place in Italy only (Sardinia and Tuscany).

5.2.2.1.2. International legislation

GFCM has always kept red coral in its work plan and discussed measures to ensure sustainable harvesting of red coral since the 1980s; an improved effort was started in 2010 when a new work plan for red coral was approved. With this in mind, GFCM has organized several meetings on the topic (seven technical workshops, five of which very recently, from 2010 to 2017) with a participatory approach, so as to involve all stakeholders in the discussion (for details see Annex 1; GFCM, 2018).

Still, in 2017 red coral became part of the GFCM 'mid-term strategy (2017–2020) for the sustainability of Mediterranean and Black Sea fisheries, which was launched to define a course of decisive action aimed at reverting the alarming trend in the status of commercially exploited stocks.

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[28] Actually, during the drafting of the present document, Italy officially adopted a new 'National Plan for the Management of Red Coral'. It entered into force in January 2019. It implements the relevant EU Regulations and GFCM Reccommendations already adopted as well as other specific national measures (MINISTERO DELLE POLITICHE AGRICOLE ALIMENTARI, FORESTALI E DEL TURISMO DECRETO 21 dicembre 2018. Disposizioni nazionali sulla raccolta del corallo rosso *Corallium rubrum*. GU Anno 160 N°1, 2 gennaio 2019).

[29] Rotating harvest regime: the stock is divided into subareas, whose harvesting is staggered over a period of years, thus allowing depleted stocks to recover before harvesting restarts. Additional information on this management scheme is available in Cau *et al* (2013). Arab fishermen apparently also practised a 9-year rotating closure period in the tenth century (Grigg, 1989). Several countries and regions already had closures of coral fisheries in their fisheries legislation; notably, a 5-year closure established for coral fisheries in Sardinia in the 1970s and a 25-year closure provided for in Spanish legislation. The disadvantages of this approach include that the considerable information on population parameters required for the application of these models, which are not yet available for red coral populations in many areas. Moreover, this management scheme, by providing for the closure of a coral fishery for a long time (several decades can be necessary for juveniles to grow to adequate size), imposes very high surveillance costs and leads to an increasing incentive for illegal fishing. Furthermore, this practice could disrupt the gene flow through coral populations, and also potentially interrupt migrations of other species that take refuge in coral populations or otherwise benefit from its presence.

Aligned with the United Nations Sustainable Development Goals (SDGs), the GFCM mid-term strategy seeks to improve Mediterranean and Black Sea fisheries and contributes to the sustainable development of coastal states. Target 4 (Minimize and mitigate unwanted interactions between fisheries and marine ecosystems and environment) mentions the adoption of a comprehensive regional management plan for red coral in the GFCM competence area as one of its key activities.

Regarding management measures, four decisions on red coral harvesting have been adopted by the GFCM since 2011, with the most recent adopted in 2017 (Recommendation GFCM/41/2017/5 on the establishment of a regional adaptive management plan for the exploitation of red coral in the Mediterranean Sea formally entered into force only on 1 April 2018).

The recommendations on gear, minimum size, depth, data collection, designated ports, use of ROV for scientific purposes (Annex 1, Table 9) are binding for GFCM CPCs. In any country fishing red coral, its exploitation must be regulated according to GFCM decisions or in accordance with stricter national measures.

In addition, in 2014, GFCM agreed to adopt specific "Guidelines for the management of Mediterranean red coral populations" based on the document Adaptive Management Plan for Red Coral (*Corallium rubrum*) in the GFCM Competence Area Part I, II and III (Cau *et al*, 2013), to promote consensus on management measures to be applied in GFCM Countries in order to avoid overexploitation of red coral populations. The Guidelines were conceived to facilitate the preparation of a precautionary, provisional and adaptive regional management plan, as these provided elements to maintain the status quo of the resource in the absence of data to perform a formal assessment of the stocks at a regional scale (Annex 1).

The compulsory management measures provided in GFCM decisions (e.g. minimum landing size and minimum depth for fishing activities) are implemented at the national level. In some instances, more restrictive minimum size and stricter depth limits have been applied. Minimum landing size is not observed in one country; for two countries, by way of the derogation provided by paragraph 4 of GFCM/35/2011/2 on the exploitation of red coral in the GFCM Competence Area, the authorized exploitation of red coral at less than 50 m has been developed as an appropriate national management framework. This ensures an authorization system and the exploitation of only a limited number of red coral banks.

The implementation of additional management measures (licenses, quotas, spatial and temporal restrictions) to limit effort and harvesting, and to enforce MCS of red coral fisheries were also reviewed. Table 12 and Table 13 of Annex 1 summarize which of the additional (non-compulsory) measures proposed in the 2014 GFCM Guidelines are implemented by GFCM Countries (i.e. certified logbooks, observers on board or at landing, traceability mechanisms).

The regular assessment of the information submitted by GFCM countries in response to the aforesaid recommendations falls under the mandate of the GFCM Scientific Advisory Committee on Fisheries (SAC) for the provision of advice to the Commission. The data presented by GFCM CPCs do not yet include information requested through Recommendation GFCM/41/2017/5 on the establishment of a regional adaptive management plan for the exploitation of red coral in the Mediterranean Sea, which formally entered into force only on 1 April 2018.

Reliable data concerning the harvesting of red coral are essential in order to understand the level of exploitation of the resource. Moreover, developing a quantitative model for an exploited population is a standard precondition for drawing up a management plan for the resource. In this context, a dedicated session on red coral stock assessment to review different models that were applied to the assessment of red coral was organized by the GFCM in 2014, within the framework of the GFCM Working Group on Stock Assessment of Demersal Species (WGSAD). The models reviewed were the Schaeffer model to estimate biomass at MSY, Beverton and Holt model to estimate size at MSY, and Leslie-Lewis matrix model (GFCM, 2014b). The results of the Schaeffer model in Sardinia showed that the current biomass level was higher than B_{MSY}, therefore indicating that the resource would be at a sustainable status. However, concerns were raised over the data series used in the analysis, taken from red coral fishery logbooks and not from fishery-independent data. The WGSAD did not validate the assessment and did not issue any scientific advice, as it was considered fundamental to integrate into the assessment additional fishery-independent data into the model (GFCM, 2014b). The application of the Beverton and Holt yield-per-recruit model led to different conclusions in the investigated countries. The data from Spain (Garcia-Rodriguez and Massò, 1986b; Tsounis *et al*., 2007) revealed that these renewable resources were exploited in a non-sustainable and inefficient way, while the data from Sardinia (Follesa *et al*., 2013) highlighted an adequate management of the resource as when the age at first capture was significantly lower than the average age of 30 years of harvested colonies.

This discrepancy highlights drawbacks of the model outputs; as a consequence, the group decided to wait for some more testing of this model before using these results for advice (GFCM, 2014b). Finally, the Leslie-Lewis matrix models, as applied up to now, provided description of population dynamics, but in order to obtain a diagnosis of the stock status they should be coupled with catch-at-age data (GFCM, 2014).

5.2.2.2. non-CITES species (other species)

Corallium sp. nov., *Hemicorallium regale* (*H. laauense*) have been harvested commercially in the past years in American waters (CITES AC29 Doc.22A and its annexes) but their fishing has remained dormant since 2002 or even before (*Corallium* sp. nov., Midway deep coral, harvesting ended in the early 1990s). In 2008, however, at least two coral fishing boats from Taiwan Province of China were sighted (Fisheries Agency of Japan, 2008 in Annex 2), and the FAO global capture database reports landings of 0.4 tonnes for China in 2012. Further investigation is therefore required since it is possible that fisheries are still operating in this sea area (Annex 2).

While coral harvesting on high seas seamounts adjacent to the United States Pacific islands EEZ is subject to international fleets and is unregulated and unmonitored, in the United States of America federal waters precious coral management plans exist and include any coral of the genus *Corallium.*[30] FMP are described in the previous paragraph.

5.3. Conservation

5.3.1. Antipatharian species

The five antipatharian species occurring in the Mediterranean Sea are protected by international conventions: in Annex II of the SPA/BD Protocol (Protocol concerning the Specially Protected Areas and Biological Diversity in the Mediterranean) of the Barcelona Convention (list of endangered or threatened species) and Appendix III of the Bern Convention (list of the protected fauna species).

Protection is also granted by national laws in several other countries (CITES AC29 Doc.22; Otero *et al*., 2017).

In India and Indonesia, measures for the protection and conservation of Antipatharia have been introduced (Annex 2, Table 10).

[30] It probably refers to the old classification scheme, when many species in the family Coralliidae were classified in this genus.

5.3.2. Coralliidae species

C. rubrum is included in several international legal instruments aimed at the conservation and protection of the species and its habitat

- Within the framework of the Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean, known as the Barcelona Convention, a specific Strategic Action Program for the Conservation of Biodiversity in the Mediterranean (SAP BIO) Coralligenous AP (Action Plan) was adopted by the Contracting Parties (2008). It aims to put forward a work programme to conserve the Coralligenous ecosystem. Although it does not have a binding legal character, this action plan was adopted as a regional strategy setting priorities and activities to be undertaken. Among the major threats affecting the Coralligenous community, special care is accorded to the commercial exploitation of red coral (*C. rubrum*), whose stocks have strongly declined in most areas. An adequate management of this extremely valuable and long-lived species is considered necessary. Moreover, *C. rubrum* is included in Annex III of the SPA/BD Protocol of the Barcelona Convention – that is in the list of species whose exploitation is regulated.
- The Convention on the Conservation of European Wildlife and Natural Habitats (19.IX.1979) known as the Bern Convention, which aims to conserve wild flora and fauna and their natural habitats, especially those species and habitats whose conservation requires the co-operation of several states. Particular emphasis is given to endangered and vulnerable species, including endangered and vulnerable migratory species. In particular, *C. rubrum* is listed in Appendix III of the Bern Convention (list of the protected fauna species).
- European Council Directive 92/43/EEC of 21 May 1992 (and following amendments Directive 97/62/EC, Regulation (EC) No 1882/2003, Directive 2006/105/EC), known as the Habitats Directive, which is intended to help maintain biodiversity in the Member States by defining a common framework for the conservation of wild plants and animals, and habitats of Community interest. *C. rubrum* is listed in Annex V of the European Union Habitats Directive (List of the "Animal and plant species of Community interest whose taking in the wild and exploitation may be subject to management measures").

Protection for *C. rubrum* is also granted by national laws in several countries (Annex 1 and CITES AC29 Doc.22).

In China, precious corals are classified as Class 1 Protected Species under the Wildlife Protection Law, so their fishing is prohibited. The Wildlife Protection Law (enacted in 1988, amended in 2016) provides for the protection of precious wildlife in danger of extinction, specifies their management according to a classification system, and prohibits their unlawful capture and the destruction of their habitat. The 2016 amendment reinforced protection measures, intensified provisions against exploitation, and increased penalties (Okamura, 2016). In Article 340 of the Penal Code (1997), penalties of up to 10 years' imprisonment, fines, and the confiscation of property were established for unlawfully capture, killing or selling of endangered wildlife or Class 1 Protected Species. In 2002, the Supreme People's Court ruled that more serious offences were punishable by life imprisonment or the death penalty (Lui, 2007).

In the Philippines, according to Section 91 of The Philippine Fisheries Code of 1998 (Republic Act No. 8550), the collection, possession, sale and exportation of precious corals are prohibited in the Philippines except for research purposes. Violations of this provision are punished by 6 months' to 2 years' imprisonment and/or fines of PHP 2 000 to PHP 20 000,[31] as well as confiscation of the corals and fishing vessel.

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[31] PHP (Philippines Peso) 1 = USD 0.019 (OANDA, 1 April 2019)

After a Japanese Fisheries Agency survey using an ROV discovered precious corals at a trawl-fishing ground in the southeastern part of Koko Seamount (south of lat. 34°57′N, east of long. 171°54′E, north of lat. 34°50′N, at depths of 400 m or more), the sea area was tentatively closed to fishing from 2009. Procedures were also established in the event that corals (including black corals) are caught unintentionally in fishing nets (Fisheries Agency of Japan, 2008). These measures have been upheld by the North Pacific Fisheries Commission (NPFC), which was established in 2015 and manages fisheries active on the high seas of the Northwest Pacific (with some exceptions). As of December 2017, Japan, mainland China and Taiwan Province of China (Chinese Taipei) were among the members of the NPFC. Direct fishing of the orders Alcyonacea, Antipatharia, Gorgonacea (including Coralliidae) and Scleractinia is prohibited by Article 13(5) of the Convention on the Conservation and Management of High Seas Fisheries Resources in the North Pacific Ocean (Shiraishi, 2018).

These orders have been designated as indicator species for Vulnerable Marine Ecosystems (VMEs), with the Scientific Committee designated to address bycatch issues. Currently, the NPFC stipulates within the resolution on VMEs (CMM 2016-05 For Bottom Fisheries and Protection of VMEs in the Northwestern Pacific Ocean) that if bottom fishing vessels encounter cold water corals (Orders Alcyonacea, Antipatharia, Gorgonacea, and Scleractinia) of more than 50 kg in one gear retrieval, the vessel shall not resume fishing activities and must relocate more than 2 nautical miles from the area (Shiraishi, 2018).

6. SUGGESTIONS FOR IMPROVING MANAGEMENT, CONSERVATION AND TRADE

6.1. Biology

6.1.1. Black corals

6.1.1.1. New studies on the biology and taxonomy of commercially valuable species, particularly in harvested areas

Considering the lack of knowledge on many (most) antipatharian species of commercial importance, priority should be given to taxonomic studies and biological investigations (i.e. mortality, both natural and fishery-related, growth rate, reproduction, recruitment, biomass, density, population structure, connectivity, genetics) to gather useful information for the proper identification of harvested species, with a special focus on understanding their biological features in relation to management.

6.1.2. White, pink and red corals

6.1.2.1. New studies on the biological features of C. rubrum *per harvested areas*

In the Mediterranean context, the need to carry out scientific research studies on red coral has emerged on several occasions (GFCM 2010, 2011, 2013, 2014a, 2014b, 2017b); only small areas of red coral populations (mainly in Italy, France and Spain) have been the subject of some studies in the last three decades. The data – such as biomass, recruitment and mortality rate – necessary to construct a population dynamics model to estimate future resources and landings is almost non-existent and growth rate was studied mainly in a few ‘shallow’ populations.

The urgency of launching a Mediterranean scientific project was widely recognized (GFCM, 2017b), aiming to fill several knowledge gaps on understanding the different traits of red coral life history, essential knowledge in support of any red coral management measure. Priority is given to the collection of useful data for the provision of advice in support of management, especially as regards adopted GFCM recommendations on red coral. The combination of fishery-dependent (e.g. analysis of catch) and fishery-independent sources of information (e.g. surveys on a multi-annual basis) will ensure a regular monitoring of the resource. Guidelines to facilitate the harmonization and standardization of data collection protocols are desirable outputs of the programme.

The implementation of a Mediterranean research programme on red coral, with the characteristics outlined above, is specifically mentioned in Recommendation GFCM/41/2017/5 on the establishment of a regional adaptive management plan for the exploitation of red coral in the Mediterranean Sea (see Paragraph 28). The GFCM Secretariat, with the support of the SAC, is called upon to provide terms of reference including costs, services and other requirements to support, through a call for tender, the implementation of a research programme on red coral in the Mediterranean Sea, which should be launched in 2018 or 2019.

The data collected through the Scientific Programme will allow an assessment of the adequacy of the management measures in place, and/or whether there is a need to modify them according to the best available scientific information.

6.1.2.2. New studies on the biological features of Pacific Coralliidae per species and harvested areas

Scientific research studies on precious coral resources in the Pacific are scant. In particular, the data on the Asian species – such as biomass, recruitment, and mortality rate – necessary to construct a population dynamics model to estimate future resources and landings is almost non-existent; only growth rate has been defined (Annex 2).

Particular attention should be devoted to ascertain the possible causes of the very high percentages of dead coral colonies collected in the waters of Taiwan Province of China and Japan, in particular to investigate and quantify the impacts of dragging gears on the coral populations and associated species.

6.2. Fishery

6.2.1. Black corals

6.2.1.1. Data collection on black corals fisheries worldwide, per species/areas

Apart from in the United States of America (Hawaii), data on fishing effort and amounts of black corals harvested are lacking. Priority should be given to data collection on black coral fisheries worldwide, with detailed information per species/area.

6.2.2. White, pink and red corals

6.2.2.1. C. rubrum

6.2.2.1.1. Improvement of fishery-related data

Reliable data concerning *Corallium rubrum* harvesting are essential in order to understand the level of exploitation of the resource.

Considering that it has been recognized that the FAO global capture database for Mediterranean red coral present some shortcomings (see previous paragraphs and Annex 1 for details), the data available in the FAO database is understood as an estimate of the industrial annual production for that given year, and not the actual annual harvest of the resource.

In order to get beyond such limitations and to ensure the sustainable exploitation of red coral in the Mediterranean Sea, the GFCM started to put in place a series of measures in 2012, including the establishment of its own data collection protocol to obtain data on the annual landings of red coral from the national administrations of its member countries. The GFCM data collection (Annex 1) for red coral is coherent with the binding decisions adopted in recent years and it ensures that the minimum data required to assess the status of the fishery is collected. Nonetheless, the analysis of the data received from 2013 to 2016 highlighted that often the data related to catches: the size structure and the biology of the species are in general not sufficient to provide an overall assessment on the status of red coral populations. In some cases, important compulsory data, such as the overall quantities of red coral caught per year or the percentage of undersized colonies, has not yet been made available to GFCM.

It is essential for successful management that GFCM member states comply with the aforesaid decisions and submit precise production figures, including the percentage of undersized colonies. Future GFCM actions should address the issue of ensuring the proper reporting of obligatory data (Annex 1).

The management plan should be periodically revised by SAC, to assess its efficacy. Eventual changes should be made in accordance with up-to-date information received by CPCs, and the results of scientific studies.

6.2.2.2. Pacific Coralliidae

6.2.2.2.1. Improvement of fishery-related data

Data concerning precious coral production is essential in order to determine the condition of resources; however, statistics are not always disclosed by countries (Annex 2). Even if data are not made public, action must be taken to verify (validate) landings in order to make them useful in formal stock assessments.

6.2.2.2.2. Fishing techniques

Studies are required on the coral nets used in Japan and Taiwan Province of China, and their impact on precious corals and the environment. Coral nets, which consist of several flat nets attached to a stone weight or horizontal bar, are thought to disturb the sea floor especially when many fishing boats repeatedly drag nets over a limited area (Annex 2). Research on the way coral nets impact the sea floor is therefore necessary.

Detailed experimental trials should be undertaken to estimate the actual fishing effort of dredging gears and their eventual damage to the environment, as compared to robotic harvesting or any eventual new fishing technique.

More selective and less impactful fishing gears and practices are required. However, detailed studies and experimental tests must be compulsory before the authorization of any new harvesting technique. The full sustainability for the resource and the environment should be scientifically proven well in advance of any massive use.

6.3. Management

6.3.1. Black corals

6.3.1.1. Development, implementation, and enforcement of national management plans in harvesting countries per species

Apart from the United States of America (Hawaii), comprehensive black coral management plans (including stock assessment) do not exist in any other country. Where harvesting of these species is taking place, priority should be given to the development, implementation and enforcement of national management plans, as well as to supporting scientific studies that can help monitor and refine the effectiveness of such management plans. The plans should include the following: principles, goals and broad objectives; operational objectives, reference points and decision rules; recovery strategy; management measures; MCS system; implementation and enforcement mechanisms; reviewing the system and timeframe; stakeholder role and involvement; and conservation and ecosystem-related issues.

6.3.2. White, pink and red corals

6.3.2.1. C. rubrum

6.3.2.1.1. Implementation of GFCM Management Plan

The full implementation of the GFCM measures by member states, including the newly established management plan (Recommendation GFCM/41/2017/5 on the establishment of a regional adaptive management plan for the exploitation of red coral in the Mediterranean Sea) is expected to be sufficient to counteract or prevent overfishing, with a view to ensuring long-term yields while maintaining the size of red coral populations within biologically sustainable levels. In addition to the measures already implemented, according to the new 2017 management plan, CPCs shall: establish an individual system of daily and/or annual catch limitation; will maintain the fishing effort at the levels authorized and applied in recent years for the exploitation of red coral; and will temporarily close the area concerned to any red coral fishing activity when undersized specimens of red coral (i.e. colonies whose basal diameter is lower than 7 mm) exceeds 25 percent of the total catch harvested from a given red coral bank for a given year.

6.3.2.1.2. Enforcement of existing management measures

Only the actual level of implementation of the GFCM management plan at the national level will determine the success or failure of such a series of regional measures. In parallel, actions to eliminate IUU fishing of red coral should be also foreseen.

Nevertheless, it must be noted that red coral harvesting is currently active in only a few countries, while in most Mediterranean countries red coral is fully protected. Any unsustainable harvesting pressure or IUU fishing is expected to have mainly local effects, with the loss of red coral banks, occurring in red coral's shallower coastal habitat (i.e. between 50 and 140 metres); the GFCM has been actively working with its member countries for years to ensure that this will not eventually occur (Annex 1).

6.3.2.1.3. Species protection (spatial closures)

Several spatial and temporal closures have already been established at the national level (Annex 1). In addition to the existing closures, countries actively harvesting red coral must introduce additional closures for the protection of red coral, on the basis of the scientific advice available, by the end of 2018 (Recommendation GFCM/41/2017/5).

6.3.2.1.4. New management measures

The possibility to implement additional management measures was discussed during two recent GFCM Workshops (GFCM, 2014a, 2017b).

Among other possibilities, the implementation of programmes involving scientific observers on board, tracking devices, validation mechanisms for logbooks, has been recognized as highly effective but not always easily feasible, mostly because of the economic costs connected to implementation (Annex 1, Table 13). However, countries should endeavour to make harvesting sustainable in the long term.

6.3.2.2. Pacific Coralliidae

6.3.2.2.1. RFMO and Pacific management plan

Coralliidae corals are distributed over sea areas across different countries and international waters (United States of America, Japan, Taiwan Province of China and southern China). International cooperation is essential for the conservation of marine life and management that extends beyond the boundaries of each nation's territorial waters or EEZ.

In order to properly manage precious coral resources and to eliminate IUU fishing the establishment of a regional fishery management organization (RFMO) composed of representatives from the different countries is highly recommended (Annex 2).

Similar to what has been done in the Mediterranean region, the development, implementation and enforcement of a Pacific management plan for Coralliidae is an option worth considering.

6.3.2.2.2. Species protection

Similarly to *C. rubrum*, the increase of MPAs for the recovery of stocks is recommended.

6.4. Trade

6.4.1. Data on trade

6.4.1.1. Black corals

Trade data have been available since the inclusion of all species of black corals in Appendix II of CITES. However, in many cases the records refer to the genus or even the order because of difficulties in the identification of taxa (both raw colonies and finished products).

In any case, apart from CITES, data on trade should be obtained and strictly monitored by all the harvesting countries within the framework of fishery management plans.

6.4.1.2. White, pink and red corals

6.4.1.2.1. C. rubrum*: Data collection and eventual implementation of traceability mechanisms*

Traceability mechanisms for *C. rubrum* should be envisaged: from the time the coral is landed and sold as raw material, to its manufacture, until it reaches the retailer as a finished product. These mechanisms would allow certification that the red coral was collected in compliance with Mediterranean or national regulations and it would be also effective in eradicating red coral IUU fishing (Annex 1).

6.4.1.2.2. Pacific Coralliidae: Data collection and eventual implementation of traceability mechanisms per species

Given the difficulty to distinguish between different typologies of corals in imports/exports (precious corals, semi-precious corals, stony corals or even shells) a convenient method of data collection needs to be devised in order to compile accurate trade statistics (Annex 2).

The production area of the different species should be consistently disclosed at every stage, from the time the coral is landed and sold as raw material until it finally reaches the retailer as a finished product. This would not only enable traceability and certify that the coral was collected in compliance with the regulations of each production area, it would be effective in eliminating illegally fished coral; this would, in turn, help deter IUU fishing practices such as poaching (Annex 2).

The feasibility of using eco-labels to certify that the fishing methods and efforts have proven environmentally sustainable is a further option to explore (Annex 2).

6.4.2. *Efficacy of CITES listing for conservation and management*

In the following paragraphs two issues are briefly described: the effects of listing Antipatharia in Appendix-II of CITES, and whether or not a listing in Appendix II of CITES would improve the conservation of Coralliidae species.

6.4.2.1. Antipatharia listing

Black corals were added to Appendix II of CITES in 1981. What effects did the CITES listings have on fisheries? There is little evidence of an improvement in the conservation status of black corals and there is still a considerable lack of information on their harvesting and trade.

Black coral harvest in Hawaii is considered sustainable for about 60 years and represents an example where CITES listing and a fisheries management plan have coexisted. Two years after the CITES Appendix II listing, in 1983, a Fishery Management Plan was developed by the WPFMC and the State of Hawaii and it has been enforced at the local level in cooperation with local industry (Grigg, 2010a, b). Some effects of adding CITES regulations have been: 1) increased costs for the procurement of raw and finished black coral products; 2) increased costs of selling finished black coral products internationally; and 3) a negative impact on domestic sales, since increased procurement costs are passed on to the consumer, thereby reducing demand (Grigg, 2010a; Tsounis et al., 2010). Grigg (2010a) highlights that the requirement for CITES provisions has on occasion hindered the scientific investigation on black corals, and has been a costly administrative burden on the local industry in Hawaii, arguing that the long-term sustainability of the black coral fishery and the industry are the result of local fishery regulations and enforcement programmes in the State of Hawaii (Grigg, 2010a), even though documentation of international trade in black coral has been made more transparent.

In general, the CITES permitting process for black corals has created an administrative and costly burden for the trade of listed species and has done little to improve their conservation in Hawaii (Grigg, 2010a). It is important to point out that the black coral fishery in Hawaii is a unique case study, as the fishery is regulated by FMP. In other parts of the world where such FMP do not exist, CITES regulations may still be an important conservation measure protecting populations from overexploitation. The important point to remember is that CITES regulations should not be used as the only conservation measure: enforced and developed FMP may be much more effective at conserving populations. Additionally, it should be noted that while CITES regulations are applicable to all of the –250 black coral species currently described, only a small fraction of those are commercially harvested. Thus, other negative impacts of CITES regulations, such as increased difficulties for scientific studies on this group, should be addressed by providing researchers with easier ways to adhere to CITES regulations (e.g. simplifying existing procedures for scientific exchanges or researches). CITES has recognized this need and is working on it. Initially it canvassed Parties on where permitting requirements posed a significant barrier to their research. Responses that referred more generally to CITES permitting procedures highlighted the length of time to obtain a permit, the complexity and burdensome administrative processes and the fact that delays in issuing permits, permit costs, the registration of scientific organisations in accordance with Article VII paragraph 6, of the Convention and Resolution Conf. 11.15 (Rev. CoP12) meant processing of permits could lead to degradation of samples. Recognizing the need for putting in place simplified procedures to movements of legitimate scientific samples of Appendix II listed species is a current focus of CITES (CITES, 2018b)

6.4.2.2. <u>Coralliidae listing</u>

The inclusion by China (effective 1 July 2008) of four species of Coralliidae (*Corallium japonicum, Pleurocorallium elatius, P. konojoi, and P. secundum*) in Appendix III of CITES requires all international traders to obtain permits and certificates for import/export and placed a heavy administrative burden on the industry (Grigg, 2010b) while not reducing IUU or deterring poachers (Annex 2). CITES Appendix III has widely acknowledged problems with implementation (see draft resolution in CITES, 2018) and has observed that many Parties are not effectively implementing the provisions of the Convention with regard to Appendix III. There is also a danger that such a listing may have counterproductive responses from the market, since it could be considered as evidence of rarity, which may therefore stimulate the desire to purchase and sales of these corals, thus also increasing the request on fishers to supply the raw materials for the trade (Annex 2; Chang, 2013).

In the case of nationally led legislative processes, in Germany *C. rubrum* was given a temporary, strict protection (January 1987–June 1997) through the listing of the species in Annex 1 of Germany's Federal Ordinance on Species Conservation. This resulted in a total prohibition of any commercial trade into Germany both from EU and non-EU Member States. In that period German customs was confronted with commercial imports of pre-manufactured and manufactured products made of other *Corallium* species with significant (irresolvable) identification problems at the species level for enforcement officials. This was particularly true of pre-manufactured products, jewellery, or products made from coral powder (Jelden, 2010). The species is not listed anymore in the country.

In addition, proposals for listing Coralliidae species in CITES have presented on three occasions: in 1987 at CITES CoP6 by Spain, limited to *C. rubrum*; in 2007 at CITES CoP14 by the United States of America, limited to the genus *Corallium*; and in 2010 at CITES CoP15 by the European Union and its Member States, and United States of America, aimed at the family Coralliidae. Arguments in support of listing have stated that because of commercial fishery boom and bust cycles caused by inadequate fishery management, a CITES listing would provide a vital tool to strengthen conservation through trade monitoring, further control of resources (Tsounis et al., 2010; Chang, 2013 and references therein) and in identifying scientific and research needs (Bruckner et al., 2009). In addition, Bruckner et al. (2009) argued the value of a CITES listing in raising awareness within the general public, industry, and other stakeholders that sustainability was a concern with this resource.

When there has been discussion on the value of listing precious corals, the FAO Expert Panel delivered a consensus decision that the species did not meet the CITES criteria and CITES Parties rejected all of the proposals. The FAO Expert Advisory Panel that reviewed the CITES Proposals for CoP14 in 2007 (FAO, 2007) noted that:

> ...despite being harvested since prehistoric times, the Mediterranean population of *C. rubrum* is still widespread. Small but mature colonies have high local densities. Nevertheless, mature colonies are now smaller than the minimum size for harvest and a problem is that large colonies have an important role in providing recruitment. These problems need to be addressed by the implementation and enforcement of suitable local management measures.

In 2010, for CITES CoP15, the FAO Expert Advisory Panel (FAO 2010) reiterated the view of the FAO (2007) assessment:

> The Panel does not recommend a CITES Appendix II listing for *Coralliidae* spp. Nevertheless, since international trade is a driver of their harvesting, if such a listing resulted in a tightening of their management, it could lead to an improvement in their status. However, this improved status would be bought at the cost of a considerable administrative overheads, and Government efforts would be better employed in enacting and enforcing appropriate local management regimes.

> The Panel cautions that if Coralliidae were included in Appendix II, aspects of the implementation would be problematic, particularly the identification at the species level of processed products and providing a suitable protocol for pre-convention specimens. The Panel noted that a very large number (many thousands) of small, individual specimens is in trade, meaning that a significant amount of paperwork would be required to track all items in trade.
>
> The Panel was convinced that the Coralliidae do require to be managed within EEZs and in areas beyond national jurisdiction in a fashion which takes account of their long life and their ecological role. The Panel considered that these long-lived species require appropriate and effective local management such as harvest restrictions and rotational closures and protected areas to facilitate their sustainable harvest.

In fact, the FAO views triggered renewed efforts from GFCM. In a few years, these have established comprehensive management measures for *C. rubrum*, and led to improved legislation for Pacific species (see Annexes 1 and 2).

The possible listing of all Coralliidae species in Appendix II of CITES could encounter similar problems as the black coral listing, in addition to new challenges:

1. Only a few species are commercially harvested in the world and species identification is very difficult, with even fake coral sometimes hard to differentiate from worked coral (Grigg, 2010a; Tsounis *et al.,* 2010). The identification of coral items relies on their colour. However, there are large colour variations within the same species. For the implementation of an export system based on CITES, a reliable method of species identification needs to be established (Annex 2).
2. Many Coralliidae specimens traded in the world today were harvested years ago and are drawn from stockpiles and are therefore exempt from CITES (Grigg, 2010a). This also the case for when jewellery under warranty is sent back to the manufacturer for repair and is held in customs due to missing permits (Tsounis *et al.,* 2010).
3. The discovery and mining of large beds of fossilised coral that would be covered by CITES provisions but that would have no link to the conservation of the species (Tsounis *et al.,* 2010)
4. Many species have distribution boundaries that extend beyond country boundaries, thus obscuring the location of collection (Grigg, 2010a). It should be noted that CITES has provisions for introduction from the high seas, however full implementation of these measures lags somewhat behind CITES adoption of these provisions.
5. A further potential problem is that a listing in Appendix II of CITES may give the impression that species are under management control, thereby defusing efforts at the local level to enact legislation and enforcement programmes (Grigg, 2010a; Tsounis *et al.,* 2010).
6. Additionally, from a socio-economic and cultural perspective opponents of the listing have argued that the CITES listing of precious corals may endanger the livelihood of local people, who depend on the precious coral handicraft industry, and damage to the traditional culture (Chang, 2013 and references therein).

Actually, at the international level, there is no doubt that precious coral resources need much stronger management practices and would greatly benefit from the creation of new spatial and fishery protections. However it is suggested that these actions (management and conservation) are best accomplished through action at the local level through cooperation of science and fishery governance measures, including surveillance, enforcement and prosecution (Grigg, 2010a).

On the other hand, advocates of trade controls argued that because corals grow slowly, poor fishery management methods will cause damage to corals that require decades to correct and for the corals to recover (Tsounis *et al*., 2010). They have indicated management failures in many communities or countries are caused by inadequate fishery management or an inability to reject short-term, high-profit economic incentives. They also advocate CITES listing as a vital tool for trade monitoring and control of resources (Tsounis *et al*., 2010; Chang 2013 and references therein).

The benefits of a CITES Appendix-II listing for Coralliidae were also recognized. In particular (Bruckner *et al*., 2009):

1. The listing would provide trade data that are not currently available; fill some gaps in knowledge in regards to the international volume of trade in Coralliidae. It would highlight areas where scientific and trade research should be conducted to better inform management and enforcement.
2. It would raise awareness within the general public, industry, and other stakeholders that sustainability is a concern with this resource. The aim of a listing is to prevent unsustainable use of the resource, which would ultimately benefit the long-term viability of the industry that depends on it.

However, it was also stressed that for any listing to be effective, CITES authorities must collaborate with relevant fishery management bodies, as it can be seen in the current case of CITES promoting complementary management between the biodiversity conservation and fishery sectors in the management and conservation of sharks and rays (CITES, 2016). Whilst CITES provides a mechanism to control international trade, it is fishery management that ultimately determines the sustainability of exploitation of this resource (Bruckner *et al*., 2009).

The important advances in fishery management that have occurred since 2010 in the Mediterranean, through GFCM, and in the Pacific though stricter domestic legislations, constitute important steps towards the sustainability of harvest – and the subsequent trade of these important marine resources.

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APPENDIX - ORDER ANTIPATHARIA: SPECIES, GROWTH RATES, AND DENSITIES VALUES

Table A– Species included in the Order Antipatharia (source WORMS Word register of marine species 2018); old combination, ** putative new combination; ° junior synonym; °° synonymy.*

Taxon		notes
Phylum	Cnidaria	
Class	Antozoa	
Subclass	Hexacorallia	
Order	Antipatharia	
Family	Antipathidae Ehrenberg, 1834	
Genus	*Allopathes* Opresko & Cairns, 1994	
Species	*Allopathes denhartogi* Opresko, 2003	
Species	*Allopathes desbonni* (Duchassaing & Michelotti, 1864)	*Antipathes desbonni* (Duchassaing & Michelotti, 1864)*; *Cirrhipathes desbonni* Duchassaing & Michelotti, 1864*
Species	*Allopathes robillardi* (Bell, 1891)	*Antipathes robillardi* Bell, 1891*
Genus	*Antipathes* Pallas, 1766	
Species	*Antipathes aculeata* (Brook, 1889)	
Species	*Antipathes arborea* Dana, 1846	
Species	*Antipathes assimilis* (Brook, 1889)	*Antipathella assimilis* Brook, 1889*
Species	*Antipathes atlantica* Gray, 1857	
Species	*Antipathes brooki* (Whitelegge & Hill, 1899)	*Antipathella brooki* Whitelegge, 1899*
Species	*Antipathes caribbeana* Opresko, 1996	
Species	*Antipathes ceylonensis* (Thomson & Simpson, 1905)	*Antipathella ceylonensis* Thomson & Simpson, 1905*
Species	*Antipathes chamaemorus* Pax & Tischbierek, 1932	
Species	*Antipathes chota* Cooper, 1903	
Species	*Antipathes clathrata*	*Arachnopathes clathrata* (Pallas, 1766)°°
Species	*Antipathes contorta* (Brook, 1889)	*Antipathella contorta* Brook, 1889*
Species	*Antipathes craticulata* Opresko, 2015	
Species	*Antipathes curvata* van Pesch, 1914	
Species	*Antipathes delicatula* Schultze, 1896	
Species	*Antipathes dendrochristos* Opresko, 2005	
Species	*Antipathes densa* Silberfeld, 1909	
Species	*Antipathes dichotoma* Pallas, 1766	*Antipathes aenea* von Koch, 1889°; *Antipathes mediterranea* Brook, 1889°°
Species	*Antipathes dofleini* Pax, 1915	
Species	*Antipathes dubia* (Brook, 1889)	

Species	*Antipathes elegans* (Thomson & Simpson, 1905)	*Antipathella elegans* Thomson & Simpson, 1905*
Species	*Antipathes ericoides*	*Arachnopathes ericoides* (Pallas, 1766)°°
Species	*Antipathes erinaceus* (Roule, 1905)	*Aphanipathes erinaceus* Roule, 1905°°
Species	*Antipathes flabellum* Pallas, 1766	(nomen dubium)
Species	*Antipathes fragilis* Gravier, 1918	*Antipathes flexibilis* Gravier, 1919°°
Species	*Antipathes fruticosa* Gray, 1857	*Aphanipathes fruticosa* (Gray, 1857)*
Species	*Antipathes furcata* Gray, 1857	
Species	*Antipathes galapagensis* Deichmann, 1941	
Species	*Antipathes gallensis* Thomson & Simpson, 1905	
Species	*Antipathes gracilis* Gray, 1860	*Antipathella brooki* Johnson, 1899*; *Antipathella gracilis* (Gray, 1860)*
Species	*Antipathes grandiflora* Silberfeld, 1909	
Species	*Antipathes grandis* Verrill, 1928	
Species	*Antipathes grayi* Roule, 1905	*Tylopathes grayi* Roule, 1902°°
Species	*Antipathes griggi* Opresko, 2009	
Species	*Antipathes herdmanni* Cooper, 1909	
Species	*Antipathes hypnoides* (Brook, 1889)	
Species	*Antipathes indistincta* (van Pesch, 1914)	*Aphanipathes indistincta*°°
Species	*Antipathes irregularis* (Thomson & Simpson, 1905)	*Antipathella irregularis* Thompson & Simpson, 1905*
Species	*Antipathes irregularis* Cooper, 1909	
Species	*Antipathes irregularis* Verrill, 1928	
Species	*Antipathes lenta* Pourtalès, 1871	
Species	*Antipathes lentipinna* Brook, 1889	
Species	*Antipathes leptocrada* Opresko, 2015	
Species	*Antipathes longibrachiata* van Pesch, 1914	
Species	*Antipathes minor (Brook, 1889)*	*Antipathella minor* Brook, 1889°°
Species	*Antipathes nilanduensis* Cooper, 1903	
Species	*Antipathes orichalcea* Pallas, 1766	
Species	*Antipathes pauroclema* Pax & Tischbierek, 1932	
Species	*Antipathes plana* Cooper, 1909	
Species	*Antipathes plantagenista* (Cooper, 1903)	*Aphanipathes plantagenista* Cooper, 1903*
Species	*Antipathes pseudodichotoma* Silberfeld, 1909	
Species	*Antipathes regularis* Cooper, 1903	
Species	*Antipathes rhipidion* Pax, 1916	
Species	*Antipathes rubra* Cooper, 1903	
Species	*Antipathes rubusiformis* Warner & Opresko, 2004	
Species	*Antipathes salicoides* Summers, 1910	

Species	*Antipathes sarothrum* Pax, 1932	
Species	*Antipathes sealarki* Cooper, 1909	
Species	*Antipathes sibogae* (van Pesch, 1914)	*Aphanipathes sibogae* van Pesch, 1914*
Species	*Antipathes simplex* (Schultze, 1896)	
Species	*Antipathes simpsoni* (Summers, 1910)	
Species	*Antipathes speciosa* (Brook, 1889)	*Antipathella speciosa* Brook, 1889°°
Species	*Antipathes spinulosa* (Schultze, 1896)	
Species	*Antipathes taxiformis* Duchassaing, 1870	
Species	*Antipathes ternatensis* Schultze, 1896	
Species	*Antipathes thamnoides* (Schultze, 1896)	*Aphanipathes thamnoides* Schultze, 1896*
Species	*Antipathes tristis* (Duchassaing, 1870)	
Species	*Antipathes umbratica* Opresko, 1996	
Species	*Antipathes viminalis* Roule, 1902	
Species	*Antipathes virgata* Esper, 1788	
Species	*Antipathes zoothallus* Pax, 1932	
Genus	*Cirrhipathes* de Blainville, 1830	
Species	*Cirrhipathes anguina* (Dana, 1846)	*Antipathes anguina* Dana, 1846*
Species	*Cirrhipathes contorta* van Pesch, 1910	
Species	*Cirrhipathes densiflora* Silberfeld, 1909	
Species	*Cirrhipathes diversa* Brook, 1889	
Species	*Cirrhipathes gardineri* Cooper, 1903	
Species	*Cirrhipathes hainanensis* Zou & Zhou	
Species	*Cirrhipathes indica* Summers, 1910	
Species	*Cirrhipathes musculosa* van Pesch, 1910	
Species	*Cirrhipathes nana* van Pesch, 1910	
Species	*Cirrhipathes propinqua* Brook, 1889	
Species	*Cirrhipathes rumphii* van Pesch, 1910	
Species	*Cirrhipathes secchini* Echeverria, 2002	
Species	*Cirrhipathes sieboldi* Blainville, 1834	nomen nudum
Species	*Cirrhipathes sinensis* Zou & Zhou, 1984	
Species	*Cirrhipathes spiralis* (Linnaeus, 1758)	
Species	*Cirrhipathes translucens* van Pesch, 1910	
Genus	*Hillopathes* van Pesch, 1914	
Species	*Hillopathes ramosa* (van Pesch, 1910)	*Cirrhipathes ramosa* van Pesch, 1910*
Genus	*Pseudocirrhipathes* Bo *et al.* 2009	
Species	*Pseudocirrhipathes mapia* Bo *et al.* 2009	

Genus	*Pteropathes* Brook, 1889	
Species	*Pteropathes fragilis* Brook, 1889	*Antipathes simpsoni* (Summers, 1910)*
Genus	*Stichopathes* Brook, 1889	
Species	*Stichopathes abyssicola* Roule, 1902	
Species	*Stichopathes aggregata* van Pesch, 1914	
Species	*Stichopathes alcocki* Cooper, 1909	
Species	*Stichopathes bispinosa* Summers, 1910	*Cirrhipathes flagellum* Brook, 1889°
Species	*Stichopathes bournei* Cooper, 1909	
Species	*Stichopathes ceylonensis* Thomson & Simpson, 1905	
Species	*Stichopathes contorta* Thomson & Simpson, 1905	
Species	*Stichopathes dissimilis* Roule, 1902	
Species	*Stichopathes echinulata* Brook, 1889	
Species	*Stichopathes euoplos* Schultze, 1903	
Species	*Stichopathes eustropha* Pax, 1931	
Species	*Stichopathes filiformis* Gray, 1868	
Species	*Stichopathes flagellum* Roule, 1902	
Species	*Stichopathes gracilis* Gray, 1857	
Species	*Stichopathes gravieri* Molodtsova, 2006	
Species	*Stichopathes indica* Schultze, 1903	
Species	*Stichopathes japonica* Silberfeld, 1909	
Species	*Stichopathes longispina* Cooper, 1909	
Species	*Stichopathes luetkeni* Brook, 1888	*Stichopathes lutkeni* Brook, 1889 (alternative spelling)
Species	*Stichopathes maldivensis* Cooper, 1903	
Species	*Stichopathes occidentalis* Gray, 1860	
Species	*Stichopathes papillosa* Thomson & Simpson, 1905	
Species	*Stichopathes paucispina* (Brook, 1889)	*Cirrhipathes paucispina* Brook, 1889*
Species	*Stichopathes pourtalesi* Brook, 1889	
Species	*Stichopathes regularis* Cooper, 1909	
Species	*Stichopathes richardi* Roule, 1902	
Species	*Stichopathes saccula* van Pesch, 1914	
Species	*Stichopathes semiglabra* van Pesch, 1914	
Species	*Stichopathes setacea* Gray, 1860	
Species	*Stichopathes seychellensis* Cooper, 1909	
Species	*Stichopathes solorensis* van Pesch, 1914	
Species	*Stichopathes spiessi* Opresko & Genin, 1990	
Species	*Stichopathes spinosa* Silberfeld, 1909	

Species	*Stichopathes variabilis* van Pesch, 1914	
Family	*Aphanipathidae* Opresko, 2004	
Subfamily	*Acanthopathinae* Opresko 2004	
Genus	*Acanthopathes* Opresko 2004	
Species	*Acanthopathes hancocki* (Cooper, 1909)	*Aphanipathes hancocki* Cooper, 1909*
Species	*Acanthopathes humilis* (Pourtalès, 1867)	*Antipathes humilis* Pourtalès, 1867*
Species	*Acanthopathes somervillei* (Cooper, 1909)	*Aphanipathes somervillei* Cooper, 1909*
Species	*Acanthopathes thyoides* (Pourtalès, 1880)	*Antipathes thyoides* Pourtalès, 1880*
Species	*Acanthopathes undulata* (van Pesch, 1914)	*Aphanipathes undulata* van Pesch, 1914*
Genus	*Distichopathes* Opresko 2004	
Species	*Distichopathes disticha* Opresko, 2004	
Species	*Distichopathes filix* (Pourtalès, 1867)	*Antipathes filix* Pourtalès, 1867*; *Aphanipathes filix* (Pourtalès, 1867)*
Genus	*Elatopathes* Opresko 2004	
Species	*Elatopathes abietina* (Pourtalès, 1874)	*Antipathes abietina* Pourtalès, 1874*
Genus	*Rhipidipathes* Milne Edwards & Haime, 1857	
Species	*Rhipidipathes colombiana* (Opresko & Sánchez, 1997)	*Aphanipathes colombiana* Opresko & Sanchez, 1997*
Species	*Rhipidipathes reticulata* (Esper, 1795)	*Antipathes reticulata* Esper, 1795*
Subfamily	*Aphanipathinae* Opresko, 2004	
Genus	*Aphanipathes* Opresko, 2004	
Species	*Aphanipathes pedata* (Gray, 1857)	*Antipathes pedata* Gray, 1857*
Species	*Aphanipathes reticulata* van Pesch, 1914 (temporary name)	
Species	*Aphanipathes salix* (Pourtalès, 1880)	*Antipathes salix* Pourtalès, 1880*
Species	*Aphanipathes sarothamnoides* Brook, 1889	
Species	*Aphanipathes verticillata* Brook, 1889	
Genus	*Asteriopathes* Opresko, 2004	
Species	*Asteriopathes arachniformis* Opresko, 2004	
Species	*Asteriopathes colini* Opresko, 2004	
Species	*Asteriopathes octocrada* Opresko, 2015	
Genus	*Phanopathes* Opresko, 2004	
Species	*Phanopathes cancellata* (Brook, 1889)	*Aphanipathes cancellata* Brook, 1889*
Species	*Phanopathes expansa* (Opresko & Cairns, 1992)	*Antipathes expansa* Opresko & Cairns, 1992*
Species	*Phanopathes rigida* (Pourtalès, 1880)	*Antipathes rigida* Pourtalès, 1880*
Species	*Phanopathes zealandica* Opresko, 2015	
Genus	*Pteridophates* Opresko, 2004	
Species	*Pteridopathes pinnata* Opresko, 2004	

Species	*Pteridopathes tanycrada* Opresko, 2004	
Genus	*Tetrapathes* Opresko, 2004	
Species	*Tetrapathes alata* (Brook, 1889)	*Aphanipathes alata* Brook, 1889*
Species	*Tetrapathes latispina* Opresko, 2015	
Family	*Cladopathidae* Kinoshita, 1910	
Subfamily	*Cladopathinae* Kinoshita, 1910	
Genus	*Chrysopathes* Opresko, 2003	
Species	*Chrysopathes formosa* Opresko, 2003	
Species	*Chrysopathes gracilis* Opresko, 2005	
Species	*Chrysopathes micracantha* Opresko & Loiola, 2008	
Species	*Chrysopathes oligocrada* Opresko & Loiola, 2008	
Species	*Chrysopathes speciosa* Opresko, 2003	
Genus	*Cladopathes* Brooks, 1889	
Species	*Cladopathes plumosa* Brook, 1889	
Genus	*Trissopathes* Opresko, 2003	
Species	*Trissopathes pseudotristicha* Opresko, 2003	
Species	*Trissopathes tetracrada* Opresko, 2003	
Species	*Trissopathes tristicha* (van Pesch, 1914)	
Subfamily	*Hexapathinae Opresko, 2003*	
Genus	*Heteropathes* Opresko, 2011	*Heliopathes* Opresko, 2011°
Species	*Heteropathes americana* (Opresko, 2003)	*Heliopathes americana* Opresko, 2003*
Species	*Heteropathes heterorhodzos* (Cooper, 1909)	*Antipathes heterorhodzos* Cooper, 1909*; *Bathypathes heterorhodzos* (Cooper, 1909)*; *Heliopathes heterorhodzos* (Cooper, 1909)°°
Species	*Heteropathes opreski* de Moto, Braga-Henriques, Santos & Ribero, 2014	
Species	*Heteropathes pacifica* (Opresko, 2005)	*Heliopathes heterorhodzos* (Cooper, 1909)*
Genus	*Hexapathes* Kinoshita, 1910	
Species	*Hexapathes alis* Molodtsova, 2006	
Species	*Hexapathes australiensis* Opresko, 2003	
Species	*Hexapathes heterosticha* Kinoshita, 1910	
Species	*Hexapathes hivaensis* Molodtsova, 2006	
Subfamily	*Sibopathinae* Opresko, 2003	
Genus	*Sibopathes* van Pesch, 1914	
Species	*Sibopathes gephura* van Pesch, 1914	
Species	*Sibopathes macrospina* Opresko, 1993	
Family	*Leiopathidae* Haeckel, 1896	

Genus	*Leiopathes* Haime, 1849	
Species	*Leiopathes acanthophora* Opresko, 1998	
Species	*Leiopathes annosa* Wagner & Opresko, 2015	
Species	*Leiopathes bullosa* Opresko, 1998	
Species	*Leiopathes expansa* Johnson, 1899	
Species	*Leiopathes glaberrima* (Esper, 1788)	*Antipathes glaberrima**
Species	*Leiopathes grimaldii* Roule, 1902	
Species	*Leiopathes montana* Molodtsova, 2011	
Species	*Leiopathes secunda* Opresko, 1998	
Species	*Leiopathes valdiviae* (Pax, 1915)	*Antipathes valdiviae* Pax, 1915*
Family	*Myriopathidae* Opresko, 2001	
Genus	*Antipathella* Brook, 1889	
Species	*Antipathella aperta* (Totton, 1923)	*Antipathes aperta* Totton, 1923*
Species	*Antipathella fiordensis* (Grange, 1990)	*Antipathes fiordensis* Grange, 1990*
Species	*Antipathella strigosa* (Brook, 1889)	*Antipathes strigosa* (Brook, 1889)*
Species	*Antipathella subpinnata* (Ellis & Solander, 1786)	*Antipathes subpinnata* Ellis & Solander, 1786*
Species	*Antipathella wollastoni* (Gray, 1857)	*Antipathes wollastoni* Gray, 1857*; *Aphanipathes wollastoni* (Gray, 1857)*
Genus	*Cupressopathes* Opresko, 2001	
Species	*Cupressopathes abies* (Linnaeus, 1758)	*Antipathes abies* (Linnaeus, 1758)*
Species	*Cupressopathes cylindrica* (Brook, 1889)	*Antipathes cylindrica* Brook, 1889*
Species	*Cupressopathes gracilis* (Thomson & Simpson, 1905)	*Antipathes gracilis* Thomson & Simpson, 1905*
Species	*Cupressopathes paniculata* (Esper, 1796)	*Antipathes paniculata* Esper, 1797*
Species	*Cupressopathes pumila* (Brook, 1889)	*Antipathes pumila* Brook, 1889*
Genus	*Myriopathes* Opresko, 2001	
Species	*Myriopathes antrocrada* (Opresko, 1999)	*Antipathes antrocrada* Opresko, 1999*
Species	*Myriopathes bifaria* (Brook, 1889)	*Antipathes bifaria* Brook, 1889*
Species	*Myriopathes catharinae* (Pax, 1932)	*Antipathes catharinae* (Pax, 1932); *Aphanipathes catharinae* Pax, 1932*
Species	*Myriopathes japonica* (Brook, 1889)	*Antipathes japonica* Brook, 1889*
Species	*Myriopathes lata* (Silberfeld, 1909)	*Antipathes lata* Silberfeld, 1909*
Species	*Myriopathes myriophylla* (Pallas, 1766)	*Antipathes myriophylla* Pallas, 1766*
Species	*Myriopathes panamensis* (Verrill, 1869)	*Antipathes panamensis* Verrill, 1869*
Species	*Myriopathes rugosa* (Thomson & Simpson, 1905)	*Aphanipathes rugosa* Thomson & Simpson, 1905°°
Species	*Myriopathes spinosa* (Carter, 1880)	*Hydradendrium spinosum* Carter, 1880*
Species	*Myriopathes stechowi* (Pax, 1932)	*Aphanipathes stechowi* Pax, 1932*
Species	*Myriopathes ulex* (Ellis & Solander, 1786)	*Antipathes ulex* Ellis & Solander, 1786*

Genus	*Plumapathes* Opresko, 2001	
Species	*Plumapathes fernandezi* (Pourtalès, 1874)	*Antipathes fernandezi* Pourtalès, 1874*
Species	*Plumapathes pennacea* (Pallas, 1766)	*Antipathes pennacea* Pallas, 1766*
Genus	*Tanacetipathes* Opresko, 2001	
Species	*Tanacetipathes barbadensis* (Brook, 1889)	*Aphanipathes barbadensis* Brook, 1889*
Species	*Tanacetipathes cavernicola* Opresko, 2001	
Species	*Tanacetipathes hirta* (Gray, 1857)	*Antipathes hirta* Gray, 1857*
Species	*Tanacetipathes longipinnula* Loiola & Castro, 2005	
Species	*Tanacetipathes spinescens* (Gray, 1857)	*Antipathes spinescens* Gray, 1857*
Species	*Tanacetipathes squamosa* (Koch, 1886)	*Antipathes squamosa* Koch, 1886*
Species	*Tanacetipathes tanacetum* (Pourtalès, 1880)	*Antipathes tanacetum* Pourtalès, 1880*
Species	*Tanacetipathes thalassoros* Loiola & Castro, 2005	
Species	*Tanacetipathes thamnea* (Warner, 1981)	*Antipathes thamnea*Warner, 1981*; *Tanacetipathes paula* Perez & Costa, 2005°
Species	*Tanacetipathes wirtzi* Opresko, 2001	
Family	*Schizopathidae* Brook, 1889	
Genus	*Abyssopathes* Opresko, 2002	
Species	*Abyssopathes anomala* Molodtsova & Opresko, 2017	
Species	*Abyssopathes lyra* (Brook, 1889)	*Bathypathes lyra* Brook, 1889*
Species	*Abyssopathes lyriformis* Opresko, 2002	
Genus	*Alternatipathes* Molodtsova & Opresko, 2017	
Species	*Alternatipathes alternata* (Brook, 1889)	*Bathypathes alternata* Brook, 1889*
Species	*Alternatipathes bipinnata* (Opresko, 2005)	*Umbellapathes bipinnata* Opresko, 2005*
Genus	*Bathypathes* Brook, 1889	
Species	*Bathypathes bayeri* Opresko, 2001	
Species	*Bathypathes bifida* Thompson, 1905	
Species	*Bathypathes conferta* (Brook, 1889)	
Species	*Bathypathes erotema* Schultze, 1903	
Species	*Bathypathes galatheae* Pasternak, 1977	
Species	*Bathypathes patula* Brook, 1889	
Species	*Bathypathes platycaulus* Totton, 1923	
Species	*Bathypathes robusta* (Gravier, 1918)	*Stichopathes robusta* Gravier, 1918*
Species	*Bathypathes seculata* Opresko, 2005	
Genus	*Dendrobathypathes* Opresko, 2002	
Species	*Dendrobathypathes boutillieri* Opresko, 2005	
Species	*Dendrobathypathes fragilis* Opresko, 2005	
Species	*Dendrobathypathes grandis* Opresko, 2002	

Species	*Dendrobathypathes isocrada* Opresko, 2002	
Genus	*Dendropathes* Opresko, 2005	
Species	*Dendropathes bacotaylorae* Opresko, 2005	
Species	*Dendropathes intermedia* (Brook, 1889)	*Antipathes intermedia* (Brook, 1889)*; *Antipathella intermedia* Brook, 1889*
Genus	*Lillipathes* Opresko, 2002	
Species	*Lillipathes lillei* (Totton, 1923)	*Antipathes lilliei* Totton, 1923*
Species	*Lillipathes quadribrachiata* (van Pesch, 1914)	*Bathypathes (Eubathypathes) quadribrachilata* van Pesch, 1914*; *Bathypathes quadribrachiata* van Pesch, 1914*
Species	*Lillipathes ritamariae* Opresko & Breedy, 2010	
Species	*Lillipathes wingi* Opresko, 2005	
Genus	*Parantipathes* Brook, 1889	
Species	*Parantipathes dodecasticha* Opresko, 2015	
Species	*Parantipathes euantha* (Pasternak, 1958)	*Bathypathes euantha* Pasternak, 1958*
Species	*Parantipathes helicosticha* Opresko, 1999	
Species	*Parantipathes hirondelle* Molodtsova, 2006	
Species	*Parantipathes laricides* van Pesch, 1914	
Species	*Parantipathes larix* (Esper, 1788)	*Antipathes larix* Esper, 1788*
Species	*Parantipathes robusta* Opresko, 2015	
Species	*Parantipathes tetrasticha* (Pourtalès, 1868)	*Antipathes tetrasticha* Pourtalès, 1868*
Species	*Parantipathes wolffi* Pasternak, 1977	
Genus	*Saropathes* Opresko, 2002	
Species	*Saropathes margaritae* Molodtsova, 2005	
Species	*Saropathes scoparia* (Totton, 1923)	*Bathypathes scoparia* Totton, 1923*
Genus	*Schizopathes* Brook, 1889	
Species	*Schizopathes affinis* Brook, 1889	
Species	*Schizopathes amplispina* Opresko, 1997	
Species	*Schizopathes crassa* Brook, 1889	
Genus	*Stauropathes* Opresko, 2002	
Species	*Stauropathes arctica* (Lütken, 1871)	*Antipathes arctica*; Bathypathes arctica* (Lütken, 1871)*
Species	*Stauropathes punctata* (Roule, 1905)	*Antipathes punctata* (Roule, 1905)*; *Tylopathes punctata* Roule, 1905*
Species	*Stauropathes staurocrada* Opresko, 2002	
Genus	*Taxipathes* Brook, 1889	
Species	*Taxipathes recta* Brook, 1889	
Genus	*Telopathes* MacIsaac & Best, 2013	

Species	*Telopathes magnus MacIsaac & Best, 2013*	
Genus	*Umbellapathes* Opresko, 2005	
Species	*Umbellapathes helioanthes Opresko, 2005*	
Species	*Umbellapathes tenuis (Brook, 1889)*	*Bathypathes tenuis* Brook, 1889*
Family	*Stylopathidae* Opresko, 2006	
Genus	*Stylopathes* Opresko, 2006	
Species	*Stylopathes adinocrada* Opresko, 2006	
Species	*Stylopathes americana* (Duchassaing & Michelotti, 1860)	*Antipathes americana* Duchassaing & Michelotti, 1860*
Species	*Stylopathes columnaris* (Duchassaing, 1870)	*Antipathes columnaris* (Duchassaing, 1870)*
Species	*Stylopathes litocrada* Opresko, 2006	
Species	*Stylopathes tenuispina* (Silberfeld, 1909)	*Antipathes tenuispina* (Silberfeld, 1909)*; *Parantipathes tenuispina* Silberfeld, 1909*
Genus	*Triadopathes* Opresko, 2006	
Genus	*Tylopathes* Brook, 1889	
Subgenus	*Tylopathes (Paratylopathes)* Roule, 1905	
Species	*Tylopathes (Paratylopathes) atlantica* Roule, 1905	
Species	*Tylopathes crispa* Brook, 1889	
Species	*Tylopathes dubia* Brook, 1889	*Tylopathes dofleini*Brook, 1889°°
Species	*Tylopathes elegans* Brook, 1889	
Species	*Tylopathes glutinata* (Totton, 1923)**	*Antipathes glutinata* Totton, 1923
Species	*Tylopathes hypnoides* Brook, 1889	

Table B Growth rate and longevity data - Antipatharian species (source Wagner et al. 2012, modified)

Species	Location	Depth(m)	Vertical Growth rate (cm/yr)	Radial growth rate (mm/yr)	Longevity	Method	Reference
Antipathes dendrochristos	California	106	1.5	0.121	140	Growth ring counts, 210Pb and 14C dating	Love *et al*. (2007)
Antipathes grandis	Hawai‘i		2.92		100	Time-series of tagged colonies	Grigg (1974)
Antipathes griggi	Hawai‘i		5.86		40	Time-series of tagged colonies	Grigg (1974)
Antipathes grandis	Hawai‘i	45–48	6.42			Time-series of tagged colonies	Grigg (1976)
Antipathes grandis	Hawai‘i	45–48	6.12			Time-series of tagged colonies	Grigg (1976)
Antipathes griggi (as *A. grandis*)	Hawai‘i	Laboratory	1.2			Time-series of branches in aquaria	Grigg (1964)
Antipathes griggi	Hawai‘i	50		0.13–1.14	12–32		Roark *et al*. (2006)
Antipathes griggi			4–11.6			Time-series of tagged	Montgomery (2006)
Antipathes sp. (as *A. dichotoma*)	Palau	24–34	4.52–9.32			Measurements of colonies on artificial structure of known age	Grigg (1975)
Cirrhipathes sp.	Madagascar				60-70	Bomb 14C	Terrana *et al*. 2017
Cirrhipathes cf. *anguina*	Indonesia	20	≤159			Time-series of tagged colonies	Bo *et al*. 2009c
Stichopathes cf. *maldivensis*	Indonesia	20	≤15.6			Time-series of tagged colonies	Bo *et al*. 2009c
Stichopathes lutkeni *Jamaica*	Jamaica	18	76.65			Measurements of colonies on artificial	Warner (2005)
Stichopathes spp. (n 1⁄4 2)	Puerto rico	22–36	3–7			Time-series of tagged colonies	Goenaga (1977) Goenaga (1977)
Stichopathes gracilis	Curacao	>10	46.8–84.76			Time-series of tagged colonies	Noome and Kristensen (1976)
Aphanipathes salix	St.Croix	15–46		0.7		Time-series of tagged colonies	Olsen and Wood (1980)
Leiopathes sp. (as *L.glaberrima*)	Hawai‘i	450		0.005	2377	14C dating	Roark *et al*. (2006) Leiopathes sp.

Leiopathes sp.	Hawai'i	400–500		0.005–0.013	350–4250	14C dating	Roark *et al.* (2009)
Leiopathes sp.	SE United States	304–697		0.008–0.022	520–2100	4C dating and growth rings counts	Prouty *et al.* (2011)
Leiopathes glaberrima	SE United States	307–697		0.014–0.015	198–483	210Pb dating and growth ring counts	Williams *et al.* (2006, 2007)
Antipathella fiordensis (as *Antipathes aperta*)	New Zealand	5–25	3.9			Colony size-frequency distributions	Grange and Singleton (1988)
Antipathella fiordensis	New Zealand		1.3–1.8	0.0984	300	Growth ring counts and time series measurements of tagged colonies	Grange and Goldberg (1993)
Antipathella fiordensis (as *underscribed* sp.)	New Zealand	5–35	2.9			Growth ring counts	Grange (1985)
Plumapathes pennacea	Gulf of Mexico	38	2.92			Measurements of colonies on artificial structure of known age	Boland and Sa Boland and Sammarco (2005)
Plumapathes pennacea	Jamaica	30	5.7	0.92	35	Measurements of colonies on artificial structure of known age	Oakley (1988)
Plumapathes pennacea	St. Croix	15–46		0.81		Time-series measurements of tagged colonies	Olsen and Wood (1980)
Stauropathes arctica	Newfoundland Labrador	812–876	1.22–1.36	0.033–0.066	33–66	Bomb 14C	Sherwood and Edinger (2009
Leiopathes glaberrima	Hawai'i			4–35 mum yr	4,265	Radiocarbon dating	Roark *et al.* 2009
Leiopathes glaberrima	Gulf of Mexico	>30		8–22µm yr	1020–2380		Prouty *et al.* 2001
Leiopathes glaberrima	Sardinia (CW Mediterranean)	200–210			2356 ± 40 Basal Part	Radiocarbon dating	Bo *et al.* 2015

Table C Density values - Antipatharian species (source Wagner et al. 2012, modified)

Species	Family	Location	Depth(m)	Density (col/m^2)	Reference
Antipathes cf *atlantica*	Antipathidae	Providencia Island, Caribbean	30–50	≤1.47	Sanchez *et al*. (1998)
Antipathes atlantica	Antipathidae	Caribbean coast off Colombia	15–50		Sanchez *et al*. (1999)
Antipathes caribbeana	Antipathidae	Providencia Island, Caribbean	30–50	≤2.19	Sanchez *et al*. (1998)
Antipathes caribbeana	Antipathidae	Caribbean coast off Colombia	15–50	0–0.9	Sanchez *et al*. (1999)
Antipathes cf. *dichotoma*	Antipathidae	Palau	6–75	≤0.25	Grigg (1975)
Antipathes gracilis	Antipathidae	Carrabbean coast off Colombia	15–50	0–0.1	Sanchez *et al*. (1999)
Antipathes grandis	Antipathidae	Hawai	40–146	0–1.5	Grigg (1974)
Antipathes grandis, A.griggi	Antipathidae	Hawai	35–110	0–1	Grigg (2001), Grigg (2004)
Antipathes griggi	Antipathidae	Hawai	40–70	0.05 (mean)	Grigg (1976)
Antipathes griggi	Antipathidae	Hawai	58–70	0–0.047	Wagner *et al*. (2011a)
Antipathes pennacea Antipathes carribeana	Antipathidae	Carrabbean coast off Mexico	60	0.005–1.300	Padila and Lara 2003
Stichopathes paucispina	Antipathidae	E.N. Pacific	>1000	<1	Genin *et al.* (1986), Opresko and Genin (1990)
Stichopathes spp.	Antipathidae	Caribbean coast off Colombia	15–50	0.1–3.8	Sanchez *et al*. (1999)
Stichopathes spp	Antipathidae	Providencia Island, Caribbean Providencia Island, Caribbean	30–50	≤7.32	Sanchez *et al*. (1998)
Stichopathes spp (N=2)	Antipathidae	Puerto rico	15–70	≤0.5	Goenaga (1977)
Antipatharian spp. (N=16)	Antipathidae, Myriopathidae	Indonesia	5.45	≤0.5	Tazioli *et al*. (2007)
Stichopathes spiessi	Antipathidae	Eastern North Pacific	550–1150	≤20	Opresko and Genin (1990)
Antipatharian spp. (N=7)	Antipathidae, Myriopathidae	Jamaica	≤35	0.1–2.5	Warner (2005)
Antipathes caribbeana	Antipathidae	Cozumel	20–75	0.5 (mean)	Padilla and Lara (2003)
Aphanipathes salix	Antipathidae	St.Croix	15–46	0.0092–0.0314	Olsen and Wood(1980)
Antipathella subpinnata	Myriopathidae	Mediterraenan	≤100	≤5.2	Bo(2008), Bo *et al*. (2008), Bo *et al*. (2009b)
Leiophates sp.	Leiophatidae	Hawai	375–450	0.002–0.003	Grigg(1988)
Leiopathes glaberrima	Leiophatidae	Mediterranean	210	0.3–2.5	Bo *et al*. 2015

ANNEX 1: Mediterranean regional report on the biology, fishery and trade of precious corals

Compiled by the GFCM Secretariat, April 2018

PREPARATION OF THIS DOCUMENT AND DATA SOURCE

This document was prepared by the Secretariat of the General Fisheries Commission for the Mediterranean (GFCM) of the Food and Agriculture Organization of the United Nations (FAO) as a contribution to the FAO response to CITES Decisions 17.190 to 17.193 on precious corals (Antipatharia Order and Coralliidae family), and provides information on the conservation status of, and trade in red coral, in the Mediterranean Sea including: i) summary of available data and information on the biology, population status, use and trade of red coral in the Mediterranean Sea as well as identification of gaps in such data and information; ii) summary and comparison of existing management measures at national level, also taking into account existing relevant regional guidelines as provided by the GFCM; and iii) proposals on the actions needed to enhance the conservation and sustainable use of precious corals.

The GFCM is a regional fisheries management organization (RFMO) established under the provisions of Article XIV of the FAO Constitution. The main objective of the GFCM is to ensure the conservation and the sustainable use, at the biological, social, economic and environmental level, of living marine resources as well as the sustainable development of aquaculture in the Mediterranean and in the Black Sea. The GFCM is currently composed of 24 Members (23 member countries and the European Union), which contribute to its autonomous budget to finance its functioning, and 3 Cooperating non Contracting Parties.

In order to provide the most updated figures on the status of red coral fishery in the Mediterranean Sea, three main sources of data were used to compile this report: i) the document 'Adaptive management plan for red coral (*Corallium rubrum*) in the GFCM Competence Area', prepared by Angelo Cau, Rita Cannas, Flavio Sacco and Maria Cristina Follesa in 2013; ii) public information emanating from data submitted by GFCM Contracting Parties and Cooperative Non Contracting Parties (CPCs) as a result of requests emanating from relevant GFCM decisions (*Recommendation GFCM/35/2011/2 on the exploitation of red coral in the GFCM Competence Area*; *Recommendation GFCM/36/2012/1 on further measures for the exploitation of red coral in the GFCM area*; *Recommendation GFCM/40/2016/7 concerning the authorization of the use of remotely operated vehicles within the framework of national scientific research programmes on red coral*); and iii) complementary sources of information such as meeting reports, questionnaires or any other information submitted by countries to the GFCM.

CONTENTS

1. SPECIES TARGETED BY THE FISHERY

The term precious corals collectively describes those species of coral (species that belong to the Phylum Cnidaria and have a skeleton made of calcium carbonate or limestone) whose skeletal axis is used as a gemstone to make ornaments and jewelry. Eight species of the Subclass Octocorallia, family Coralliidae, including one undescribed species, are currently used in this way. One of these species – red coral, *Corallium rubrum* – is exploited in Mediterranean waters.

2. BIOLOGY OF RED CORAL

2.1. Taxonomy

The biology of the red coral, *Corallium rubrum*, differs in many aspects from that of other commercially exploited marine organisms. Many of its peculiar features, summarized in the following sections, can have important implications for the effectiveness of management measures addressing its fishery.

The current accepted classification for the Mediterranean red coral is as follows:

Phylum	Cnidaria Hatschek, 1888
Class	Anthozoa Ehrenberg, 1834
Subclass	Octocorallia Haeckel, 1866
Order	Alcyonacea Lamouroux, 1816
Suborder	Scleraxonia Studer, 1887
Family	Coralliidae Lamouroux, 1812
Genus	*Corallium* Cuvier, 1798
Species	*Corallium rubrum* Linnaeus, 1758

Consistent with recent mitochondrial, mitogenomic, and nuclear data (Ardila *et al.*, 2012; Figueroa and Baco 2014; Tu *et al.*, 2015a; Tu *et al.*, 2016; Uda *et al.*, 2013), the traditional classification of Coralliidae in two genera (*Corallium* and *Paracorallium*) has been reconsidered: the genus *Paracorallium* is not considered a valid taxon anymore, given that its species are all nested within *Corallium*, therefore *Paracorallium* has to be subsumed into *Corallium* as its junior synonym (Ardila *et al.*, 2012). The family has been divided into three genera as proposed by Gray (1867): *Corallium, Hemicorallium* and *Pleurocorallium* (Ardila *et al.*, 2012; Figueroa and Baco 2014; Tu *et al.*, 2015a; Tu *et al.*, 2016).

Along with *C. rubrum*, the family Coralliidae comprises 42 valid species reported in Table , with the indication of both the traditional (van Ofwegen 2004) and the new classification (Tu *et al.*, 2016), to facilitate the reader accustomed to the past nomenclature (possibly still used in trade records).

Table 1 – Species included in the Family Coralliidae (in bold are shown the species of commercial interest).

van Ofwegen 2004	Tu *et al.*, 2016
***Corallium abyssale* Bayer, 1956**	*Hemicorallium abyssale* Bayer, 1956
	Hemicorallium aurantiacum Tzu, Dai and Jeng 2016
***Corallium bathyrubrum* Simpson & Watling, 2011**	*Hemicorallium bathyrubrum* Simpson & Watling, 2011
***Corallium bayeri* Simpson & Watling, 2011**	*Hemicorallium bayeri* Simpson & Watling, 2011
	Pleurocorallium bonsaiarborum Tzu, Dai & Jeng 2016
***Corallium borneanse* Bayer**	*Pleurocorallium borneense* Bayer
***Corallium boshuense* Kishinouye, 1903**	*Hemicorallium boshuense* Kishinouye, 1903
***Corallium carusrubrum* Tu, Dai & Jeng, 2012**	*Pleurocorallium carusrubrum* Tu, Dai & Jeng, 2012
***Corallium ducale* Bayer**	*Hemicorallium ducale* Bayer
	Pleurocorallium clavatum Tzu, Dai & Jeng 2016
***Corallium elatius* Ridley, 1882**	***Pleurocorallium elatius* Ridley, 1882**
***Corallium gotoense* Nonaka, Muzik & Iwasaki, 2012**	*Pleurocorallium gotoense* Nonaka, Muzik & Iwasaki, 2012
	Hemicorallium guttatum Tzu, Dai & Jeng 2016
***Corallium halmaheirense* Hickson, 1907**	*Hemicorallium halmaheirense* Hickson, 1907
***Corallium imperiale* Bayer**	*Hemicorallium imperiale* Bayer
***Corallium johnsoni* Gray, 1860**	*Pleurocorallium johnsoni* Gray, 1860
***Corallium kishinouyei* Bayer, 1996**	*Pleurocorallium porcellanum* Pasternak, 1981
***Corallium konojoi* Kishinouye, 1903**	***Pleurocorallium konojoi* Kishinouye, 1903**
***Corallium laauense* Bayer, 1956**	***Hemicorallium laauense* Bayer, 1956**
***Corallium maderense* (Johnson, 1899)**	*Hemicorallium maderense* (Johnson, 1899)
***Corallium medea* Bayer, 1964**	*Corallium medea* Bayer, 1964
***Corallium niobe* Bayer, 1964**	*Hemicorallium niobe* Bayer, 1964
***Corallium niveum* Bayer, 1956**	*Pleurocorallium niveum* Bayer, 1956
***Corallium occultum* Tzu-Hsuan Tu, Alvaro Altuna & Ming-Shiou Jeng, 2015**	*Pleurocorallium occultum* Tzu-Hsuan Tu, Alvaro Altuna & Ming-Shiou Jeng, 2015
	Pleurocorallium norfolkicum Tzu, Dai & Jeng 2016
***Corallium porcellanum* Pasternak, 1981**	*Pleurocorallium porcellanum* Pasternak, 1981
***Corallium pusillum* Kishinouye, 1903**	*Pleurocorallium pusillum* Kishinouye, 1903
***Corallium regale* Bayer, 1956**	***Hemicorallium regale* Bayer, 1956**
***Corallium reginae* Hickson, 1907**	*Hemicorallium reginae* Hickson, 1907
***Corallium rubrum* (Linnaeus, 1758)**	***Corallium rubrum* (Linnaeus, 1758)**
***Corallium secundum* Dana, 1846**	***Pleurocorallium secundum* Dana, 1846**
***Corallium sulcatum* Kishinouye, 1903**	***Hemicorallium sulcatum* Kishinouye, 1903**
***Corallium taiwanicum* Tu, Dai & Jeng, 2012**	*Hemicorallium taiwanicum* Tu, Dai & Jeng, 2012
***Corallium tricolor* (Johnson, 1899)**	*Hemicorallium tricolor* (Johnson, 1899)
***Corallium uchidai* Nonaka, Muzik & Iwasaki, 2012**	*Pleurocorallium uchidai* Nonaka, Muzik & Iwasaki, 2012
***Corallium vanderbilti* Boone, 1933**	Not included
***Corallium variabile* (Thomson & Henderson, 1906)**	*Hemicorallium variabile* (Thomson & Henderson, 1906)
***Paracorallium inutile* (Kishinouye, 1903)**	***Pleurocorallium inutile* (Kishinouye, 1903)**
***Paracorallium japonicum* (Kishinouyi, 1903)**	***Corallium japonicum* (Kishinouyi, 1903)**
***Paracorallium nix* (Bayer, 1996)**	*Corallium nix* (Bayer, 1996)
***Paracorallium salomonense* (Thomson & Mackinnon, 1910)**	*Corallium salomonense* (Thomson & Mackinnon, 1910)
***Paracorallium stylasteroides* (Ridley, 1882)**	*Corallium stylasteroides* (Ridley, 1882)
***Paracorallium thrinax* (Bayer & Stefani in Bayer, 1996)**	*Pleurocorallium thrinax* (Bayer & Stefani in Bayer, 1996)
***Paracorallium tortuosum* (Bayer, 1956)**	*Corallium tortuosum* (Bayer, 1956)

Twelve species have only been described and formally named in recent years (Nonaka *et al.*, 2012; Simpson and Watling 2011; Tu *et al.*, 2016; Tu *et al.*, 2015b; Tu *et al.*, 2012). However, several undescribed species are still reported to exist within the Coralliidae.

2.2. Distribution of red coral

Corallium rubrum is a sciaphilous endemic species to the Mediterranean and the neighboring Atlantic coasts that occurs primarily around the central and western basin (5–350 m, although more commonly at 30–200 m) with smaller populations in deeper water in the eastern basin (60–200 m) and off the coast of Africa around the Canary Islands, southern Portugal and around the Cape Verde Island (Chintiroglou *et al.*, 1989; Marchetti, 1965; Weinberg, 1976; Zibrowius *et al.*, 1984; Boavida *et al.*, 2016; Cattaneo-Vietti *et al.*, 2016; Ghanem *et al.*, 2018) (Figure 1). Nonetheless, a very recent paper by Çinar *et al.* (2018) suggested that the distributional range of *C. rubrum* within the Mediterranean Sea could reach Anamur, Levantine Sea (Turkey), as the eastern point of its distribution.

Figure 1 – Map of red coral distribution in the Mediterranean Sea, FAO (2018)

The bathymetric limits of red coral distribution have been greatly expanded: from 350 to 800–1000 m depth in the Central Mediterranean (Strait of Sicily and Malta waters) (Costantini *et al.*, 2010; Freiwald *et al.*, 2009; Taviani *et al.*, 2010; Knittweis *et al.*, 2016). In the Mediterranean, *C. rubrum* inhabits subtidal rocky substrates and is one of the dominant and important components of Mediterranean "coralligenous" species assemblages (Ballesteros, 2006), coexisting with other gorgonians, large sponges, and other benthic invertebrates. The coralligenous biocoenosis, that hosts most of red coral populations, is classified as a 'Priority habitat for conservation', that is a habitat whose conservation is required because of its vulnerability, heritage value, rarity, aesthetic and economic values (Relini and Giaccone, 2009). More in general the coralligenous outcrops represent one of the hotspot for species diversity, and are considered a benthic habitat of high conservation interest (Ballesteros, 2006; UNEP/MAP, 2009), and also of special concern, as they are increasingly suffering impacts of a range of anthropogenic disturbances. In particular, the gorgonian corals play a very important role within the coralligenous biocoenosis as ecosystem engineers by providing structural complexity and biodiversity. The precious red coral is found also outside the coralligenous, in the semi-dark caves (and upper enclaves) with the facies with *C. rubrum*.

Several studies (see the following sections) concordantly suggest the occurrence of different typologies of *C. rubrum* populations:

- **shallow-water populations** (depth range between 15 and about 50 m), dwelling on caves, crevices, overhangs and protected interstices. Many of them are reported to be overexploited, in consequence of the pressure exerted by SCUBA diving since the 1950s, able to be picking of red coral in areas inaccessible to dredges. According to the Recommendation GFCM/35/2011/2 Paragraph 4, shallow-water populations are now fully protected from exploitation, to allow for their recovery.
- **deep-water populations** (depth range of between 50 and about 130 m), typically dwelling on open surfaces. The commercial harvesting is now focused on these populations. In the past, they were harvested by the use of dredging gears (now forbidden), while nowadays by divers down up to depths of 100–130 meters.
- **deepest-water populations** (depth range 130 to about 1000 m); poorly known. No legal harvesting is realized on those populations, but considering they are very sparse (very low densities) they are said of not commercial interest (not permitting any profitable exploitation).

2.3. Threats and environmental issues

The red coral is considered one of the most vulnerable resources in the Mediterranean Sea because it is a long-lived species, with a slow growth, low fecundity and limited dispersal capabilities and consequently strong genetic differentiation of neighboring populations at spatial scales of 10s of meters.

Because of its high economic value, the red coral has been exploited since ancient times. Fishing impacts are worsened by natural stressors and climate change, especially in shallow water where also mass mortality events have been documented since the late 1990s, attributed to a fungal and protozoan disease and linked to temperature anomalies (Bramanti *et al.*, 2005; Cerrano *et al.*, 1999; Garrabou *et al.*, 2009; Garrabou *et al.*, 2001; Perez *et al.*, 2000; Romano *et al.*, 2000). In particular, the Mediterranean red coral is expected to be particularly susceptible to acidification effects linked to climate change, due to the elevated solubility of its Mg-calcite skeleton. Experimental studies have shown, for the first time, evidence of detrimental ocean acidification effects on this valuable and endangered coral species (Bramanti *et al.*, 2013; Cerrano *et al.*, 2013).

Besides harvesting, other types of anthropogenic disturbances threat red coral populations: pollution, tourism, recreational diving, incidental takes, or habitat degradation (Boavida *et al.*, 2016; Cau *et al.*, 2015a; Cau *et al.*, 2017; Coma *et al.*, 2004; Garrabou *et al.*, 1998; Lastras *et al.*, 2016; Linares *et al.*, 2003; Linares *et al.*, 2010b).

Currently, the most destructive impact affecting coralligenous communities is the action of trawling gear (Cinelli and Tunesi, 2009). In the past, special trawling used to collect the precious red coral by the so-called "Italian Bar" or "Saint Andrew Cross" proved to be highly destructive, causing degradation of large areas of coralligenous (RAC/SPA, 2008). Negative impacts on the coralligenous can be also caused by traditional artisanal fishing (long-lines, trammel nets, pots for catching lobsters), recreational fishing, diving activities and anchoring (RAC/SPA, 2008). At present, fishing on coralligenous formations is prohibited, although there are also reports of illegal fishing (Cattaneo-Vietti *et al.*, 2017).

As most of the builder species are long-lived, have low recruitment and complex demographic patterns, destruction of the coralligenous structure is critical as their recovery will probably take several decades or even centuries (Ballesteros, 2006; RAC/SPA 2008). For instance, after 20 to 30 years of protection within French and Spanish MPAs, red coral colony sizes did not reach the values of pristine populations (Garrabou and Harmelin 2002; Linares *et al.*, 2010a; Tsounis *et al.*, 2006b) suggesting that full recovery will require decades of effective protection (Linares *et al.*, 2010a; Torrents and Garrabou 2011; Tsounis *et al.*, 2006b). Given the longevity of red coral (more than 100 years) (Garrabou and Harmelin 2002; Marschal *et al.*, 2004; Tsounis *et al.*, 2007), it seems reasonable to speculate that a few decades of protection may not be sufficient to reach the size of pristine populations.

Recently, harvesting effects and the recovery processes of *C. rubrum* have been investigated using 5–7 year photographic series on two populations located on the coast of Marseilles (France, north-western Mediterranean) (Montero-Serra *et al.*, 2015). At the end of the study only a partial recovery was observed. The general pattern of low recruitment and high mortality of new recruits demonstrated limited effects of reproduction on population recovery while low mortality of partially harvested adults and a large proportion of colonies showing new branches highlighted the importance of re-growth in the recovery process (Montero-Serra *et al.*, 2015).

A multi-annual data set (comparing population structure in 1964, 1990 and 2012) was collected from the most important shallow-water coral populations in the Ligurian Sea (Bavestrello *et al.*, 2015). The population is at present within the marine protected area (MPA) of Portofino but it was actively harvested up to the 70s. The data indicated a strong size increase of the colonies with time, in colony biomass, and a slight decrease in density (switch of red coral populations from a 'grass plain-like' towards a ‘forest-like’ structure). These results indicate that the creation of MPAs is a winning strategy in the conservation of this precious species (Bavestrello *et al.*, 2015).

A recent transplantation experiment for *C. rubrum* was highly successful over a relatively short term due to high survival and reproductive potential of the transplanted colonies. However, demographic projections predict that from 30 to 40 years may be required for fully functional *C. rubrum* populations to develop (Montero-Serra *et al.*, 2017).

2.4. Reproduction

Many aspects of the reproductive biology of red coral have been studied but there are still many knowledge gaps and uncertainties that prevent scientists to understand at full all the factors that drive the reproduction of the species. To date, except for the historical work of Lacaze-Duthiers (Lacaze-Duthiers, 1864), few reports on red coral reproduction are published for shallow water colonies (Santangelo *et al.*, 2003; Torrents *et al.*, 2005; Tsounis 2005; Tsounis *et al.*, 2006a,b; Vighi, 1970, 1972; Weinberg, 1979). Since 2013, new knowledge on the fecundity of deep-sea colonies is available for Italy and Spain (Porcu *et al.,* 2017; Priori *et al.*, 2013).

Recently, a genetic study demonstrated the occurrence of an XX/XY genetic sex-determination (GSD system) in *C. rubrum*; this is the first record of GSD for a non-bilaterian species (Pratlong *et al.*, 2017).

C. rubrum presents a limited capability for asexual reproduction. The sexual status of the population appears completely gonochoric at both the colony and the polyp level (Porcu *et al.*, 2017; Santangelo *et al.*, 2003; Tsounis *et al.*, 2006; Vighi 1970, 1972). Red coral is an iteroparous species, undergoes internal fertilization, and broods larvae internally (planulator). Gonadal development follows an annual cycle with a synchronized release in summer (Porcu *et al.*, 2017; Santangelo *et al.*, 2003; Tsounis *et al.*, 2006a; Vighi 1970). The embryonic period lasts about 30 days (Lacaze-Duthiers 1864; Vighi 1970). Colonies do not fuse together (in nature) and each adult colony therefore is likely to have originated from a singular planula (Stiller *et al.*, 1984; Weinberg, 1979). Larvae exist in the water column for a few hours to days before setting near parent’s colonies. It is therefore very likely that single populations are genetically isolated.

Martínez-Quintana *et al.*, (2015) studied the pelagic larval duration (PLD), buoyancy and larval vertical motility behaviour. PLD ranged from 16 days (95 percent survival) to 42 days (5 percent survival). Larvae exhibited negative buoyancy with a free fall speed decreasing linearly with age, at a velocity varying from -0.09 ± 0.026 cm s^{-1} on day 1 to -0.05 ± 0.026 cm s^{-1} on day 10. *C. rubrum* larvae maintained active swimming behaviour for 82 percent of the time. This larval motility behaviour, combined with the extended PLD, confers on *C. rubrum* larvae an unsuspectedly high dispersive potential in open waters.

Data on fecundity is summarized in Table 2. Fertile female polyp of shallow water colonies produced from 1 to 4 mature oocytes per year, while each fertile male polyp produced 6±3.5 spermiaries on average (Santangelo *et al.*, 2003). A range from 1 to 11 sexual products per year is registered in Sardinian seas (Porcu *et al.*, 2017). Female fecundity and fertility are constant over time, but decrease after March, despite the fact that oocytes do not turn into planulae during this period (Santangelo *et al.*, 2003). Female’s fecundity ranged between 0.05–3 mature oocytes per year has been found for the deep-water colonies of Tuscany Archipelago (Priori *et al.*, 2013), while average fecundity values of 0.87 and 2.09±0.65 of female sexual products per polyp have been recorded for deep colonies caught in Spain and Sardinia, respectively (Porcu *et al.*, 2017; Santangelo *et al.*, 2015).

Table 2 - Colonies fecundity in several Mediterranean shallow- and deep-water populations; fecundity is expressed as number of oocytes/ polyp /year

Colonies	Fecundity/sex	Depth / substrate	Population examined	Reference
Shallow	3–6/female	-	Italy (Liguria)	(Vighi 1970; Vighi 1972)
	0.87/female	-	Italy (Calafuria)	(Santangelo *et al.,* 2003)
	0.9–1.6/female	-	Spain (Medes islands)	(Tsounis *et al.,* 2006a; Tsounis *et al.,* 2006b)
	1.0–3.2 / female	-	France (Provence)	(Torrents and Garrabou 2011)
	2.1±1.5 / female	39–42 m	France (Provence)	(Torrents and Garrabou 2011)
	2.7±1.6 / female	15–22 m	France (Provence)	(Torrents and Garrabou 2011)
	2.7±1.6/ female	Cave entrance	France (Provence)	(Torrents and Garrabou 2011)
	1–12/male	-	Italy (Liguria)	(Vighi 1970; Vighi 1972)
	6/male	-	Italy (Calafuria)	(Santangelo *et al.,* 2003)
	1.9–3.9/male	-	Spain (Medes islands)	(Tsounis *et al.,* 2006a; Tsounis *et al.,* 2006b)
	5.0±1.7/ male	39–42 m	France (Provence)	(Torrents and Garrabou 2011)
	5.7±3.0/ male	15–22 m	France (Provence)	(Torrents and Garrabou 2011)
	5.7±3.0 / male	Cave entrance	France (Provence)	(Torrents and Garrabou 2011)
	0.64±0.33/female	Marine reserve at 30–35 m	Italy (Portofino)	(Bramanti *et al.,* 2014)
	0.42±0.30/female	Marine reserve at 30–35 m	Spain (Cape de Creus)	(Bramanti *et al.,* 2014)
	0.93/female	20–50 m	Italy (Livorno and Genoa)	(Santangelo *et al.,* 2015)
	1.14±0.61/female	38–40 m	Italy (Sardinian sea)	(Porcu *et al.,* 2017)
	1.34±0.86/male	38–40 m	Italy (Sardinian sea)	(Porcu *et al.,* 2017)
Deep	0.05–3/female	50–130 m	Italy (Tyrrhenian sea)	(Priori *et al.,* 2013)
	0.87/female	60–120 m	Spain (Cap De Creus)	(Santangelo *et al.,* 2015)
	2.09±0.65/female	96–115 m	Italy (Sardinian sea)	(Porcu *et al.,* 2017)
	3.38±2.14/male	96–115 m	Italy (Sardinian sea)	(Porcu *et al.,* 2017)

The size has positive effect on reproductive potential. The reproductive output increases exponentially with colony size. Generally, large colony contained more oocytes and sperm sacs and may produce a hundred or more of planulae than the small ones (some tens of planulae) (Santangelo *et al.*, 2003; Torrents *et al.*, 2005; Tsounis *et al.*, 2006a). The actual age of first reproduction is probably 7–10 years corresponding to colonies about 24.0 mm in eight, 3.6 mm in basal diameter and 0.6 g in wet weight (Torrents *et al.*, 2005). Female colonies reach fertility at a minimum age of approximately 10 years (corresponding to a diameter of 2 mm), although reproductive parameters (percentage of fertile colonies and fertility) are significantly lower in small size than in medium and large colonies (Torrents *et al.*, 2005). Male colonies, on the other hand, may develop gametes much earlier. The youngest fertile male colony of the sample with a basal diameter of 1.2 mm is supposed to be not older than six years (Gallmetzer *et al.*, 2010).

Priori *et al.*, (2015) quantified the effects of the partial predation exerted by the gastropod *Pseudosimnia carnea* on the reproductive features of *Corallium rubrum*: colony fecundity was reduced by 81 percent with a consequent reduction in population reproductive output and thereby limited resilience to intense commercial harvesting.

2.5. Recruitment

Recruitment is one of the main process determining both population structure and dynamics (Caley *et al.*, 1996). The rate of recruitment can greatly vary in space and time. It could be different if it occurs in natural or artificial substrates. There are few studies on the recruitment process in red coral populations (Table 3).

Table 3 - Recruitment rate measured in different time and areas of the Mediterranean

Colonies	Recruitment rate	Substrate	Population examined	Reference
Shallow	0–32 recruits/m^2	Shallow waters (40 m depth)	Spain (Medes Islands)	(Linares *et al.,* 2000)
	1.6±1.96 recruits/dm^2	Semi natural substrate on a vertical cliff	Spain (Medes Island)	(Bramanti *et al.,* 2007)
	0.4–0.6 recruits/dm^2	Semi natural substrate on a vertical cliff	France (Monaco)	(Cerrano *et al.,* 1999)
	0–12 recruits/m^2	Semi natural substrate on lateral wall	France (Marseille)	(Garrabou and Harmelin 2002)
	1.3 recruits/dm^2	Semi natural substrate in a cave	France (Marseille)	(Garrabou and Harmelin 2002)
	0.178 recruits/dm^2	Semi natural substrate in a cave	France (Marseille)	(Garrabou and Harmelin 2002)
	6.24±4.26 recruits/dm^2	Semi natural substrate on a vertical cliff	Italy (Calafuria)	(Bramanti *et al.,* 2005)
	1.1±1.4 recruits/dm^2	Semi natural substrate on a vertical cliff	Italy (Elba Island)	(Bramanti *et al.,* 2007)
	3.88±0.68 recruits/dm^2	Semi natural substrate on marble tiles	mean (Medes I., Elba, Calafuria)	(Santangelo *et al.,* 2012)
	0.56±0.21 recruits/dm^2	Semi natural substrate on marble tiles	Spain (Medes Island)	(Santangelo *et al.,* 2012)
	6.06±1.75 recruits/dm^2	Semi natural substrate on marble tiles	Italy (Calafuria)	(Santangelo *et al.,* 2012)
	4.66±1.01 recruits/dm^2	Semi natural substrate on marble tiles	Italy (Elba Island)	(Santangelo *et al.,* 2012)
	17.5±4.7 recruits/400 cm^2	Shallow waters (30–35 m depth) of MPA	Italy (Portofino)	(Bramanti *et al.,* 2014)
	5.6±2.8 recruits/400 cm^2	Shallow waters (30–35 m depth) of MPA	Spain (Cape de Creus)	(Bramanti *et al.,* 2014)
	0.56–6x10^2 m^2	Shallow waters (20–50 m depth)	Italy (NW Mediterranean	(Santangelo *et al.,* 2015)
	0.98±1.40/400 cm^2	Shallow waters (15–22 m) vertical wall	France (Riou island)	(Montero-Serra *et al.,* 2015)
	0.21±0.53/400 cm^2	Shallow waters (15–22 m) cave like tunnel	France (Maire island	(Montero-Serra *et al.,* 2015)

Saturation of space and interference competition appears to be a major controlling factor in red coral recruitment (Garrabou *et al.*, 2001; Santangelo *et al.*, 1988). This could explain why recruitment rates were higher after harvesting events (Linares *et al.*, 2000; Santangelo *et al.*, 1997).

2.6. Growth rate

Corallium rubrum is long-lived but how "long" its life span can be, it is still controversial and object of research. Some researches talk about >100 years (Garrabou *et al.*, 2002; Roark *et al.*, 2006) but this has to be confirmed. Up to now three different methods have been applied to determine the age of coral colonies:

- Petrographic method (Abbiati *et al.*, 1992; Garcia-Rodriguez and Massò, 1986a; Santangelo *et al.*, 1993) ;
- Organic matrix staining of thin sections from the base of colonies (Marshal *et al.*, 2004), which allows one to read annual growth rings;
- Direct measurements of new settled colonies of known age, a not destructive approach to the study of colony growth rate, based on artificial or semi-natural substrates on which new settled

colonies can be followed during their growth for a determined time interval (Bramanti *et al.*, 2005; Cerrano *et al.*, 1999; Garrabou and Harmelin, 2002).

Unfortunately, the dark bands highlighted with the first method (Petrographic method) were not annual, so colony age was underestimated. Only "organic matrix staining method" allowed reading growth rings that were checked to be annual by calcein labelling in vivo (Caley *et al.*, 1996). In general, colonies exhibit low growth rate that varies among location, depths, and habitats (Abbiati *et al.*, 1992; Bramanti *et al.*, 2005; Cerrano *et al.*, 1999; Garcia-Rodriguez and Massò 1986a; Garrabou and Harmelin, 2002) (Table 4).

Table 4 - Growth rates (mm yr-1) in Mediterranean shallow- and deep-water populations of C. rubrum

Colonies	Growth rate (mm yr-1)	Method used*	Population examined	Reference
Shallow Mediterranean	1.32	P	Spain (Gerona)	(Garcia-Rodriguez and Massò 1986)
	0.24±0.06	OMS	Spain (Cape de Creus)	(Vielmini *et al.,* 2010)
	0.35±0.15	OMS	France (Marseille)	(Marschal *et al.,* 2004)
	0.24±0.05	D (semi natural substrate in a cave)	France (Marseillle)	(Garrabou and Harmelin 2002)
	0.91	P	Italy (Calafuria, Livorno)	(Abbiati *et al.,* 1992)
	0.62±0.19	D (semi natural substrate in vertical cliff)	Italy (Calafuria, Livorno)	(Bramanti *et al.,* 2005)
	0.62	D (natural substrate in a cave)	Italy (Portofino)	(Cerrano *et al.,* 1999)
	0.22±0.04	OMS	Italy (Portofino)	(Vielmini *et al.,* 2010)
	0.2	OMS	Italy (Ligurian Sea)	(Gallmetzer *et al.,* 2010)
	0.68±0.02**	D (marble tiles)	Italy (Calafuria, Livorno)	(Santangelo *et al.,* 2012)
	0.59±0.02**	D (marble tiles)	Italy (Elba, Tuscany)	(Santangelo *et al.,* 2012)
	0.241±0.061	OMS	Italy (Portofino)	(Bramanti *et al.,* 2014)
	0.237±0.062	OMS	Spain (Cape de Creus)	(Bramanti *et al.,* 2014)
	0.236±0.027	OMS	Spain (Riou Island)	(Chaabane *et al.,* 2016)
	0.172±0.007	OMS	Spain (Medes island)	(Chaabane *et al.,* 2016)
	0.276±0.065	OMS	Italy (Portofino)	(Chaabane *et al.,* 2016)
Deep Mediterranean	0.26	OMS	Italy (Tyrrhenian Sea)	(Priori *et al.,* 2013)
	0.24±0.08	OMS	Italy (North-Central Tyrrhenian Sea)	(Benedetti *et al.,* 2016)
	0.26±0.07	OMS	Italy (North Tyrrhenian Sea)	(Benedetti *et al.,* 2016)
	0.21±0.08	OMS	Italy (Central Tyrrhenian Sea)	(Benedetti *et al.,* 2016)
Deep Atlantic	0.23±0.06	OMS	Atlantic (SW Portugal)	(Boavida *et al.,* 2016)

Galli *et al.*, (2016) applied a mechanistic numerical model to describe the growth of a *C. rubrum* colony (polyps number, polyp and gametes biomass, skeletal inorganic and organic matter) as a function of food availability and seawater temperature. They found that large colonies are more likely to undergo negative growth in shallow waters, where they experience both high temperatures and food shortage at the same time during late summer, and are thus less likely to be found near the upper distribution limits of *C. rubrum*. In deeper waters, larger colonies may have less constraints due to both quite stable temperature and food input. If this temperature effect on size distribution is superimposed on the harvesting effect, the occurrence of large colonies could be further limited to deeper waters as mean sea temperatures increase. Increasing trend of harmful heat waves events would also increase this risk (Galli *et al.*, 2016).

Boavida *et al.*, (2016) report a very slow growth rate (0.23 mm/year) in Atlantic red corals dwelling at 60–100 m depth in southern Portugal, when compared to Mediterranean specimens.

After settlement, the growth rate of shallow water colonies is about 1 mm/year for the base diameter, and 10 mm year-1 for the height (Cattaneo-Vietti and Bavestrello, 1994). However, after 4–5 years, the growth virtually stops and become negligible (Bavestrello *et al.*, 2010). The growth rate of deep-water colonies seems to be narrower (from 0.21 to 0.26 mm) (Benedetti *et al.*, 2016; Priori *et al.*, 2013). Similar values of growth rate have been registered for Atlantic deeper colonies (Boavida *et al.*, 2016).

2.7. Population structure and density

Red coral population density varies from place to place, according to depth and exploitation (Rossi *et al.*, 2008; Tsounis *et al.*, 2006b; Angiolillo *et al.*, 2016; Cau *et al.*, 2016). Schematically, we can distinguish two different spatial situations: 1) coastal populations, occurring up to 50 m depth, characterized by high density (up to 1000 colony/m^2) and small colony size (until 5 cm of height) (Table 5, Table 6); 2) deeper populations, extending up to 200 m depth and more, are characterized by low density and high colony size (Tables 6, 7, 8) probably because of lower intra-specific competition in deep areas (Angiolillo *et al.*, 2016). At this depth range, colonies are characterized by more extensive branching patterns (Santangelo *et al.*, 2007; Angiolillo *et al.*, 2016) forming small aggregates on individual banks and hard ground areas, where colonies are concentrated on the exposed surface facing into high-current areas (Cannas *et al.*, 2010; Rossi *et al.*, 2008; Angiolillo *et al.*, 2016; Cau *et al.*, 2016).

Table 5 - Adult colonies densities (colonies/m^2) at different depths in several Mediterranean areas

Populations	Country	Density (Colonies/m^2)	Depth (m)	Reference
Shallow	France (Scandola)	70	19–22	Linares *et al.*, 2010a
	France (Banylus)	137	23–25	Linares *et al.*, 2010a
	France (Carry)	47	24–25	Linares *et al.*, 2010a
	France (NW Mediterranean)	228–606	25	Garrabou *et al.*, 2001
	Spain (Costa Brava)	3.42±4.39	20–50	Tsounis *et al.*, 2006b
	Spain (Palma de Mallorca)	55	40	GFCM, 1984
	Spain (Costa Brava)	20	60	GFCM, 1984
	Italy (Calafuria)	4322±3358	20–45	Santangelo *et al.*, 1993
	Italy (Ligurian Sea)	1050 ± 7.39; 212.5±10.38	22–40	Santangelo *et al.*, 1988
	Italy (Eastern Ligurian Sea)	2493±1299.5	30–50	Santangelo and Abbiati 1989
	Spain (Cap de Creus)	43±53	45–85	Rossi *et al.*, 2008
Deep	Italy (Calabrian coast, South)	18.04±23.6	50–105	Angiolillo *et al.*, 2009
	Italy (Calabrian coast, South)	6.42±4.6	70–130	Angiolillo *et al.*, 2009
	Italy (Calabrian coast, South)	96.57±7.5	50–200	Angiolillo *et al.*, 2009
	Italy (Tyrrhenian Sea)	12.9±7.9	50–130	Priori *et al.*, 2013
	Italy (Amalfi)	28–128	50–130	Angiolillo *et al.*, 2016
	Italy (Ischia)	20–204	0–120	Angiolillo *et al.*, 2016
	Italy (Elba)	46–132	60–95	Angiolillo *et al.*, 2016
	Italy (Sardinia)	8–300	80–170	Cau *et al.*, 2016

In deep colonies, Cau *et al.*, (2016) recently found an inverse relationship between maximum population density and mean colony height, suggesting that self-thinning processes may shape population structure. They also found demographically young populations composed of small and dense colonies dominated along rocky vertical walls, whereas mature populations characterized by large and sparsely distributed colonies were found only in horizontal beds not covered by sediment.

Nevertheless, ROV surveys showed that populations dwelling between 80 and 170 m depth can exhibit a continuous range of population density (from 2 to 75 colonies per 0.25 m^2) (Cau *et al.*, 2016; Cau *et al.*, 2015a; Cau *et al.*, 2017; Cau *et al.*, 2015b). Young populations composed of small and dense colonies dominated along rocky vertical walls, whereas mature populations characterized by large and sparsely distributed colonies were found only in horizontal beds not covered by sediment. An inverse relationship between maximum population density and mean colony height was found, suggesting that self-thinning processes may shape population structure. It has been hypothesized that, in the long term, shallow protected populations should resemble to present deep populations, with sparsely distributed large colonies (Cau *et al.*, 2016).

The recent discovery of an exceptional red coral population from a previously unexplored shallow underwater cave in Corsica (France), characterized by the coexistence of a relatively high abundance of small (<3 cm) colonies with large colonies (>10 cm in height) older than 100 years, challenge current assumptions on red coral populations. Possibly before intense exploitation, red coral lived in relatively high-density populations with a large proportion of centuries-old colonies, even at very shallow depths (Garrabou *et al.*, 2017).

Table 6 – Red coral densities in shallow waters in several Mediterranean areas.

Country	Density (colonies/m^2)	Depth (m)	Reference
Italy (Portofino)	93±22.6	25–40	(Marchetti 1965)
Spain (Palma de Mallorca)	55	40	(GFCM 1984)
Italy (Calafuria)	2000	31–36	(Santangelo *et al.*, 2007)
Spain (Costa Brava)	20	60	(GFCM 1984)
Italy (Portofino)	379.7±85.7	25–40	(Cattaneo-Vietti *et al.*, 1993)
Italy (Calafuria)	4322±3358	20–45	(Santangelo *et al.*, 1993)
Italy (Ligurian Sea)	1050 ± 7.39; 212.5±10.38	22–40	(Santangelo *et al.*, 1988)
Italy (Eastern Ligurian Sea)	2493±1299.5	30–50	(Santangelo and Abbiati 1989)
France (NW Mediterranean)	228–606	25	(Garrabou *et al.*, 2001)
France (Marseille)	400–600$^{\#}$	10–50	(Garrabou and Harmelin 2002)
Spain (Costa Brava)	3.42±4.39	20–50	(Tsounis *et al.*, 2006b)
Croatia (Adriatic sea)	5.8–18.6$^{\#}$	30–60	(Kružic and Popijac 2009)
France (Scandola)	70	19–22	(Linares *et al.*, 2010a)
France (Banyuls)	137	23–25	(Linares *et al.*, 2010a)
France (Carry)	47	24–25	(Linares *et al.*, 2010a)
Italy (Portofino)	39.7±17.8*	30–35	(Bramanti *et al.*, 2014)
Spain (Cap de Creus)	11.9±6.9*	30–35	(Bramanti *et al.*, 2014)
France (Maire island)	0.98±1.40*	15–22	(Montero-Serra *et al.*, 2015)
France (Riou island)	0.21±0.53*	15–22	(Montero-Serra *et al.*, 2015)
Italy (Portofino MPA)	227.0±37.3	25–40	(Bavestrello *et al.*, 2015)
Italy (NW Mediterranean)	0.2–3x10^3	20–50	(Santangelo *et al.*, 2015)
France (Scandola, cave b)	201	18–27	(Garrabou *et al.*, 2017)

*colonies 400 cm^2; $^+$colonies 0.25 m^2; $^{\#}$range

Table 7 – Red coral densities in deep waters in several Mediterranean areas.

Country	Density (colonies/m^2)	Depth (m)	Reference
Spain (Cap de Creus)	43±53	45–85	(Rossi *et al.*, 2008)
Italy (Calabrian coast, South)	18.04±23.6	50–105	(Angiolillo *et al.*, 2009)
Italy (Calabrian coast, South)	6.42±4.6	70–130	(Angiolillo *et al.*, 2009)
Italy (Calabrian coast, South)	96.57±7.5	50–200	(Angiolillo *et al.*, 2009)
Italy (Tyrrhenian Sea)	12.9±7.9	50–130	(Priori *et al.*, 2013)
Italy (Scogli Forio d'Ischia)	13.8±2.5	70–80	(Bavestrello *et al.*, 2014b)
Italy (Punta Imperatore)	20.8±4.9	89–130	(Bavestrello *et al.*, 2014b)
Italy (Punta Angelo 1)	146.4±46.6	58–120	(Bavestrello *et al.*, 2014b)
Italy (Punta Angelo 2)	24.0±3.7	87–96	(Bavestrello *et al.*, 2014b)
Italy (Punta Solchiaro)	96.6±8.5	50–60	(Bavestrello *et al.*, 2014b)
Italy (Punta Pizzato)	33.0±5.8	50–60	(Bavestrello *et al.*, 2014b)
Italy (Secca del Pampano)	4.1±0.6	140	(Bavestrello *et al.*, 2014b)
Italy (Secca di Zio Antonino)	4.9±1.8	120	(Bavestrello *et al.*, 2014b)
Italy (Secca de La Montagna)	2.7±0.4	130	(Bavestrello *et al.*, 2014b)
Italy (Scogli della Bocca piccola)	3.4±0.3	110–115	(Bavestrello *et al.*, 2014b)
Italy (Li Galli Is.W)	19.4±2.1	90–110	(Bavestrello *et al.*, 2014b)
Italy (-Li Galli Is.E)	5.1±0.7	78–85	(Bavestrello *et al.*, 2014b)
Italy (Torre di Grado)	13.0±2.5	55–70	(Bavestrello *et al.*, 2014b)
Italy (Grotta del Diavolo)	17.3±3.3	90–120	(Bavestrello *et al.*, 2014b)
Italy (Scoglio D'Isca)	18.5±3.8	65–84	(Bavestrello *et al.*, 2014b)
Italy (Capo di Conca)	15.5±4.6	50–65	(Bavestrello *et al.*, 2014b)
Italy (Ligurian sea, Punta del Faro)	41±12	50–100	(Bavestrello *et al.*, 2014a)
Italy (Ligurian sea, Isuela Shoal)	97±34	40–60	(Bavestrello *et al.*, 2014a)
Italy (Ligurian sea, Maledetti Shoal)	278±12	60–90	(Bavestrello *et al.*, 2014a)
Italy (Ligurian sea, Canyon Lua)	<0.1	90–140	(Bavestrello *et al.*, 2014a)
Italy (Ligurian sea, Taggia Arma)	0.03	90–160	(Bavestrello *et al.*, 2014a)
Italy (Ligurian sea, Bordighera)	35±16	60–80	(Bavestrello *et al.*, 2014a)
Italy (Tuscany, North Pianosa is.)	48±23	70–80	(Bavestrello *et al.*, 2014a)
Italy (Tuscany, Sante Shoal)	21±16	75–95	(Bavestrello *et al.*, 2014a)
Italy (Tuscany, Montecristo Shoal)	72±47	60–100	(Bavestrello *et al.*, 2014a)
Italy (Tuscany, Tuna Paradise)	41±30	80–100	(Bavestrello *et al.*, 2014a)
Spain (NW Mediterranean)	0.28–2x10^3	60–120	(Santangelo *et al.*, 2015)
Tyrrhenian Sea, Amalfi	35.5±30.1	50–130	(Angiolillo *et al.*, 2016)
Phlegrean Islands, Ischia	64.5 ±44.1	50–122	(Angiolillo *et al.*, 2016)
Tuscany Archipelago, Elba	50.5±30.6	60–94	(Angiolillo *et al.*, 2016)
Italy (Sicily, Ragusa bank, 09)	2.7±1.3	88–112	(Cattaneo-Vietti *et al.*, 2017)
Italy (Sicily, Ragusa bank, 10)	1.3±1.3	88–112	(Cattaneo-Vietti *et al.*, 2017)
Italy (Sicily, Ragusa bank, 19)	10.7±2.1	88–112	(Cattaneo-Vietti *et al.*, 2017)
Italy (Sicily, Ragusa bank, 20)	0.4±0.4	88–112	(Cattaneo-Vietti *et al.*, 2017)
Italy (Sicily, Ragusa bank, 22)	6.4±0.8	88–112	(Cattaneo-Vietti *et al.*, 2017)
Italy (Sicily, Ragusa bank, 23)	3.4±0.9	88–112	(Cattaneo-Vietti *et al.*, 2017)
Italy (Sicily, Ragusa bank, 25)	9.1±1.0	88–112	(Cattaneo-Vietti *et al.*, 2017)

*colonies 400 cm^2; $^+$colonies 0.25 m^2; $^\#$range

Extensive data on red coral population densities have been recorded in recent years for Sardinian mesophotic banks (Cau *et al.,* 2016; Cau *et al.,* 2015a; Cau *et al.,* 2017; Cau *et al.,* 2015b) (Table 8).

Table 8 –Red coral densities in deep waters in Sardinia.

Country	Density (colonies/m^2)	Depth (m)	Reference
Italy (Sardinia- S. Pietro island)	12.88±16.18	80–85	(Cau *et al.*, 2015b)
Italy (Sardinia- Capo Carbona)	54.14±32.36	90–115	(Cau *et al.*, 2015b)
Italy (Southwestern Sardinia, S1)	5.75 ± 1.46	150–170	(Cau *et al.*, 2015a)
Italy (Southwestern Sardinia, S4)	0.68 ± 0.35	120–130	(Cau *et al.*, 2015a)
Italy (Southwestern Sardinia, S5)	0.19 ± 0.04	120–130	(Cau *et al.*, 2015a)
Italy (Sardinian vertical-wall VWe1)	15.1$^+$	80–90	(Cau *et al.*, 2016)
Italy (Sardinian vertical-wall VWe2)	8.08$^+$	120–130	(Cau *et al.*, 2016)
Italy (Sardinian vertical-wall VWe3)	1.9$^+$	115–130	(Cau *et al.*, 2016)
Italy (Sardinian vertical-wall VWe4)	19.5$^+$	120–140	(Cau *et al.*, 2016)
Italy (Sardinian vertical-wall VWe5)	5.34$^+$	120–130	(Cau *et al.*, 2016)
Italy (Sardinian vertical-wall VWe6)	1.8$^+$	115–130	(Cau *et al.*, 2016)
Italy (Sardinia pinnacle PIe_1)	9.08$^+$	155–170	(Cau *et al.*, 2016)
Italy (Sardinia pinnacle PIe_2)	2.27$^+$	90–100	(Cau *et al.*, 2016)
Italy (Sardinia pinnacle PIe_3)	1.12$^+$	110–120	(Cau *et al.*, 2016)
Italy (Sardinia pinnacle PIe_4)	1.25$^+$	125–130	(Cau *et al.*, 2016)
Italy (Sardinia pinnacle PIe_5)	3.7$^+$	80–85	(Cau *et al.*, 2016)
Italy (Sardinia pinnacle PIe_6)	0.22$^+$	120–130	(Cau *et al.*, 2016)
Italy (Sardinia, Canyon E2	<0.1	90–150	(Cau *et al.*, 2017)
Italy (Sardinia, Canyon E3	1.5±0.7	90–150	(Cau *et al.*, 2017)
Italy (Sardinia, Canyon E4	0.5±0.3	90–150	(Cau *et al.*, 2017)
Italy (Sardinia, pinnacle, W1)	<0.1	80–160	(Cau *et al.*, 2017)
Italy (Sardinia, pinnacle, W2)	0.51±0.10	80–160	(Cau *et al.*, 2017)
Italy (Sardinia, pinnacle, W3)	0.1±0.01	80–160	(Cau *et al.*, 2017)

*colonies 400 cm^2; $^+$colonies 0.25 m^2; $^\#$range

2.8. Genetics and genetic stock identification

The bulk of available genetic knowledge refers to the more accessible shallow-water populations (from 15 to 60 m of depth) (Abbiati *et al.*, 1992, 1993, 1997; Aurelle *et al.*, 2011; Calderon *et al.*, 2006; Casu *et al.*, 2008; Costantini *et al.*, 2003, 2007a,b, 2011; Del Gaudio *et al.*, 2004; Ledoux *et al.*, 2010a,b). These studies found evidence of breeding isolation and population sub-structuring suggesting that larval dispersal could not be able to ensure sufficient gene flow to preserve genetic homogeneity of the species. Several studies confirmed the occurrence of genetic differentiation at spatial scales of 10s of meters (that is the effective larval dispersal range may be restricted to < 10 m). The strong genetic differentiation between nearby samples implies that the recovery of overexploited populations should be mainly due to self-recruitment. Moreover, a pattern of Isolation by Distance was described in Ledoux *et al.*, (2010b), that is the more distant the more different populations are. All studies revealed, using differentially powerful markers, significant deviations from Hardy-Weinberg equilibrium due to elevated heterozygote deficiencies, consistent with the occurrence of inbreeding (mating between consanguineous).

For deep-water populations, the number of genetic studies is very limited. Costantini *et al.*, (2011) analysed colonies along a depth gradient (from 20 to 70 m) and showed strong patterns of genetic structuring among the samples both within and between two study sites (Catalan and Ligurian Sea), with a pattern of reduction in genetic variability with depth. A threshold in connectivity was observed among the samples collected across 40–50 m depth, supporting the hypothesis that separations between shallow- and deep-water red coral populations occur. This finding could have major implications for management strategies and the conservation of commercially exploited deep red coral populations.

In Sardinian populations (range from 80 m to 120 m of depth) high level of genetic differentiation was measured, over different spatial scales from hundreds to less than 1 km (Cannas *et al.*, 2010, 2011, 2015, 2016). Results seem to indicate the existence of a strong genetic differentiation among populations over the different depths (banks from 30/60 meters vs. banks from 80/120 m), further underlying possible restrictions to gene flow along the depth gradient (Cannas *et al.*, 2016). The genetic results indicated that for precious coral populations in Sardinia the ‘deep reef refugia hypothesis', that envisages the capacity for deep corals to act as seed banks for the shallower impaired (overharvested) populations, is not supported (Cannas *et al.*, 2016).

The highest genetic diversity recorded in Sardinia for all areas and depths with respect to other Mediterranean areas indicates that the strict local management has been effective, since harvesting has not yet led to a substantial erosion of the genetic pool. Possible causes for the high levels of observed diversity in Sardinia are in relation to hydrological conditions, its geographical position and its proximity to some putative glacial refugia (Cannas *et al.*, 2016).

Strongly differentiated red coral populations in the Tyrrhenian with respect to the Algero-Proveṇal Basin were described (Cannas *et al.*, 2015). From a management perspective, the high genetic diversity and the substantial demographic stability indicate that Sardinian deep red coral populations are stable. Nevertheless, the significant spatial structuring at the scale of less than 10 km indicates that they are sustained largely by local recruitment, excluding the possibility that they can help in recovering shallower banks (Cannas *et al.*, 2015).

The genetic spatial structuring of deep-water *C. rubrum* colonies found between 58/118 m of depth within the Tyrrhenian Sea confirmed the occurrence of significant genetic differentiation at both small and large spatial scales (Costantini *et al.*, 2013).

Costantini and Abbiati (2016) analyzed genetic variability of *C. rubrum* populations dwelling between 55 and 120 m depth, from the Ligurian to the Ionian Sea along about 1500 km of Italian coastline. The estimated gene flow seems to indicate that each basin contains the major sources of its own populations, and small proportions of gene flow from other populations. Ligurian and Tyrrhenian populations showed lower genetic diversity compared to the populations from the Sardinian Sea (Cannas *et al.*, 2015; Cannas *et al.*, 2016), suggesting that they may have smaller population size and possibly have been overexploited.

Recently, (Jaziri *et al.*, 2017) provided the very first knowledge on genetic and biological features of commercial populations from the south-western Mediterranean Sea (Tunisia, North African coast; depth range 44–83 m). The main results indicate that Tunisian populations have a weak genetic structuring but significant differentiation between coastal and offshore locations. Moreover, harvesting seems not have altered yet the structure of red coral populations.

Concerning the deepest red coral populations, a few colonies (a total of 12 fragments from 5 sites from Malta and Linosa shelves) from the lower limit of red coral depth range in the Mediterranean Sea (up to 819 m of depth) were genetically characterized by Costantini *et al.* (2010). The authors found differences between shallow and deep-water samples, but the small sample size of the deep-water collections did not allow the authors to make final considerations on the degree of isolation (Costantini *et al.*, 2010).

An interesting experiment was recently performed combining transplant and genetic analyses (Ledoux *et al.*, 2015). Two populations dwelling in contrasted temperature regimes and depths (20 m and 40 m) were reciprocally transplanted. The analyses reveal partially contrasting PEIs (population-by-environment interactions) between shallow and mesophotic populations separated by approximately one hundred meters, suggesting that red coral populations may potentially be locally adapted to their environment with differential phenotypic buffering capacities against thermal stress (Ledoux *et al.*, 2015). The study questions the relevance of the deep refugia hypothesis and highlights the conservation value of marginal populations as a putative reservoir of adaptive genetic polymorphism.

The adaptive response to *C. rubrum* to thermal stress has been recently studied (Haguenauer *et al.*, 2013) on three populations from different depths (5 m, 20 m and 40 m depths). In particular, the 5 m population seems to have a higher thermal tolerance: they showed higher gene expression response (induction of HSP70 after heat shock), which might be explained by local adaptation or acclimatization. The authors suggest that, facing climate change, these shallow populations may be an irreplaceable source for evolutionary rescue and re-colonization potential, and must worth special attention and protection (Haguenauer *et al.*, 2013).

New candidate markers potentially implicated in the response to thermal stress, which could be good candidates for the study of thermal adaptation for the red coral have been proposed by (Pratlong *et al.*, 2015).

On a larger spatial scale, mitochondrial data suggest that Atlantic *C. rubrum* is genetically distinct from Mediterranean one, Atlantic red corals may have introgressed the Mediterranean ones after migration via the Algeria current (Boavida *et al.*, 2016).

On the overall the genetic studies performed within the Mediterranean Sea so far highlight a strong genetic heterogeneity even at very small spatial scales, and suggest that management of red coral has to be planned on a local base. Individual harvesting plans for each bank should also to be considered, that is each commercial stock should be previously characterized from the genetic point of view to identify population boundaries and define management units.

2.9. Mortality

As already said, the high commercial value of red coral involves that this vulnerable species is subject not only to natural mortality but also to harvesting mortality.

Natural mortality of red coral is due to competition over space with sponges and other sessile biota, dislodgement from the substrate due to the action of boring species (Harmelin *et al.*, 1984) or seismic movement (Di Geronimo *et al.*, 1994), predation by the small gastropod *Pseudosimnia carnea* and the crustacean *Balssia gasti* (Abbiati *et al.*, 1992), sedimentation increase, widespread mortality due to epidemies, and ocean acidification.

Phenomena of mass mortality events have been observed in shallow waters populations since the late 1990s, including several mass-mortality events linked to elevated temperature anomalies (Bramanti *et al.*, 2005). A mass mortality event occurred in the NW Mediterranean, from Tuscany (Italy) to Marseille (France), in summer 1999. The phenomena, in which about 80 percent of the colonies were affected, was attributed to a fungal and protozoan diseases, and linked to temperature anomalies (Cerrano *et al.*, 2000; Garrabou *et al.*, 2001; Perez *et al.*, 2000; Romano *et al.*, 2000; Van De Water *et al.,* 2016). In the same year (late summer 1999) some shallow water red coral populations have been affected by mass mortality associated to anomalous temperature increase in the Eastern Ligurian Sea (Calafuria, Italy; Bramanti *et al.*, 2005), as well as western Ligurian Sea. A recent mass mortality episode was also recorded along the Amalfi coast, around Li Galli Islands (Gulf of Salerno, South Tyrrhenian Sea), at a depth range between 80 and 100 m, where the mortality affected 80 percent of the largest colonies, estimated to be around 70 years old (Bavestrello *et al.*, 2014b). Several possible reasons for this mortality have been hypothesised, such as the formation of local down-welling currents inducing an unusual drop of the thermocline, or sudden warm water emissions (sulphur springs) in an area characterized by important volcanic activities, or local landslides generating turbidity currents along the steep slopes (Bavestrello *et al.*, 2014b).

Mortality in *C. rubrum* had a different impact, depending on the colony size. Large colonies are more resilient to natural stressors. On the contrary, small colonies prevalently suffer of higher virulence, showing higher whole-colony mortality rates. However, further studies are required to provide basic information regarding red coral population dynamics as a basis for the hypothesis on the actual recovery capability of affected populations.

In a recent study, the colonies of *Corallium rubrum* proved to be extremely resistant to the stress of transplantation, displaying high survival rates similar to those in natural populations (Montero-Serra *et al.*, 2017). Transplanted *C. rubrum* colonies were subject to the stresses of being harvested, kept out of the water in the poachers' nets, transported, maintained in aquaria for one week, and then transplanted back into natural habitat. Yet these transplanted *C. rubrum* colonies had a similar proportion of fertile colonies and even higher frequency of larvae per polyp after 4 years than observed for colonies in natural populations (Montero-Serra *et al.,* 2017).

Many species (mainly sponges, crustaceans, brachiopods, molluscs and echinoderms) have been documented living on/in or in strict association with red coral colonies (Calcinai *et al.*, 2010; Crocetta and Spanu, 2008). Some of these species, especially the parasitic ones, may increase the red coral mortality rate and therefore profoundly affect its population structure (Corriero *et al.*, 1997). Apart from being the main causes of natural mortality, sponges have the ability to damage colonies and thus reduce the commercial value of red coral (Calcinai *et al.*, 2010).

3. INSIGHTS ON TRADE

Countries with some history in red coral harvesting in the Mediterranean Sea are less than a dozen. In Albania, Malta and Monaco, there is the prohibition of harvesting red coral while Croatia, France, Italy, Montenegro, Spain and Tunisia exploit red coral under different national regulation frameworks (including the implementation of multi-annual closures to allow the recovery of exploited red coral banks). In Greece, Algeria and Morocco (Mediterranean coast), red coral fisheries are temporarily closed, although for the latter two CPCs an assessment of the closure is undergoing.

Nowadays, Torre del Greco manufactures (Italy) supply with Mediterranean raw corals from France, Italy, Morocco, Spain, and Tunisia and in the past (when available) also from Albania, Algeria, and Greece. The imported raw coral is not usually re-exported but worked on site.

It is difficult to determine the exact number of people harvesting red coral. In general, they can be assigned to two categories: people materially collecting colonies (divers) and people indirectly involved in it (the crew and owners of the boats equipped for the diving, people involved in supplying services for the at sea, SCUBA diving); while for the first ones some rough estimates can be tentatively presented, the number of the last category is almost impossible to define. In the last years many countries have regulated the harvesting of red coral by defining a fixed number of authorized divers (or boats); the number of divers and the individual quota, etc. according to relevant GFCM management decisions.

According to the provisional analysis made by Cau *et al.* (GFCM, 2013), in 2010 about 350 divers could be involved in the legal fishing of red coral in the Mediterranean area, summing up the licenses indicated in the respective national legislations as the maximum number allowed in each year/for each area. However, those figures do not indicate the actual number of divers effectively working at sea, since often not all the licenses are granted in each year/area or not all the divers with a valid license work.

In general, in the Corsica and Sardinia ‘young’ divers (<34 years old) are absent, with the prevalence of those in the range 56–65 years. As *Corallium rubrum* harvesting mainly concentrates on the warmer summer months (May–October), it means that most of the divers are effectively working part time, and often invest into other businesses as well (Tsounis *et al.*, 2010). Regarding the number of people indirectly involved in the harvesting, it is impossible to be determined from the available data; for instance, in some countries divers can work in couple and hence share the crew or own the boat they use. The collecting of such kind of information and more in general the realization of a full socio-economic analysis of the sector should be regarded as a priority by the countries involved in red coral exploitation.

Since the Roman Empire, Italy has been the center of Mediterranean red coral harvesting, manufacturing, and trading. However, based on historic data collected and compiled by private industries, red coral has been imported and used in expensive jewelry in Taiwan Province of China (Huang and Ou, 2010). This is also confirmed by the fact that in the years 1986–1988 substantial quantities of Mediterranean red coral were said to be exported to Taiwan Province of China and Japan for processing (GFCM, 1989). *Corallium rubrum* imported into Japan, was regarded as one of the most valuable presents to state officials, and is still highly revered (Tsounis *et al.*, 2010). Even in the very recent years relevant amounts of raw Mediterranean red coral have been exported from Italy to the United States of America (CITES, 2010) and more recently to India and China taking advantage of the economic crisis.

The market price of raw red coral varies consistent with its quality and origin. According to the information provided by Borras, a Spanish coral manufacturer, during the first GFCM technical consultation in 1984 (GFCM, 1984), red coral colonies are classified in five different categories: tips, third category colonies (mainly < 7 mm of trunk diameter), second category colonies (7–9 mm of trunk diameter), first category colonies (>10 mm of trunk diameter, usually 12–14 mm) and special category colonies (>14 mm in diameter and >100 g in weight). The most profitable red coral colony should have the following characteristics: 15/12 cm in size, approximate weight of 100 g, 10–15 mm diameter of the trunk and at least 4 mm thickness at the tips. The quality (and value) can vary depending on whether the specimen was harvested as a living or dead colony (or a dead broken branch). Coral that was harvested live is the most valuable because of its deep colour and high translucency, decreasing as a dead coral ages on the ocean bottom (Cooper *et al.*, 2011). The quality of products made is also affected by damage from boring sponges of the family Clionidae, that create a series of holes in the skeleton (Cooper *et al.*, 2011).

In 1984, the market price of raw red coral in Spain varied between USD 61 to 610 /kg, considering the conversion rate for those years (Borras in GFCM, 1984). The prices of by products like twigs and/or thin trunks, pieces and impaired remains for the costume jewelry market ranged between USD 182–426 /kg (Borras in GFCM, 1984). In the same years, the market price for red coral in Italy ranged from USD 25–300 /kg, depending on the quality, size and other elements that should be assessed case by case even for colonies caught in the same area (Iacobelli in GFCM, 1984). At that time, in order to make profit, a boat (dredges were still in use) or a diver needed to collect about 200 kg of coral a year (Iacobelli in GFCM, 1984). According to the Chairman of the Association of Coral Producers of Torre del Greco, until the 1970’s all kinds and qualities of coral were easily marketed (GFCM, 1989). Later, in the mid-80s a strong request for dark red coral only of high quality was recorded, while clearer corals could hardly be sold. Certain types of corals, although available on the market at reasonable prices were not any more interesting for the processing industry, this was reflected also in decreasing of the overall quantity of coral processed at Torre del Greco in those years (GFCM, 1989). *C. rubrum* is currently still sold for relatively high prices: in Torre del Greco medium and big sized colonies are prevalently worked whose price can range from a minimum of EUR 200 to a maximum of EUR 2 000 /kg (in cases of colonies of extremely high quality). Unconfirmed information indicates that one single, large *C. rubrum* colony with a basal diameter of more than 4 cm was sold for as much as EUR 45 000 (Tsounis *et al.*, 2010). Tropical *Corallium* species are also of high value. To have a term of comparison, in 2009 the price/kg of the momo coral (*Corallium elatius*) in the Taiwanese auctions ranged from USD 840 to 2 031 /kg, while for Aka coral (*C. japonicum*) the price was USD 1 093 /kg (Huang and Ou, 2010).

4. MANAGEMENT OF RED CORAL HARVESTING IN THE MEDITERRANEAN

4.1. Regional management measures by the GFCM

Considering the high vulnerability of red coral to fishing activities, the GFCM has always kept red coral in its work plan and discussed about measures to ensure sustainable harvesting of red coral since the 80s; in this view, the GFCM has organized several meetings on the topic with a participatory approach as to involve all stakeholders in the discussion:

Technical Consultations on Red Coral of the Mediterranean
December (13–16), 1983 Palma De Mallorca, Spain
September (27–30), 1988, Torre del Greco, Italy
Transversal workshop on Red Coral
September (16–17), 2010, Alghero, Italy
Second Transversal workshop on Red Coral
October (5–7), 2011, Ajaccio, Corsica, France
Workshop on Regional Management Plan for Red Coral
January (21–22), 2014, Brussels, Belgium
Working groups on stock assessment of demersal and small pelagic species (including assessment of red coral populations)
November (24–27), 2014, GFCM HQ, Rome, Italy
Workshop on red coral
March (9–10), 2017, Gammarth, Tunisia

Still, in 2017 red coral became part of the GFCM 'mid-term strategy (2017–2020) towards the sustainability of Mediterranean and Black Sea fisheries' launched to define a course of decisive actions aimed at reverting the alarming trend in the status of commercially exploited stocks. Aligned with the UN Sustainable Development Goals (SDGs), the GFCM mid-term strategy seeks to improve Mediterranean and Black Sea fisheries and contribute to the sustainable development of coastal States. The overall objective of the mid-term strategy is to improve, by 2020, the sustainability of Mediterranean and Black Sea fisheries, by achieving five targets and related outputs and activities. Target 4 (Minimize and mitigate unwanted interactions between fisheries and marine ecosystems and environment) foresees measures taken to minimize and mitigate negative impacts of fisheries on marine biodiversity, especially in relation to vulnerable species and ecosystems, as well as to mitigate negative anthropogenic effects on fisheries. Target 4 will be achieved by the following outputs: i) Reduced bycatch rates in Mediterranean and Black Sea fisheries, and ii) Healthier marine ecosystems and more productive fisheries. Under the scope of the latter, the adoption of a comprehensive regional management plan for red coral is mentioned as one of the key activity.

Regarding management measures, four decisions on red coral harvesting have been adopted by the GFCM since 2011, with the later recently adopted in 2017 (entry into force date 1 April 2018). The recommendations outlined below (Table 9) are binding to GFCM Contracting Parties and Cooperating Non Contracting Parties (CPCs).

Table 9 – GFCM recommendations addressing red coral harvesting

Year	Recommendation	Main measures	Reference
2011	GFCM/35/2011/2 on the exploitation of red coral in the GFCM Competence Area	• Only permitted gear for red coral harvesting is hammer used by a scuba diver; use of remotely operated vehicle (ROV) is prohibited • Prohibition of exploiting (shallow) red coral populations at depth < 50 m. • Mandatory to record and report to GFCM daily catches and fishing effort by area and depths (e.g. number of fishing days, numbers of diving, etc.)	Link
2012	GFCM/36/2012/1 on further measures for the exploitation of red coral in the GFCM area	• Minimum landing diameter is 7 mm at the trunk measured within 1 centimeter from the base of the colony (tolerance of undersized colonies of 10%) • Red coral landed only in a limited number of designated ports communicated to GFCM	Link
2016	GFCM/40/2016/7 concerning the authorization of the use of remotely operated vehicles within the framework of national scientific research programmes on red coral	• Use of ROV is limited to observation for scientific purposes and authorized until 31 December 2017 • Commercialization of red coral harvested within research programmes is forbidden	Link
2017	GFCM/41/2017/5 on the establishment of a regional adaptive management plan for the exploitation of red coral in the Mediterranean Sea	• Individual system of daily and/or annual catch limitation by Countries • Fishing effort not to increase • When undersized specimens of red coral (i.e. colonies whose basal diameter is lower than 7 mm) exceeds 25 percent of the total catch harvested from a given red coral bank in a given year, Countries close the area to fishing • Countries harvesting red coral shall introduce additional closures for the protection of red coral on the basis of the scientific advice available and not later than 1 January 2019 • Use of ROV is limited to observation for scientific purposes and authorized until 31 December 2020	Link

In addition, in 2014, the Commission agreed to adopt specific "Guidelines for the management of Mediterranean red coral populations" based on the document GFCM, 2013. Adaptive Management Plan for Red Coral (*Corallium Rubrum*) in the GFCM Competence Area (Part I, Part II and Part III). Prepared by Cau A. Cannas R., Sacco F., and Follesa M.C. to promote consensus on management measures to be applied in GFCM Countries in order to avoid overexploitation of red coral populations. The Guidelines were conceived to facilitate the preparation of a precautionary, provisional and adaptive regional management plan, as these provided elements to maintain the status quo of the resource in the absence of data to perform a formal assessment of the stocks at a regional scale. The core features of these Guidelines are presented in Appendix.

4.2. Management measures and status of red coral fishery in GFCM countries

GFCM CPCs traditionally exploiting red coral banks in their territorial waters are: Algeria, Croatia, France, Greece, Italy, Morocco, Spain and Tunisia. Among these, Greece, Algeria and Morocco have temporarily closed the fishery according to their national legislations; if in any of these Countries fishing of red coral would start again, its exploitation must be regulated according to GFCM decisions or to stricter national measures.

The regular assessment of the information submitted by GFCM Countries in response to the aforesaid recommendations falls under the mandate of the GFCM Scientific Advisory Committee on Fisheries (SAC) for the provision of advice to the Commission. Here below we present a summary of the data received from 2013 to 2017, including an assessment of the implementation of the GFCM management measures. The data presented do not yet include information requested through *Recommendation GFCM/41/2017/5 on the establishment of a regional adaptive management plan for the exploitation of red coral in the Mediterranean Sea,* which has formally entered into force only on 1 April 2018.

4.2.1. Implementation of management measures

The compulsory management measures provided in GFCM decisions (e.g. minimum landing size and minimum depth for fishing activities) are generally implemented at the national level. In eight countries, fishing activities are currently forbidden, while one country is applying a more restrictive minimum size, and one country is applying a stricter depth limit. Minimum landing size is not observed in one country, while two countries by way of derogation from paragraph 4 of GFCM/35/2011/2 *on the exploitation of red coral in the GFCM Competence Area* authorized exploitation of red coral at less than 50 m as an appropriate national management framework has been developed ensuring an authorization system and that only a limited number of red coral banks are exploited. The tables below (Table 10 and 11) provide summaries of the information collected on different aspects of the national measures in place by country.

Table 10 - Management data requested by the GFCM Secretariat

N	Question/Field
1	Reference to national/regional legislations
2	How many banks/areas have been identified?
3	How many of them are harvested?
4	Are there "no take zones" for red coral?
5	If yes: specify the number
6	Are those within a Marine Protected Area (MPA) or any type of marine reserve?
7	If yes: provide the name(s)
8	Is there any Marine Protected Area (MPA) or Marine Reserve specifically aimed at protecting Red Coral?
9	If yes: provide the name(s)
10	Is there alternate areas system in place?
11	If yes: how many areas?
12	How long intervals?
13	Number of harvesting licenses in the country
14	Number of active licenses (current year)
15	Cost of license fee per year
16	Permitted quota (annually or daily) per active license
17	Period in which harvest of red coral is allowed
18	Legal minimum size (diameter measured within one centimeter from the base of the colony)
19	Percentage of allowance (Percentage of total weight allowed for undersized red coral colonies)
20	Depth range allowed
21	Is the harvest declared by divers?
22	Is the logbook mandatory?
23	Is the harvest recorded by authorities?
24	How is the minimum size controlled?
25	Is there a programme of observers on board?
26	Is there a programme of observers on landing site?
27	Designated ports for landing red coral
28	Are sale notes mandatory?
29	Are data on the buyer and seller registered?
30	Number of ROV authorizations for prospection purposes
31	Is there any biological sampling programme in place?
32	Is there any research project in place?

Table 11 - Data on management received by the GFCM Secretariat

	Country						
Question	**Croatia**	**Greece**	**France**	**Italy**	**Morocco**	**Spain**	**Tunisia**
1	national	national	national and local	local	national	national and local	national
2	X	5	X*	50 (T); X (S)	3	4(e)	9
3	X	0	X*	X (T); 8 (S)	3	4	7
4	no	no	yes	yes	yes	yes	yes
5	na	na	40	20 (T); 9 (S)	2	no specific number	2
6	X	no	yes	no (T); yes (S)	no	yes	no
7	na	na	Nat.Plan**	yes	na	X	na
8	X	no	yes	no (T); yes(S)	yes	no	no
9	na	na	3	na (T); 2	2	na	na
10	X	yes	yes	no (T); yes(S)	yes	no	no
11	X	5	Nat.Plan**	na (T); 6	2	na	na
12	X	25 y	X	na (T); 34 y	5–10 y	na	na
13	53	0	30	17(T); 25(S)	10	53 (37e+16i)	27
14	11	0	30	2(T); 12 (S)	9	41(25e+16i)	26
15	0	2934.7 EUR	0	EUR 516.2 (T); 1500 (S)	variable[§§]	0	X
16	200 kg/li/y	No limit	50 kg HP	X(T); 2.5 kg/day (S)	400 kg/li/y	300 kg/li/y[§§§]	no quota
17	Apr–Nov	Apr–Dec	All year	All year(T); Jun–Sep (S)	All year	May–Oct	All year
18	X	7 mm	7mm (8 HP)	8mm (T); 10mm (S)	7 mm	7mm	>7mm
19	X	none	10%	5%	10%	10% (5% C)	10%
20	>50 m	>=50 m	Nat.Plan**; >=50 m Co	>60m(T); >50m (S)	40–80m	no limitation	>=50m
21	X	yes	yes	yes	X	yes	yes
22	yes	yes	yes	no (T); yes (S)	yes	yes	yes
23	yes	yes	yes	no (T); yes (S)	yes	yes	yes
24	X	Ministry of shipping and island policy	controls on board and at landing	X(T); Maritime Authority at designated ports (S)	In landed coral	At landing	X
25	X	no	no*	no (T); not yet (S)	no	no	no
26	X	no	no*	no	no	no	no
27	X	9***	Natl.Plan**	X(T); 9(S)	1 (Atlantic)	28	2
28	yes	yes	no*	yes	yes	yes	yes
29	yes	yes	X*	X(T); yes (S)	yes	yes	yes
30	X	prohibited	8	0(T); 2–8(S)[§]	no	0	0
31	X	X	no	no (T); yes (S)	no	no	X
32	X	X	no	Yes	no	no	X

Legend: **X**= unanswered; **na**= not applicable; **T**= Tuscany; **S**= Sardinia; **HP**= Haute Pyrenées; **C**= Cataluña; **Co=** Corsica; **e**= aguas exteriors; **i**= aguas interiors; **kg/li/y=** kilos per licence per year;

Notes: *answer: not mandatory for GFCM recommendations; ** Natl.Plan= Confert French National Management Plan; ***only codes provided not full names; §= 8 in 2013, 2 in 2014 and 2015; §§ variable, according to the tonnage of the ship; §§§= summing up both catches of interior and external waters.

4.2.2. Exploitation of red coral

The compulsory measures provided in the above mentioned GFCM decisions request CPCs exploiting Mediterranean red coral to submit a series of compulsory and optional information on harvesting activities to the GFCM Secretariat. Compulsory information includes: area of exploitation (GSA and Statistical Grid), area name, name of landing port, annual catch of red coral, average diameter (mm) of the colonies harvested; percentage in weight of undersized (i.e. <7 mm) colonies; effort expressed as annual number of fishing days or of dives; range depth of harvesting.

Figure 2 show the annual harvest quantity as reported to GFCM. Trend analysis per country by both yield/year and yield/dive for indicate that total landings of red coral are decreasing in some countries but are stable in others. Information related to catches, the size structure and the biology of the species is overall not sufficient to provide an overall assessment on the status of red coral populations. In general, important compulsory data, such as the average colony diameter and the percentage of undersized colonies, has not yet been made available, and an improvement of information is expected as a result of the entry into force of Recommendation GFCM/41/2017/5.

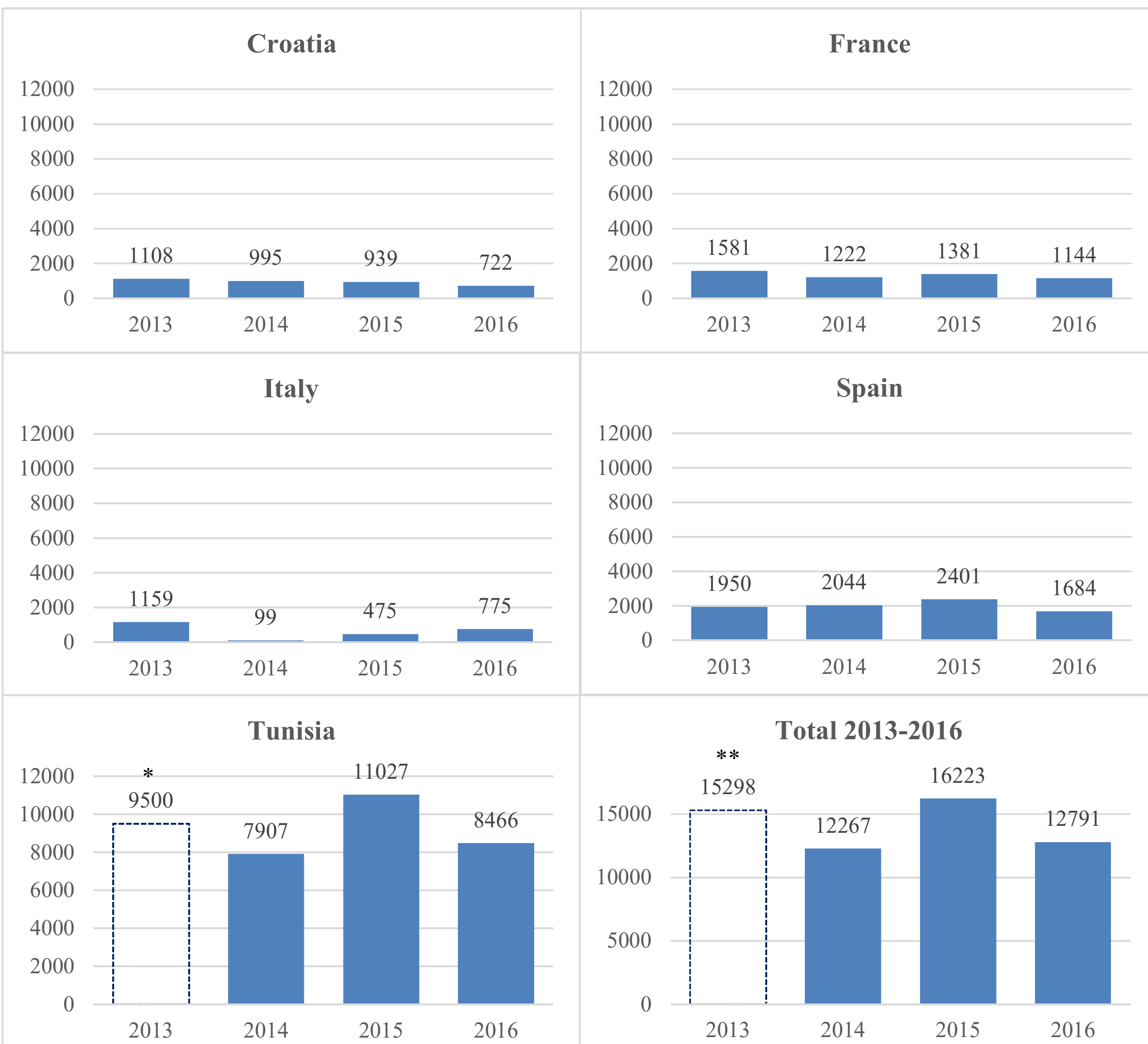

*Figure 2 – Data on annual red coral harvest from 2013 to 2016 received by the GFCM Secretariat (harvest values expressed in kilograms, *= experts' estimated value the as official production data was not made available to the GFCM; **=the value includes the estimation for Tunisia 2013)*

4.2.3. Additional measures to limit the effort and to enhance monitoring, control and surveillance (MCS)

The implementation of additional management measures to limit effort and harvesting and to enforce MCS of red coral fishery were also reviewed. Tables 12 and 13 summarize which of the additional (not compulsory) measures proposed in the 2014 GFCM Guidelines are implemented by GFCM Countries.

Table 12 – Additional technical management measures adopted by Countries to limit fishing effort and harvest of red coral (from the 2014 GFCM Guidelines for the management of Mediterranean red coral populations).

Management measure	Specific measure	Rank effectiveness	Rank feasibility	Comment	GFCM[1] countries implementing the measure
Limits to fishing capacity	Licensing systems	High	High		**7/7**
Limits to catches	Individual annual quotas	High	High	Based on scientific data	**5/7**
	Individual daily quota	High	High	Might force divers to do more trips per year, but allows efficient inspections	**1/7**
Spatial restrictions	License restricted to certain areas	High	High		**7/7**
	Establish refugia, or permanently closed zones	High	High	Deep populations found in virgin status should be kept as refugia and specific MPAs to protect red coral could be established	**?/7**
Temporal restrictions	Seasonal harvest restriction	High	High	Facilitates control of effort	**4/7**
	Rotation periods in different banks	Medium	Medium	Recovery rates of red coral are low – coral needs 25 to 30 years to reach the minimum legal size – and geographical variation is not well known.	**4/7**

Legend: ?= Partial data or data not available

[1] GFCM countries involved in red coral harvesting: Algeria, Croatia, France, Greece, Italy, Morocco, Spain and Tunisia.

Table 13 – Additional MCS measures adopted by countries to control the red coral fishery (from the 2014 GFCM Guidelines for the management of Mediterranean red coral populations)

MCS measure	Proposal	Rank effectiveness	Rank feasibility	Purpose	GFCM countries implementing the measure
Logbook	Logbook	High	High	To register the catches and related data by dive on a daily basis	**7/7**
Designation of ports	Designation of ports	High	High	Provide the designated ports with the necessary facilities and personnel	**7/7**
Observers on board	Scientific observers on board	High	Low	To control size, transshipment and sales prior to landing	**1/7**
Patrolling unit		High	Medium	To control depth, licenses, gear, size	**?/7**
Certification of logbook at landing sites		High	High	Logbook must be certified at landing to verify it contents with the actual landed catches	**?/7**
	Advance warning	High	High	A phone call to the port when the vessel is approaching	**?/7**
	Tracking device on board	High	Medium	To control that harvest takes place only at appropriate sites	**?/7**
Traceability mechanisms	Sales note with details of the seller, the buyer and a code for each lot sold	Medium	Medium	To control the origin (legal and geographical) of corals, and address poaching. Certified coral from legal fisheries might have added value.	**6/7**

Legend: **?=**Partial data or data not available

5. WAY FORWARD TO ENHANCE THE SUSTAINABLE EXPLOITATION OF RED CORAL IN THE MEDITERRANEAN

5.1. Collection of fishery data

Reliable data concerning red coral harvesting are essential in order to understand the level of exploitation of the resource. Yet in 2010, it was suggested that for the FAO Area 37, i.e. the Mediterranean Sea, the FAO Global Production Statistics was inaccurate and that it presented some shortcomings as red coral data are consistently submitted since mid-1980s by a major red coral import-export and production of jewelry wholesaler, and not by national official sources as for other exploited species (see GFCM, 2010, 2011, and 2017b). The data available in the FAO database is therefore understood as an estimate of the industrial annual production for that given year and not the actual annual harvest of the resource.

In order to go beyond such limit and to ensure the sustainable exploitation of red coral in the Mediterranean Sea, in 2012, the GFCM started to put in place a series of measures, including the establishment of its own data collection protocol to obtain annual landings data of red coral from the national administrations of its member countries. The GFCM data collection (GFCM, 2018) for red coral is coherent with the binding decisions adopted in the last years and it ensures that the minimum data required to assess the status of the fishery is collected. Nonetheless, the analysis of the data received from 2013 to 2016 highlighted that often the data related to catches, the size structure and the biology of the species are in general not sufficient to provide an overall assessment on the status of red coral populations. In some cases, important compulsory data, such as the overall quantities of red coral caught per year or the percentage of undersized colonies, has not yet been made available to GFCM.

It can be recognized that GFCM, as the RFMO of the Mediterranean Sea, has given high priority to the management of red coral and as results of the effort done, it put in place a series of measures to guarantee the sustainable exploitation of red coral, including the newly adopted decision *GFCM/41/2017/5 on the establishment of a regional adaptive management plan for the exploitation of red coral in the Mediterranean Sea* and the inclusion of red coral as a priority within the GFCM ‘mid-term strategy (2017–2020) towards the sustainability of Mediterranean and Black Sea fisheries’. There can therefore be no doubt that GFCM has taken into serious account the management of the red coral fishery and that it has issued several binding decisions in this regard, however it cannot be neglected that to enable an accurate assessment of the fishery, GFCM member States must comply with the aforesaid decisions and submit precise production figures, including the percentage of undersized colonies. Future GFCM actions should also address the issue of ensuring the proper reporting of obligatory data.

5.2. Possibilities for sustainable exploitation

The full implementation of the GFCM measures by member States, including the newly established management plan (*Recommendation GFCM/41/2017/5 on the establishment of a regional adaptive management plan for the exploitation of red coral in the Mediterranean Sea*) is expected to be sufficient counteract or prevent overfishing with a view to ensuring long-term yields while maintaining the size of red coral populations within biologically sustainable levels. In addition to the measures already implemented, according to the new 2017 management plan, CPCs shall establish an individual system of daily and/or annual catch limitation; will maintain the fishing effort at the levels authorized and applied in recent years for the exploitation of red coral and will temporarily close the area concerned to any red coral fishing activity when undersized specimens of red coral (i.e. colonies whose basal diameter is lower than 7 mm) exceeds 25 percent of the total catch harvested from a given red coral bank for a given year. In addition, notwithstanding the spatio-temporal closures already established at the national level, countries actively harvesting red coral must introduce additional closures for the protection of red coral on the basis of the scientific advice available by the end of 2018.

It is apparent how the GFCM is attempting its best to achieve such ambitious goals by putting at disposal all the necessary background and indications to its member countries to adjust exploitation rates and fishing capacity to sustainable levels, however only the actual level of implementation at national level will determine the success or the failure of such series of regional measures. In parallel, actions to eliminate illegal, unreported and unregulated (IUU) fishing of red coral shall be also foreseen. Similarly, traceability mechanisms could be envisaged: from the time the coral is landed and sold as raw material to the manufactures until it reaches the retailer as a finished product. These mechanisms would allow to certify that the red coral was collected in compliance with Mediterranean or national regulations and it would be also effective in eradicating red coral IUU fishing.

Nevertheless, it must be noted that only in few countries red coral harvesting is currently active while in most of Mediterranean countries red coral is fully protected and no trade has ever occurred, therefore resulting in very unlikely that Mediterranean red coral populations could be mainly threatened by harvesting activities. Any unsustainable harvesting pressure or IUU fishing is expected to have mainly local effects, with the loss of red coral banks, occurring in red coral's shallower coastal habitat (i.e. between 50 and 140 meters), and the GFCM has been actively working since years with its member countries to ensure that this will not eventually occur.

5.3. Advances in research

In GFCM, the need to carry out scientific research studies on red coral has emerged on several occasions; only small areas of red coral populations in Italy, France and Spain were object of some studies in the last three decades. The data – such as biomass, recruitment, and mortality rate – necessary to construct a population dynamics model to estimate future resources and landings is almost non-existent and growth rate was studied only in few 'shallow' populations. During the last GFCM Workshop on Red Coral in March 2017 (GFCM, 2017b, c), the experts remarked the urgency of launching a Mediterranean scientific project aimed to fill several knowledge gaps on understanding the different traits of red coral life history, essential knowledge in support of any red coral management measure. The experts agreed on the main features of a research programme on red coral in the Mediterranean Sea: i) clear objectives defined in advance; ii) priority given to the collection of useful data for the provision of advice in support of management, especially as regards adopted GFCM recommendations on red coral, iii) need to combine fishery-dependent (e.g. analysis of catch) and fishery-independent sources of information (e.g. surveys on a multiannual basis) to ensure a regular monitoring of the resource, iv) countries where the commercial harvesting of red coral occurs should be involved while countries where red coral populations are known to exist are invited to participate, v) guidelines to facilitate the harmonization and standardization of data collection protocols should be an output of the programme. It was also agreed that any Mediterranean scientific research programme on red coral should focus on five main aspects:

1. Demography: Studies on size/branching patterns, density, abundance, biomass, recruitment, growth, reproduction, physiology, environmental parameters, habitat and biodiversity.
2. Ecology of red corals: Genetics – connectivity, interactions with other species, effects of pollution, climate change. Use of existing and new genetic markers to investigate the connectivity among red coral populations and the possible effects of fisheries genetic erosion due to harvesting (fisheries genetics) and to assess the geographic origin of red coral (genetic traceability). Connectivity studies could be implemented with numerical simulations of oceanic currents as well as with studies on the larval stage.
3. Stock assessment: Investigation on methodologies for assessing the status of red coral populations, including by compiling historical data.

4. Surveys at sea: Development of large and small-scale bathymetric surveys to map Mediterranean red coral populations using standardized methodologies
5. Stock recovery and restoration: Study of the dynamics of recovery of fished colonies, development of restoration techniques.
6. Socio-economics: Socio-economic analysis of the sector and development of economic indicators. External aspects affecting the fishery.

The implementation of a Mediterranean research programme on red coral in support of the provision of advice of the SAC, with the characteristics above outlined, is specifically mentioned in *Recommendation GFCM/41/2017/5 on the establishment of a regional adaptive management plan for the exploitation of red coral in the Mediterranean Sea,* Paragraph 28, in which the GFCM Secretariat, with the support of the SAC, is called to provide terms of reference, including costs, services and other requirements to support, through a call for tender, the implementation of a research programme on red coral in the Mediterranean Sea, that should be launched in 2018 or 2019.

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APPENDIX

Core elements of the 2014 GFCM Guidelines for the management of Mediterranean red coral populations

[…]

OBJECTIVES

4. Following the *Guidelines on a general management framework and presentation of scientific information for multiannual management plans for sustainable fisheries in the GFCM area*[3], the Guidelines for the management of Mediterranean red coral populations aim to provide elements to maintaining stock size, counteract overfishing (reported to occur in many areas, especially for shallow populations) and prevent it in areas where the resource is not fully exploited, while ensuring long-term sustainable yields.

Operational objectives:

5. Provisional operational objectives (Oob) should be based on the existing binding recommendations of the GFCM (Rec. GFCM/35/2011/2 and Rec. GFCM/35/2012/1), in particular:

- **Oob1**: To control that the legal size limit for harvesting red coral colonies is enforced at the GFCM level;
- **Oob2**: To maintain the same catch level as that of the three previous years in order to keep the fishery working while waiting for a consistent assessment of red coral populations based on sound scientific information.

INDICATORS, REFERENCE POINTS AND DECISION RULES

6. In order to measure management performance in the achievement of objectives, an indicator and corresponding reference points (RP) should be defined for each Oob.

7. Each RP has three associated values:

- **Target reference point** (TRP), corresponding to a situation considered as desirable and to be achieved on average;
- **Limit reference point** (LRP), indicating a situation that is undesirable and to be avoided at all costs;
- **Threshold or Precautionary reference point** (PRP), i.e. a threshold from which initial actions can be taken to reduce the risk of breaking the limit.

8. Specific actions to be taken in order to keep RPs to sustainable levels or to drive them back to the target, shall be decided by each country.

9. In line with point 9 of the above mentioned GFCM Guidelines, targets, thresholds and limit reference points should defined along with a range of potential management actions based on available scientific and socioeconomic data on the resource. However, considering the peculiarity of the red coral resource and the structural lack of reliable and up-to-date data on actual yields and populations status in many areas of the distribution range, it is worth pointing out that the reference points that are frequently used in fisheries management (as advised at points 11–13 of the GFCM Guidelines) can hardly be applied to red coral at present. The reference points proposed in these guidelines reflect the paucity of information and should be regarded as provisional ones. A revision could be made on the basis of SAC advice and GFCM discussions.

[3] Other decisions OTH-GFCM/36/2012/1 in the Compendium of GFCM decisions

10. Each Oob is associated to a decision rule. The decision rule serves to trigger a management action. The action to be taken will depend on the position of the indicator that is relevant to the reference point. The current guidelines leave the selection of those actions up to the countries and advises to use measures that are rated as efficient and take into account the socioecomic impacts of the proposed measures.

Operational objective 1: To control that the legal size limit for harvesting red coral colonies is enforced at the GFCM level

11. The indicator for this objective is the mean size (basal diameter) of landings. The value of the **target** reference point for **Oob1** is proposed to be defined on the basis of the current size limit set by GFCM Recommendations which foresee a 10 percent allowance in live weight for undersized colonies. The **limit** reference point for **Oob1** could defined as double of the TRP, which means that 20 percent of live weight of undersized coral colonies in landings is considered as the limit situation to be avoided at all costs. A **threshold** at 15 percent could be established as an early warning, indicating that the values are approaching of the limit and that actions should be triggered in order to reduce the risk of breaking the limit reference point (LRP).

Table 1: Decision control rules and actions for Oob1 of the regional management plan for red coral

Decision control rules	Actions to be triggered
Percentage of undersized colonies = 0%	▪ No action
0%<Percentage of undersized colonies ≤ 10%	▪ Recommend stricter control
10%<Percentage of undersized colonies ≤ 15%	▪ Recommend stricter control ▪ Survey to evaluate the actual size structure
Percentage of undersized colonies > 15%	▪ Recommend stricter control ▪ Survey to evaluate the actual size structure ▪ Control harvesting ▪ Evaluate the possibility of close the fishing

12. The revision made during the Workshop in Brussels (January 2014) led to a proposal to reduce the values of the three reference points. Consensus was reached to modify the values of the target reference point to a **0** percent allowance in weight of colonies under 7 mm basal diameter; the precautionary (or threshold) level has been set to **10** percent and the limit to **15** percent. No action is needed between 0 and 10 percent, but once this value is reached the actions suggested in the proposal should be triggered.

13. Actions should be triggered when the value of the indicator overpasses the target reference point, and that besides those, in cases where the limit (15 percent) is overpassed, additional actions to control harvesting would be suggested before resorting to the extreme of considering the closure of the fishery.

14. The timeframe and geographical scale of the actions to be taken when the threshold or the limit RPs are overpassed shall be decided by the Members provided that in the annual dataset to be transmitted to the GFCM, the country's total landings comply with the 2012 Recommendation (minimum legal size 7 mm with 10 percent tolerance based on total annual weight). Nevertheless, since stocks distribution is patchy and very local, countries should ensure that this average size will be also respected on a daily (or weekly) basis for all fishing grounds through the establishment of a systematic (daily or weekly) control of catches at ports.

Operational objective 2: To keep red coral harvesting at sustainable levels

15. The indicator for this objective is the value of total catches (landings) in the GFCM area, consistent with the GFCM Guidelines on multiannual management plans. The target reference point for Oob2 shall be the average yield of the three previous years, and precautionary and limit reference points shall be established at an increase of 10 percent and 20 percent in total landings. The target

had been established assuming that average catches for the three previous years (as reported to the FAO global capture database) were at a sustainable level, but in several occasions it was pointed out the need to base Operational objective 2 on formal stock assessments (e.g. maximum sustainable yield models, etc.) under the supervision of SAC as long as data become available to perform such exercise. Moreover, it was noted that many fisheries were data poor and not ready for such an analysis. In this regard, the urgent need to collect catch data and transmit them to the GFCM Secretariat was recognized. The importance of fisheries independent data coming from scientific surveys that are less affected by a subjective selectivity of fishers towards large colonies was stressed.

16. Given those evidences, the implementation of Oob2 could be adopted at a second stage. The proposed framework for an adaptive revision over a three-year period has been seen as feasible and convenient in order to collect the necessary data and implement, in the future, a more detailed and efficient management system relying on Oob2 as well. Notwithstanding, it was also recommended that countries adopt it gradually to the shortest delay if scientific data at the national level would allow doing so.

FISHERIES MANAGEMENT MEASURES

17. According to the recommendations in force, the following technical management measures are currently applied in the whole region (Table 2).

Table 2: Technical measures already in force in existing GFCM recommendations

Management tools	Current measures at the regional level
Depth restrictions	Prohibition to collect coral at depths shallower than 50 m
Gear restriction	The only permitted gear is manual hammer by scuba diving
Minimum landing size	7 mm basal diameter (only 10% of tolerance in weight is allowed for undersized colonies)

18. Other potential measures that could be applied are presented in Table 3.

Table 3: Ranking of potential technical measures to be adopted by countries to limit effort and catches

Management tools	At the regional level	Rank effectiveness	Rank feasibility	Comments
Limits to fishing capacity	Licensing systems	High	High	
Limits to catches	Individual annual quotas	High	High	Based on scientific data
	Individual daily quota	High	High	Might force divers to do more trips per year, but allows efficient inspections
Spatial restrictions	License restricted to certain areas	High	High	
	Establish *refugia*, or permanently closed zones	High	High	Deep populations found in virgin status should be kept as *refugia* and specific MPAs to protect red coral could be established
Temporal restrictions	Seasonal harvest restriction	High	High	Facilitates control of effort
	Rotation periods in different banks	Medium	Medium	Recovery rates of red coral are low – coral needs 25 to 30 years to reach the minimum legal size – and geographical variation is not well known.

FISHERIES MONITORING, CONTROL AND SURVEILLANCE (MCS)

19. To ensure compliance with the measures to be adopted in the management plan, concerned Members shall be responsible for implementing the adopted management measures in their jurisdictional waters.

20. Control and surveillance should be provided by national authorities. The list of MCS measures assessed in terms of effectiveness and feasibility is presented in Table 4 and may be used as a guidance for national authorities.

[…]

IMPLEMENTATION AND ENFORCEMENT MECHANISMS

24. Members should take measures to ensure that the provisions of any management plan for red coral are covered under their national legislation. The implementation and rule enforcement mechanisms of the management plan should be defined through legislation and regulations at the national level, taking into account the specificities of the national legal frameworks as well as economic, social, and cultural aspects.

[…]

ANNEX 2: Asian regional report on the biology, fishery and trade of precious and semi-precious corals

Nozomu IWASAKI,
Professor, Faculty of Geo-Environmental Science,
Rissho University, Japan

CONTENTS

III Possibilities and proposals for sustainable exploitation

I PRECIOUS CORALS

1. SPECIES TARGETED BY FISHERY

Among coral species – i.e. species that belong to the Phylum Cnidaria and have a skeleton –, the term "precious corals" collectively describes those whose skeletal axis is used as a gemstone. Eight species of the subclass Octocorallia, family Coralliidae, including one undescribed species, are currently used in this way. Four of these – Japanese red coral (aka), pink coral (momo), white coral (shiro), and miss coral – inhabit Asian waters. Three other species, one of which has yet to be given a scientific name, are found in the coastal waters of Hawaii and around Midway Atoll. These three are no longer collected commercially although they have been fished in the past (Table 1).

Tu *et al.* (2012) recorded two new species of Coralliidae that were collected in Taiwanese coastal waters: *Pleurocorallium carusrubrum* and *Hemicorallium taiwanicum. P. carusrubrum* was collected in a designated coral-fishing sea area north of Keelung at a depth of 156 m. Due to its pink axis, it is reported to be circulating on the market as pink coral (Jeng, 2015). It is a matter of concern that *H. taiwanicum* was collected southwest of Taiwan Province of China at depths of 736–1040 m (outside the designated coral-fishing areas).

Corals other than Coralliidae that are used as gemstones are described as semi-precious corals. These will be discussed later (See Section II, 1.).

2. RESEARCH ON BIOLOGY AND ECOLOGY

2.1. Taxonomy

The red coral that inhabits the coastal waters of Japan and Taiwan Province of China was previously identified as a member of the genus *Paracorallium*, while pink coral and white coral were classified along with Mediterranean red coral as *Corallium* (Bayer and Cairns, 2003). However, it was proposed that this classification be reconsidered since one of the morphological characteristics that distinguishes *Paracorallium* – i.e. deep axial pits with prominently beaded margins located under the autozooid polyps – is not always found in Japanese red coral (Iwasaki and Suzuki, 2010). Recent advances in gene analysis enabled Uda *et al.* (2011, 2013) to examine the mitochondrial genome sequences of the four species and confirm that, while pink coral and white coral share the same gene order arrangement, that of Japanese red coral is identical to the gene order arrangement of the Mediterranean species. Tu *et al.* (2015) subsequently distinguished 3 genera of Coralliidae (*Corallium, Hemicorallium, Pleurocorallium*) based on molecular phylogenetic analysis of Coralliidae in general using eight mitochondrial genes and a region of the nuclear genome. According to this gene-based classification system, both Japanese and Mediterranean red corals are now classified as *Corallium* (*C. japonicum* and *C. rubrum* respectively), while pink coral and white coral are classified as *Pleurocorallium* (*P. elatius* and *P. konojoi* respectively).

Due to the difference in their distribution and size, there are two known varieties of both *P. elatius* and *P. konojoi* (Nonaka *et al.*, 2006; Nonaka and Muzik, 2010). The results of the molecular phylogenetic analysis performed by Tu *et al.* (2015) on these species also indicated that neither of them is monophyletic: the two species are intermingled. For this reason, the classification of these species needs to be reexamined from both morphological and genetic points of view.

No taxonomic records are kept of the deep-sea corals collected by Japanese and Taiwanese fishing boats around Midway and the Emperor Seamounts. Undescribed species have been circulating on the market for many years. This problem requires urgent attention since identification of species is imperative for their protection and the maintenance of resources. Furthermore, in a survey conducted by remotely operated vehicles (ROVs), Coralliidae sp. – a species of Coralliidae with the same axial composition as precious corals – was collected from the waters around the Ogasawara Islands at a depth of 1 420–1 620 m (Hasegawa *et al.*, 2010). Coralliidae sp. has also been observed around Koko Seamount (Emperor Seamounts) at a depth of 425 m (Fisheries Agency of Japan, 2008). Taxonomic studies are needed on the Coralliidae spp. that inhabit these deep-sea areas.

Finished coral products circulating on the market are differentiated according to the colour of their axis. However, each species exhibits a wide colour variation, which makes it difficult to identify species based on colour alone. For example, Japanese red coral may be anywhere from pale pink to dark red (Table 1). Identification methods using the DNA and trace elements present in the axis therefore need to be established.

2.2. Distribution and density

Japanese red, pink and white corals are distributed from the coastal waters of southern Japan to the northern South China Sea. They inhabit Japanese and Taiwanese coastal waters at depths of around 80 m to 320 m (Table 1). Nonaka and Muzik (2009) report that all three species live at greater depths, and pink coral colonies are larger, in the southern sea areas of the Ryukyu Archipelago (Amami, Okinawa, Ishigaki) than in the sea areas further north (off Kagoshima). However, since the specimens used in that particular study were collected by ROVs for commercial purposes, the findings do not necessarily reflect natural conditions.

C. japonicum is also known to inhabit the northern Philippines, while *P. elatius* and *P. konojoi* can be found in the coastal waters of Vietnam. However, no taxonomic research has been conducted on the colonies living in these sea areas. Taxonomic studies are needed in order to clarify distribution areas and identify variations within the species.

Until recently, *Hemicorallium sulcatum* (miss coral) was known to inhabit only the coastal waters of the Boso Peninsula (Chiba Prefecture, Japan). However, huge quantities of coral collected in Taiwanese waters in the 1980s and 90s have now been identified as *H. sulcatum* (Tu *et al.*, 2012). Close examination is therefore required to investigate the possibility that miss coral is also being collected in Japanese waters.

Four species of precious coral produced in China are listed in CITES Appendix III. One of these, *Pleurocorallium secundum* (listed in CITES Appendix III as *Corallium secundum*), is found in Hawaiian coastal waters and around Midway Atoll (Grigg, 1974). It is reported to have been collected off Lanyu Island (Taiwan Province of China) (Huang and Ou, 2010) (Fisheries Agency, Council of Agriculture, Executive Yuan, Taiwan Province of China), however further investigation is required since no taxonomic studies have been reported.

Table 1. Characteristics of precious coral species

Species	Common name	Distribution	Depth range (m)	Form and size of axis	Color of axis	Fishing method	CITES Listing
Corallium rubrum	red coral	Mediterranean, Eastern Atlantic from Portugal to Senegal	10 to more than 1000 mainly 30–120	Fan-shaped, sometimes tree shaped;25 cm in height, maximum 50cm	Uniformly red	SCUBA diving	
Corallium japonicum	Japanese red coral	Southern Japan to north South China Sea and Philippines	75–300	Fan-shaped; 30cm in height	Whitish pink to dark red, with white spot in the center of the cross-section of axis	Tangle net, ROV	III (China)
Pleurocorallium elatius	pink coral**	Southern Japan to north South China Sea and Vietnam	100–320	Fan-shaped; 90cm in height, maximum 110cm	Whitish pink to faint red, with white spot in the center of the cross-section of axis	Tangle net, ROV	III (China)
Pleurocorallium konojoi	white coral	Southern Japan to north South China Sea and Vietnam	76–300	Fan-shaped; 30cm in height	White	Tangle net, ROV	III (China)
Hemicorallium sulcatum	miss coral pink coral	Japan (Bosō) and Taiwan Province of China	400–450	Fan-shaped; 24cm in height	Pink	Tangle net, harvest in Taiwan Province of China	
Coralliidae sp.*	deep-sea coral mid coral	Midway and Emperor Seamount Chain	900–1500	Fan-shaped	Pink is marbled and sometimes flecked with red	No harvest at the present time	
Hemicorallium regale	pink coral	Hawaii and Midway	350–600	Fan-shaped	Dark pink to red	No harvest at the present time	
Pleurocorallium secundum	deep-sea coral mid coral	Hawaii and Midway	350–475	Fan-shaped; 75cm in height	Pale pink	No harvest at the present time	III (China)

* Undescribed species

** Pink corals collected in Taiwan Province of China may include *Pleurocorallium carusrubrum* (see text, section I, 1)

In order to establish the distribution density of precious corals in Japanese waters, ROVs were used to investigate 20 sites in Kochi, Kagoshima and Okinawa prefectures from 2009 to 2012. According to the research, the average distribution density of red, pink and white corals was 0.45 ± 0.71 colonies/100 m^2, 0.19 ± 0.69 colonies/100 m^2, and 0.07 ± 0.17 colonies/100 m^2 respectively, with an average of 0.74 ± 1.10 colonies/100 m^2 for the three species combined. Abundant pink and white coral beds were also discovered, with distribution densities of 4.79 colonies/100 m^2 and 1.72 colonies/100 m^2 respectively (Iwasaki, unpublished).

Despite the wide distribution area of precious corals, investigations of distribution density to date have been limited to pinpoint surveys, which are insufficient in order to obtain the biomass data required for adequate resource management. An ROV was used to investigate the distribution of precious corals in Tosa Bay, Kochi Prefecture, in 2012, but the results of the survey have not been disclosed. Coral fishing trials were begun in Wakayama Prefecture in 2014 (See Section I, 3.3.1.), but no data concerning fishing areas or landings is available to the public. Furthermore, large-scale poaching was rampant around the Ogasawara Islands in 2014, but, since no distribution data exists for the period prior to the poaching incidents, it is impossible to estimate the extent of the damage (See Section I, 3.3.4.). Surveys concerning the distribution of precious corals (including density and biomass) are essential for the protection of these species, so it is desirable that each sea area be investigated without delay.

2.3. Population structure and sustainable exploitation

Iwasaki *et al.* (2012) investigated the age composition of red coral at a depth of 200 m off the Amami Islands in Kagoshima Prefecture, Japan. In a sea area that had never been fished, colonies estimated to be 20–50 years old were observed, as well as a few aged 60–70 years. Meanwhile, in a nearby sea area, colonies aged 30–50 years had been collected selectively by an ROV, leaving only smaller ones of 10–20 years. This indicates that Japanese red coral can reach a harvestable size if fishing is suspended for at least 10–20 years after each operation (Figure 1). The study demonstrated that sustainable fishing can be achieved by establishing non-fishing periods and implementing selective fishing methods using ROVs to collect only colonies of a certain size.

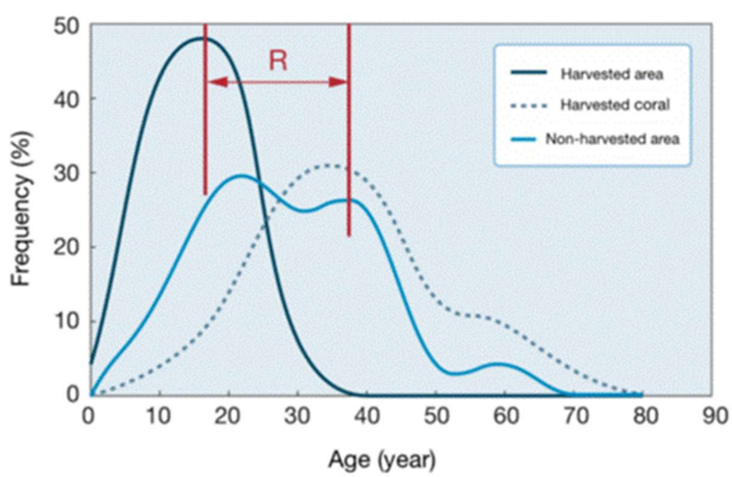

Figure 1. The size–age structure of Japanese red coral in a harvested and a non-harvested area off the Amami Islands, southern Japan: The size peak of corals harvested coincides with that of corals in the non-harvested area, indicating that corals of that size are not caught in the non-harvested area. Meanwhile, the harvested area has no corals of that size, demonstrating that it would take younger colonies 10–20 years to reach the harvestable size (R). (cited from Iwasaki web page)

2.4. Axis formation and growth rate

Computed tomography was used to investigate the early stage of axis formation in Japanese red, pink and white corals (Urushihara *et al.*, 2016). Computed topography is a computation process that enables three-dimensional reconstruction of an image from transmitted X-ray images of numerous specimens obtained using high-penetration X-rays.

This enabled researchers to observe the formation of a large protrusion where sclerites had adhered to the tip of the axis. Smaller protrusions were also seen on the surface of the axis tip. These observations supported the axis formation hypothesis obtained from research on Mediterranean red coral that the aggregation of sclerites and their adherence to the tip of the axis is concurrent with the formation of the surface (Allemand, 1996).

Three methods have been developed to estimate the growth rate of Japanese precious corals: high-precision analysis of the colours of the growth rings in a transverse section of the axis using a high-resolution digital microscope (Luan *et al.*, 2013); analysis of the two-dimensional distribution of infrared absorption spectra using high-luminance infrared synchrotron radiation micro-beams (Iwasaki *et al.*, 2014); and age determination using the naturally occurring radionuclide lead-210 (^{210}Pb) (Hasegawa and Yamada, 2010). Table 2 shows that similar results were obtained with the different methods.

Observation often reveals that the growth rings present in a transverse section of the axis are off centre: they are larger on one side than on the other. When a white coral sample exhibiting this phenomenon was analysed using the lead-210 dating method, different growth rates were observed on opposite sides of the section, one side being 0.13 mm/year and the other 0.33 mm/year (Yamada, unpublished).

If we assume the average growth rate of the axis to be 0.3 mm/year in diameter, it would take around 50 years to achieve an axial diameter of 15 mm. Since some pink coral colonies living on the ocean floor have an axial diameter of over 50 mm, it must have taken them 170 years at least to reach that size.

Table 2. Radial growth rates of Japanese precious coral species

Species	Common name	Growth ring method (mm/year)	Synchrotron radiation-infrared spectroscopy method (mm/year)	Pb-210 method (mm/year)
Corallium japonicum	Japanese red coral	0.10–0.14	0.110	
Pleurocorallium elatius	pink coral	0.15	0.147	0.15
Pleurocorallium konojoi	white coral	0.22	0.109	0.13 (radius of minor axis) 0.33 (radius of major axis)
References		Luan et al., 2013	Iwasaki et al.,2014	Hasegawa and Yamada, 2010 Yamada, unpublished

2.5. Reproduction

The corals in Japanese and Mediterranean waters differ in features such as the location of their gonads and the part of their body where fertilization occurs. In Mediterranean red coral the gonads are formed in the autozooids. Fertilization takes place in the autozooids of female colonies, after which planula larvae develop and are released into the sea. By contrast, in Japanese red, pink and white corals, the gonads are formed in the siphonozooids. In Japanese red coral, fertilization does not take place in the siphonozooids: the oocytes and sperm are discharged into the sea (Nonaka *et al.*, 2015; Sekida *et al.*, 2016). Furthermore, planula larvae have not been observed in Japanese precious corals.

Sekida *et al.* (2016) developed a method of using a liquid chelating agent to decalcify the Japanese red coral axis and sclerites, rendering the coenenchyme semi-transparent and enabling observation of the gonads. This method was used to measure the gonad size of red coral samples collected monthly from 2006 to 2014 from the coastal waters of Kochi, researchers were able to establish that the gonads are formed during the period from March to May. Also, since the number of colonies with gonads was seen to decrease sharply from July onwards, the eggs and sperm are thought to be released all at once in July. From March 2012, in response to an interim report of these findings, the government of Kochi Prefecture has prohibited coral fishery in Kochi during the months of June and July in order to protect precious corals during the spawning season (See Section I, 3.3.1.).

The reproduction season of Japanese red, pink, and white corals around Okinawa is estimated to be from May to August (Nonaka *et al.*, 2015). On the other hand, while the exact location is unknown, Kishinouye (1904) reports that the eggs and seminal vesicles of mature red coral colonies collected in March were larger and more numerous than those of colonies collected in September. These findings call for close examination of each sea area since they indicate that the timing of the reproduction season may differ from place to place. In addition, effective resource management requires data concerning the age at which corals attain sexual maturity and the correlation between colony size and maturity rate, but these parameters are still unknown.

2.6. Genetic variation

In order to conserve each precious coral species, the genetic diversity of the species must be ensured. For this reason, research was conducted on red coral (*C. japonicum*) collected from Japanese coastal waters. Since mitochondrial genomes generally evolve more quickly than nuclear genomes, they are used as DNA markers to detect genetic diversity. However, the mitochondrial genomes of octocorals, including *C. japonicum*, are known to evolve more slowly than their nuclear genomes. This makes it impossible to detect genetic diversity within a species when only a short region of the mitochondrial DNA is compared. Uda *et al.* (2011, 2013) therefore analysed the red coral samples by comparing a region amounting to 38 percent of their whole mitochondrial genome. Not only were eleven different types of mitochondrial DNA detected, but these were found to be evenly distributed over a wide sea area rather than localised in patches, indicating that the genetic diversity of *C. japonicum* is assured to a certain extent.

Next generation sequencing (NGS) is currently being used to obtain single nucleotide polymorphisms (SNPs) from *C. japonicum* in order to analyse the genetic diversity of coral populations in each sea area and the gene flow between them (Iwasaki, Uda, Iguchi and Suzuki, unpublished). It is hoped this research will be useful both in maintaining genetic diversity and in selecting sea areas for protection in order to ensure adequate sources of larvae.

2.7. Trace components in the axis

Accurate species identification of raw materials is essential in order to monitor and control the trade of precious corals. The red coral and pink coral collected in Japanese and Taiwanese waters has a white spot in the centre of a transverse section of the axis, which enables identification of its habitat. However, it is impossible to differentiate between processed items in which the white spot has been removed. Research was therefore conducted in order to identify species and locality by analysing trace elements found in the precious coral axis. Using inductively coupled plasma atomic emission spectrometry (ICP-AES) to analyse trace elements found in the axis, Hasegawa *et al.* (2012) pointed out values in magnesium/calcium (Mg/Ca) and barium/calcium (Ba/Ca) ratios that were characteristic of their locality, thus establishing that it is possible to distinguish between precious corals from Japanese coastal waters, the Mediterranean and Midway. However, no inter-species variation was detected in the composition ratios, so this method did not enable differentiation between different species from the same location.

Since the previously described method involves grinding and dissolving specimens, it is not suitable for the analysis of valuable processed items. For this reason, other methods of analysis that do not damage the specimen – such as synchrotron radiation X-ray fluorescence spectrometry (SR-XRF) and attenuated total reflection (ATR) using Fourier transform infrared spectroscopy (FT-IR) – have been tried, but none of these has enabled identification of either habitats or species (Hasegawa *et al.*, 2010; Iwasaki *et al.*, 2014). A non-destructive technique that permits identification of species and habitats therefore needs to be established. Research on the DNA contained in the organic matter in the axis is also essential in order to differentiate between species.

Regarding the trace elements and organic matter present in the axis, research has also been conducted on the correlation between these and the coral's growth rings, environment and colour (Luan *et al.*, 2014; Yoshimura *et al.*, 2011; Tamemori *et al.*, 2014; Yoshimura *et al.*, 2015; Yoshimura *et al.*, 2017).

2.8. Artificial breeding

The ability to farm precious corals and conduct various experiments on them would increase our knowledge of their biological characteristics and ecology. Research to establish breeding techniques is now underway at the Marine Ecology Research Institute in Japan (Iwasaki, unpublished).

Since precious corals are slow-growing species, even if they could be farmed, it would be unrealistic to breed and hope to raise them in a facility on land. Researchers are therefore attempting to propagate precious corals by creating an environment conducive to larvae settlement in their natural habitat.

Various tiles made of vinyl chloride or ceramics were placed for over a year in a sea area inhabited by precious corals to see whether larvae would settle on them. (Figure 2, Iwasaki webpage). Unfortunately, no precious coral larvae settlement was observed, but numerous hydrozoan colonies were found to have settled on the vinyl chloride tiles. Since no other hydrozoan habitation has been observed in the area, it is clear that artificial plates can promote the settlement of coral larvae (Iwasaki, unpublished). Kochi Coral Fisheries Association, an alliance of coral fishermen in Kochi Prefecture, is experimenting with attaching Japanese red coral fragments to an artificial substratum and placing it on the sea floor (The Kochi Shimbun, 2016).

Figure 2. Left: Settlement plates for larvae set close to red coral colonies. Right: Hydrozoan coral colonies on a vinyl chloride tile (cited from Iwasaki web page).

3. FISHERY

3.1. Fishing methods

The methods currently used for coral fishery are tangle nets and ROVs, although manned submersibles have been used in the past in Japan and Vietnam.

Coral tangle-nets are traditional fishing tools that have been used in Japan since coral fishery began there in 1871. In Kochi today, several coarse-mesh nets of just over 1 m in length are attached to either a stone weight (30-plus kg) (Figure 3) or a chain weight. When stone weights are used, three stones are tied to a rope, with five nets attached to each stone. In the case of a chain weight, four or five nets are attached at regular intervals to a horizontal bar. These are dragged along the sea floor, and corals are entangled in the nets. This method does not allow selective fishing of precious corals according to either species or size.

Since tangle nets disturb the top layer of the sea floor, they affect deep-sea life and their habitat. Iwasaki conducted a survey to investigate the extent of the impact. In a dragging experiment conducted at a depth of 30–40 m, a chain-weight net was observed to move in a linear fashion across the sea floor with the attached nets remaining closed (Figure 4). In another experiment, using a stone-weight net at a depth of 10 m, the scratch marks left by the net on the sea floor were no longer visible after eight days. The impact of this net was also studied by comparing the number of meiobenthos present in the sediment before and after the dragging experiment. There were fewer copepods in the top 1 cm of the sediment both immediately after the experiment and the following day, but they had recovered 14 days later. As for nematodes, while they had recovered on the surface after 14 days, the impact of the experiment was still evident at a depth of 3 cm (Iwasaki, unpublished). Although the impact of a single operation is localised and limited, when multiple fishing boats drag nets repeatedly over a small area, the impact and time needed for recovery are expected to be considerable.

ROVs enable fishermen to select the species and size of the corals they collect. On the other hand, unless restrictions are imposed, there is a danger that small, immature colonies and those living in places that are inaccessible to tangle nets may be fished to depletion.

Figure 3. Coral net using a stone weight (cited from Iwasaki web page).

Figure 4. A coral net using a chain weight being dragged, each small net remaining closed throughout the experiment (cited from Iwasaki web page).

3.2. Fishing grounds and methods

3.2.1. Japan

Coral fishery exists in the following Japanese sea areas: off Hahajima in the Ogasawara Islands (Tokyo-to); the Kii Peninsula, Wakayama (Wakayama Prefecture); Cape Ashizuri, Cape Muroto (Kochi Prefecture); Uwajima (Ehime Prefecture); the Goto Islands (Nagasaki Prefecture); the Uji

Islands, Mishima, Tanegashima, Yakushima, Toshima, the Amami Islands (Kagoshima Prefecture); and the Ryukyu Islands (Okinawa Prefecture).

Operations are conducted by ROVs in Kagoshima and Okinawa Prefectures and by coral tangle-nets in the other areas. See Section I, 3.3.1. for the numbers of licences and fisheries in operation.

3.2.2. Taiwan Province of China

According to "Regulations Governing Fishing Vessels that Also Engage in Coral Harvesting" coral fishing is permitted in five sea areas – northeast of Keelung, east of Yilan County, south of Orchid Island, south of the Hengchun Peninsula, and west of Kaohsiung (Figure 5) – and licences are issued to a maximum of 60 fishing vessels.

The same type of coral nets are used in Taiwan Province of China as in Japan, but the scale of operations is larger in Taiwan Province of China. In Kochi, Japan, fishing boats of up to 20 tonnes operate with one net per boat, while in Taiwan Province of China boats of 20–29 tonnes (about 4–6 fishermen) operate with 6–12 nets per boat (Chen, 2012, 2015).

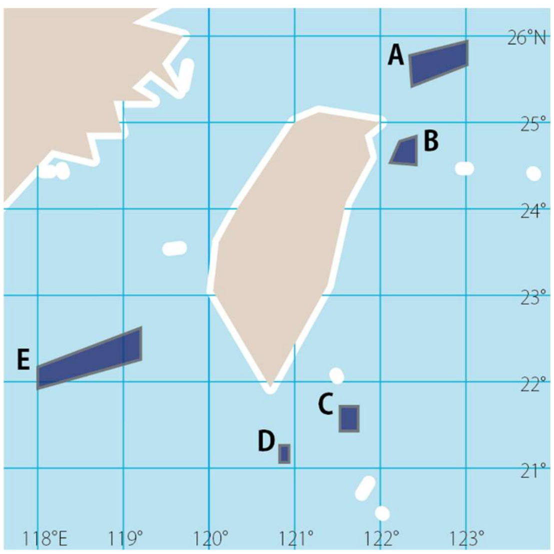

Figure 5. The five authorised fishing grounds for Taiwan Province of China's precious coral fisheries. The location data of the fishing grounds referred to Huang and Ou, 2010.

3.2.3. China

Precious coral fishery is prohibited in Chinese waters. However, Chinese production of Japanese red, pink and white corals is recorded in the FAO Global Production Statistics (See Section I, 3.3.3.).

A pink coral bed was found off Hong Kong SAR in 1963, and this led to the discovery of a large fishing ground for pink coral, which was fished by Japanese and Taiwanese boats from 1964 to around 1971.

3.2.4. Vietnam

In 2008, a manned submersible was used for coral fishing in the waters south of Ho Chi Minh, and the following year 135 kg of precious corals were traded at a bid market in Japan (The Kochi Shimbun, 2009). According to a poll conducted by coral traders, the submersible is currently inoperative and no longer in use.

3.2.5. Southeast Asia

Coral fishing is prohibited in the Philippines. In 1971, a Japanese fishing boat discovered precious coral in the northern Philippines and this was fished for a time. According to Wells (1981), there were abundant precious coral beds off the Batan Islands and these were poached by Japanese and Taiwanese fishing boats.

There are records of Japanese fishery operations around Saipan and Palao in 1963, but the details are unknown.

3.2.6. Midway Atoll

The discovery of precious coral around Midway Atoll by a Japanese fishing boat in 1964 (or 1965 according to one source) triggered the beginning of coral fishery there. Taiwanese boats later moved in and coral fishery thrived for a while, but profits eventually declined to the point that operations came to a halt around 1990. However, at least two Taiwanese coral fishing boats were sighted in 2008 (Fisheries Agency of Japan, 2008), and the FAO Global Production Statistics report landings of 0.4 tonnes for China in 2012. Further investigation is therefore required since it is possible that fisheries are still operating in this sea area.

3.3. Fishery management

3.3.1. Japan

Coral fishing in Japan is regulated by each prefecture, and licences are issued by the respective governors. Coral fishing is currently prohibited by nine regional authorities (Chiba, Kanagawa, Shizuoka, Aichi, Mie, Tokushima, Miyazaki, Oita, and Kumamoto Prefectures) and permitted by seven regional authorities (Tokyo metropolitan area, Wakayama, Kochi, Ehime, Nagasaki, Kagoshima and Okinawa Prefectures). Licensing requirements and fishing methods are regulated by each prefecture's fishery adjustment rules and instructions from the respective sea area Fishery Adjustment Committees.

The regulations imposed by local authorities are governed by policy framework at the national level. In 2015, the Fisheries Agency of Japan (FAJ) issued technical advice to the local authorities for management of domestic precious coral fishery based on the Local Autonomy Act. The relevant prefectural governments were requested to take measures to reinforce appropriate management of precious coral fishery, including the following:

1) limiting fishing effort to current levels
2) clearly designating fishing zones in order to conserve precious corals in non-fishing zones (while avoiding clear designation of non-authorised zones since this might provoke poaching activities)
3) monitoring through VMS, GPS, etc. and maintaining records of the movements of authorised fishing vessels
4) ensuring accurate catch reporting based on each prefecture's fishery adjustment rules

In Ogasawara (Tokyo), adding to the public regulations, restrictions were self-imposed by the local fishermen in 2014. For this reason, the regulations differ among regional governments (Table 3). In Kagoshima and Okinawa Prefectures, non-selective fishing using tangle nets is prohibited, but selective fishing using ROVs is allowed.

In Kochi Prefecture, fishing regulations were tightened from 2012 in response to mounting global consensus on the need to protect precious corals. While fishing in January and February had already been banned, research on spawning seasons (Sekida *et al.*, 2016) prompted the local government to extend the ban to include the spawning season of June and July. Limits were also set on fishing zones, catch quotas (up to 0.75 tonnes of live coral a year), fishing hours and coral size. Although the number of authorised fishing boats increased (See Section I, 4-3-4), stricter limits were set on the fishing periods and areas so that the total fishing effort remained unchanged.

While on one hand concerted efforts are being made to tighten regulations, soaring precious coral prices (See Section I, 4.3.2.) have triggered a revival of coral fishery. The number of fishing boats using coral nets has continued to grow. In 2012, coral fishing was resumed off the Ogasawara Islands after a gap of 10 years, and sales volume has also increased in correlation with rises in market prices (Figure 6). In Wakayama Prefecture, where coral fishery had never existed before, 50 vessels were authorised to conduct trial operations in 2014, and in 2015 fishery permission was granted to a company jointly managed by 12 fishery cooperatives. This led to 152 registered vessels, with a limit of 50 vessels operating per day. Currently, the number of licences is 359 in Kochi (2018), 24 in Tokyo (eight in operation, 2016), five in Nagasaki (one in operation, 2017), four in Ehime (2017), one in Wakayama (actually 123 boats, 2017), one in Kagoshima (ROV, 2017), and three in Okinawa (one in operation, ROV, 2017) (Agriculture, Forestry and Fishery Division, Tokyo Metropolitan Government, 2017; public hearings for each prefecture, websites, etc.).

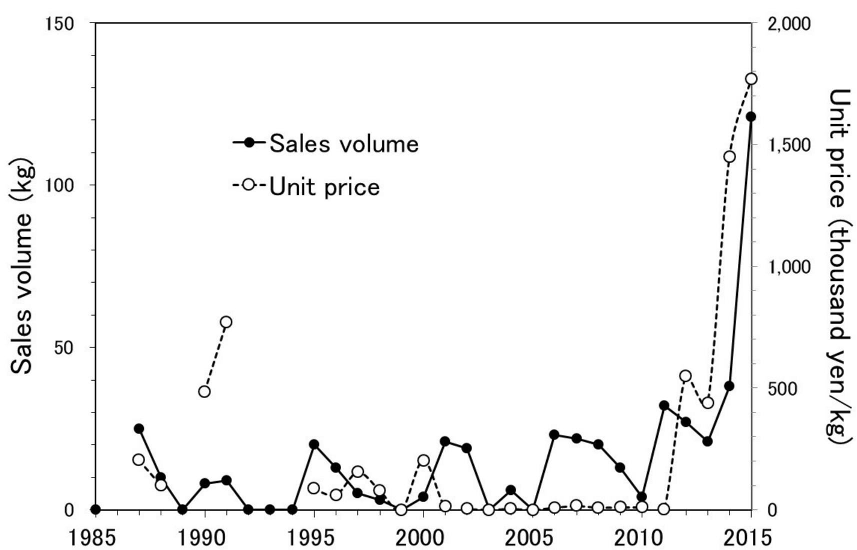

Figure 6. Changes in sales volume and average unit prices for precious corals collected off Ogasawara Is., 1985–2015 (Source: Agriculture Forestry and Fishery Division, Bureau of Industrial and Labor Affairs, Tokyo Metropolitan Government, 1987–2017). Note: Data includes reef-building corals.

Table 3. Regulations of coral fishing by local governments in Japan (public hearings for each prefecture, websites and The Fisheries Agency of Japan)

		Chiba	Tokyo	Kanagawa	Shizuoka	Aichi	Mie	Wakayama	Tokushima
Regulations	**Coral fishery**	Prohibited except for research purposes	License system	Prohibited except for research purposes	Prohibited except for research purposes	Prohibited from 1 June 2017 to 31 August 2018 (revised every year)	Prohibited except for research purposes	License system	Prohibition from 1 April 2017 to 1 March 2018, expect purposes of research (revised every year)
	Species	Live and dead bodies of Japanese red coral, pink coral and white coral		Live and dead bodies of Japanese red coral, pink coal and white coral	Coral	Live and dead bodies of Japanese red coral, pink coal and white coral	Live and dead bodies of Japanese red, pink and white corals	Japanese red, pink and white corals	Live and dead bodies of Japanese red, pink and white corals
	Fishing method		Tangle net (length of beam; less than 7 m, number of beam; 1)					Using a set of tangle nets (length of a tassel net; 1.5 m or less, length of beam; 5 m or less) only per a vessel. Prohibition of trawling by power	
	Fishing vessels		Less than 20 tons						
	Number of licenses		24 (8 in operation, 2016)					1; actually 152 vessels,2017 (see text section I,3-2-1)	
	Operationg period		From 1 April to 31 March except for 1–30 June; From sunrise to sunset					From 1 September to 31 January; From sunrese to sunset	
	Operating area							Restricted	Restricted
	Quantity of catch							Limit of 150 kg live corals per one operating period	
	Size of catch							Release of live coral less than 3 cm in length and 7 mm in diameter attached to rock	
	Recording							Recording of ship track by GPS, fishing points and catch volume	
	Reporting		Yearly reporting of days of operation, quantity of catch and sale and value of sale					Submission of the above record within 10 days	
	Unintended fishing								
	Penalty	Violation of instructions from the Fishery Adjustment Committee: imprisonment of one year or less, a fine of 500 000 yen or less, detention or petty fine	Unlicensed fishery: imprisonment of three year or less or a fine of 2 000 000 yen or less Violation of condition on license: imprisonment of six months or less, a fine of 100 000 yen or less or cumulative imposition of the foregoing	Violation of instructions from the Fishery Adjustment Committee: imprisonment of one year or less, a fine of 500 000 yen or less, detention or petty fine	Violation of instructions from the Fishery Adjustment Committee: imprisonment of one year or less, a fine of 500 000 yen or less, detention or petty fine	Violation of instructions from the Fishery Adjustment Committee: imprisonment of one year or less, a fine of 500 000 yen or less, detention or petty fine	Violation of instructions from the Fishery Adjustment Committee: imprisonment of one year or less, a fine of 500 000 yen or less, detention or petty fine	Unlicensed fishery: imprisonment of three year or less or a fine of 2 000 000 yen or less Violation of condition on license: imprisonment of six months or less, a fine of 100 000 yen or less or cumulative imposition of the foregoing	Violation of instructions from the Fishery Adjustment Committee: imprisonment of one year or less, a fine of 500 000 yen or less, detention or petty fine
Disclosure of information	**Number of licenses**		Open					Open	
	Number of operating vessels		Open						
	Quantity of catch		Quantity of catch including reef-building corals: Not open to the public Sale volume and price including reef-building corals: Open					Not open to the public	Not open to the public

		Kochi	Ehime	Miyazaki	Oita	Kumamoto	Nagasaki	Kagoshima	Okinawa
Regulations	**Coral fishery**	License system	Prohibition expect purposes of research and fisherman who continues to coral fishing before 2015	Prohibition expect purposes of research	Prohibition from 1 January 2018 to 31 December 2018 except purpose of research (revised every year)	Prohibition in Amakusa water	License system	License system	License system
	Species	Japanese red, pink and white corals	Live and dead bodies of Japanese red, pink and white corals	Live and dead bodies of Japanese red, pink and white corals	Live and dead bodies of Japanese red coral, pink coral and white corals	Live and dead bodies of Japanese red coral, pink coral and white corals	Japanese red, pink and white corals	Japanese red, pink and white corals	Japanese red, pink and white corals
	Fishing method	Using a set of tangle net, prohibition of trawling by power Limit of length of main rope (Muroto, within 300m; Ashizuri, within 375 m)	Tangle net					Selective harvesting method	Selective harvesting method such as ROV
	Fishing vessels	Vessel with less tahan 20 tons	Power vessel with less than 5 tons						
	Number of licenses	359 (March 2018)	4 (2017)				5 (1 in operation, 2017)	1 (2017)	3 (1 in operation, 2017)
	Operationg period	From 1 March to 31 May and from 1 August to 31 December; From sunrise to 3 pm	From 1 March to 31 May and from 1 August to 31 December; From sunrise to 3 pm						
	Operating area	Restricted （Muroto, Ashizuri), 200 m or less in depth	Restricted （Uwa)			Prohibition in Amakusa water	Restricted	Restricted (Uji, Mishima, Tanegashima, Yakushima, Toshima, Amami), 200 m or less in depth	
	Quantity of catch	Limit of 750 kg live corals per one operating year							
	Size of catch	Release of live coral less than 3cm in length and 7 mm in diameter with stone							
	Recording	Recording ship track by GPS							
	Reporting	Report on live coral catch every month and report on coral sale every year	Monthly report on fishing area and quantity of catch				Monthly report on quantity of catch	Yearly report on quantity of catch	Report on catch and sales volumes
	Unintended fishing		Prohibition of the possession and sale	Prohibition of the possession and sale	Prohibition of the possession and sale				
	Penalty	Unlicensed fishery: imprisonment of three year or less or a fine of 2 000 000 yen or less Violation of condition on license: imprisonment of six months or less, a fine of 100 000 yen or less or cumulative imposition of the foregoing	Violation of instructions from the Fishery Adjustment Committee: imprisonment of one year or less, a fine of 500 000 yen or less, detention or petty fine	Violation of instructions from the Fishery Adjustment Committee: imprisonment of one year or less, a fine of 500 000 yen or less, detention or petty fine	Violation of instructions from the Fishery Adjustment Committee: imprisonment of one year or less, a fine of 500 000 yen or less, detention or petty fine	Violation of instructions from the Fishery Adjustment Committee: imprisonment of one year or less, a fine of 500 000 yen or less, detention or petty fine	Unlicensed fishery: imprisonment of three year or less or a fine of 2 000 000 yen or less Violation of condition on license: imprisonment of six months or less, a fine of 100 000 yen or less or cumulative imposition of the foregoing	Unlicensed fishery: imprisonment of three year or less or a fine of 2 000 000 yen or less Violation of condition on license: imprisonment of six months or less, a fine of 100 000 yen or less or cumulative imposition of the foregoing	Unlicensed fishery: imprisonment of three year or less or a fine of 2 000 000 yen or less Violation of condition on license: imprisonment of six months or less, a fine of 100 000 yen or less or cumulative imposition of the foregoing
Disclosure of information	**Number of licenses**	Open					Open	Open	Open
	Number of operating vessels						Not open to the public		
	Quantity of catch	Restricted	Not open to the public				Not open to the public	Not open to the public	Not open to the public

3.3.2. Taiwan Province of China

In Taiwan Province of China too, regulations have been tightened due to increasing global support for the protection of precious corals. Coral fishing is managed according to the "Regulations Governing Fishing Vessels that Also Engage in Coral Harvesting", established in January 2009. In addition to restricting the number of fishing vessels, fishing zones, catch quotas and landing ports, the regulations require fishermen to keep a logbook of fishing operations and to have an observer on board the boat (Huang and Ou, 2010) (Table 4). Fishermen who violate these regulations are penalised with confiscation of their fishing equipment and revocation of their licence.

Prior to that, Taiwan Province of China had already been reducing the number of fishing vessels in stages since 1979 – by not authorising more than the already existing 150 vessels in 1983, then setting the number at 56 in 2009 and 60 in 2010 (Chen, 2012).

Table 4. Regulation of coral fishing in Taiwan Province of China by the Regulations Governing Fishing Vessels that Also Engage in Coral Harvesting (2009)

Regulatory objectives	Description
Number of licensed vessels	60 vessels
Quantity of catch	Limit of 200 kg/vessel/year, with a total allowable catch of 6 metric tons for the fishery
Number of operating days	Limit of 220 days/vessel/year
Operating area	5 areas (see Fig.6)
Unloading	Designated 3 harbors with inspection of amount of catch and daily fishing record
Sale of catches	Open auction at one designated place
Report	Daily fishing record and on-board observers
Quantity of export	120 kg/vessel/year including raw corals, products and semi-finished goods

3.3.3. China

In April 2008, the Chinese government requested the CITES Secretariat to list four precious coral species (pink coral, Japanese red coral, white coral and *P. secundum*) in Appendix III specifically for China, and this took effect on July 1 the same year. As previously mentioned, however, there is some doubt as to whether *P. secundum* inhabits Chinese waters.

In China, precious corals are classified as Class 1 Protected Species under the Wildlife Protection Law, so fishery is prohibited. The Wildlife Protection Law, (enacted in 1988, amended in 2016) provides for the protection of precious wildlife in danger of extinction, specifies their management according to a classification system, and prohibits their unlawful capture and the destruction of their habitat. The 2016 amendment reinforced protection measures, intensified provisions against exploitation, and increased penalties (Okamura, 2016). In Article 340 of the Penal Code (1997), penalties of up to 10 years imprisonment, fines, and the confiscation of property were established for unlawfully capturing, killing or selling endangered wildlife or Class 1 Protected Species. In 2002, the Supreme People's Court ruled that more serious offences were punishable by life imprisonment or the death penalty (Lui, 2007).

Despite this strict prohibition of precious coral collection, Chinese production is reported in the FAO Global Production Statistics from 2011 to 2015 (Table 5). Thorough investigation is required in order to determine whether this is a matter of faulty statistics or whether the corals were collected outside Chinese waters.

Table 5. Annual landings (tonnes) of precious corals in China collected from Northwest Pacific (Source: FAO Global Production Statistic)

Year	Japanese red coral	Pink coral	White coral	Total
1950–2010	0	0	0	0
2011	0	1	0.15	1.15
2012	0.3	20	0.3	20.6
2013	0.3	12	0	12.3
2014	0.36	13	0	13.36
2015	0.08	0.7	0	0.78

3.3.4. The Philippines

According to Section 91 of The Philippine Fisheries Code of 1998 (Republic Act No. 8550), the collection, possession, sale and exportation of precious corals are prohibited in the Philippines except for research purposes. Violations of this provision are punished by six months to two years imprisonment and/or fines of PHP (Philippine peso) 2 000 to 20 000, as well as forfeiture of the catch and fishing vessel.

Nonetheless, precious coral necklaces purported to be from the Philippine islands of Mindanao were seen offered for sale by jewellery retailers in Manila and Makati in February 2016 (Emi Kainuma, personal communication). According to Nozomu Iwasaki's identification, however, the necklaces were not made from precious coral, but from dyed bamboo coral or mollusc shell. In July 2010, items in a Japanese red coral specialty store in Manila Airport were being offered for sale as Philippine coral (See Section II, 5.3.). This situation requires closer investigation since it indicates the possibility that precious corals are being collected and sold in the Philippines.

3.3.5. The Emperor Seamounts (high seas)

After a Japanese Fisheries Agency survey using an ROV discovered precious coral at a trawl-fishing ground in the south-eastern part of Koko Seamount, (south of lat. 34°57′N, east of long. 171°54′E, north of lat. 34°50′N, at depths of at least 400 m), the sea area was tentatively closed to fishery from 2009 (Figure 7). Procedures were also established for the event that corals (including black corals) are caught unintentionally in fishing nets (Fisheries Agency of Japan, 2008). These measures have been upheld by the North Pacific Fisheries Commission (NPFC), which was established in 2015. The NPFC also prohibits fishing in waters under its jurisdiction for cold-water corals (Alcyonacea, Antipatharia, Gorgonacea, Scleractinia), which are defined as vulnerable marine ecosystems (VMEs) (North Pacific Fisheries Commission, 2017).

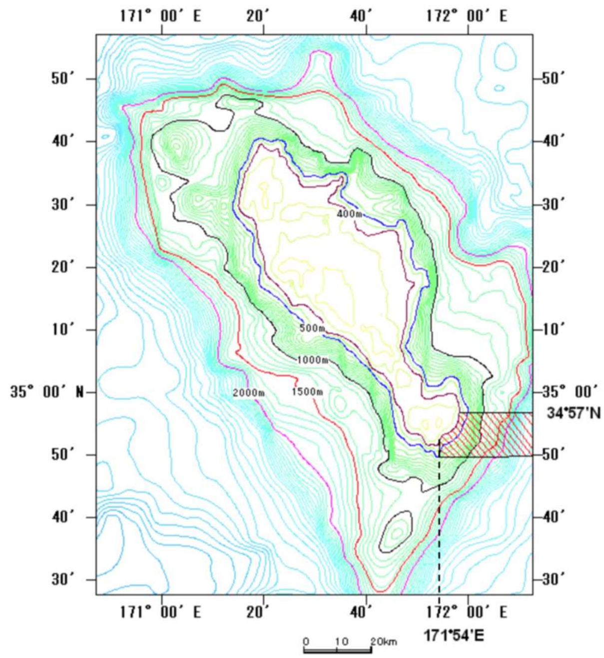

Figure 7. Tentatively closed area in Koko Seamount for protection of precious corals (cited from Fisheries Agency of Japan, 2008, Appendix P).

3.4. Poaching and illegal fishing

From around 2010, the thriving Chinese economy caused a precious coral boom (See Section I, 4.3.2.), which resulted in poaching by Chinese boats around the Ogasawara Islands in 2014. From 15 September 2014 to 22 January 2015, their number amounted to 2111 vessels, peaking at 212 a day. Poaching occurred even within Japan's territorial waters (Nagano, 2013; Chang, 2015), (Machida *et al.*, 2014; Japan Coast Guard, 2015), let alone in its exclusive economic zone (EEZ). The Japan Coast Guard arrested ten vessels and the government strengthened penalties in order to control the situation. For example, the penalty for illegal operations in Japanese territorial waters was increased from a maximum of three years imprisonment and/or a fine not exceeding JPY (Japanese yen) 4 million to a maximum of three years imprisonment and/or a fine not exceeding JPY 30 million. Also, in December the Japanese and Chinese governments agreed to a definitive and consistent crackdown, employing stronger measures such as strict penalties for offenders, in order to stamp out illegal coral fishing by Chinese boats. As a result, no more poaching incidents were reported near the Ogasawaras after late January 2015 (Ministry of Agriculture, 2015; Japan Coast Guard, 2015; Ministry of Agriculture, Forestry and Fisheries, Japan, 2015; Ministry of Foreign Affairs of Japan web page).

However, there has been no end to poaching by Chinese fishing boats. In 2015, for example, Chinese authorities prosecuted 19 fisherman who had poached 30 kg of Japanese red coral (CNY (Chinese Yuan)10 million, JPY 190 million) over a period of ten months from around the Senkaku Islands (The Asahi Shimbun, 2015). Again, in July 2017, Chinese fishing boats were apprehended in Japan's EEZ off Nagasaki Prefecture for collecting coral without authorisation from the Minister of Agriculture, Forestry and Fisheries (Kyushu Fisheries Coordination Office, Fisheries Agency website). Neither is poaching by Chinese fishing boats limited to Japanese waters: waters surrounding Taiwan Province of China are also being targeted (Chang, 2015). Chang (2015) estimates the quantity of illegally collected coral to be in excess of 21 tonnes.

In response to these incidents, the Fisheries Agency of Japan investigated the waters off the Ogasawara Islands (2015), Okinawa (2015) and the sea area southwest of Kyushu (2017). Around the Ogasawara Islands, while fishing equipment believed to be associated with illegal Chinese operations was found on the sea floor and damage to precious corals was confirmed, precious coral habitation was observed even in the illegally fished sea areas and no traces of major alteration to the sea floor or ghost fishing (corals captured in abandoned nets) were identified (Fisheries Agency, Japan web page). However, since no data had been collected concerning distribution and population density before the poaching incidents, it is impossible to assess the actual extent of the damage.

Both in Japan and Taiwan Province of China, domestic coral fishing boats operating in violation of laws and regulations have also been exposed (Huang and Ou, 2010; The Mainichi, 2012).

4. THE PRESENT STATE OF PRODUCTION AND RESOURCES

4.1. FAO statistics and limitations of Japan's precious coral production statistics

It is impossible to accurately identify Japan's total coral production and its fluctuations because data on coral landings in most regions (Wakayama, Ehime, Kochi, Nagasaki, Kagoshima and Okinawa prefectures) are not disclosed, while those of other regions are incomplete (the data from Tokyo not only refer to sales rather than landings but also include reef-building corals) (Figure 6). The Food and Agriculture Organization of the United Nations (FAO) compiles Global Production Statistics, in which annual production for each country and species is recorded. But it is unclear if Japan's data include all of those from undisclosed areas, and discrepancies are found when the FAO statistics on Japan are compared with those from other sources. For example, when the FAO statistics for 1983–1991 are compared with Grigg's data (1993), some years are roughly in agreement while others differ by as much as about 5 tonnes (1984; Figure 8). In addition, the FAO reports production of only 0.1 tonnes for Japan for 1997, while the quantity traded at bid markets in Kochi prefecture that year amounted to as much as 1.4 tonnes (Figure 9). Nonetheless, for 2003–2006, the total quantities traded at bid markets throughout Japan (Kochi, Kagoshima, and Okinawa prefectures) reported by the Kochi Department of Fisheries (2007) differ by only 0.2–0.9 tonnes from the FAO's data, and the two sources report similar yearly fluctuations (Figure 8). The FAO Global Production Statistics can therefore be seen as a fairly accurate reflection of Japan's production and its changes in recent years.

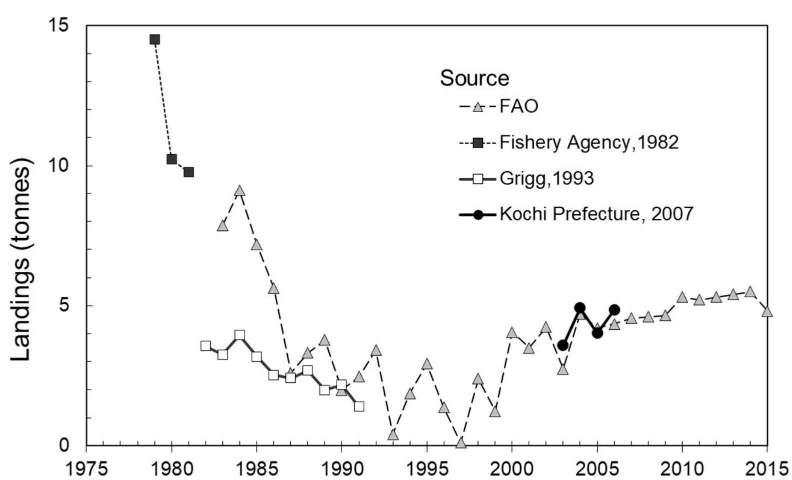

Figure 8. Changes in Japanese precious coral landings (Source: Fisheries Agency, Japan, 1982; FAO; Grigg, 1993; Kochi Department of Fisheries, 2007).

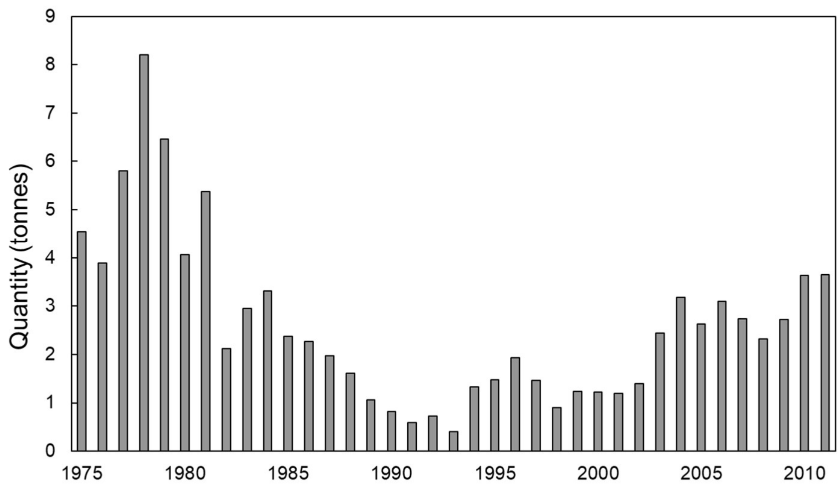

Figure 9. Changes in bid market quantities of Kochi precious corals (Source: Kochi-ken shōseika chūshōkigyō shidōshitsu, 1978; Iwasaki, 2010; Shosakai, 2013; Sukumo Precious Coral Cooperative).

4.2. Coral production in Asia according to FAO statistics

The FAO Global Production Statistics indicate that Japan's annual production declined sharply from 14.9 tonnes in 1979 to 2.6 tonnes over a period of about 10 years. It then increased and decreased repeatedly, recovering to 4.05 tonnes in 2000. In recent years, production has remained at around 5 tonnes (Figure 10). As for changes in sea areas other than Japanese coastal waters, Taiwan Province of China landed overwhelming quantities in the 1970s and 80s, with nine out of those 20 years surpassing 100 tonnes. Production then declined sharply and has stayed at around 2–3 tonnes a year since 2009 (Figure 10). In the Mediterranean, 98 tonnes were landed in 1978, but this gradually decreased to 19 tonnes in 1998 before gradually increasing again to exceed 50 tonnes in recent years (Figure 10).

Currently, the two major production areas are the seas near Japan and Taiwan Province of China, and the Mediterranean. During the seven years from 2010 to 2016, annual production in each area did not change significantly. The average annual production in Japan, Taiwan Province of China, and Mediterranean countries was 5.2, 3.0, and 51.8 tonnes respectively, and Japan accounted for 7.3 percent of the global total (Table 6).

Records are available for China from 2011 despite the fact that precious corals are designated as protected species and fishery is therefore prohibited in Chinese waters (See Section III, 1). It is unknown whether the statistics are faulty or whether certain coral fishing is legally allowed.

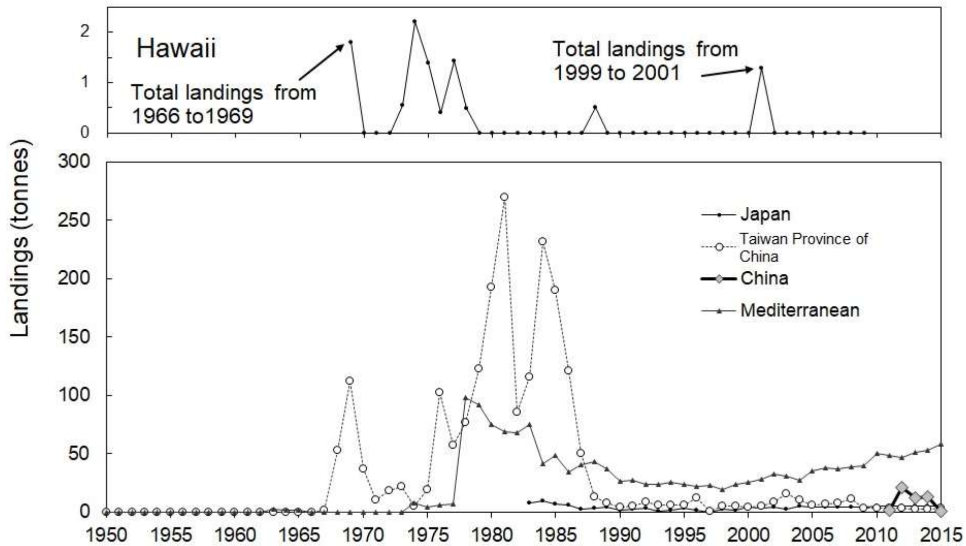

Figure 10. Changes in precious coral landings by sea area (Source: FAO Global Production Statistics; Grigg, 1993, 2010).

Table 6. Annual landings (metric tons) of precious corals （Source: FAO）

Year	Northwest Pacific			Mediterranean and Atlantic	Total
	Japan	Taiwan Province of China	China*		
2010	5.3	2.93	-	54.11	62.3
2011	5.2	3.21	1.15	52.29	61.9
2012	5.3	3.25	21.0	50.58	80.1
2013	5.4	2.82	12.3	54.30	74.8
2014	5.5	2.51	13.36	55.25	76.6
2015	4.8	3.63	0.78	60.31	69.5
2016	4.6	2.59	0.54	60.92	68.7
Average	5.2	3.0	7.0	55.4	70.6
Percentage	7.3	4.2	9.9	78.5	100.0

* Although coral fishing is banned in China, landings were reported there.

4.3. Recent trends in bid-market trade volumes of precious corals landed in Kochi (Japan)

4.3.1. Precious coral statistics for Kochi Prefecture

Kochi Prefecture is the birthplace of coral fishing and also the principal production area in Japan, so its statistics for precious coral are relatively well maintained. Production statistics are available from 1897 to 1925 (Figure 11). However, the yearly fluctuations during that period do not necessarily reflect the quantities of resources. In those days, when attractive fishing grounds were discovered in other prefectures, fishing boats moved away from Kochi and production there decreased (Ogi, 2010a). There are no statistics available for after World War II, but from 1975 the bid-market trade volumes of precious coral landed in Kochi Prefecture can be used. Bid markets are held in Kochi prefecture several times a year, and almost all the precious coral landed in the prefecture is brought there by the fishermen and sold to processors and distributors. Unlike fresh fish, precious coral can be preserved after it is landed. This means sellers can hold on to it when market prices are low, so there could be some difference in the timing of when it is landed and when it is traded at a bid market.

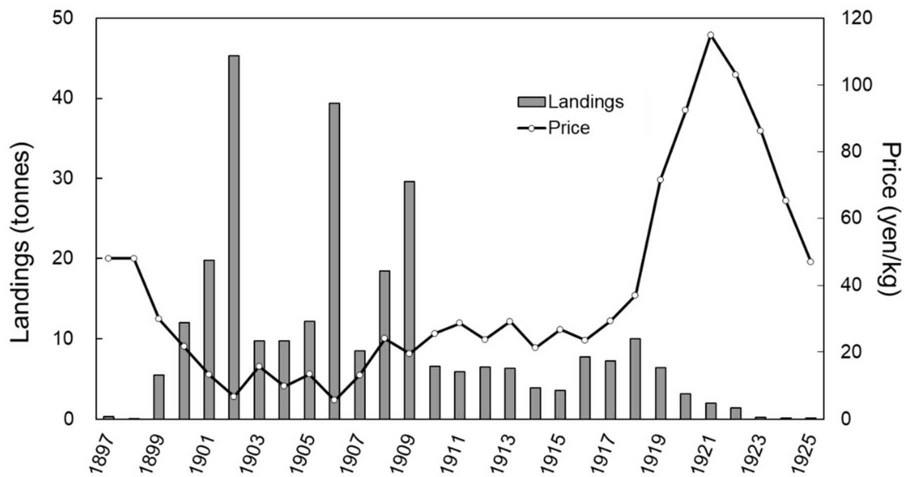

Figure 11. Changes in landings and average unit prices for precious corals collected off the coast of Kochi Prefecture, 1897–1925 (Source: Komatsu, 1970; Ogi, 2010a).

4.3.2. Changes in trade volumes and average prices

Annual production from the waters off Kochi prior to World War II peaked at 45.3 tonnes in 1902. After production of 29.6 tonnes in 1909, the quantity declined rapidly to an annual average of 5.8 tonnes (1911–1922; Figure11). The largest quantity traded after the war was 8.2 tonnes in 1978, but this was followed by a period of continuous decline, reaching a low of 0.4 tonnes in 1993 (Figure 8). The average annual quantity traded in recent years was 3.6 tonnes (2010–2011) (Figure 9), which amounts to only 20 percent of the average annual production of 17.5 tonnes in the peak period (1899–1920). Although quantities traded at bid markets do not necessarily reflect quantities of resources, it is clear that resources have diminished compared with the peak periods.

Furthermore, Nakanishi (2010) points out that the small quantities traded in the first half of the 1990s were due to fishermen turning from coral to alfonsino (*Beryx decadactylus*) fishing. Few fishermen specialised in coral at the time; many of them were also engaged in other types of fishing. Their turning to alfonsino in pursuit of higher profits is thought to have caused the decrease in precious coral trades (Chang *et al.*, 2013; Nakanishi, 2010).

Average bid market prices rose sharply from 2010 after gradually increasing from 1990 (Sakita, 2016). This was due to increased demand from wealthy Chinese as their economy boomed (Machida *et al.*, 2014; The Nikkei, 2010). The price of Japanese red coral, which is preferred in China, soared: in 2015 the unit price of raw coral reached an historic high (585 g, JPY 28.89 million, JPY 49 385 /g), with one piece being traded for the highest price ever recorded (3 326 g, JPY 91.2 million) (Miyazaki, 2015; Four Seasons of Jewelry, 2015). While the average unit price has followed a downward trend since then, prices remained high in 2017.

4.3.3. Proportions of live and dead coral

For sale as raw material, precious coral is classified not only by species and colour but also according to whether it is dead or alive (Iwasaki, 2010). Live coral refers to colonies that are still alive when they are collected while dead coral consists of skeletal axes that have been left as remains on the sea floor. Live coral prices have always been higher.

Precious corals grow very slowly. The growth rate of the pink coral axis is only 0.3 mm/year in diameter (See Section I, 2.4.). It is therefore not easy for precious coral resources to recover after they are fished. The condition of resources can be assessed by looking at the proportion of live and dead coral in the landings.

According to the records, when coral fishing began in the waters off Kagoshima (around 1902–1903), the percentage of live coral in the landings was as high as 81.6 percent for Japanese red coral, 67.1 percent for pink coral, and 86.3 percent for white coral (Ogi, 2010a). In a fishing survey conducted near the Ogasawara Islands in 1935, live coral made up 67.0 percent of the 2 844 g red coral sample and 65.5 percent of the 917 g white coral sample. For fishing grounds in southwestern Kochi prefecture (Hata district), there are records for around 1919, roughly 40 years after coral fishing began there in 1871. These records indicate that live colonies made up 20 percent of the catch at that time, a significant drop from earlier years (Ogi, 2010a). From 1989 to 2011, the percentage of live coral in the quantities traded at bid markets averaged 11.1 percent, down from 20 percent around 1910. Even since resource management was reinforced in the fishing grounds near Taiwan Province of China, the proportion of live colonies in the catch is reported to be gradually decreasing there too (See Section I, 4.4.) (Chen, 2012).

These statistics suggest that live colonies make up a large proportion of the catch in newly discovered fishing grounds, and the proportion decreases as fishing continues. The following paragraph assesses the trends in precious coral resources in the coastal waters of Kochi, based on this hypothesis.

4.3.4. Resource trends reflected in proportions of live and dead coral

The total average percentage of all types of dead coral (red coral, red coral twigs, pink coral and white coral) in the annual quantities traded at bid markets in the Kochi area from 1989 to 2011 was 88.9 percent. In particular, dead red coral accounted for 68.1 percent of the annual quantities traded, indicating that coral fishing in the Kochi region consists largely of collecting red coral skeletal remains (Figure 12).

Regarding changes in the proportion of dead coral, there was no significant difference from around 1990 to the early 2000s, but from 2010 to 2011 the proportion of dead coral declined rapidly (Figure 12) while the quantity of live coral traded at bid markets increased (Figure 13–Figure 16). As for the changes in this period by type, the quantity of live coral increased for red coral, red coral twigs and white coral, resulting in a sharp rise in the overall proportion of live coral (Figure 13–Figure 16). The timing of the sharp increase in live coral coincides with a jump in the number of vessels authorised for coral fishing, encouraged by soaring prices (Figure 17). We can infer from this that, due to the large number of boats carefully searching within the sea areas designated for coral fishing, the discovery rate of live colonies increased, causing the quantities of live coral traded at bid markets to rise sharply.

The average annual quantities of live red coral, red coral twigs and white coral traded during the dramatic upswing of 2010–2011 were significantly larger than the quantities traded in 2005–2009. This can be seen as an indication that coral resources are being subjected to fishing pressure due to the greater number of fishing boats in operation (Figure 18).

Pink coral is the species with not only the smallest total quantities traded at bid markets but also the smallest quantities and proportions of live coral (Figure 15). While pink coral accounted for 20.6 percent of all precious coral production during the peak period of 1904–1920 (Ogi, 2010b), it made up 12.4 percent of the annual quantity traded at bid markets in 1989–2011. Also, the average annual quantity of live pink coral traded in the same period was a mere 4.5 kg, an average share of just 1.9 percent of pink coral landings. This can be seen as a clear indication that the pink coral biomass has decreased compared with the peak period.

Academic resource studies on precious corals are long overdue. With the exception of their growth rate, few of the biological parameters necessary for assessing the state of existing resources (e.g., biomass and recruitment, growth rate, mortality rate) have been defined. This report therefore investigates the resource situation using the quantities and proportions of live coral traded at bid markets. Until recently, since over 80 percent of precious coral trades to date have been dead coral, the burden on the biomass and renewal of resources was thought to be insignificant (Chang *et al.*, 2013). However, the increase in fishing boats since 2010 in response to soaring prices is thought to have intensified fishing pressure, particularly on pink and white coral resources.

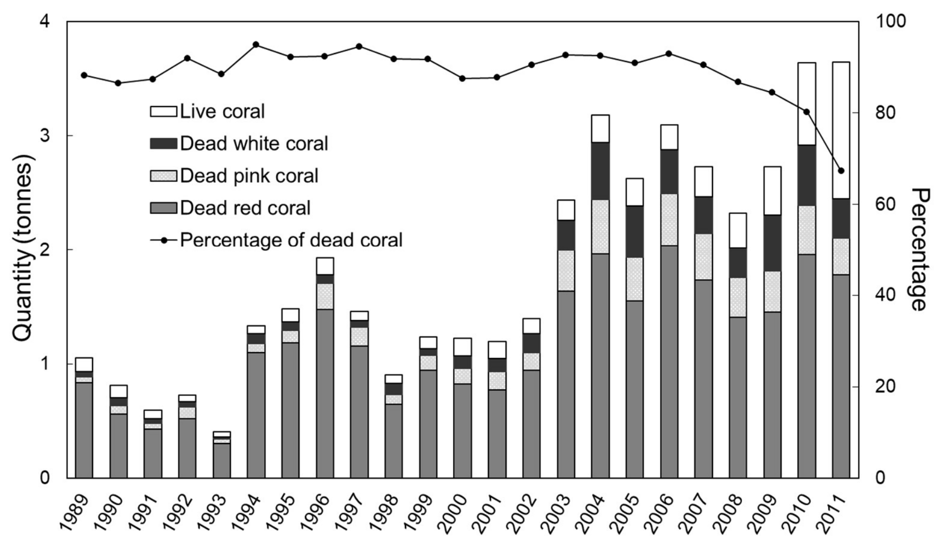

Figure 12. Proportions of live and dead coral in quantities of Kochi precious corals traded at bid markets (Source: Shosakai, 2013).

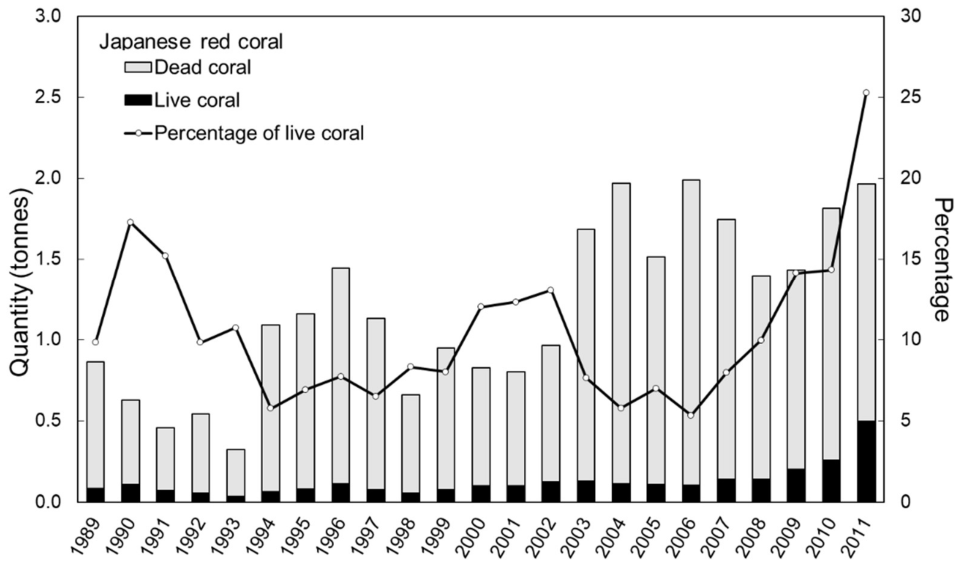

Figure 13. Proportions of live and dead coral in quantities of Kochi red coral traded at bid markets (Source: Shosakai, 2013).

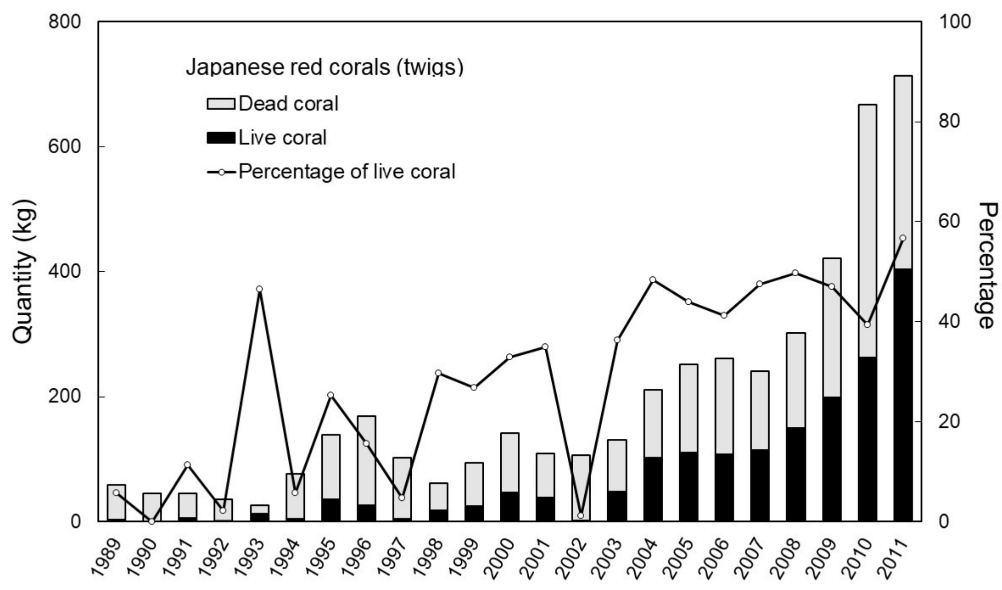

Figure 14. Proportions of live and dead coral in quantities of Kochi red coral twigs traded at bid markets (Source: Shosakai, 2013).

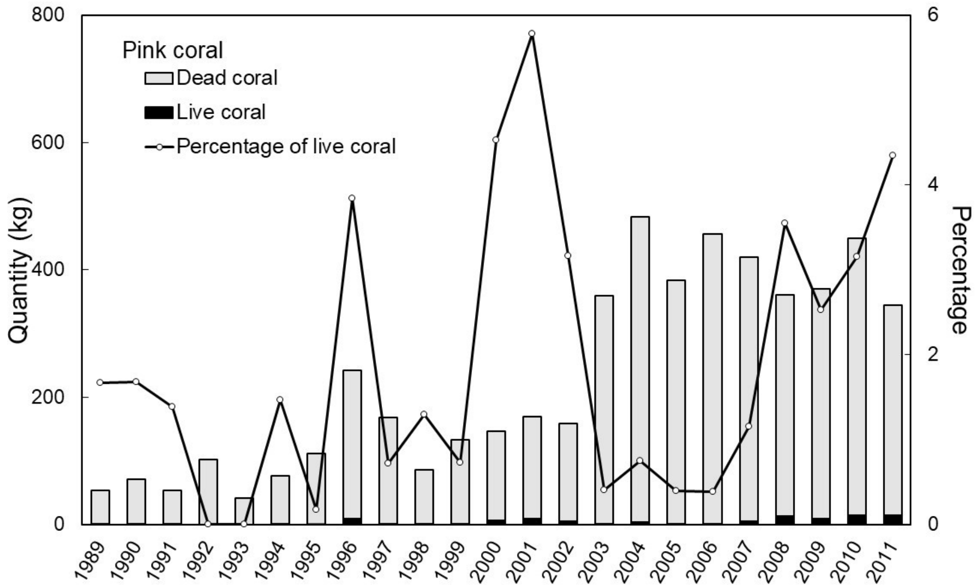

Figure 15. Proportions of live and dead coral in quantities of Kochi pink coral traded at bid markets (Source: Shosakai, 2013).

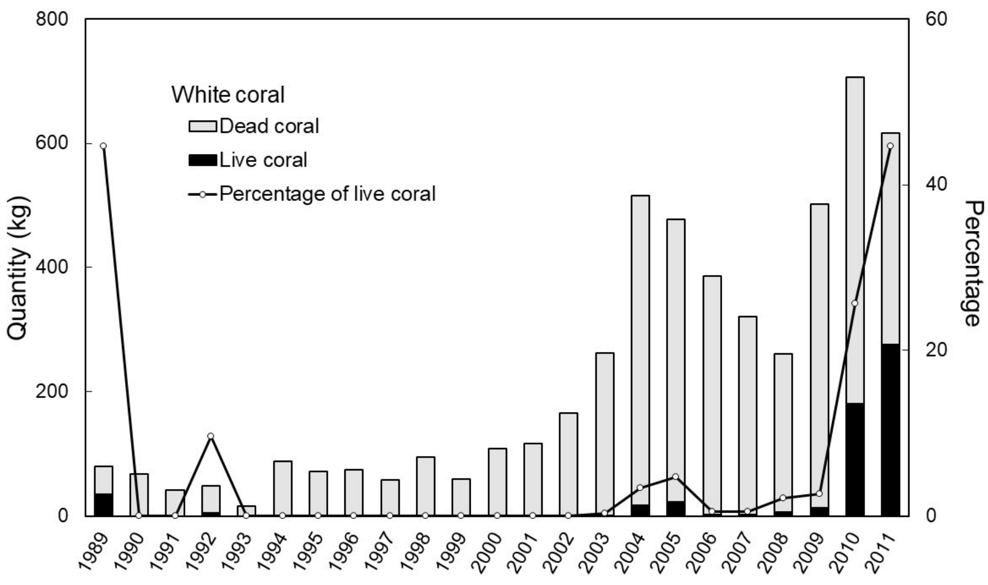

Figure 16. Proportions of live and dead coral in quantities of Kochi white coral traded at bid markets (Source: Shosakai, 2013).

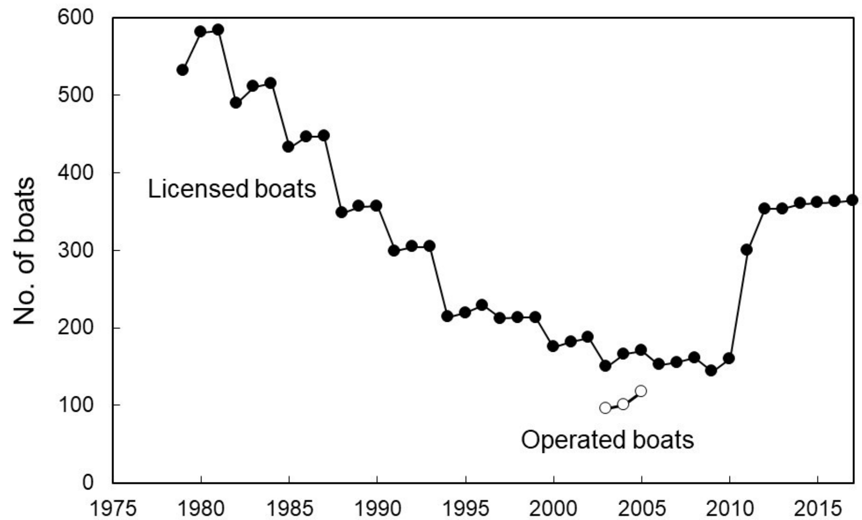

Figure 17. Changes in the number of licensed coral fishing boats and boats in operation for Kochi corals (Source: Kochi Department of Fisheries, 2007).

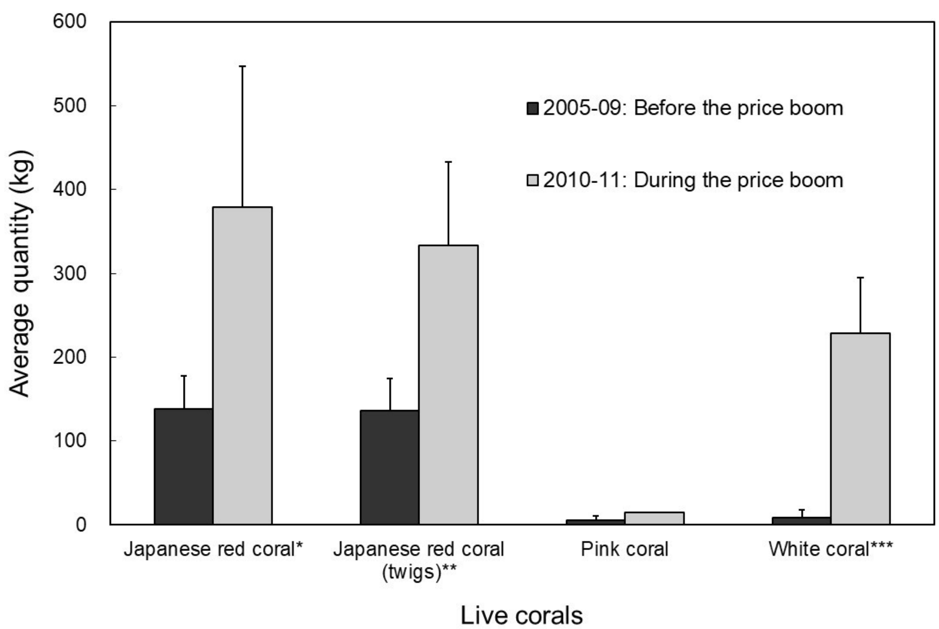

*Figure 18. Comparison of quantities of live coral traded at bid markets before and after the precious coral price boom *p<0.05, **p<0.01, ***p<0.001, t-test (Source: Shosakai, 2013).*

4.4. Taiwan Province of China

Since Taiwan Province of China began coral fishing in 1924, annual production has peaked three times, once in the late 1960s and twice in the 1980s (Figure 19) (Huang and Ou, 2010; Chang *et al.*, 2013). Production exceeded 100 tonnes in the first peak, and 200 tonnes in the second and third. However, most of this was collected near Midway Atoll so these figures do not represent the catch from Taiwanese coastal waters. Huge hauls from Midway caused coral prices to fall and fishing boats eventually withdrew from the area. Since then, annual production has remained below 20 tonnes.

In 2009, regulations governing coral fishing were tightened (See Section I, 3.3.2.). Coral fishing boats were required to record details of their operations in a logbook (dates and times, latitudes and longitudes, precious coral species and weight, etc.) and landings were inspected. According to the sum total of this data, average annual production for the period 2009–2012 was 2.9–3.2 tonnes (Figure 2a in Chen, 2015). Pink coral made up the greater part of the catch (63–86 percent), followed by miss coral (1–9 percent) and red coral (3–4 percent) (Chen, 2012, 2015).

In 2009, production of live coral was 109 kg, but this fell to 40 kg the following year and then followed a gradual upward trend (Figure 2b in Chen, 2015). The percentage of live coral in the landings remained low at 2–5 percent. Pink coral (momo) accounted for 60–90 percent of the live coral, followed by red coral (aka) at 6–21 percent (Figure 4b in Chen, 2015). Landings of live pink coral, and especially red coral, varied in different sea areas: in sea area E, while over 11 kg of live red coral was landed in 2009, there were no landings in subsequent years (Figure 7 in Chen, 2015) (Chen, 2012, 2015). Huang and Ou (2010) point out that the small proportions of live colonies in the catch indicate deteriorating resources that are ill-adapted for further exploitation. They therefore call for an immediate reduction in the number of authorised vessels and operating days.

There are five designated fishing grounds in Taiwanese coastal waters, covering a total area of 7 811 km^2. This is an average of 120 km^2 per fishing boat, which makes for relatively dense fishing operations. In addition, the operations of 60 percent of the authorised vessels are concentrated in one sea area. It is feared that, over time, this situation will further deplete coral resources in the designated fishing grounds (Huang and Ou, 2010).

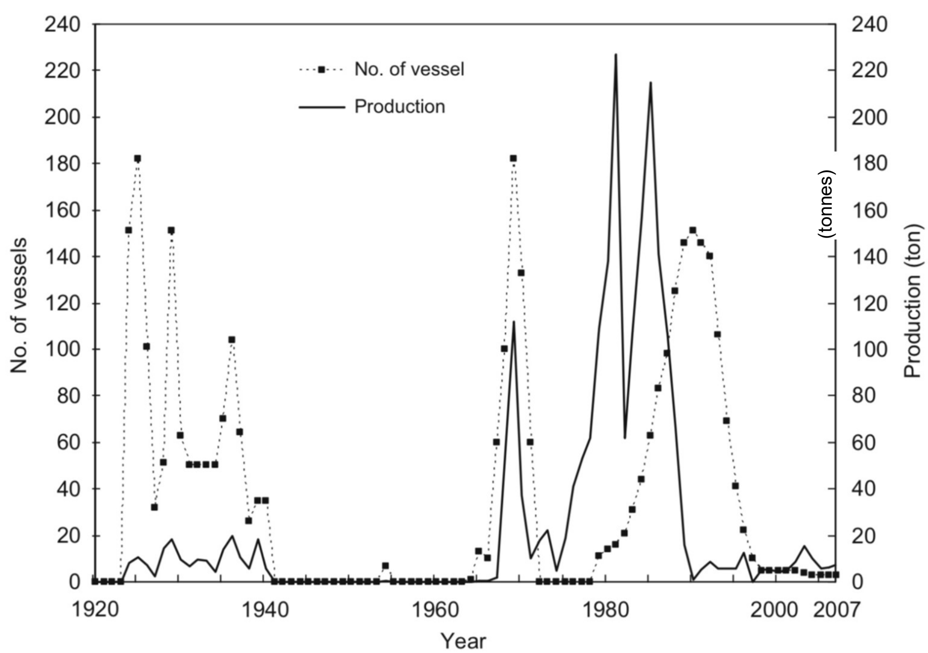

Figure 19. Number of fishing vessels and production of Taiwan Province of China's precious coral fisheries, 1924–2007 (cited from Huang and Ou, 2010, Figure 1).

5. TRADE

5.1. Issues in foreign trade statistics

5.1.1. HS codes

All foreign trade items are classified and statistics kept according to the Harmonized Commodity Description and Coding System (HS). The first six digits (designating the chapter, heading and subheading) of the HS code are standard throughout the world and the optional four-digit suffix is a domestic code that is decided individually by each member country. None of the first six internationally standardised digits corresponds specifically to precious corals, and the suffix they are allocated varies from country to country. It is therefore difficult to obtain accurate import and export data for precious corals.

For example, when precious corals are exported from Japan, the six-digit HS code that applies to them covers not only all species of coral and similar materials but also mollusc and crustacean shells, echinoderms and cuttlebone. In Japan raw coral is allocated the suffix 200, so the code for raw coral is 0508.00.200. However, no distinction is made between precious and reef-building corals; they are all lumped together under the same code.

In this report, all exports of raw coral from Japan are assumed to be precious corals (see Section I, 6.2.). This is because reef-building corals are primarily distributed around the Ryukyu Islands (Okinawa) and the Ogasawara Islands, and the collection of these corals has been prohibited in Okinawa since 1972 and is also restricted in Ogasawara. On the other hand, raw coral imports include both precious and reef-building corals. The precious coral imports reported in the following paragraphs were therefore identified from production areas and prices (reef-building coral prices are low). For processed and manufactured items, precious corals share HS Code 960190 with not only all species of coral but also bone, tortoise shell, horn, antlers, mother-of-pearl and other animal carving materials.

5.1.2. Disparity between landings and exports

There are large disparities between the data for Japanese precious coral landings and raw coral exports, indicating flaws in the statistics. The export quantity is much larger than the landings (Figure 10, Figure20), for example in 2015 the former and the later are 34.9 and 4.8 tonnes, respectively. Takahashi (2010) points out differences between Taiwan Province of China's reported exports to Japan and Japan's reported imports from Taiwan Province of China, noting that there are problems not only in the statistics but in the customs controls in both nations as well. (See Section I, 5.2.2.)

5.2. Japanese imports and exports

5.2.1. Japan's raw coral imports and exports

This section outlines the trends in precious coral imports and exports after World War II, based on Trade Statistics of Japan, published by the Japanese Ministry of Finance.

After World War II, commercial trading resumed in 1947. Japan began exporting raw coral to the United States of America and Italy the following year and to India in 1949. Exports to Italy were dominant in both quantity and value until around 1980 (Figure 20, Figure 21). In the 1980s, export quantities to Taiwan Province of China increased along with an increase in the nation's share of Japan's exports. There was a sharp increase in the quantity of exports to the Republic of Korea in 1990 and to the United States of America in 2002. From 2010 to 2015, the proportions exported to the three major countries in terms of quantity were 68.0 percent to the United States of America, 16.3 percent to the Republic of Korea, and 12.3 percent to Taiwan Province of China, while Taiwan Province of China accounted for an overwhelming 88 percent of the total value of exports during that period.

The export prices (averages for 1988–2015) for the Republic of Korea and the United States of America were low at JPY 795 /kg and JPY 1 527 /kg respectively, indicating that large quantities of low-priced items were exported to those countries (Figure 22). Meanwhile, export prices to Taiwan Province of China, Hong Kong SAR, and European countries were over JPY 190 000 /kg, and small quantities at unit prices exceeding JPY 1 million /kg were exported, particularly to Switzerland and France.

As for imports of raw coral, from the mid-1950s to the early 1970s, imports from Okinawa, then under United States of America military occupation, accounted for the major share of both quantity and value (Figure 23, Figure 24). However, in the mid-1960s, when the catch in Okinawa decreased, raw coral from mainland Japan was exported to Okinawa (Figure 20). From the mid-1970s to the present, imports from Taiwan Province of China have dominated.

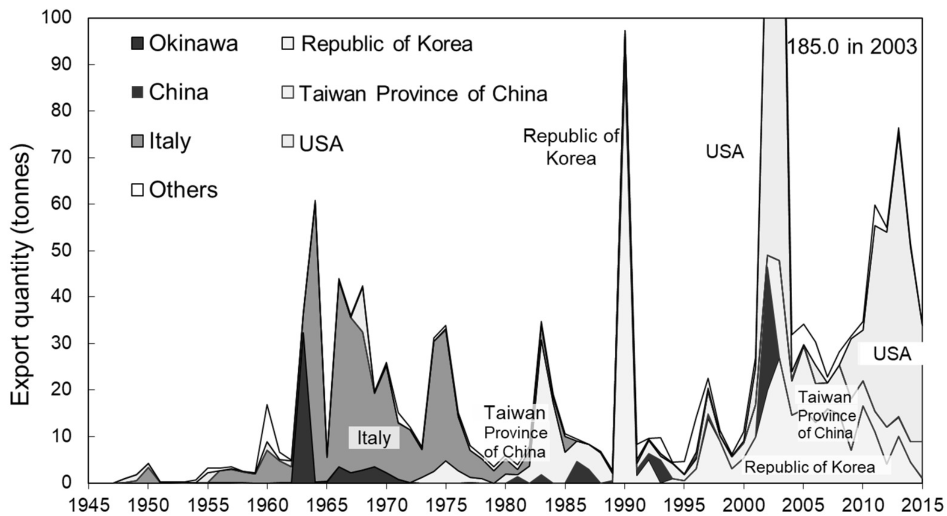

Figure 20. Changes in Japan's export quantities of raw precious corals (Source: Customs and Tariff Bureau, Japan; Japan Tariff Association).

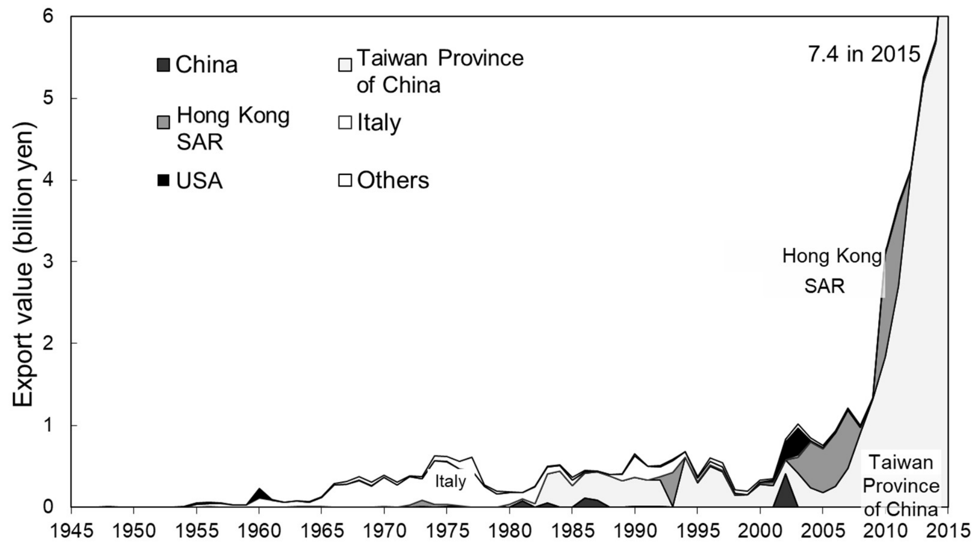

Figure 21. Changes in Japan's export values of raw precious corals (Source: Customs and Tariff Bureau, Japan; Japan Tariff Association).

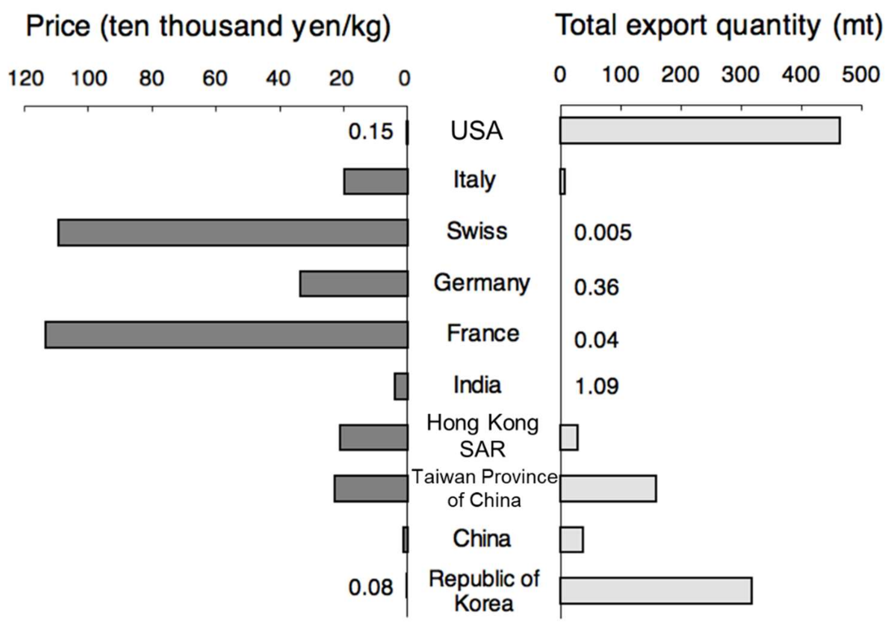

Figure 22. Japan's total export quantities and average unit prices to major countries, 1988–2015 (Source: Customs and Tariff Bureau, Japan).

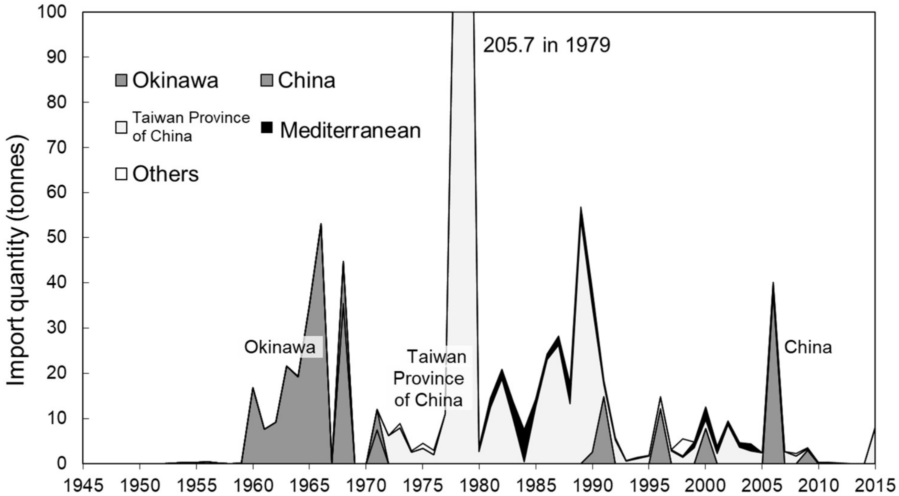

Figure 23. Changes in Japan's import quantities of raw precious corals (Source: Customs and Tariff Bureau, Japan; Japan Tariff Association).

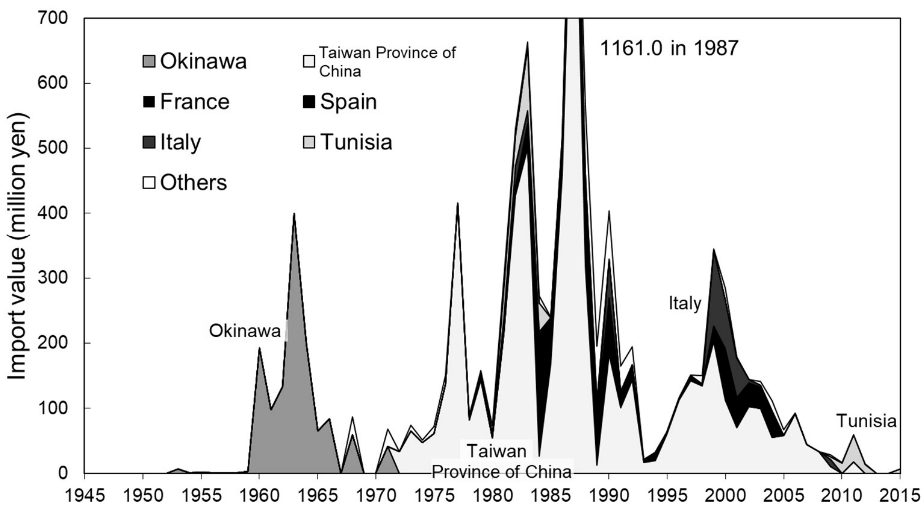

Figure 24. Changes in Japan's import values of raw precious corals (Source: Customs and Tariff Bureau, Japan; Japan Tariff Association).

5.2.2. Imbalance between Japan's imports and exports of raw coral and processed items

There is a large disparity between Japan's precious coral landings and exports. A comparison of yearly fluctuations in landings and exports reveals that exports are consistently two-digit figures while landings are single figures (Figure 8, Figure 20). Let us consider the total quantity of precious coral stock that existed in Japan during three periods for which domestic production can be estimated. The total stock can be thought of as precious coral landings plus imports of raw coral and processed items, minus exports of raw coral and processed items. From 1979 to 1981, the period when the largest postwar raw coral imports were recorded, the average stock was 79.4 tonnes (Figure 25). However, excessively large exports in 2003–2006 and 2010–2015 resulted in average stocks of minus 99.6 tonnes and minus 80.5 tonnes respectively. When the stock is a positive figure, we can assume the coral was kept in Japan for processing or used to fill domestic demand. When the figure is negative, it is possible that already existing stock was exported. However, given the fact that landings were never more than a few tonnes, the huge average annual export figures of 80–90 tonnes can be said to be unreasonably excessive.

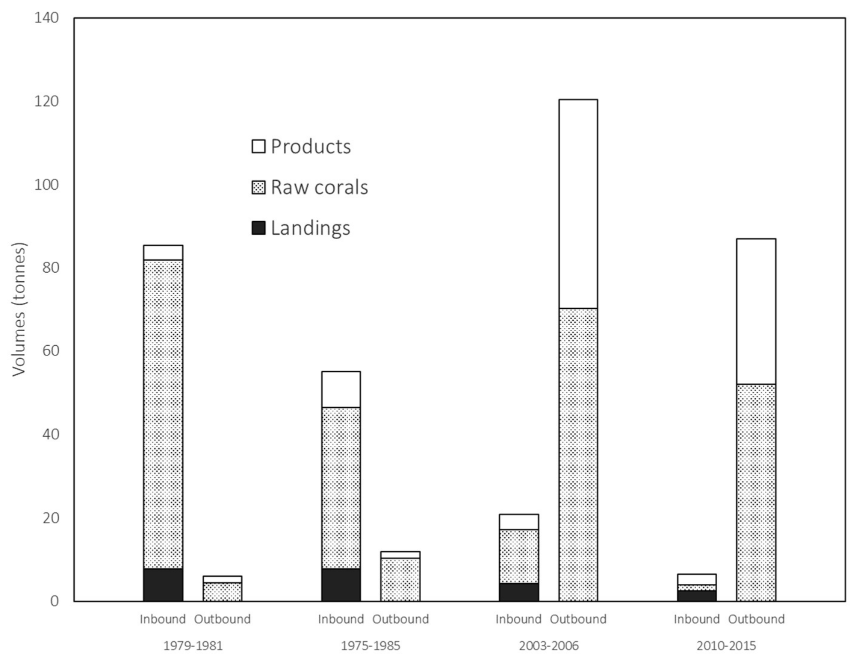

Figure 25. Inbound and outbound quantities of precious corals for Japan. Inbound: landings, imports of raw coral and coral products. Outbound: exports of raw coral and coral products.

5.2.3. Recent changes in Chinese demand and Japanese exports

Precious coral prices soared in the years 2010–2015. Japanese exports to Taiwan Province of China, Hong Kong SAR, and China during this period consisted of more valuable, higher-priced items compared to 2003–2006, prior to the price rise (Table 7). Meanwhile, the low-priced items that were being exported to the Republic of Korea and the United States of America further decreased in value. In particular, prices for the United States of America plunged to one-fifth, causing export quantities to increase. In this way, prices polarised further after the boom.

Figure 26 shows the monthly quantity and value of Japan's exports to Taiwan Province of China in 2015. According to reports, raw coral bid markets were held in Kochi Prefecture in March, May, 30 August–3 September and November. The trade values for those months are as follows: JPY 2 158 million for March (Kochi coral JPY 1 632 million, Wakayama coral JPY 526 million); unknown for May; JPY 900 million for August–September (30 August–1 September); and JPY 1 178.84 million for November (The Kochi Shimbun, 2015a; The Asahi, 2015; NHK, 2015). The quantity of Japan's exports to Taiwan Province of China spiked in the months bid markets were held or the month after, and the export values also reflect the bid market trade values.

Table 7. Annual averages of export quantity and price of raw corals in the years 2003–06 and 2010–15 (Source: Customs and Tariff Bureau, Japan)

Exporting countries from Japan	2003–06		2010–15	
	Price (yen/kg)	Quantity (kg)	Price (yen/kg)	Quantity (kg)
Republic of Korea	1 100	17 545	665	7 830
China	-	0	380 619	11
Taiwan Province of China	21 692	12 622	784 341	5 700
Hong Kong SAR	175 398	2 765	360 417	1 042
India	20 448	211	123 761	46
Italy	99 412	156	1 355 878	10
United States of America	2 490	34 183	526	36 854
All countries	12 636	70 290	93 896	52 058

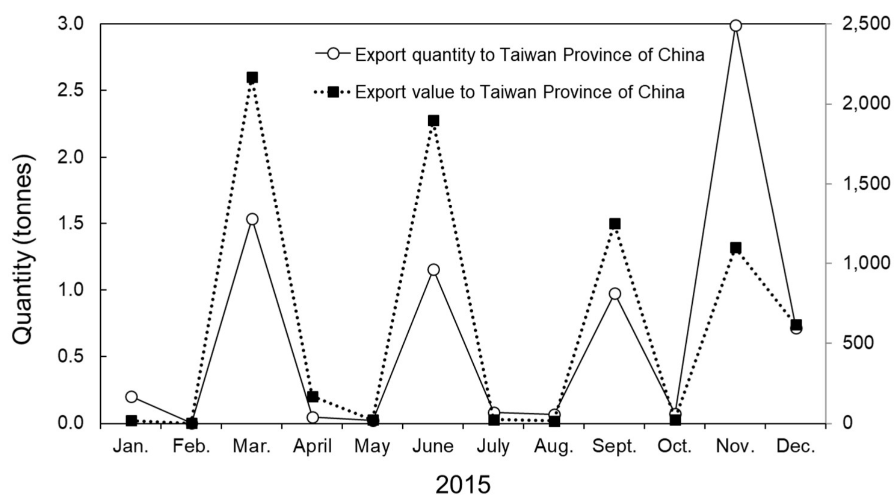

Figure 26. Monthly export quantities form Japan to Taiwan Province of China in 2015 (Source: Customs and Tariff Bureau, Japan; Japan Tariff Association) Note: Bid markets held for Kochi corals in March, May, 30 August–3 September and November, for Wakayama corals in March, and for Kagoshima and Okinawa corals in December.

5.3. Circulation in Taiwan Province of China

Precious coral collected in Taiwan Province of China is sold at bid markets in Suao. It is then processed domestically (in part in Vietnam) and sold either to Chinese tourists visiting Taiwan Province of China or at fortnightly trade shows held in China. China is therefore the principal market for Taiwanese precious coral (Figure 27) (Chang, 2015).

Precious coral prices rose steeply on the back of the booming Chinese economy, increasing from USD 900 /kg in 2009 to USD 7 500 /kg in 2014. The prices began to rise in July 2008 when Taiwan Province of China passed legislation opening the gate to visitors from mainland China. The number of Chinese visitors shot up from 329 000 in 2008 to 2 875 000 in 2013 (Chang, 2015). According to Huang and Ou (2010), the average unit price for bid-market trades in 2009 was USD 1 395 /kg.

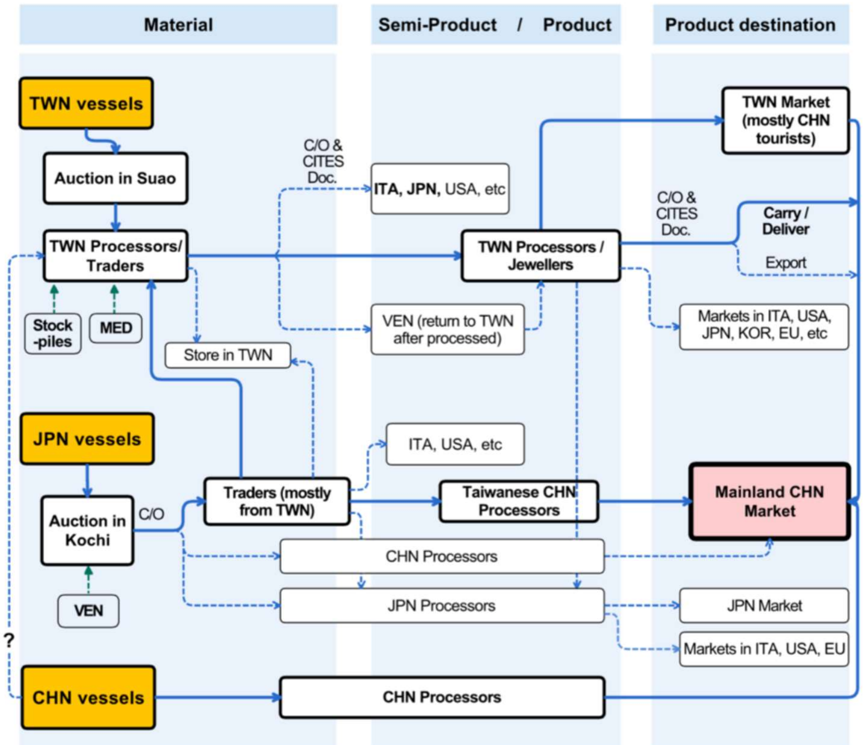

Figure 27. Current supply chain of Northwestern Pacific precious corals. Boxes indicate holders of precious corals as raw materials, semi-processed or finished products, and lines indicate the routes. Major holders and routes are represented by bold boxes or lines. Although not shown here, there might be buyers from different countries in each market. The country codes are TWN for Taiwan Province of China, CHN for China, EU for the European Union, ITA for Italy, JPN for Japan, KOR for the Republic of Korea and VEN for Vietnam. MED is for countries around the Mediterranean Sea (cited from Chang, 2015, Figure 1).

5.3.1. Issues with HS codes in Taiwan Province of China

The following is an overview of Taiwan Province of China's imports and exports based on the Global Trade Atlas (HIS Markit) database. Like the Japanese HS codes previously discussed, the HS codes used in Taiwan Province of China apparently make no distinction between precious corals, similar materials and reef-building corals (See Section I, 5.1.1.). Statistics must therefore be treated with caution. For example, from 2004 to 2016, the average annual import quantities of HS Code 0508001100 (coral and similar materials) was 1 457 tonnes, of which the overwhelming majority was imported from the Philippines (Figure 28). Considering the low unit price of USD 0.1 /kg and the fact that precious coral fishing is prohibited in the Philippines, we can assume that the imported materials were not precious corals. For this reason, countries from which low-priced coral items are imported have been excluded from this report.

Furthermore, Taiwanese HS codes differentiate between "worked coral materials" (9601904100) and "articles of coral" (9601904200). Since the details of these classifications are unknown, rather than treating them separately, the sum of the two categories is described in the following paragraphs as coral products.

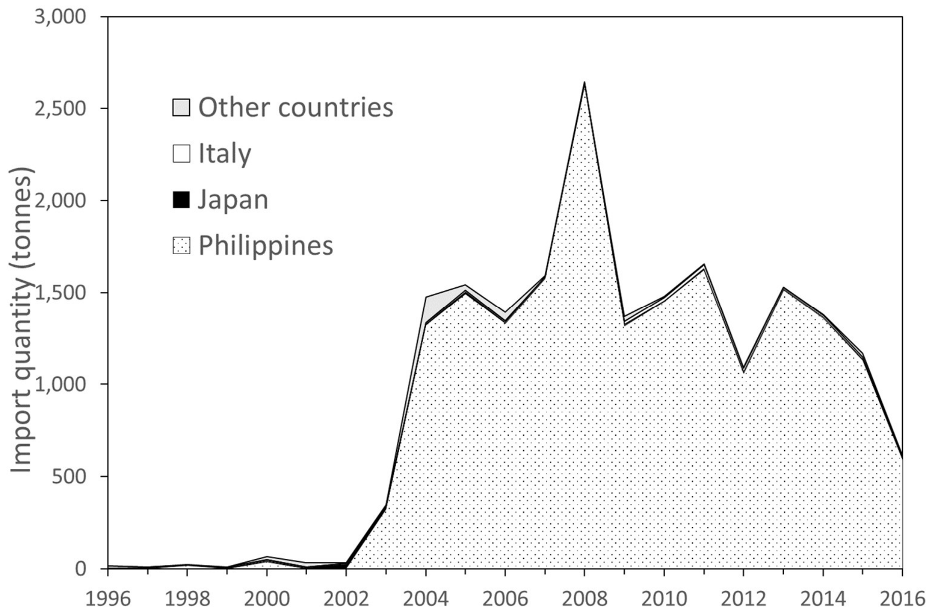

Figure 28. Changes in import quantities of raw coral and similar materials in Taiwan Province of China (Source: Global Trade Atlas)

5.3.2. Taiwan Province of China's imports and exports

In the years leading up to 2009, the annual quantity of raw coral exports from Taiwan Province of China amounted to 20–30 tonnes and even exceeded 140 tonnes at one time. However, this decreased to 2–3 tonnes from 2009, when stricter regulations governing precious coral fishing were implemented (Figure 29). In 2015, Japan was the recipient of almost all Taiwan Province of China's exports, which amounted to 31 tonnes. However, the unit price was low at USD 0.3 /kg, which makes it difficult to identify whether these were low-quality precious corals, similar materials or reef-building corals. In 2009, the value of Taiwan Province of China's exports decreased along with the quantity. In 2010, the annual export value was USD 306 717, a record low since 1996. However, export values recovered from 2011 on the back of the steep price rise, reaching a record high of USD 2 755 889 in 2016 (Figure 29).

Taiwan Province of China imported 50 tonnes of raw coral (excluding countries from which imports were priced below USD 10 /kg) in 2009, but annual imports subsequently decreased to under 10 tonnes in 2016 (Figure 30). While the quantity of imports declined, the value increased, peaking in 2015 at USD 66 997 158, the highest annual value since 1996. In recent years (2011–2015), the majority of Taiwan Province of China's imports have come from Japan and Italy, with the two countries accounting for a total of 89 percent of the import quantity during that period (Japan 30.3 percent, Italy 58.5 percent).

Export quantities of coral products followed a downward trend from 1996 before taking a steep dive from 2006 to 2010. In 2010, exports amounted to 2.14 tonnes, which is less than one tenth of the quantity recorded in 1985 (27.1 tonnes). However, while the quantity of exports declined dramatically, the value increased due to the price boom (Figure 31). For the period 2011–2015, Japan accounted for 28.9 percent of the quantity of Taiwan Province of China's exports, followed by Hong Kong SAR at 20.5 percent. Exports to China were low at 0.5 percent. As for export value, Japan made up 57.5 percent, while Hong Kong SAR and China accounted for 8.5 percent and 1.0 percent respectively.

Taiwan Province of China's imports of coral products have fluctuated largely from year to year. While quantities have followed a downward trend since 2012, values have increased (Figure 32). The import value of around USD 400 000 recorded in 2008 had shot up to USD 28 600 000 by 2016. A quarter of all Taiwan Province of China's coral product imports came from Vietnam. This is because coral is processed in Vietnam (Chang, 2015). During the decade 1996–2006, Taiwan Province of China's total exports of raw coral to Vietnam came to only 1.3 tonnes (1.1 tonnes in 1996 and 0.2 tonnes in 2006). However, exports of coral products to Vietnam during this time amounted to 31.2 tonnes. On the other hand, Taiwan Province of China also imported 37.4 tonnes of coral products from Vietnam. Thus, there is a correlation between coral product imports and exports in and out of Taiwan Province of China and Vietnam. We can deduct from this that Taiwan Province of China exports semi-processed coral products to Vietnam for finishing as processed items, and these are then imported back into Taiwan Province of China.

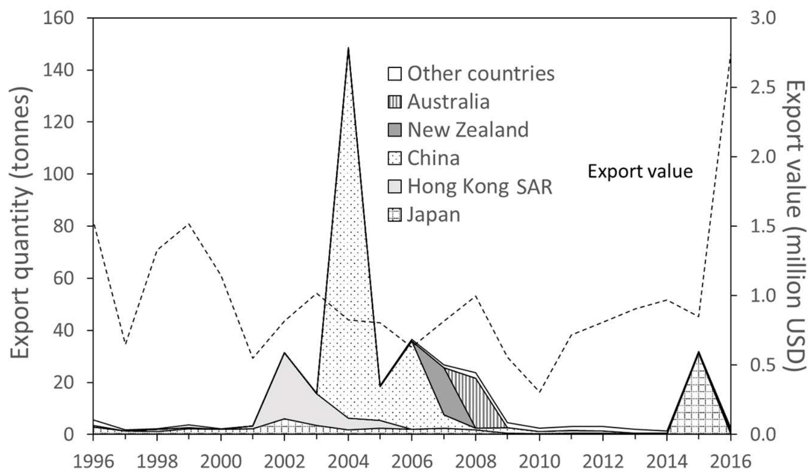

Figure 29. Changes in export quantities and values of raw coral and similar materials in Taiwan Province of China (Source: Global Trade Atlas)

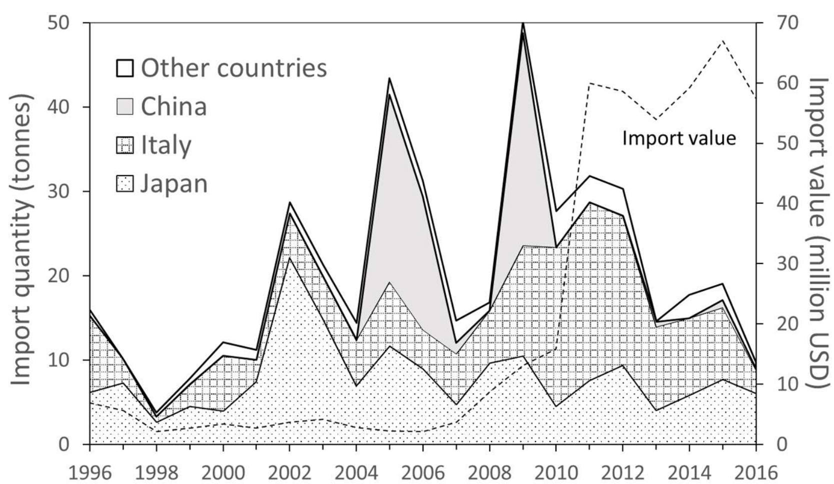

Figure 30. Changes in Taiwan Province of China's import quantities and values of raw coral and similar materials, excluding imports from countries exporting low-priced coral (under USD 10 /kg) to Taiwan Province of China (Source: Global Trade Atlas)

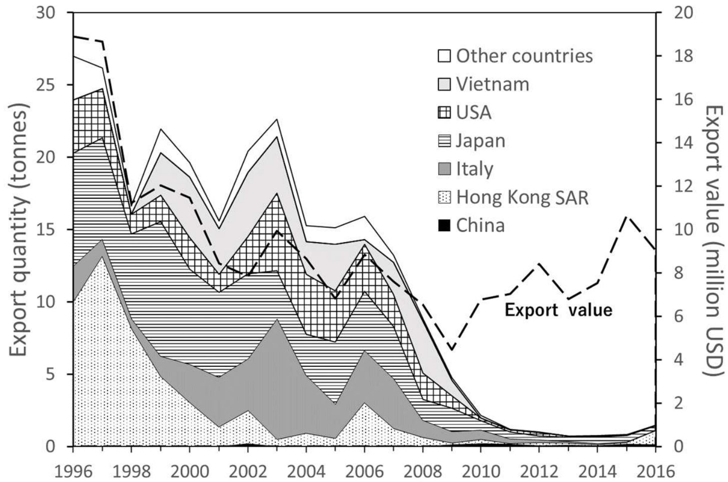

Figure 31. Changes in Taiwan Province of China's export quantities and values of coral products (Source: Global Trade Atlas)

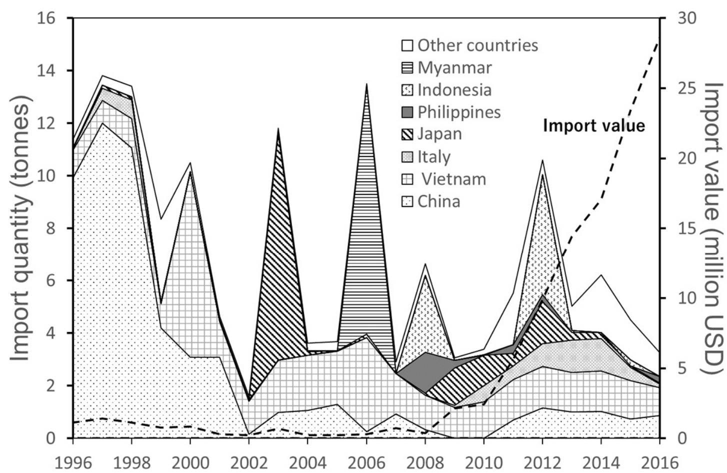

Figure 32. Changes in Taiwan Province of China's import quantities and values of coral products (Source: Global Trade Atlas)

5.3.3. Imbalance between Taiwan Province of China's imports and exports of raw coral and processed items

Figure 33 compares inbound volumes (landings plus imports of raw coral and coral products) with outbound volumes (exports of raw coral and coral products) for each five-year period from 1996. While there was little difference between the inbound and outbound volumes in 1996–2000, the outbound volume was greater in 2001–2005. The outbound volume subsequently decreased to approximately a third of the inbound volume in 2011–2015. This can be seen as a reflection of vigorous domestic consumption in Taiwan Province of China, as discussed below. This is the opposite situation to that of Japan's excessive exports (See Section I, 5.2.2. and Figure 26).

Chang (2015) studied the situation of precious coral circulation in Taiwan Province of China in recent years and pointed to China as the principal market. According to that report, precious coral prices rose rapidly on the back of the prosperous Chinese economy, shooting up from USD 900 /kg in 2009 to USD 7 500 /kg in 2014. The price boom began in July 2008 when Taiwan Province of China opened its doors to tourists from China.

Precious coral collected in Taiwan Province of China is sold at bid markets in Suao before being processed domestically (in part in Vietnam) and sold to Chinese tourists in Taiwan Province of China (Figure 27). In 2015, tourists from mainland China numbered over 4 000 000 and accounted for 40 percent of all tourists visiting Taiwan Province of China. While a change in policy towards China passed by a new government in 2016 resulted in a reduction in Chinese tourists, the number remained high in 2017, with around 2 700 000 tourists visiting Taiwan Province of China from China that year (Figure 34) (Tourism Bureau, M.O.T.C. Republic of China (Taiwan Province of China)). Tourists visiting Taiwan Province of China in tour groups from China in 2016 spent an average of USD 208.10 per person per day, and around 65 percent, or USD 136.16, of this was assigned to shopping. Furthermore, around 36 percent of their shopping expenditure, or USD 48.46, was on jewelry or jade (Tourism Bureau, M.O.T.C. Republic of China (Taiwan Province of China)). Compared to overseas tourists in Taiwan Province of China, whose average shopping expenditure came to USD 58.24 per person per day, tourists from China are highly motivated to buy jewelry. While it is not clear from the statistics, it is likely that many of them are buying coral items (Chang, 2015; field surveys conducted by Nozomu Iwasaki in February 2014 and March 2016).

Furthermore, precious coral is sold not only to Chinese tourists in Taiwan Province of China but also at fortnightly trade shows held at various places in China (Figure 27) (Chang, 2015).

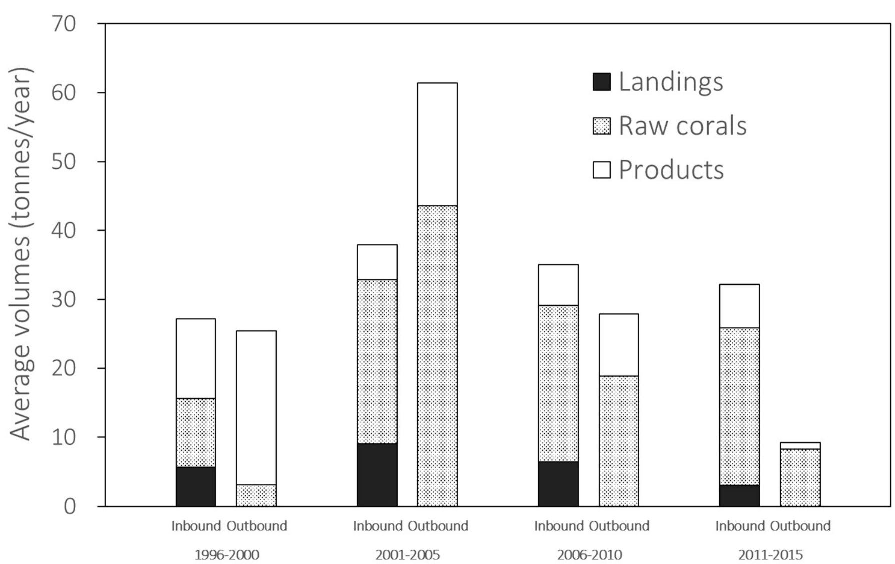

Figure 33. Inbound and outbound quantities of precious corals for Taiwan Province of China. Inbound: landings, imports of raw coral and coral products. Outbound: exports of raw coral and coral products.

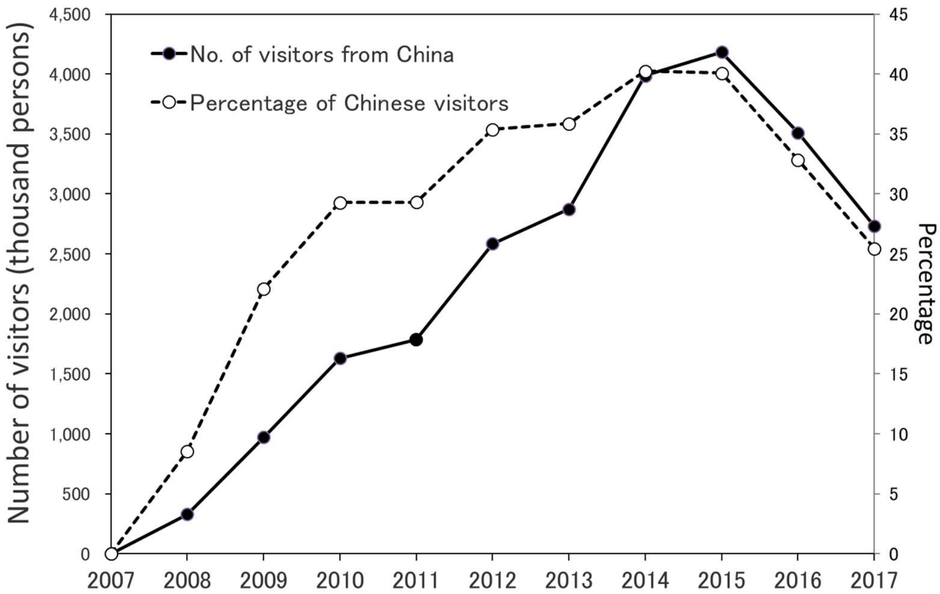

Figure 34. Changes in number of visitors from China and percentage of Chinese visitors in total number of visitors to Taiwan Province of China (Source: Tourism Bureau, M.O.T.C. Republic of China (Taiwan Province of China), Tourism Statistics Database)

6. TRADE WITHIN ASIA

6.1. Issues concerning the use of Global Trade Atlas

The following is an overview of raw coral trade in and out of Asia based on the Global Trade Atlas (HIS Markit) database. Average annual imports and exports during the years 2011–2015 have been used as an index.

Several points must be taken into account when using the statistics reported here. For countries such as China and Hong Kong SAR where no statistics are kept for coral, the statistics of trading partner countries have been used. In some cases, even for the same country, only import or export statistics are available, but not both. For these reasons, the figures do not always add up. For the Philippines, for example, only export statistics are available. In addition, since the statistics do not differentiate between reef-building and precious corals, these have been identified according to unit prices. Furthermore, although Tibet is a consumer nation for precious coral (The Nikkei, 2010), it is impossible to ascertain the trade situation there since no statistics are available.

6.2. Precious coral imports and exports

Figure 35 and Table 8 show quantities of exports to various countries from Japan and Taiwan Province of China, where precious corals are fished. Asian countries (China, Hong Kong SAR, India, Republic of Korea) and Italy are the principal recipients of exports from Japan and Taiwan Province of China, and, with the exception of Japan's exports to the Republic of Korea, the unit price of exports to these countries is high (USD 85.1–12 373.4 /kg) (Table 9).

The United States of America accounts for three quarters of Japan's average annual exports of 55.5 tonnes, and the Republic of Korea is the second largest recipient. However, the unit prices are low at USD 4.7 and USD 5.9 respectively (Table 9). While the specific contents of these exports are unknown, since reef-building corals are traded at even lower prices, as explained in the next paragraph, we can infer that these exports are low-priced precious corals.

As for the relationship between Japan and Taiwan Province of China (Figure 35), both of which harvest precious corals, while Japan exports high-priced items to Taiwan Province of China (USD 7 905.4 /kg, Table 9), it imports low-priced items in return (USD 19.9 /kg, Table 11).

India imports 36.5 tonnes annually from Asian and European countries at unit prices of USD 94.6–839.3 /kg (Table 10, 11), indicating that India is an important consumer of precious coral.

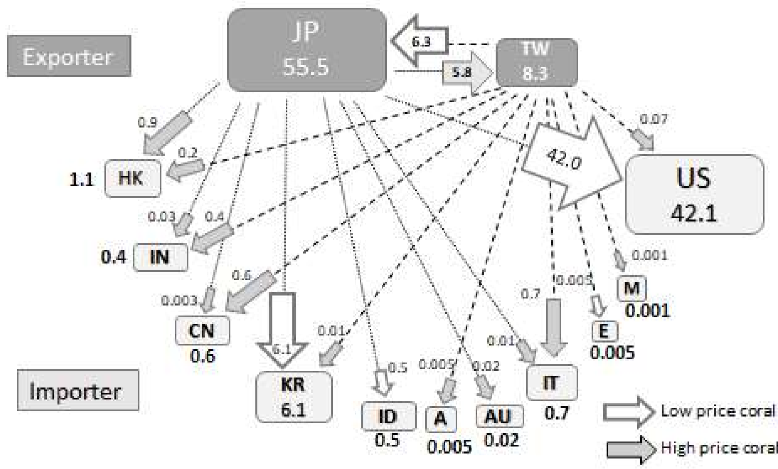

Figure 35. Flow of export quantities of coral from Japan and Taiwan Province of China.
Arrows: white arrows show flow of coral with a unit price of up to USD 80 /kg; dark arrows show flow of coral priced at over USD 80 /kg; figures show average annual quantity of coral (tonnes/year)
Abbreviations show the names of the countries and regions: JP, Japan; TW, Taiwan Province of China; HK, Hong Kong SAR; IN, India; CN, China; KR, Republic of Korea; ID, Indonesia; A, other Asian countries; AU, Australia; IT, Italy; E, Other European countries; M, Other Middle East countries; US, United States of America

6.3. Imports and exports of corals other than precious corals

Indonesia, India, the Philippines, Malaysia and Thailand export low-priced coral to various countries, with annual exports from Indonesia, in particular, exceeding 1 000 tonnes (Table 8). In view of the extremely low unit price of USD 0.4–3.3 /kg (Table 9), it is likely that these are reef-building corals rather than precious corals.

Both Japan and Taiwan Province of China import large quantities from the Philippines, averaging 481 tonnes and 1 339 tonnes respectively annually (Table 10). Since these are extremely low-priced items with a unit price of USD 0.2 /kg and USD 0.1 respectively, they are highly likely to be reef-building corals (Table 11).

The statistics used for this report make no distinction between precious corals, semi-precious corals and reef-building corals. In order to build an accurate picture of precious coral circulation, that of semi-precious and reef-buildings corals also needs to be clarified.

Table 8. Average export quantities (kg/year) of raw corals and similar materials from 2011 to 2015

			Exporter							Total
			Japan*	Taiwan Province of China*	India	Thailand	Malaysia	Philippines	Indonesia	
Importer	Asia	Japan*		6 332.0		62.4			53 749.0	60 143.4
		Taiwan Province of China*	5 769.2						792.2	6 561.4
		Hong Kong SAR	893.6	187.2	46 515.6	2.8			11 945.2	59 544.4
		India	27.6	400.4		30.4			99.8	558.2
		China	2.6	638.8	1 800.0	17 281.0			21 072.2	40 794.6
		Republic of Korea	6 096.0	8.2				9 700.0	10 596.4	26 400.6
		Vietnam				0.6	5 600.0			5 600.6
		Thailand							4.2	4.2
		Malaysia							4 567.8	4 567.8
		Philippines								0.0
		Indonesia	471.0							471.0
		Lao People's Democratic Republic				2 210.6				2 210.6
		Singapore					20 855.4		3 835.4	24 690.8
		Others		5.0		48.0			173.4	139.7
	Europe	Italy	9.8	659.6				3 875.0	41 914.4	46 458.8
		Germany				16.4		369.0	113 585.4	113 970.8
		Turkey						6 000.0	6 471.2	12 471.2
		United Kingdom of Great Britain and Northern Ireland						13 500.0	127 558.0	141 058.0
		Others		4.6		4.0		369.0	195 594.4	132 733.5
	Middle East			1.2		769.8			22 633.8	11 911.5
	Africa								14 255.2	2 375.9
	North America	United States of America	42 024.6	73.6				8 349.0	410 508.2	460 955.4
		Canada							31 920.8	31 920.8
	South America								12 624.6	1 803.5
	Oceania	Australia	227.0			37.0			130.4	394.4
		New Zealand							390.0	390.0
Total			55 521.4	8 310.6	48 315.6	20 463.0	26 455.4	42 162.0	1 084 422.0	1 188 131.1

* Fishery countries

Red figures; unit price is more than USD 80 /kg

Table 9. Average unit price (USD/kg) of export raw corals and similar materials from 2011 to 2015

			Exporter						
			Japan*	Taiwan Province of China*	India	Thailand	Malaysia	Philippines	Indonesia
Importer	Asia	Japan*		13.9		0.8			2.5
		Taiwan Province of China*	7 905.4						4.3
		Hong Kong SAR	2 034.9	85.1	1.7	0.1			3.1
		India	1 129.4	93.8		4.3			4.6
		China	4 003.4	434.3	3.0	1.6			0.3
		Republic of Korea	5.9	1 162.6				0.2	2.1
		Vietnam				221.3	0.4		
		Thailand							4.8
		Malaysia							3.4
		Philippines							
		Indonesia	20.0						
		Lao People's Democratic Republic				0.6			
		Singapore					1.1		5.4
		Others		17.0		0.0			3.4
	Europe	Italy	12 373.4	467.1				0.1	2.1
		Germany				13.6		11.4	2.4
		Turkey						0.2	4.8
		United Kingdom of Great Britain and Northern Ireland						0.3	2.6
		Others		9 040.8		74.7		11.4	2.4
	Middle East			1 002.0		0.4			2.9
	Africa								3.7
	North America	United States of America	4.7	11.6				0.6	4.6
		Canada							3.2
	South America								4.5
	Oceania	Australia	56.9			6.1			14.6
		New Zealand							5.5
Total			985.6	102.2	1.8	1.5	1.0	0.4	3.3

* Fishery countries

Red figures; unit price is more than USD 80 /kg

Table 10. Average import quantities (kg/year) of raw corals and similar materials from 2011 to 2015

			Importer								Total
			Japan*	Taiwan Province of China*	India	Republic of Korea	Thailand	Malaysia	Singapore	Indonesia	
Exporter	Asia	Japan*		6 888.2	43.8	9 617.0				1 820.0	18 369.0
		Taiwan Province of China*	1 625.2		516.0	29.4	70.0				2 240.6
		Hong Kong SAR		74.2	9 235.4	0.2					9 309.8
		India		466.2							466.2
		China		299.6	193.4	803.0	1 643.6		600.0		3 539.6
		Republic of Korea					2.8				2.8
		Vietnam	40.2		0.6	272.0					40.8
		Thailand			24.2			2 790.0			3 086.2
		Malaysia			1.2						1.2
		Singapore			22.0						22.0
		Philippines	480 764.0	1 338 576.4		140 259.0		24 747.2			1 984 346.6
		Indonesia	1 162.6	4 000.0	34.6	4 294.6	14.2	793.0	32.8		10 331.8
	Europe	Italy		13 293.4	20 077.2	0.2					33 370.8
		France		0.6	289.8				0.8		291.2
		Germany		24.4	213.0	11.6					249.0
		Spain		426.0	407.8						833.8
		Others		2.2	0.6	0.2	2.0				5.0
	Middle East	Tunisia	24.4	1 052.0	2 602.2						3 678.6
		Morocco		111.6	2 524.8						2 636.4
		Others			216.4	3.0					219.4
	N & S America			10.2	123.2	36.4					169.8
	Oceania & Pacific		304.4			19.8					324.2
	Other NES			63.6	10.6	21.0					95.2
Total			483 920.8	1 365 288.6	36 536.8	155 367.4	1 732.6	28 330.2	633.6	1 820.0	2 073 630.0

* Fishery countries

Red figures; unit price is more than USD 80 /kg

NES; not elsewhere specified

Table 11. Average unit price (USD/kg) of import raw corals and similar materials from 2011 to 2015

			Importer							
			Japan*	Taiwan Province of China*	India	Republic of Korea	Thailand	Malaysia	Singapore	Indonesia
Exporter	Asia	Japan*		7 635.1	839.3	12.6				19.9
		Taiwan Province of China*	19.9		147.6	426.8	5.3			
		Hong Kong SAR		56.1	10.0	1 525.0				
		India		219.3						
		China		1 600.2	109.5	0.7	1.5		0.3	
		Republic of Korea					1.4			
		Vietnam	27.2		293.0	10.4				
		Thailand			447.8			0.5		
		Malaysia			222.3					
		Singapore			242.7					
		Philippines	0.2	0.1		0.2		1.0		
		Indonesia	6.5	0.4	111.7	1.6	6.8	6.1	2.5	
	Europe	Italy		418.7	94.6	6 817.0				
		France		4 059.7	153.5				136.0	
		Germany		1 195.7	251.7	18.5				
		Spain		1 078.5	207.1					
		Others		504.5	317.3	69.0	32.0			
	Middle East	Tunisia	4 745.4	365.9	324.4					
		Morocco		934.7	202.7					
		Others			208.7	8.1				
	N & S America			128.7	315.3	36.2				
	Oceania & Pacific		11.9			84.1				
	Other NES			278.1	265.0	80.0				
Total			0.5	43.9	104.1	1.1	2.2	1.1	0.6	19.9

* Fishery countries

Red figures; unit price is more than USD 80 /kg

NES; not elsewhere specified

II BLACK CORALS AND SEMI-PRECIOUS CORALS

1. SPECIES IN CIRCULATION AND THEIR DISTRIBUTION

The skeletal axis of some members of Phylum Cnidaria is utilised as semi-precious coral (Table 12). Members of subclass Hexacorallia, order Antipatharia, are commonly known collectively as black corals. Antipatharians, which make up an order of over 235 recognised species including seven families and 43 genera, are distributed throughout the world (Wangner *et al.*, 2012). Among these, eight genera, according to Green and Shirley (1999), or 11 species, according to Parrish and Baco (2007), are in circulation on the market, but, with the exception of those produced in Hawaii, the names of the species are unknown. Red dyed coral of the family Isididae is circulating as bamboo coral or mountain coral. *Melithaea ochracea,* which is very porous, is filled with resin and traded under various names, including apple coral. The axis of most of these corals is aragonite or keratinous material, which distinguishes them from the calcite axis of precious corals.

Since the taxonomic classification of some items circulating on the market is still unknown, future research is awaited. Small items that have been cut and polished have lost the morphological characteristics that distinguish them, so identification is difficult. For this reason, non-destructive methods of analysis, such as Fourier-transform infrared spectroscopy (FTIR) and X-ray fluorescence spectrometry (XRF) are being used in research to identify items such as black corals and bamboo corals (Espinoza *et al.*, 2012; Iwasaki *et al.*, 2014). With a view to practical application, for customs clearance for example, methods of analysis need to be established and more analysis examples are needed, including those of artificial materials such as resin (See Section I, 2.7.).

2. RESEARCH ON BIOLOGY AND ECOLOGY

There is a dearth of knowledge concerning the biology and ecology of black and semi-precious corals in Asia. Concerning their distribution, some research has been conducted in Hawaii and the Pacific islands (Parrish and Baco, 2007), Japan (Kubota *et al.*, 2007; Kubota and Uchida, 2008) (Imahara *et al.*, 2014), and China (Shu-Hua *et al.*, 2009).

The only location where research has been conducted on semi-precious corals as a resource for fishery is in Hawaiian coastal waters (Grigg, 1993, 2001; Parrish and Baco, 2007); no such research has been reported in Asia.

Basic research is required for the protection and conservation of black and semi-precious coral resources.

3. FISHERY

Black corals are distributed throughout the world and have been utilised since ancient times in jewelry and medicine. They are collected in Asia, Hawaii, Latin America, the Caribbean Sea, the Mediterranean Sea and the Red Sea (Wagner *et al.*, 2012). Most are thought to be small-scale fishing operations (Grigg, 1993), but the details are unknown.

Black corals that are caught unintentionally in bottom-trawl fishing nets are known to be utilised (Wagner *et al.*, 2012). Investigation is needed to determine whether specialised fishery exists for black corals or whether they are collected only as bycatch in other types of fishery.

Table 12. List of Cnidarians used for jewelry

Grigg, 1993; Green and Shirley, 1999, Iwasaki and Suzuki, 2010

Subclass	Order	Family	Species	Common name	Distribution *	Characteristics of axis	CITES Appendix listing
Octocorallia	Alcyonacea	Coralliidae	see Table 1				III (Chinese 4 species)
		Melithaeidae	*Melithaea ochracea*	Apple coral, sponge coral	Japan, Malaysia, Indonesia, Australia	Porous, with keratinous nodes, red	
		Primnoidae	*Primnoa pacifica* *Callogorgia gilberti* *Narella* sp. *Calyptrophora* sp.	Alaska gold coral	North Pacific	Central part in the center of the cross-section of axis is carbonate, outer perimeter pate is Keratinous, without node, covered with scale shaped sclerites	
		Isididae	*Lepidisis olapa** *Acanella* sp.	Bomboo coral, mountain coral	Hawaii	Aragonite, white, with keratinous nodes, dyed with red colouring	
	Helioporacea	Helioporidae	*Heliopora coerulea* ‡	Blue coral	Pacific and Indian Ocean	Aragonite, blue	II (Helioporidae spp.)
Hexacorallia	Zoantharia	Parazoanthidae	*Savalia (=Gerardia)* sp. **	Gold coral	Hawaii, Emperor Seamount Chain	Keratinous, beige	
	Antipatharia ‡	Antipathidae	*Antipathes* cf. *curvata* (=*dichotoma*) ** *A. grandis* ** *Antipathes* spp. † *Myriopathes ulex* ** *Cirrhipathes anguina* ** *Aphanipathes* spp. † *Cladopathes* spp. † *Sibopathes* spp. † *Bathypathes* spp. † *Hexapathes* spp † *Leiopathes* spp. † *Schizopathes* spp. †	Black coral	Hawaii, Philippines Sea , Bay of Bengal Hawaii, Tonga Hawaii, Philippines Sea , Bay of Bengal Hawaii, Tonga, Japan	Keratinous, containing iodine, black	II (Antipatharia spp.)
Hydroidolina	Anthoathecata	Stylasteridae	*Stylaster* sp. ‡ *Distichopora* sp. ‡	Lace coral		Aragonite, porous	II（Stylasteridae spp.）

* The Biological Information System for Marine Life (BISMaL), Ocean Biogeographic Information System (OBIS)

** Collected in Hawaii water（Grigg, 1993; Parrish and Baco, 2007）

‡ Collected in Philippines water (Wells, 1981)

† Green and Shirley (1999)

3.1. Hawaii

Commercial black coral fishery has existed to a certain degree since 1958, and its landings from 1999 to 2005 amounted to 3 182 kg (Grigg, 2001, 2010). Gold coral fishery also existed from 1974 (Grigg, 2010). Their management plan was established by the Western Pacific Regional Fishery Management Council (Grigg, 2010; Parrish and Baco, 2007).

3.2. Indonesia

Black coral fishery has existed since at least the 18th century, and Bentley (1998) reports small-scale fishery, consisting of diving using an air compressor.

Grigg (2001) reports that the United States of America (primarily Hawaii) imported 26 000–617 000 processed black coral items a year from Taiwan Province of China in the 1980s, and during that time 70 tonnes of raw black coral, mostly from the Philippines, was processed in Taiwan Province of China.

3.3. The Philippines

Wells (1981) reports that black coral was collected around the Batan Islands, Palawan and elsewhere and mainly processed in Zamboanga for the tourist market and export. Blue coral, or *Heliopora,* is also fashioned into beads, and lace coral (*Stylaster* sp.*, Distichopora* sp.) is processed in Cebu for export.

In the past, much of the world's black coral supply was landed in the Philippines, but the collection of stony corals and the trade of precious and semi-precious corals were prohibited there in 1977 and stocks were required to be sold within 3 years (Wells, 1981; Tsounis, 2010).

3.4. China

When Yasuoki Tamura (Emeritus Professor, Kochi University) visited the workshop of a coral processing plant in Qionghai, Hainan province, in August 2006, the raw material he saw being processed there had been collected in China and Indonesia (Yasuoki Tamura, personal communication). The processed coral items were not precious coral, but dyed bamboo coral (Chang *et al.*, 2015).

4. INTERNATIONAL TRADE

Green and Shirley (1999) and Green (1999) report trade links between the largest importers and exporters of black coral from 1985 to 1997. Among Asian nations, Taiwan Province of China, the Philippines, China, Hong Kong SAR and Thailand were the principal exporters, and Japan and the Republic of Korea were importers (Table 13).

Three Asian trading corporations dealing in black coral are listed on Gmdu.net (global trades from here): Arts & Jewelry Collection (Hong Kong) Co.; MV Global Trading (Indonesia); and Dongguan Jiuyuan Jewelry Co., Ltd. (China). Huang and Ou (2010) cite a case of a Dominican company named Dominican Companies importing black coral into Cuba from Taiwan Province of China in 2005.

Seventy percent of the hard corals listed in CITES Appendix II that were traded internationally during the years 2000–2010 were supplied by Indonesia. Other suppliers included Fiji (10.3 percent), Tonga (5.3 percent), Australia (4.5 percent), and the Solomon Islands (4.2 percent) (Wood *et al.*, 2012).

Asian and Pacific nations are therefore playing an important role as suppliers. Close investigation is required to rule out the possibility that black and semi-precious corals are also being supplied.

Table 13. Trade links between the largest importers and exporters of black coral (cited from Green and Shirley, 1999, Table 12)

Importers	Trade from exporters recorded as number of pieces (1000s)										Trade from exporters recorded in units of weight (tonnes)						
	Taiwan Province of China	Philippines	China	Dominican Republic	Hong Kong SAR	USA	Cayman Islands	Thailand	Mexico	Total (1000s)	Taiwan Province of China	Philippines	Dominican Republic	Hong Kong SAR	USA	Cayman Islands	Total (tonnes)
USA	5 793	283		13	11		5	7	5	6 117	4	6	0.3	0.1		0.3	10.7
Japan	354	1			0.4	0.5				356	17				0.01		17
France	76	0.7								77							
Cuba	235									235							
Republic of Korea			40							40							
Germany	0.5	25				0.06				25.6							
Netherlands		15								15							
Greece		9								9							
Cayman Islands	0.1					7				7.1					0.8		0.8
Total	6 458	334	40	13	11.4	7.6	5	7	5	6 881	21	6	0.3	0.1	0.81	0.3	28.5

Notes: It was not possible to convert between weight and number of pieces for black coral so trade is expressed separately in tonnes and number of pieces traded, as recorded under CITES for the years 1985–1997. A large amount of black coral was traded with permits with did not specify the exporting (14 600 pieces and 421 tonnes) or importing nation (9 986 pieces).

5. DOMESTIC MARKETS

Black and semi-precious corals are utilised and in circulation on domestic markets in various Asian countries. Surveys are needed to assess the actual situation concerning fishery, processing and trade in each country.

5.1. Japan

In the past, items made from the black corals *Paracalyptrophora* sp. and Halipteridae sp. were produced as a traditional industry of Tottori Prefecture, but it is not known whether this is still the case. The raw materials were apparently collected as bycatch in bottom trawling nets (Oshima, 2017). Black coral was also processed in Shimane Prefecture, Hokkaido, and Nagasaki Prefecture, but the present situation is unknown.

In an exhibition held by International Jewelry Tokyo in January 2018, two or three Japanese traders were exhibiting bamboo coral (confirmed by Nozomu Iwasaki). In a poll conducted in March 2018 in Kochi city, three coral retailers dealing in black coral stated that all their stock was imported from Hawaii before all Antipatharians were listed in CITES Appendix II in 1981 (confirmed by Nozomu Iwasaki).

5.2. Taiwan Province of China

In December 2005 and September 2015, black coral was being traded at the Jianguo Holiday Jade Market in Taipei (confirmed by Nozomu Iwasaki).

5.3. The Philippines

In Dipolog, Mindanao, an infusion made by steeping black and bamboo coral fragments, along with resin and pyrite, in coconut oil is used to relieve pain and ward off evil (Iwasaki A. and Iwasaki N., 2011).

In February 2016, jewelry retailers in Manila and Makati were selling Japanese red coral jewelry they claimed to be from the Philippine islands of Mindanao (Emi Kainuma, personal communication). However, after purchasing three of the necklaces, Nozomu Iwasaki identified them as dyed bamboo coral and mollusc shell. In 2010, a retail store in Manila Airport was also selling Japanese red coral items produced in the Philippines (confirmed by Nozomu Iwasaki), but it is not known whether these were genuine red coral or imitations (See Section I, 3.3.4.).

5.4. Indonesia

In addition to fashion accessories, black coral is used in medications for joint pain and infertility (Bentley, 1998).

5.5. Thailand

In December 2003, Nozomu Iwasaki confirmed that dyed bamboo coral was being sold in Chiang Mai (Iwasaki *et al.*, 2014).

5.6. India

In March 2017, Masayuki Nishie confirmed in a survey of Pondicherry that dyed bamboo coral was being sold there (Iwasaki *et al.*, 2014; Iwasaki A. and Iwasaki N., 2011).

6. ILLEGAL OPERATIONS

News sources often report illegal collection of black corals, which are protected species listed in CITES Appendix II. Since illegal operations are proof of demand, cases of violations need to be reviewed not only from the point of view of species protection and resource conservation but also for the effective implementation of CITES. Several cases reported in the Philippines and India are discussed below.

6.1. The Philippines

In 2011, a large-scale poaching incident occurred in Cotabato, southern Mindanao. The damage caused by the poachers extended to a sea area five times the size of Metro Manila and resulted in the death of over 21 000 black coral colonies and 161 sea turtles (Philippine Daily Inquirer, 26 May 2011, 28 May 2011; Cebu daily News, 14 June 2011). A presidential spokesperson issued a warning concerning the illegal collection and export of black corals in the Philippines and called for protection of the corals, including a boycott of processed items. According to officials, black corals are collected in the Moro Gulf and exported to China and Europe (Philippine Daily Inquirer, 3 June 2011).

In May 2011, customs officials seized black coral and stuffed green turtles to the value of PHP 35 million, which a Chinese trader had shipped from Cotabato to Manila with the intention of smuggling them out of the country (Philippine Daily Inquirer, 14 June 2011).

In April 2014, twelve Vietnamese nationals aboard a Malaysian-registered fishing vessel were arrested for poaching black corals, hawksbill turtles, grouper and other marine creatures off Pangutaran Island in Sulu (Department of Environment and Natural Resources, Philippines web page).

6.2. India

All corals, including Antipatharians (black corals), are protected under the Indian Wildlife Protection Act, 1972. However, these are collected and traded illegally. Researchers who conducted a survey of items in circulation on the market identified 22 coral species, although no black corals were among them (Prakesh *et al.*, 2017).

7. PROTECTION IN THE HIGH SEAS OF THE PACIFIC OCEAN

Fishing for cold-water corals (Alcyonacea, Antipatharia, Gorgonacea, Scleractinia) is prohibited in the sea area under the jurisdiction of the North Pacific Fisheries Commission (NPFC) (See Section I, 3.3.5.).

8. QUESTIONNAIRE-SURVEY OF MARINE-LIFE RESEARCHERS

From 2004 to 2008, a questionnaire-survey was conducted of around 40 marine-life researchers from Southeast Asian and Pacific countries regarding the fishery and utilization of precious and semi-precious corals (Iwasaki, unpublished). Their responses were as follows:

Vietnam:	Precious corals are fished (See Section I, 3.2.4.).
Fiji:	Black corals are found in western and southern Fiji, particularly at depths of 25–30 m in the Yasawa and Kadavu reefs. Black coral earrings, pendants, bracelets, etc. are sold at the Handicraft Centre in Suva.
Vanuatu:	Black corals are distributed at depths of 5–20 m off Mystery Island (Aneityum Island). Semi-precious corals are collected with iron rods. They are sold as religious objects and tourist souvenirs.
The Philippines:	Fishing is prohibited, but the regulations are neither respected nor adequately enforced around many of the islands, so corals are fished by individual fishermen and fishing companies.
Malaysia:	Fishing is prohibited.
Thailand:	Collection, sale and possession is prohibited (Tsounis, 2010). Coral is sold in the form of medicines and fashion accessories, such as bracelets, etc. *Antipathes*, a genus of black coral, lives in shallow waters in the Gulf of Thailand and the Andaman Sea at depths of up to 10 m.

III POSSIBILITIES AND PROPOSALS FOR SUSTAINABLE EXPLOITATION

1. SPECIES PROTECTION AND RESOURCE CONSERVATION

Corals are now recognised as vulnerable marine ecosystems, including some endangered species, and various countries have introduced measures for their protection and conservation (Table 14).

In March 2017, the Japanese Ministry of the Environment included Japanese red coral, pink coral and white coral on the Red List of Threatened Species under the classification "Near Threatened". While these species' risk of extinction is not significant at present, the reason for the classification was given as increased fishing pressure as a result of dwindling resources for commercial fishing, greater numbers of authorised fishing vessels and poaching by foreign boats. Since Japanese red coral is not threatened with extinction but depletion of resources is feared, the Fisheries Agency publication "Basic Data on Rare Wild Aquatic Life in Japan" classifies it under Category 3: Diminishing Species (species that are clearly decreasing in number) (Fujioka, 1996).

In China, precious corals (*Corallium* spp.) are classified as Class 1 Protected Species and coral fishing is prohibited (See Section I, 3.3.3.).

For the conservation of vulnerable marine ecosystems (VMEs), the North Pacific Fisheries Commission (NPFC) prohibits fishing for cold-water corals (Alcyonacea, Antipatharia, Gorgonacea, Scleractinia) in the sea area under its jurisdiction and also prohibits bottom-trawl fishing in C-H Seamount and the south-eastern part of Koko Seamount (North Pacific Fisheries Commission, 2017) (See Section I, 3.3.5.).

Precious corals and black corals have still not been included on the Red Lists of Taiwan Province of China, the Republic of Korea, the Philippines, India and Sri Lanka (National Red List web page).

2. POSSIBILITIES FOR SUSTAINABLE EXPLOITATION

Iwasaki *et al.* (2012) demonstrated that sustainable exploitation can be achieved by implementing selective fishing methods using ROVs to collect only colonies of a certain size and establishing non-fishing periods to allow small colonies time to grow (See Section I, 2.3.). For non-selective fishing using tangle nets, dividing fishing grounds into sections and fishing them in rotation is thought to be effective. The combination of regulations, such as suspension of fishing operations during the breeding season and the establishment of marine protected areas (MPA), further enables effective resource management (Iwasaki and Suzuki, 2010; Chang *et al.*, 2012).

In the current climate of insufficient scientific fishery management, a new initiative has been launched by coral fishermen aiming for sustainable operations. In March 2017, coral fishery (red, pink and white corals) using ROVs in Okinawa and Kagoshima gained chain-of-custody (CoC) certification from Marine Eco-Label Japan (MEL), a body that certifies fisheries proactively seeking protection of resources and ecosystems. The selective fishing method of collecting only colonies of a certain size received a positive evaluation from MEL for its contribution to the conservation of precious coral resources and their habitat (Japan Fisheries Resource Conservation Association web page).

Table 14. Precious and black coral species listed in CITES Appendix and the other lists and domestic regulations

	Corals	CITES Appendix	Japan		China	India	Indonesia	Philippines	North Pacific Fisheries Commission
			Red list	Basic Data of Rare Wild Aquatic Organisms	The wildlife under special state protection	The Indian Wildlife (Protection) Act 1972	Regulation No. 7 concerning the preservation of wild plants and animals	Philippine Fisheries Code & Presidential Decree No.1219	CMM 2017-05
Precious coral	Japanese red coral *Corallium japonicum*	III (China)	Near Threatened	3rd rank, declining species	1st class *				
	Pink coral *Pleurocorallium elatius*	III (China)	Near Threatened		1st class *				
	White coral *Pleurocorallium konojoi*	III (China)	Near Threatened		1st class *			Prohibition of fishing and exportation	Prohibition of fishing **
	Miss coral *Hemicorallium sulcatum*								
	Deep-sea coral, mid coral *Pleurocorallium secundum*	III (China)							
Black coral	Antipatharia spp.	II				Schedule I	Genus *Antipathes* is listed		
Remarks					Prohibition of coral fishing	Rigorous protection	UNEP-WCMC. 2014		C-H Seamount and southeastern part of Koko Seamount are closed

* Registered as *Corallium* spp.

** Alcyonacea, Antipatharia, Gorgonacea, Scleractinia

3. ISSUES AND PROPOSALS

3.1. Advances in research

In Japan and Taiwan Province of China, scientific research studies on precious coral resources are long overdue. The data – such as biomass, recruitment, and mortality rate – necessary to construct a population dynamics model to estimate future resources and landings is almost non-existent; only growth rate has been defined.

Studies are also needed on the coral nets used in Japan and Taiwan Province of China and their impact on precious corals and the sea floor environment. Coral tangle-nets, which consist of several flat nets attached to a stone weight or horizontal bar, are constructed differently from bottom-trawl nets. While tangle nets move across the sea floor in a more-or-less linear fashion, bottom-trawl nets spread out over a wider surface. In this regard, tangle nets are thought to cause less disturbance on the sea floor. However, the impact of numerous fishing boats repeatedly dragging nets over a limited area is expected to be considerable (See Section I, 3.1.). Research is therefore required on the way coral nets impact the sea floor.

Basic biological information concerning, for example, the taxonomy and distribution of species such as black corals and bamboo corals is lacking. The actual situation regarding fishery of these species is also unknown. There is an urgent need to initiate biological and ecological research in order to ascertain the extent of fishing activities for the evaluation of sustainable use.

3.2. Collection of fishery and trade data

Data concerning precious coral production is essential in order to determine the condition of resources, however no statistics are compiled in Japan. Fisheries are required to report their landings to the prefecture, but the information is not disclosed. For this reason, the FAO Global Production Statistics for Japan are inaccurate (See Section I, 4.1.). To enable access to accurate production figures, action must be taken to provide access to the data.

While precious coral fishery is prohibited in Chinese waters, Chinese production of red, pink and white corals is included in the FAO Global Production Statistics (Table 5; See Section I, 3.3.3.). Close examination of the actual situation in China is required.

The current HS codes make it difficult to ascertain imports and exports of precious corals and black corals (See Section I, 5.1.1.). Since the black corals and precious corals listed in CITES Appendices II and III require certification for customs clearance, it would be possible to collect data on the country of origin, species and volume at the same time. A convenient method of data collection needs to be devised in order to compile accurate trade statistics.

3.3. Reinforcement of resource management

The small proportion of live coral landed in Kochi Prefecture (Japan) and Taiwan Province of China indicates deterioration of resources. The increased number of fishing boats in operation due to the precious coral boom has put pressure on precious coral resources in Kochi Prefecture (See Section I, 4.3.4. and 4.4.). For their protection and conservation, urgent action is required to reduce the number of vessels. In Kochi Prefecture, the number should be reduced at least to pre-2010 levels. Fishing activities also need to be controlled by setting upper limits on the permitted number of total fishing days (the number of boats multiplied by the number of days in operation).

3.4. Establishment of Regional Fisheries Management Organizations (RFMOs)

The distribution range of precious corals includes the coastal waters of Japan, Taiwan Province of China and southern China. International cooperation is essential for marine life conservation and management that extends beyond the boundaries of each nation's territorial waters or EEZ.

In the East China Sea, which is the principal precious coral fishing ground, the Japan-China Fishery Agreement specifies three sea areas: a Japan-China provisional waters zone, an intermediate zone and a zone south of 27°N. In the Japan–China provisional waters zone, both nations are permitted to operate within limits set by mutual agreement on vessel numbers and catch quotas. No distinction is made between precious corals and other species, and there are no individual catch quotas according to species. In the intermediate zone and the sea area south of 27°N, operations are permitted without mutual agreement. Elsewhere in the East China Sea, Japanese and Taiwanese fishing boats are operating together under a bilateral agreement on private fishing activities (Fisheries Agency, Japan web page).

In order to eliminate illegal, unreported and unregulated (IUU) fishing and conserve precious coral resources, a regional fishery management organization composed of representatives from Japan, Taiwan Province of China and China needs to be established to oversee operations in the East China Sea and other sea areas fished by the three nations (Chang *et al.*, 2012; Chang, 2015).

3.5. CITES Appendix listing and challenges

From around 2010, when precious coral prices skyrocketed on the back of the booming Chinese economy, the expansion of Japanese fishing grounds and the greater number of vessels in operation have increased the burden on precious coral resources (See Section I, 5.2.3.). Poaching by Chinese fishing vessels increased temporarily. This classic example of species or resources threatened by economic activity seems a likely candidate for CITES Appendix listing. However, there are several issues concerning the validity and effectiveness of this.

Despite the fact that Chinese precious corals are listed in Appendix III, there was large-scale poaching by Chinese fishing boats around the Ogasawara Islands in 2014 (See Section I, 3.4.). This is testimony to the fact that a CITES Appendix listing does not deter poachers. It may even be counter-productive since it is evidence of rarity and may therefore boost consumer demand and increase fishing pressure on resources. Furthermore, regulations at the fishery level are more effective than trade regulations when it comes to conserving resources. For coral reef conservation, Tissot *et al.* (2010) suggests that the large volume of corals circulating on the markets despite being listed in CITES Appendix II (1.5 million live stony corals, over 2 million kg of dead coral, 30–50 metric tonnes of red and black corals) indicates a need for alternative approaches. Further discussion is needed on these issues to ascertain the validity and effectiveness of CITES Appendix listing.

CITES documentation and a certificate of origin issued by the Taiwanese Fisheries Agency and Bureau of Trade are required for the exportation of coral from Taiwan Province of China to China. However, while exports to China continue to increase, applications for the necessary CITES documentation have decreased. Due to the connections (guanxi) typical of Chinese society, coral products can still be transferred from Taiwan Province of China into the Chinese market via private connections between brokers. In addition, larger numbers of Chinese tourists buying precious corals in Taiwan Province of China makes customs inspection of coral products more difficult (Chang, 2015). Since many coral items are small, it is relatively easy to carry or send them out of the country without following the procedure laid down by CITES. The feasibility of implementing CITES-based import-export procedures therefore needs to be verified.

When importing or exporting items listed in Appendix II or III, the species must be identified. The identification of coral items relies on their colour: a red axis is identified as red coral, a pink axis is pink coral and a white axis is white coral. However, there are large colour variations within the same species. For example, Japanese red coral can be anywhere from dark blackish red to pale whitish red. This often makes it difficult to differentiate between red coral and pink coral. The red and pink corals produced in Japan and Taiwan Province of China have a white spot in the centre of a transverse section of the axis that distinguishes them from Mediterranean red coral, but identification is difficult in processed items when the white spot has been removed. Although identification of coral species is unreliable, the problem would be solved if all species of Coralliidae were simply listed as Coralliidae spp. in the CITES Appendix, as proposed at the 14th and 15th meetings of the Conference of the Parties to the Convention on Biological Diversity (COP 14 and 15). However, bamboo corals, whose white axis is dyed red and utilised as semi-precious coral, are not listed in the Appendix so they can be traded according to regular import-export procedures. For the implementation of an export system based on CITES, a reliable method of species identification needs to be established (See Section I, 2.7.).

Four species of precious coral produced in China are listed in CITES Appendix III. The distribution area of one of those species, *Pleurocorallium secundum* (listed as *Corallium secundum*) is the coastal waters of Hawaii and Midway Atoll (Section I, 2.2.). Close examination is required to ensure the accuracy of information concerning species listed in the Appendix III.

3.6. Proposals for ethical coral

Regulations governing coral fishery differ between Japan and Taiwan Province of China. Differences also exist in the management methods of regional authorities. I propose that the production area of precious corals should be disclosed at every stage, from the time the coral is landed and sold as raw material until it finally reaches the retailer as a finished product. This would not only enable traceability and certify that the coral was collected in compliance with the regulations of each production area, it would effectively eliminate illegally fished coral, which would, in turn, help deter IUU fishing such as poaching.

A system needs to be established whereby coral produced by fisheries with MEL certification (See Section III, 2) is labeled accordingly. I propose that coral collected by legal fishing methods with consideration for resources and the environment be traded under the name “ethical coral”. Above all, in order to achieve these initiatives, fishing pressure must be reduced and its excessive burden lifted from resources.

CONTRIBUTORS

Chih-Shin CHEN, Associate Professor, Dean of Institute of Marine Affairs and Resources Management, National Taiwan Ocean University, Republic of China

Joanna Dorothy HARE, Associate Professor, Faculty of Humanities and Social Sciences, Kochi University, Japan

Janet S. ESTACION, OIC Director, Institute of Environmental and Marine Sciences, Silliman University, Philippines

Emi KAINUMA, Associate Professor, Faculty of Geo-Environmental Science, Rissho University, Japan

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This document has been prepared by the Food and Agriculture Organization of the United Nations (FAO), in accordance with a request from CITES (CoP Decision 17.191 on Precious corals, for consideration at the 30th meeting of the Animals Committee). The report concerns precious (red, pink, white and black) coral species within the hexacoral order Antipatharia, and the octocoral family Coralliidae. According to the requirements of CITES Decision 17.191, the study considers all available data and information on the biology, population status, use and trade in each species, including the identification of gaps in such data and information. It contains information on the management and harvest regulation schemes for these coral species, with the aim of considering the effectiveness of their management and conservation. The report intends to inform the CITES parties of the status of the management and trade of precious corals, in order to provide guidance on the actions needed to enhance the conservation and sustainable use of precious corals.

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