PLASMAR

Bases para la planificación sostenible de áreas marinas en la Macaronesia

Analysis of the professional fishing sector in Macaronesia under MSFD

Finding the Balance of Blue Growth Sustainable Development within Ecosystem Approach (Act. 2.1.1 c&d)

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I. Introduction

Report carried out within the framework of the '*Activity 2.1.1.c&d. Finding the balance of Blue Growth Sustainable Development within Ecosystem Approach'. Establishment of the BG index. Standard framework for EIA*. The purpose of this report is to know the influence of the professional fishing sector of the Canary Islands, Madeira and Azores, on the descriptors and criteria of Good Environmental Status within the framework of the Marine Strategy Framework Directive.

Glossary

ALDFG: Abandoned, lost or otherwise discarded fishing gears.

B: Biomass.

B_{MYS}: Biomass needed to provide maximum sustainable yield.

BG: Blue Growth.

CECAF: Committee for the Central Easten Atlantic Fisheries.

CFP: The *Common Fisheries Policy of European Union.*

CPUE: The catch per unit of effort.

DCOIT: Chemical compound named '4,5-dichloro-2-n-octyl-4-isothiazolin-3-one'.

DMEM: MSFD in Spanish.

Di: Descriptor number "i" of GES (MSFD).

DiCj: Criterio number "j" in quality descriptor number "i" (MSFD).

DQEM: MSFD in Portuguese.

DRAM: Regional Directorate for the Sea Affairs in Azores Islands.

E: Exploitation Rate.

ECOAQUA: Instituto Universitario de Acuicultura y Ecosistemas Marinos Sostenibles of the ULPGC (Research Institute of ULPGC).

EIA: Environmental impact assessment.

EMODnet: European Marine Observation and Data Network.

ENV: Environmental.

ERDF: European Regional Development Fund.

EU: European Unión.

EwE: Ecopath with Ecosim.

F: Fishing mortality.

$F_{0.1}$: The fishing mortality rate at which the marginal yield -per-recruit is only 10 percent of the marginal yield-per-recruit on the unexploited stock.

$F_{40\%}$: Type of fishing mortality. It is the fishing mortality rate that reduces the spawning stock per recruit to 40 percent of that which would exist in the absence of fishing.

FAO: Food and Agriculture Organization of the United Nations.

FiB index: The Fishing in Balance index.

Fmax: Type of fishing mortality (F). Considering the long-term yield per recruit (Y/R), as a function of F: Fmax is the point of the curve, Y/R against F, where Y/R is maximum.

Fmed: Type of fishing mortality. Fishing mortality rate F corresponding to a SSB/R (recruitment) equal to the inverse of the 50th percentile of the observed R/SSB.

F_{MSY}: Type of fishing mortality (F). F_{MSY} is defined as being the value of F which produces the maximum yield in the long-term. It is necessary to select an S-R relation to estimate F_{MSY}. This point is different from Fmax.

GES: Good Environmental Status (MSFD).

GESAMP: The Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection.

GMR Canarias, S.A.U.: Gestión del Medio Rural de Canarias, sociedad anónima unipersonal.

ICCAT: The International Commission for the Conservation of Atlantic Tunas.

ICES: International Council for the Exploration of the Sea.

IEO: Instituto Español de Oceanografía (Spanish Institute of Oceanography).

IMPOSEX index: Index that measures ‘the imposition of the male sex in female individuals of the species for contaminating effects’.

IUCN: International Union for the Conservation of Nature.

IUU: Illegal, unreported and unregulated fishing.

JEI: Juvenile exploitation index.

LBI: Length-based indicators.

LCA: Life Cycle Assessment.

LRPs: Limit Reference Points.

MA: Marine activity.

MAPA: Spanish Ministry of Agriculture, Fisheries and Food.

MARPOL: The International Convention for the Protection of Marine Pollution from Ships.

MCRS: Minimum conservation reference size.

MPA: Marine protected area.

MSFD: Marine Strategy Framework Directive.

MSY: Maximum Sustainable Yield

MTL: Mean Trophic Level.

NE Atlantic: Northeast Atlantic.

OSPAR: The Convention for the Protection of the Marine Environment of the North-East Atlantic.

P/B: Production per unit of biomass (EwE).

PLASMAR: Acronym of the project named ‘Bases para la planifiación sostenible en areas marinas de la Macaronesia’.

POMAC: Programa Operativo de Cooperación Territorial Madeira-Açores-Canarias.

QD: Quality Descriptor of GES (MSFD).

REPESCAN: Scientific report named ‘Memoria científico-técnica final sobre el Estado de los Recursos Pesqueros de Canarias’.

SCRS: Standing Committee on Research and Statistics.

SPR: The spawning biomass per recruit.

SSB: Spawning stock biomass.

TAC: Total Allowable Catch.

TCMTB: Chemical compound named 2-(thiocyanomethylthio) benzothiazole.

TL: Length total.

TRPs: Target Reference Points

ULPGC: Universidad de las Palmas de Gran Canaria (University of the Canary Islands).

UNEP: United Nations Environment Programme.

WGDEEP: Working Group on the Biology and Assessment of Deep-sea Fisheries Resources (ICES).

II. Methodology

The work is based on a bibliographic review following instructions common to all the partners of the PLASMAR Project and aimed at the sectors under analysis in each case. The instructions were as follows:

State of art on maritime activity environmental impacts and possible env. solutions, should be gather mainly through the review of scientific and technical publications; if possible, to deliver a workshop with local/international experts and/or interviews with experts. It will be followed structure given by Marine Strategy Framework Directive on Good Environmental Status (GES). Need to identify maritime activity environmental pressures/impacts related to the GES (Quality Descriptor/criteria elements). In the analysis need to be considered construction phase (e.g. raising aquaculture facility, building offshore wind energy facility) and operational phase. For this task is necessary to fill 14 tables (one table per Quality Descriptor + 4 tables on biodiversity) that are included below. Tables include criteria elements, in detail described in ***COM 2017/848/EU on Good Environmental Status (GES)****:*

Example:

QD2 Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems							
QD	Criteria (element)	CODE Criteria	Env. Impact	Env. impact spatial extent	MA pressure solutions	Impact mitigation measures	Monitoring method
QD2	Newly-introduced non-indigenous species.	D2C1					
	Abundance and spatial distribution of established non-indigenous species, contributing significantly to adverse effects on particular species groups or broad habitat types	D2C2 — Secondary					
	Proportion of the species group or spatial extent of the broad habitat type which is adversely altered due to non-indigenous species, particularly invasive non-indigenous species.	D2C3 — Secondary					

Table fields values:

1. ***__Environmental impact,__*** *values:* ***YES/NO****; if* ***YES*** *please fill rest of table fields. Please explain impacts (significant adverse effects) on the environment in additional text below the table, including the references and sources of information, describing as much as possible (through review of available* ***scientific and technical publications****) the following factors:*

    - ***Description of the impact - significant adverse effect on the environment (if more than one, please include relevant one or all);***
    - ***Direct / indirect impact;***
    - ***Probability /Intensity/complexity of the impact;***

- ***Expected onset, duration, frequency and reversibility of the impact;***
- ***Expected cumulation with other types of adverse effects*** *linked to this maritime activity,*
- *Currently* ***relevant for the Macaronesia,*** *or expected to be relevant in the future (due to expected development of the MA).*

2. ***Environmental impact spatial extent****,values:*

    a. ***Impact area is less than operative maritime activity area***
    b. ***Impact area equal to operative maritime activity area;***
    c. ***Impact area broader than operative maritime activity area;***

    Please explain in additional text below the table, including references and sources of information. If available, provide detail on magnitude of impact and the quantitative values on spatial extent of operation area (surface, volume) for the MA, and spatial extent of impact area.

3. ***Maritime Activity MA pressure solution,*** *values:* ***YES/NO****;*

    if ***YES*** *please explain identified solution in additional text below the table, including the references and sources of information. When possible, and according to the review of available* ***scientific and technical publications****, identify:*

    a. *If solution is* ***envisaged to avoid, prevent, reduce or offset the pressure;***
    b. ***If measure is a reasonable alternative in terms of technical complexity, cost and expected success in reduction of impact;***
    c. ***If the MA pressure solution is relevant for the Macaronesia.***

4. ***Impact mitigation measures,*** *values:* ***YES/NO****;*

    if ***YES*** *please explain measures in additional text below the table, including the references and sources of information. When possible, and according to the review of available* ***scientific and technical publications****, identify:*

    a. *If solution is* ***envisaged to avoid, prevent, reduce or offset the impact/adverse effect;***
    b. ***If measure is a reasonable alternative in terms of technical complexity, cost and expected success in reduction of impact;***
    c. ***If the impact mitigation measure is relevant for the Macaronesia.***

5. ***Monitoring method*** *available: values:* ***YES/NO****;*

    if ***YES*** *please describe briefly the method in additional text below the table, including the references and sources of information, and identify:*

    a. ***The viability of the monitoring method*** *in terms of cost-effectiveness, complexity and relevance for the Macaronesia.*
    b. *Should monitoring start before the construction phase or with the operational phase?*

Acceptable value for all fields ***No data & information, needed further research****. This can only be used when there are no available publications or other related information.*

Search of technical and scientific literature should be done on global level for each MA. Only the information relevant for the Macaronesia (applying or that might apply in the future) is to be documented and included in the analyses (tables and reference text).

With this study, should be applied "medium" level of details, e.g. analysis should document examples with species, but only if relevant. It is not necessary to deliver analysis for each specie, or each possible event - please apply common sense, e.g. providing information for specie groups, types of events, only when relevant.

III. Results

1 Analysis >> QD1 Species groups of birds, mammals, reptiles, fish and cephalopods (relating to Descriptor 1)

QD1 Species groups of birds, mammals, reptiles, fish and cephalopods (relating to Descriptor 1)							
QD	**Criteria (element)**	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
QD1	The mortality rate of birds, mammals, reptiles and non-commercially-exploited species of fish and cephalopods from incidental by-catch is below levels which threaten the species, such that its long-term viability is ensured.	D1C1	yes	less	yes	yes	yes
	The population abundance of the species is not adversely affected due to anthropogenic pressures, such that its long-term viability is ensured. Member States shall establish a set of species representative of each species group, selected according to the criteria laid down under 'specifications for the selection of species and habitats', through regional or subregional cooperation. These shall include the mammals and reptiles listed in Annex II to Directive 92/43/EEC and may include any other species, such as those listed under Union legislation (other Annexes to Directive 92/43/EEC, Directive 2009/147/EC or through Regulation (EU) No 1380/2013) and international agreements such as Regional Sea Conventions.	D1C2	yes	broader	yes	yes	yes
	The population demographic characteristics (e.g. body size or age class structure, sex ratio, fecundity, and survival rates) of the species are	D1C3	yes	need further research	need further research	need further research	need further research

	indicative of a healthy population which is not adversely affected due to anthropogenic pressures. Primary for commercially-exploited fish and cephalopods and secondary for other species						
	The species distributional range and, where relevant, pattern is in line with prevailing physiographic, geographic and climatic conditions. Primary for species covered by Annexes II, IV or V to Directive 92/43/EEC and secondary for other species.	D1C4	yes	need further research	need further research	need further research	need further research
	The habitat for the species has the necessary extent and condition to support the different stages in the life history of the species. Primary for species covered by Annexes II, IV and V to Directive 92/43/EEC and secondary for other species	D1C5	yes	need further research	need further research	need further research	need further research

1.1 D1C1

By-catch can be defined as "part of a catch of a fishing unit taken incidentally in addition to the target species towards which fishing effort is directed and some or all of it may be returned to the sea as discards" (FAO, 1999).

These types of catches affect, to a large extent, species without commercial interest such as mammals, turtles and birds (Alverson et al., 1994), which could not only affect the sustainability of these species but also cause an impact in the structure and functioning of the ecosystem, since other fish and invertebrates that are not targeted by the fishery are caught and their populations can be compromised.

The accidental capture of birds in longlines and gillnets is one of the causes of the population decline of several species of albatrosses and shearwaters (Brothers et al., 1999; Anderson et al., 2011) to the point where some of them are listed as Vulnerable or Critically Endangered on the Red List of the World Conservation Union (IUCN). However, there is barely information for the Macaronesian region despite being a breeding place for several species of the family Procellariidae, including the largest population of Calonectris diomedea in Ilhas Selvagens. In the three archipelagos, the overlap between the fishing zones and the distribution and feeding areas of some bird species is scarce, which may explain the absence or low proportion of captured specimens (González-Solís et al., 2007; Lloret Capote et al., 2012; García-Barcelona et al., 2013; Reyes-González et al., 2017).

The fishing gear that seems to have a greater impact on bird populations is the longline, as they approach fishing vessels while fishemen are cast the gear. When the birds try to catch the bait they are hooked, dying when they sink. The longline fleet that usually operates in the Mediterranean and is focused on the capture of swordfish and sharks can cause an impact on bird populations, since during the winter months they usually fish in waters surrounding Canary Islands. In the work developed by García-Barcelona et al., (2013), data collected on board referring to the bycatch of this fleet by personnel of the Spanish Institute of Oceanography (IEO) was analyzed. However, no fishing activity was registered with incidental capture of birds in the Canary archipelago and there is also no information regarding incidental catches of birds caused by other fishing gear. For the archipelagos of the Azores and Madeira, no records have been found that allow estimating the mortality of birds associated with the longline or with other fishing gears (Morato, 2012).

Considering the aforementioned, and in the absence of more conclusive data, we can assume that the impact on bird populations in the Macaronesian region is less than that which occurs in other nearby regions such as the Mediterranean. However, it is difficult to estimate the duration of the impact, its reversibility or the repercussions it may have on the ecosystem in these regions. Likewise, it is complicated to assess the pressure that fishing activity exerts on bird communities in the Macaronesian region , as we do not have enough information due to the scarcity of data on the distribution of seabird species, their populations status as well as detailed data about bycatch or discards. However, if we focus on the longline fleet, we can observe differences in fishing strategies with respect to other European zones that could be minimizing the adverse effects on bird populations. For example, the bait used is larger than that used in other regions and sometimes fishermen used sharks as bait, which are not so attractive to birds. Likewise, a smaller number of hooks are used, which in turn are larger and heavier than those used in other areas, which facilitates their sinking (García-Barcelona et al., 2013).

Different strategies have been proposed to alleviate the effect of bycatch caused by longlines (Brothers et al., 1999; ICES, 2013; ACAP, 2014; MISTIC SEAS, 2016), and the following three seem to obtain the best results, especially if they are used simultaneously: (i) adding ballast to the lines of hooks to favor the sinking, (ii) realizing the draft night with minimum illumination and (iii) the use of tori lines (Birdlife International, 2009).

One of the greatest threats to the survival of marine mammal populations is related to their interaction with fisheries and other activities related to aquaculture (FAO, 2018), and the impact is not only related to the harvested individuals but also to those that are injured, since it is unknown if they will survive (Reeves et al., 2013). In the three archipelagos, interactions occur mainly during tuna fishing, when fishermen use longlines and hand lines (Silva et al., 2002, 2011; Hale et al., 2011; Nicolau et al., 2013; Morales et al., 2015). In the document developed by Morato (2012), no deaths of marine mammals were recorded in the Archipelagos of Madeira and Azores due to by-catch, since all the specimens captured were returned alive to the sea. However, Covelo and Martínez (2001) confirmed the mortality of some specimens of cetaceans in these two archipelagos, after finding indications of mortality due to an interaction with the fishing gears in stranded specimens. In the Canary Islands there are records of marine mammal strandings due to fishing gear (Covelo and Martínez, 2001; Arbelo et al., 2013; Morales et al., 2015), although, the impact of fishing currently appears to be low or moderate. It is not an easy task to know the real pressure that fishing exerts on the populations of marine mammals in these regions, as well as the spatial extent of the impact since it is does not exist an in-depth knowledge of the spatial distribution of each fleet.

The majority of recorded dead of turtles in the Canary Islands are due to lesions associated with human activities such as boat-strike injuries , entanglement in derelict fishing nets, ingestion of hooks and monofilament line and crude oil ingestion (Orós et al., 2005; Camacho et al., 2013). The loggerhead sea turtles foraging in Azorean waters are incidentally caught in drifting longlines and most of them were caught when the fisheries were directed at swordfish and blue shark, obtaining the highest catch rates around the eastern islands (Balazs and Pooley, 1994; Bolten et al., 2000; Ferreira et al., 2001, 2011; Morato, 2012), but plastic debris and lost or discarded fishing gears also cause the death of the turtles (Barreiros and Raykov, 2014; Pham et al., 2017). Bottom set lines in the coastal waters of Madeira are reported to take an estimated 500 pelagic immature loggerheads each year (Dellinger and Encarnaçâo, 2000). Previous studies show a high impact on the populations of *Caretta caretta* in the three archipelagos; however it is not the only impacted specie. A recent study concludes that if high-fishing-pressure areas are overlapped with leatherback habitat use, it could be possible to determine the long-term susceptibility of leatherback turtle to bycatch in longline fisheries (Fossette et al., 2014); in this work, the offshore waters of Canary Islands were defined as high-risk area.

Threats to cephalopods come from multiple sources such as climate change, fishing pressure, habitat degradation, among others; so it is very difficult to establish a baseline level which might indicate their good environmental status. Because of this and the wide abundance fluctuations typically seen in cephalopods, the first recommended step will be to distinguish which impacts come from anthropogenic pressures (i.e. fishing) and which ones from natural changes.

Elasmobranches are a common component of the bycatch and discard from fisheries (Megalofonou et al., 2005; Zhou and Griffiths, 2008; James et al., 2016; Bonanomi et al., 2017) being also vulnerable to overexploitation due to their life strategies. Leafscale gulper shark (*Centrophorus squamosus*), smooth lanternshark (*Etmopterus pusillus*), and Portuguese dogfish (Centroscymnus coelolepis), among other elasmobranchs, are incidentally captured and discarded in the longline fisheries targeting black scabbardfish (*Aphanopus carbo*) in Azores and Madeira archipelagos (Ramos et al., 2013). Leafscale gulper shark is assessed as critically endangered and the other two elasmobranch species have been recently assessed as Endangered in the Red List of European marine fish (Nieto et al., 2017). Deep-sea sharks were also observed to be the most important component of the incidental catch of the black scabbardfish longline fishery in the Canarian archipelago, being *Zameus squamulosus*, *Deania hystricosa*, *Centrophorus squamosus*, *Centroscymnus coelolepis* and *Etmopterus princeps* the most frequently species present in the catches (Pajuelo et al., 2010). Given that black scabbardfish and deep-water sharks apparently spatially overlap it would be advisable to develop management strategies in order to ensure the sustainability of the deep-water sharks populations in these fishing grounds.

Council Directive 92/43/EEC of 21 May 1992 and Council Directive 97/62/EC consider the loggerhead sea turtle (*Caretta caretta*), the bottlenose dolphin (*Tursiops truncates*), harbor porpoise (*Phocoena phocoena*) and the Mediterranean monk seal (*Monachus monachus*) as species of community interest whose conservation requires the designation of special areas of conservation and require Member States to undertake surveillance of the conservation status of these species, and Council Regulation (EC) No 812/2004 lays down measures concerning incidental catches of cetaceans in fisheries, amending Regulation (EC) No 88/98.

As a result of the transposition to the Spanish legal system of this Directive, cetaceans are also included in Annexes II and IV of Royal Decree 1997/1995, of 7 December, which establishes measures to guarantee biodiversity through the conservation of natural habitats, flora and fauna. Royal Decree 139/2011 provides the List of Wild Species in Special Protection Regime as well as the Spanish Catalogue of Endangered Species, where we can find species of turtles, birds, cetaceans and sharks present in the Canary Islands.

Law 15/2019 modifies Law 17/2003, of April 10, on Fishing of the Canary Islands and it forbids coming close to any cetacean species, especially whales and dolphins, and sea turtles. It also indicates that, in the event that during the fishing operations the approach to the boat of some group or isolated specimens of cetaceans occurs, the fishermen must adopt as many measures as they deem appropriate to avoid injuries or negative interactions with these species.

Council Regulation (EC) No 2347/2002 establishes specific access requirements and associated conditions applicable to fishing for deep-water stocks, including sharks. To improve the sampling of deep-water species under the EU Data Collection Regulation (DCR) further sampling requirements were specified in Commission Regulation (EC) No 1581/2004. The Council Regulation (EU) Nº1225/2010 and the Council Regulation (EU) No 1262/2012 set a zero TAC for deep-sea sharks.

The technical report coordinated by DRAM (Regional Directorate for the Sea Affairs, Azores) (MISTIC SEAS, 2016) analyses the monitoring programmes that were designed and reported by both Member States Spain (for Canaries) and Portugal (for Madeira and the Azores) related to monitoring strategies of marine mammals, marine turtles and seabirds throughout the subregion of Macaronesia. In this report, authors also propose monitoring programs with which we agree, since they allow assessing the Good Environmental Status (GES) baseline and trends using the MSFD criteria and indicators

1.2 D1C2

The evaluation of the D1C2 criterion supposes a high complexity, since it includes many species. An additional problem is the ignorance concerning the current state of the stocks and the disparity in terms of available information, since there are species for which there is hardly any information in the areas of Macaronesia. However, those species or taxonomic groups for which information does exist, this is dispersed and unequal in time and/or in the spatial component.

Regarding marine mammals, birds and turtles, the studies seem to indicate that the impact that fishing exerts on these groups in the three archipelagos is not so great as to severely compromise the abundance of the stocks of these species. In the case of the cephalopods, the difficulty of evaluating their populations status was already commented in the previous criterion, since factors of various kinds are involved; although as these are target species of the fisheries (Couce-Montero et al., 2015, 2019; Morato et al., 2016), we understand that fishing can compromise their abundance.

Canarian fisheries had been historically focused on three main target groups: benthic-demersals species, medium-sized coastal pelagics and tunas and this fishing strategy has allowed the development of a polyvalent fleet, which alternates the exploitation of the different target species according to their biological cycles and the seasonal fishing of the different tunas. Benthic and demersal species show symptoms of depletion since the early 1970s (Garcia Cabrera, 1970), but this circumstance has not led to significant changes in the exploitation strategy and the main fishing target groups are considered overexploited (González, 2008). On the other hand, medium-sized coastal pelagic species have had a lower degree of exploitation, despite their relatively greater abundance, for what they have been historically considered underexploited. Thus, Pastor and Delgado de Molina (1985) estimated that the biomass of this group of species was around 73000 tonnes throughout the archipelago, biomass that remained practically invariant until the end of the 1990s (Bordes et al., 1987, 1993, 1995, 1997, 1998, 1999).

However, the REPESCAN report (González, 2008) highlights the lack of existing information about this group of species and the state of their stocks and perhaps for this reason, it was not possible to foresee the apparent collapse of the sardine (Sardina pilchardus) during the second half of the 2000s and Atlantic chub mackerel (Scomber colias) in year 2010 in the waters of Gran Canaria, possibly as a result of fishing and adverse climatic conditions for both species.

In the Canary Islands, assessments of the state of the crustacean populations of semi-deep and deep waters have been carried out (González, 2018), concluding that pandalid shrimps (*Plesionika edwardsii*, *P. narval*, *P. williamsi*, *Heterocarpus ensifer*, H. *Laevigatus* and *H. Grimaldi*) populations are being exploited below their maximum sustainable yield (MSY). The populations of large deep-water crabs have hardly any fishing interest, so it is assumed that their populations are in good condition.

Stock assessments for tuna species in the three archipelagos are conducted by the International Commission for the Conservation of Atlantic Tunas (ICCAT), regulating these species according its recommendations; although conclusive values or reference points that indicate the current state of the different tuna populations in these regions are not available.

The black scabbardfish (*Aphanopus* spp.) are the most important resources exploited in Madeira and landings were marked by an increasing trend toward a maximum recorded in 1998; however landings are steadily decreasing since this year (Serrano Gordo, 2009; Morato, 2012). Recently it has been known that in fact the black scabbardfish landings in Madeira correspond to *Aphanopus carbo* and the sympatric specie *Aphanopus intermedius* (Biscoito et al., 2011) in a ratio 80:20 (Delgado et al., 2018). The intensive fishing of deep-sea species caused a decrease in their relative abundance in the usual fishing grounds, leading to an extensive geographical expansion of the fishery reaching as far as the Southern Azores Seamount Chain and the Canaries Economic Exclusive Zone (Delgado et al., 2018). Black scabbardfish stock structure is still unknown in European waters where but, although available information does not unequivocally support the assumption of a single stock, most available evidences support it (ICES, 2019).

Medium-sized coastal pelagic is an important fishery in Madeira being *Scomber colias* and *Trachurus picturatus* the most representative species. The landings of these species show an irregular pattern with an apparent decreasing trend; however, there is no information about the current status of the stocks (Morato, 2012). In Madeira, the average of the landings from 1986 to 1991 was three times higher than the average landings from 1992 to 2007 and the hypothesis is that the fluctuations in landings can be due to changes in availability or abundance, and not just by changes in fishing effort (ICES, 2018b). The stocks status for all the demersal fish species and invertebrates is also unknown.

In the Azores, large pelagic species (tuna and swordfish), blue jack mackerel and conger are very important, but invertebrate fishing is also very significant. Bottom longlines and handlines fisheries are focus on deep-sea species such as blackspot seabream (*Pagellus bogaraveo*), wreckfish (*Polyprion americanus*), alfonsinos (*Beryx* spp.) and blackbelly rosefish (*Helicolenus dactylopterus*). Since the mid-1990s the landings of deep-water species show a decreasing tendency, reflecting the change in the fleet behaviour towards blackspot seabream (ICES, 2019). Studies suggest that alfonsinos (da Silva and Pinho, 2007), blackbelly rosefish (Perrotta and Hernández, 2005) are intensively exploited and Pagellus bogaraveo is overexploited (Novoa-Pabon et al., 2015).

Black scabbardfish fishery is still in an experimental phase in the Azores archipelago (Machete et al., 2011) and in recent years landings have increased, and as it occurs in Madeira archipelago, *Aphanopus carbo* coexist with A. Intermedius but the current status of these stocks are unknown in Azores (ICES, 2019).

The swordfish and blue shark stocks fished in the Azores are assessed as part of the Atlantic stock by the ICCAT so their catches are limited by TACs and, according to the 2017 stock assessment, North Atlantic swordfish are not overfished and are not subject to overfishing (ICES, 2017). For the North Atlantic stock, blue shark stock is not overfished and overfishing was not occurring whereas shortfin mako shark is assumed that is in an overfished state and experiencing overfishing (ICES, 2018a).

Length-based indicators (LBI) suggest that Trachurus picturatus stock in Azores fishing grounds is being exploited above MSY, but given the characteristics of the blue jack mackerel stock and the fishery selectivity pattern, authors consider that LBI method might not be useful to evaluate stock status (ICES, 2018b). There are no stock assessments of other coastal pelagic species in the Azores.

In Azores, artisanal fishing also captures benthic and demersal fish as well as invertebrate species; however, the status of their stocks is unknown so the impact of fishing on their populations cannot be assessed.

The impact of fishing on the abundance of species with commercial interest will be discussed in detail in Descriptors 3 and 4 to avoid excessive redundancies, since we understand that the approach is more specific in these descriptors.

Law 17/2003 of April 10, developed through the Decree 182/2004, of December 21 of exclusive application in inland waters of the Autonomous Community of Canary Islands regulates the use and limitation of the different gears or fishing modalities, the species whose capture is prohibited, the minimum sizes, the closed areas and seasons and the delimitation of fishing grounds. Law 3/2001 of March 26 is mandatory in the external waters of the whole Spanish national territory.

In Spain, Order ARM/2689/2009, Order ARM/1647/2009 prohibit the capture of certain species of sharks and Council Regulation (EU) 2016/72 prohibits fishing, keeping on board, transhipping or landing certain species of elasmobranches in all waters of the European Union.

The current fisheries resource management strategy of the Azores and Madeira is based on the EU Common Fishery Policy, implemented primarily through Total Allowable Catches (TACs) for various deep-sea species. Trawling is forbidden in the Macaronesian archipelagos according Council Regulation (EC) No 1568/2005 and Council Regulation (EC) No 1954/2003 indicates that in the waters up to 100 nautical miles from the baselines of the Azores, Madeira and the Canary Islands, the Member tates concerned may restrict fishing to vessels registered in the ports of these islands, except for Community vessels that traditionally fish in those waters in so far as these do not exceed the fishing effort traditionally exerted.

Azores and Madeira Regional Governments have also implemented technical measures such as minimum landings sizes or weights, minimum mesh sizes, allowable percentage of bycatch species, area and temporal closures and ban on the use of specific fishing gears (Morato, 2012). The use of surrounding nets to capture medium-sized coastal pelagic species is regulated by Ministerial Order (Portaria) No 1102-G/2000 as amended by Ministerial Order No 346/2002. Ministerial Order (Portaria) No 543-B/2001 establishes specific measures concerning the catches, retention on board, landing and marketing of sardines. Ministerial Order (Portaria) No 27/2001, as amended by Ministerial Order No402/2002 and by Ministerial Order No1266/2004, establishes the minimum sizes for 43 species of fish, 11 species of crustaceans and 22 species of molluscs.

According Portaria No 66/2014 de 8 de Outubro de 2014, the Azores Administration implemented a management measure for the purse-seine fleet that allows only 200 kg or 300 kg per vessel, per day, depending on the island. Also states that fishing and consequent landings shall also be forbidden on weekends

Council Regulation (EC) No 2347/2002 establishes specific access requirements and associated conditions applicable to fishing for black scabbardfish and other deep-water species in the FAO CECAF 34.1.2 area. Despite the insufficient and inconclusive information about the biology and life history of black scabbardfish, some indicators suggest that abundances are declining in some areas of the Northeast Atlantic and this situation motivated the implementation of a TAC system in 2003 (ICES, 2006).

Three marine reserves of fishing interest were created in Canary Islands as places of protection for the reproduction and breeding of the target species of the fishery, thus facilitating the recovery of the resources and contributing to achieve sustainable exploitation of these species. These areas are managed jointly by the Department of Agriculture, Livestock, Fisheries and Waters of the Government of the Canary Islands and the Ministry of Agriculture, Fisheries and Food (MAPA). There are five marine protected areas located across the archipelagos of Madeira, Desertas and Selvagens. The main objective of the Desertas MPA is the protection of Mediterranean monk seals (*Monachus monachus*) and several rare and endemic species such as the Desertas petrel (*Pterodroma deserta*). The rest of marine areas serve as protection areas for species of fishing interest as well as for some bird species that have their breeding areas on these islands. In Azores, numerous habitats and ecosystem are under management: offshore hydrothermal vents and seamounts, coastal zones, as well as underwater archaeological parks that can have a functional ecology role as marine protected areas (Garcia and Barreiros, 2018).

The lack of information to be able to correctly evaluate this criterion is remarkable, since the current status of the populations that inhabit the Macaronesian ecosystems is unknown. Although there is information available on the impact of fishing on certain species, it is scarce and limited to certain areas and specific moments, which makes it difficult to assess the impact of this activity at the population level. Likewise, it is impossible to evaluate the impact of fishing on the populations at the habitat level since there is scarcely any information below 50 m. of depth and the little information available is discontinuous and in many cases it has punctual and descriptive character. For all these reasons, we recommend the following actions as monitoring methods:

- Continuous campaigns over time to evaluate the current state of stocks through acoustic campaigns, visual censuses and analytical models in the case of not being able to make direct estimates. These evaluations should include all trophic levels, from primary producers to apex predators, in order to have a more detailed knowledge of the structure of the ecosystem.
- Obtaining realistic fishery statistics detailing all the fishing gears used and the species captured with each, including information on bycatch or discards if necessary.
- Monitoring the populations of protected and endangered species, as well as the evaluation of non-indigenous species introduced by human activities to assess their impact on ecosystems.

1.3 D1C3

Fishing exploitation for prolonged periods linked to changes in environmental conditions can lead to changes in the life cycle of marine species (Smith, 1996; Jennings et al., 1999; Winemiller, 2005). Fishing removes preferentially the larger and older specimens modifying the age class structure of the populations, which can lead to a reduction in the buffering capacity of the population affecting for example, the recruitment (Perry et al., 2010; Hixon et al., 2014) or the spawning stock biomass (Ebisawa et al., 2016).

For sex-changing species, selective fishing practices can affect additional traits such as the mature population sex ratio and the timing of sexual transformation (Hawkins and Roberts, 2003; Hamilton et al., 2007). Furthermore, fishing of protogynous species can cause an inadequate number of males in the population, and thus fertilization can be affected due to the sperm limitation (Coleman et al., 1996; Alonzo and Mangel, 2004)

The age and size at which individuals of most species mature are not fixed, and this variability can be induced by different factors, including environmental changes and fishing or a combination of both; in many studies, changes in maturity have been associated with changes in stock abundance (Morgan and Colbourne, 1999). An array of studies show that changes in survival and reproductive success could be associated with fishing mortality, modifying maturation at a younger age and smaller size (Hutchings, 1993; Rowell, 1993; Stokes and Elythe, 1993; Grift et al., 2003; Olsen et al., 2005; Rijnsdorp et al., 2005)

However, there are at least 2 non-exclusive hypotheses which may account for maturity changes. The first hypothesis predicts that earlier maturation is a phenotypic plasticity response that allows fish to react to a reduction in stock size due to fisheries-induced alterations. The second hypothesis predicts that earlier maturation is due to evolutionary changes in the life histories of harvested stocks, manifesting some underlying modifications in their genetic composition (Engelhard and Heino, 2004). So, for management purposes, it is necessary to distinguish between these two possible adaptive changes to manage the resources, since mitigating adverse evolutionary changes takes many generations, whereas phenotypically plastic responses usually occur within a single generation (Reznick, 1993).

This study revealed no technical reports or scientific publications related to fishing regarding this criterion in the Macaronesian region. Taking into account the biological and geographical peculiarities of these marine ecosystems, as well as the high fishing pressure to which most of the resources are subjected, we do not rule out that changes in the population demographic characteristics due to the effect of fishing are occurring.

1.4 D1C4

This criterion refers to the distribution of species conditioned by physiographic, geographic and climatic aspects, so that fishing would have no impact. However, fishing can modify the distribution of organisms, reducing it as the biomass decreases. For example, in the case of overexploited neritic populations it can happen that the species reduce their distribution area favouring a certain proximity between the individuals, which would facilitate the reproduction, the formation of groups (schools) or allow them to adopt other strategies oriented to the protection of the species (hierarchies, defence of feeders, etc.). Another example could be the disappearance of seagrass due to overfishing. These areas act as a refuge and nursery areas for many species of fish and invertebrates which could have an impact on the reproduction and recruitment of certain species. However, it is impossible to evaluate the impact of fishing on this criterion because the distribution pattern of most species is unknown and, in addition, this may be conditioned by the impacts of other anthropogenic activities as well as environmental changes

1.5 D1C5

According Council Directive 92/43/EEC the habitat of a species means an environment defined by specific abiotic and biotic factors, in which the species lives at any stage of its biological cycle. The number of existing habitats in the three archipelagos is extensive, so we will classify them into two large groups to simplify: benthic habitats and pelagic habitats.

For the majority of habitats in the Macaronesian region, adequate information about their distribution, extension and / or status is not currently available; knowledge about the extent and status of them in the past is even more limited, which makes it difficult to assess the impact that fishing has had on habitats since there are no previous reference levels. This reference levels can be established for well-conserved areas, with very little anthropic influence, but this information is not available for all habitats and is only representative of some localities or regions, but under no circumstances at the level of demarcation. In addition, the degradation observed in certain habitats, such as coastal habitats, is mainly due to the cumulative impact of various activities carried out in the same area and other environmental impacts.

An example can be found in the seagrass beds on infralittoral sand; these habitats are spatially complex and very important for several marine organisms which depend on them in different phases of their life cycle, not only to feed but also to take shelter from predators. Impacts of fishing on these habitats include the deterioration and stripping of rhizomes and leaves, as well as the elimination of large numbers of fish in fry and juvenile stages. Likewise, the anchoring of boats, both recreational and professional, has an impact because the anchors and chains are dragged along the bottom, producing furrows where the rhizomes are removed (Espino et al., 2008). These are direct impacts; however other indirect impacts include the increase in populations of urchins (*Diadema* spp.) due to the overfishing of their predators, converting these areas into urchin barrens (Alves et al., 2003; Hernández et al., 2008a, 2008b). Nevertheless, coastal development including port developments, aquaculture, brine from desalination and sewage discharges also exert great pressures on these habitats (Espino et al., 2008).

Mobile fishing gears can reduce the habitat complexity since they have measurable impacts on its structural components. For example, trawling is the most damaging for benthic habitats and fishing impacts can be direct, such as physical impact of the trawl on the seabed and the sediment disturbance associated with it, or indirect such as modification to sediment properties and solute fluxes and changes in benthic community abundance and composition (National Research Council, 2002). Numerous studies have been addressed to evaluate the negative effects of trawling on benthic communities (Freese et al., 1999; Collie et al., 2000; Smith et al., 2000; D'Onghia et al., 2003; Gage et al., 2005; Watling, 2005; Morato et al., 2006; Maynou and Cartes, 2012) and it is prohibited according to Regulation (EC) 1568/2005 in the three archipelagos. Other fishing gears frequently used in the Macaronesia can also potentially modify the integrity of benthic habitats, such as: gillnets, trammel nets, long lines and traps (Tudela, 2004; Chiappone et al., 2005; Matsuoka et al., 2005; Sampaio et al., 2012); and the use of fixed gears in these small-scale Macaronesian fisheries makes ghost fishing by abandoned or discarded gears a potentially important problem.

There is not enough information about the distribution and extent of marine habitats in Macaronesia, their functioning, their responses to disturbances and the extent of the pressures of all human activities and besides, the information available is often imprecise and even redundant. Otherwise, to elucidate the fishing impacts on pelagic habitats is difficult since these habitats are poorly described at the scales that allow for measurements of changes based on gear uses, since most of the habitats are detailed in the depth range from the surface to 50 meters. For those reasons and taking into account the lack of information on the distribution and timing of fishing in the archipelagos, it is impossible to evaluate this criterion and we suggest that further research is needed.

1.6 References

ACAP. 2014. Bycatch Mitigation Fact Sheets. Seabird Bycatch Working Group, https://www.acap.aq/en/bycatch-mitigation/bycatch-mitigation-fact-sheets.

Alonzo, S. H., and Mangel, M. 2004. The effects of size-selective fisheries on the stock dynamics of and sperm limitation in sex-changing fish. Fishery Bulletin, 102: 1–13.

Alverson, D. L., Freeberg, M. H., Murawski, S. A., and Pope, J. G. 1994. A global assessment of fisheries bycatch and discards. FAO. FAO Fisheries Technical Paper. No. 339. Rome, FAO. 1994. 233p.

Alves, F. M. A., Chícharo, L. M., Serrão, E., and Abreu, A. D. 2003. Grazing by Diadema antillarum (Philippi) upon algal communities on rocky substrates. Scientia Marina, 67: 307–311.

Anderson, O. R. J., Small, C. J., Croxall, J. P., Dunn, E. K., Sullivan, B. J., Yates, O., and Black, A. 2011. Global seabird bycatch in longline fisheries. Endangered Species Research, 14: 91–106.

Arbelo, M., de los Monteros, A. E., Herráez, P., Andrada, M., Sierra, E., Rodríguez, F., Jepson, P. D., et al. 2013. Pathology and causes of death of stranded cetaceans in the Canary Islands (1999− 2005). Diseases of aquatic organisms, 103: 87–99.

Balazs, G. H., and Pooley, S. G. 1994. Research plan to assess marine turtle hooking mortality: results of an expert workshop held in Honolulu, Hawaii, November 16-18, 1993. NOAA Technical Memorandum NMFS.

Barreiros, J. P., and Raykov, V. S. 2014. Lethal lesions and amputation caused by plastic debris and fishing gear on the loggerhead turtle *Caretta caretta* (Linnaeus, 1758). Three case reports from Terceira Island, Azores (NE Atlantic). Marine Pollution Bulletin, 86: 518–522.

Birdlife International. 2009. European Community Plan of Action(ECPOA) for reducing incidental catch of seabirds in fisheries. Proposal by BirdLife International. Cambridge, UK. http://edepot.wur.nl/170818.

Biscoito, M., Delgado, J., González, J. A., Stefanni, S., Tuset, V. M., Isidro, E., García-Mederos, A., et al. 2011. Morphological identification of two sympatric species of Trichiuridae, *Aphanopus carbo* and A. intermedius, in NE Atlantic. Cybium, 35: 19–32.

Bolten, A. B., Martins, H. R., and Bjorndal, K. A. 2000. Workshop to design an experiment to determine the effects of longline gear modification on sea turtle bycatch rates. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-OPR- 19, 50 p.

Bonanomi, S., Colombelli, A., Notti, E., Pulcinella, J., and Sala, A. 2017. Fisheries bycatch of chondrichthyes. In Chondrichthyes-Multidisciplinary Approach. IntechOpen.

Bordes, F., Barrera, A., Ramírez, R., Gómez, J. A., Santana, J. I., Hernández-León, S., and Arístegui, J. 1987. Prospección hidroacústica para la evaluación del stock de peces pelágicos costeros de Canarias. Consejería de Agricultura, Ganadería y Pesca, Gobierno de Canarias. Centro de Tecnología Pesquera, Cabildo Insular de Gran Canaria.

Bordes, F., Barrera, A., Castillo, R., Gómez, J., Ojeda, A., Pérez, F., Álvarez, S., et al. 1993. Cartografía y evaluación de los recursos pesqueros de la plataforma y talud de Gran Canaria, Islas Canarias. Informe Técnico. Viceconsejería de Agricultura, Pesca y Alimentación.

Bordes, F., Barrera, A., Castillo, R., Gómez, J., Pérez, F., Álvarez, S., and Ojeda, A. 1995. Cartografía y evaluación de los recursos pesqueros en la plataforma y talud de Fuerteventura y Lanzarote (Islas Canarias). Informe Técnico. Viceconsejería de Agricultura, Pesca y Alimentación.

Bordes, F., Barrera, A., Carrillo, J., Castillo, R., Castro, J. J., Gómez, J., Hansen, K., et al. 1997. Evaluación acústica de los recursos epipelágicos y estudio de la capa de reflexión profunda en Lanzarote, Fuerteventura y Gran Canaria (Islas Canarias). Informe Técnico. Viceconsejería de Agricultura, Pesca y Alimentación.

Bordes, F., Almeida, C., Barrera, A., Carrillo, J., Castillo, R., Coca-Sáez, J., Gómez, J. A., et al. 1998. Prospección acústica y pesquera de los recursos pelágicos en Lanzarote, Fuerteventura y Gran Canaria (Islas Canarias).Resultados de la

Campaña “Bocaina 1197". Informe Técnico. Viceconsejería de Agricultura, Pesca y Alimentación.

Bordes, F., Uiblein, F., Castillo, R., Barrera, A., Castro, J. J., Coca, J., Gomez, J., et al. 1999. Epi-and Mesopelagic Fishes, Acoustic Data, and SST Images Collected Off Lanzarote, Fuerteventura, and Gran Canaria, Canary Islands, During Cruise" La Bocaina 04-97". Informe Técnico, Instituto Canario de ciencias Marinas.

Brothers, N. P., Cooper, J., and Løkkeborg, S. 1999. The incidental catch of seabirds by longline fisheries : worldwide review and technical guidelines for mitigation. FAO Fisheries Circular. No. 937. Rome, FAO. 1999. 100p.

Camacho, M., Calabuig, P., Luzardo, O. P., Boada, L. D., Zumbado, M., and Orós, J. 2013. Crude Oil as a Stranding Cause among Loggerhead Sea Turtles (*Caretta caretta*) in the Canary Islands, Spain (1998–2011). Journal of Wildlife Diseases, 49: 637–640.

Chiappone, M., Dienes, H., Swanson, D. W., and Miller, S. L. 2005. Impacts of lost fishing gear on coral reef sessile invertebrates in the Florida Keys National Marine Sanctuary. Biological Conservation, 121: 221–230.

Coleman, F. C., Koenig, C. C., and Collins, L. A. 1996. Reproductive styles of shallow-water groupers (Pisces: Serranidae) in the eastern Gulf of Mexico and the consequences of fishing spawning aggregations. Environmental Biology of Fishes, 47: 129–141.

Collie, J. S., Hall, S. J., Kaiser, M. J., and Poiner, I. R. 2000. A quantitative analysis of fishing impacts on shelf-sea benthos. Journal of Animal Ecology, 69: 785–798.

Couce-Montero, L., Christensen, V., and Castro, J. J. 2015. Effects of small-scale and recreational fisheries on the Gran Canaria ecosystem. Ecological Modelling, 312: 61–76.

Couce-Montero, L., Christensen, V., and Castro, J. J. 2019. Simulating trophic impacts of recreational fishing scenarios on two oceanic islands using Ecopath with Ecosim. Article submitted for publication.

Covelo, P., and Martínez, J. 2001. Varamientos de mamíferos marinos en las costas de España y Portugal entre 1996 y 1998: Atlancetus. Galemys, 13: 93–106.

da Silva, H. M., and Pinho, M. R. 2007. Small-Scale Fishing on Seamounts. In Seamounts: Ecology, Fisheries & Conservation, pp. 333–360. Ed. by T. J. Pitcher, T. Morato, P. J. B. Hart, M. R. Clark, N. Haggan, and R. S. Santos. Blackwell Publishing Ltd, Oxford, UK. http://doi.wiley.com/10.1002/9780470691953.ch16 (Accessed 24 June 2019).

Delgado, J., Amorim, A., Gouveia, L., and Gouveia, N. 2018. An Atlantic journey: The distribution and fishing pattern of the Madeira deep sea fishery. Regional Studies in Marine Science, 23: 107–111.

Dellinger, T., and Encarnaçâo, H. 2000. Accidental capture of sea turtles by the fishing fleet based at Madeira Island, Portugal. US Department of Commerce NOAA Technical Memorandum NMFS-SEFSC-443, 218.

D’Onghia, G., Mastrototara, F., Matarrese, A., Politou, C.-Y., and Mytilineou, C. 2003. Biodiversity of the Upper Slope Demersal Community in the Eastern Mediterranean: Preliminary Comparison Between Two Areas With and Without Trawl Fishing. Journal of Northwest Atlantic Fishery Science, 31: 263–273.

Ebisawa, A., Kanashiro, K., Ohta, I., Uehara, M., and Nakamura, H. 2016. Changes of group construction accompanying with growth and maturity in blue-barred parrotfish (Scarus ghobban), and influences of the fishing targeting the immature group to the stock. Regional Studies in Marine Science, 7: 32–42.

Engelhard, G. H., and Heino, M. 2004. Maturity changes in Norwegian spring-spawning herring Clupea harengus: compensatory or evolutionary responses? Marine Ecology Progress Series, 272: 245–256.

Espino, F., Tuya, F., Blanch, I., and Haroun, R. J. 2008. Los sebadales de Canarias. Oasis de vida en los fondos arenosos. BIOGES, Universidad de Las Palmas de Gran Canaria, 68 pp.

FAO. 1999. Guidelines for the routine collection of capture fishery data. FAO Fisheries Technical Paper. No. 382. Rome, FAO. 1999. 113p.

FAO. 2018. Report of the Expert Workshop on Means and Methods for Reducing Marine Mammal Mortality in Fishing and Aquaculture Operations, Rome, 20-23 March 2018. FAO Fisheries and Aquaculture Report No.1231. Rome, Italy.

Ferreira, R. L., Martins, H. R., Silva, A. A. D., and Bolten, B. 2001. Impact of swordfish fisheries on sea turtles in the Azores. Life and Marine Sciences, 18A: 75–79.

Ferreira, R. L., Martins, H. R., Bolten, A. B., Santos, M. A., and Erzini, K. 2011. Influence of environmental and fishery parameters on loggerhead sea turtle by-catch in the longline fishery in the Azores archipelago and implications for conservation. Journal of the Marine Biological Association of the United Kingdom, 91: 1697–1705.

Fossette, S., Witt, M. J., Miller, P., Nalovic, M. A., Albareda, D., Almeida, A. P., Broderick, A. C., et al. 2014. Pan-Atlantic analysis of the overlap of a highly migratory species, the leatherback turtle, with pelagic longline fisheries. Proceedings of the Royal Society B: Biological Sciences, 281: 20133065.

Freese, L., Auster, P. J., Heifetz, J., and Wing, B. L. 1999. Effects of trawling on seafloor habitat and associated invertebrate taxa in the Gulf of Alaska. Marine Ecology Progress Series, 182: 119–126.

Gage, J. D., Roberts, J. M., Hartley, J. P., and Humphery, J. D. 2005. Potential Impacts of Deep-Sea Trawling on the Benthic Ecosystem along the Northern European Continental Margin: A Review. American Fisheries Society Symposium, 41: 503–517.

Garcia, A. C., and Barreiros, J. P. 2018. Are underwater archaeological parks good for fishes? Symbiotic relation between cultural heritage preservation and marine conservation in the Azores. Regional Studies in Marine Science, 21: 57–66.

Garcia Cabrera, C. 1970. La pesca en Canarias y Banco Sahariano. Publ. Cons. Eco. Soc. Sind. Inter. Canarias. Tenerife, 168.

García-Barcelona, S., Báez, J. C., de Urbina, J. M. O., Gómez-Vives, M. J., and Macías, D. 2013. By-catch of cory’s shearwater in the commercial longline fisheries based in the Mediterranean coast and operating in East Atlantic waters: first approach to incidental catches of seabird in the area. Collect. Vol. Sci. Pap. ICCAT, 69: 1929–1934.

González, J. A. 2008. Memoria científico-técnica final sobre el Estado de los Recursos Pesqueros de Canarias (REPESCAN). Instituto Canario de Ciencias Marinas, ACIISI, Gobierno de Canarias. Telde (Las Palmas).

González, J. A. 2018. Análisis de los principales hallazgos de la pesca experimental en aguas de Canarias, destacando las lecciones aprendidas y señalando los caminos a seguir. Informe científico-técnico para el proyecto ORFISH. Co-financiado por la DG MARE (Unión Europea) y GMR Canarias, S.A.U. Fundación

Parque Científico Tecnológico de la Universidad de Las Palmas de Gran Canaria. Las Palmas de Gran Canaria. 136 pp.

González-Solís, J., Croxall, J. P., Oro, D., and Ruiz, X. 2007. Trans-equatorial migration and mixing in the wintering areas of a pelagic seabird. Frontiers in Ecology and the Environment, 5: 297–301.

Grift, R. E., Rijnsdorp, A. D., Barot, S., Heino, M., and Dieckmann, U. 2003. Fisheries-induced trends in reaction norms for maturation in North Sea plaice. Marine Ecology Progress Series, 257: 247–257.

Hale, R., Pires, R., Santos, P., and Karamanlidis, A. A. 2011. Mediterranean Monk Seal (*Monachus monachus*): Fishery Interactions in the Archipelago of Madeira. Aquatic Mammals, 37: 298–304.

Hamilton, S. L., Caselle, J. E., Standish, J. D., Schroeder, D. M., Love, M. S., Rosales-Casian, J. A., and Sosa-Nishizaki, O. 2007. Size-selective harvesting alters life histories of a temperate sex-changing fish. Ecological Applications, 17: 2268–2280.

Hawkins, J. P., and Roberts, C. M. 2003. Effects of fishing on sex-changing Caribbean parrotfishes. Biological Conservation, 115: 213–226.

Hernández, J. C., Clemente, S., Sangil, C., and Brito, A. 2008a. Actual status of the sea urchin Diadema aff. antillarum populations and macroalgal cover in marine protected areas compared to a highly fished area (Canary Islands-eastern Atlantic Ocean). Aquatic Conservation: Marine and Freshwater Ecosystems, 18: 1091–1108.

Hernández, J. C., Clemente, S., Sangil, C., and Brito, A. 2008b. The key role of the sea urchin Diadema aff. antillarum in controlling macroalgae assemblages throughout the Canary Islands (eastern subtropical Atlantic): An spatio-temporal approach. Marine Environmental Research, 66: 259–270.

Hixon, M. A., Johnson, D. W., and Sogard, S. M. 2014. BOFFFFs: on the importance of conserving old-growth age structure in fishery populations. ICES Journal of Marine Science, 71: 2171–2185.

Hutchings, J. A. 1993. Adaptive Life Histories Effected by Age-Specific Survival and Growth Rate. Ecology, 74: 673–684.

ICES. 2006. Report of the Working Group on the Biology and Assessment of Deep-Sea Fisheries Resources. ICES Document CM 2006/ACFM: 28.

ICES. 2013. Report of the Workshop to Review and Advise on Seabird Bycatch (WKBYCS), 14–18 October 2013, Copenhagen, Denmark. ICES CM 2013/ACOM:77. 79 pp.

ICES. 2017. Report of the 2017 ICCAT Atlantic swordfish stock assessment session. Atlantic SWO Stock assessment meeting. Madrid 2017.

ICES. 2018b. Report of the Working Group on Southern Horse Mackerel, Anchovy and Sardine (WGHANSA), 26–30 June 2018, Lisbon, Portugal. ICES CM 2018/ACOM:17. 659 pp.

ICES. 2018a. Report for biennial period, 2016-17 PART I (2017) - Vol. 2. International Commission for the Conservation of Atlantic Tunas, Madrid, Spain, 2018.

ICES. 2019. Working Group on the Biology and Assessment of Deep-sea Fisheries Resources (WGDEEP). ICES Scientific Reports/Rapports scientifiques du CIEM, 1(21), 988p.

James, K. C., Lewison, R. L., Dillingham, P. W., Curtis, K. A., and Moore, J. E. 2016. Drivers of retention and discards of elasmobranch non-target catch. Environmental Conservation, 43: 3–12.

Jennings, S., Greenstreet, S. P. R., and Reynolds, J. D. 1999. Structural change in an exploited fish community: a consequence of differential fishing effects on species with contrasting life histories. Journal of Animal Ecology, 68: 617–627.

Lloret Capote, A., Del Barrio Alvarellos, I., and Moreno Aranda, I. M. 2012. Estrategia Marina. Demarcación Marina Canaria. Evaluación Inicial. Parte II: Análisis de presiones e impactos. Ministerio de Agricultura, Alimentación y Medio Ambiente, Madrid, Spain. https://www.miteco.gob.es/es/costas/temas/proteccion-medio-marino/II_Analisis_Presiones_Canarias_tcm30-130933.pdf.

Machete, M., Morato, T., and Menezes, G. 2011. Experimental fisheries for black scabbardfish (*Aphanopus carbo*) in the Azores, Northeast Atlantic. ICES Journal of Marine Science, 68: 302–308.

Matsuoka, T., Nakashima, T., and Nagasawa, N. 2005. A review of ghost fishing: scientific approaches to evaluation and solutions. Fisheries Science, 71: 691–702.

Maynou, F., and Cartes, J. E. 2012. Effects of trawling on fish and invertebrates from deep-sea coral facies of Isidella elongata in the western Mediterranean. Journal of the Marine Biological Association of the United Kingdom, 92: 1501–1507.

Megalofonou, P., Yannopoulos, C., and Damalas, D. 2005. Incidental catch and estimated discards of pelagic sharks from the swordfish and tuna fisheries in the Mediterranean Sea. Fishery Bulletin, 103: 620–634.

MISTIC SEAS. 2016. Macaronesia islands standard indicators and criteria: reaching common grounds on monitoring marine biodiversity in macaronesia. Technical Report 2, 145 p.

Morales, E., Adern, N., Aldúan, M., Reyes, B., and Aguilar de Soto, N. 2015. Interacciones pesca artesanal y delfines. In: Bécares, J.; Gil-Velasco, M.; Morales, E. y Aguilar, N. 2015. Canarias con la Mar. Conservación de cetáceos y Aves marinas en Canarias (Memoria Técnica). Informe de GIC-ULL a la Fundación Biodiversidad-MAGRAMA. https://www.miteco.gob.es/es/costas/temas/proteccion-medio-marino/II_Analisis_Presiones_Canarias_tcm30-130933.pdf.

Morato, T., Watson, R., Pitcher, T. J., and Pauly, D. 2006. Fishing down the deep. Fish and Fisheries, 7: 24–34.

Morato, T. 2012. Description of environmental issues, fish stocks and fisheries in the EEZs around the Azores and Madeira. Report Prepared for the STECF Plenary Meeting, 12.

Morato, T., Lemey, E., Menezes, G., Pham, C. K., Brito, J., Soszynski, A., Pitcher, T. J., et al. 2016. Food-Web and Ecosystem Structure of the Open-Ocean and Deep-Sea Environments of the Azores, NE Atlantic. Frontiers in Marine Science, 3: 245.

Morgan, M., and Colbourne, E. B. 1999. Variation in maturity-at-age and size in three populations of American plaice. ICES Journal of Marine Science, 56: 673–688.

National Research Council. 2002. Effects of trawling and dredging on seafloor habitat. Washington, DC: The National Academies Press.

Nicolau, C., Alves, F., Ferreira, R., Henriques, F., Dinis, A., Ribeiro, C., and Freitas, L. 2013. A first assessment of the cetaceans' occurrence and threats in the offshore waters of Madeira. In 27th Conference of the European Cetacean Society, Setbal, Portugal.

Nieto, A., Ralph, G. M., Comeros-Raynal, M. T., Kemp, J., García Criado, M., Allen, D. J., Dulvy, N. K., et al. 2017. European Red List of Marine Fishes. Luxembourg: Publications Office of the European Union. 10.2779/082723.

Novoa-Pabon, A., Barros, P., Krug, H., Silva, P., and Pinho, M. R. 2015. Yield per recruit analysis for the red blackspot seabream (Pagellus bogaraveo) stock from the Azores (ICES area Xa2). ICES Document CM 2015/A:33.

Olsen, E. M., Lilly, G. R., Heino, M., Morgan, M. J., Brattey, J., and Dieckmann, U. 2005. Assessing changes in age and size at maturation in collapsing populations of Atlantic cod (Gadus morhua). Canadian Journal of Fisheries and Aquatic Sciences, 62: 811–823.

Orós, J., Torrent, A., Calabuig, P., and Déniz, S. 2005. Diseases and causes of mortality among sea turtles stranded in the Canary Islands, Spain (1998-2001). Diseases of Aquatic Organisms, 63: 13–24.

Pajuelo, J. G., González, J. A., and Santana, J. I. 2010. Bycatch and incidental catch of the black scabbardfish (Aphanopus spp.) fishery off the Canary Islands. Fisheries Research, 106: 448–453.

Pastor, X., and Delgado de Molina, A. 1985. Acoustic abundance estimation of mackerel, pilchard and bogue in Canary Islands waters, April 1984. ICES C.M. 1985/H:39; 24 pp.

Perrotta, R. G., and Hernández, D. R. 2005. Stock assessment of the bluemouth (Helicolenus dactylopterus) in Azorean waters during the 1990-2002 period, applying a biomass dynamic model. Rev. Invest. Desarr. Pesq., 17: 81–93.

Perry, R. I., Cury, P., Brander, K., Jennings, S., Möllmann, C., and Planque, B. 2010. Sensitivity of marine systems to climate and fishing: Concepts, issues and management responses. Journal of Marine Systems, 79: 427–435.

Pham, C. K., Rodríguez, Y., Dauphin, A., Carriço, R., Frias, J. P. G. L., Vandeperre, F., Otero, V., et al. 2017. Plastic ingestion in oceanic-stage loggerhead sea turtles (*Caretta caretta*) off the North Atlantic subtropical gyre. Marine Pollution Bulletin, 121: 222–229.

Ramos, H., Silva, E., and Gonçalves, L. 2013. Reduction of the deep-sea shark's by-catches in the Portuguese long-line black scabbard fishery. Final Report to the European Commission MARE/2011/06 (SI2.602201). Horta.seaExpert, Lda.

Reeves, R. R., McClellan, K., and Werner, T. B. 2013. Marine mammal bycatch in gillnet and other entangling net fisheries, 1990 to 2011. Endangered Species Research, 20: 71–97.

Reyes-González, J. M., Zajková, Z., Morera-Pujol, V., De Felipe, F., Militão, T., Dell'Ariccia, G., Ramos, R., et al. 2017. Migración y ecología espacial de las poblaciones españolas de pardela cenicienta. Monografía n.o 3 del programa Migra. SEO/BirdLife. Madrid. https://doi.org/10.31170/0056.

Reznick, D. N. 1993. Norms of Reaction in Fishes. In The Exploitation of Evolving Resources, pp. 72–90. Ed. by K. Stokes, J. M. McGlade, and R. Law. Springer Berlin Heidelberg.

Rijnsdorp, A. D., Grift, R. E., and Kraak, S. B. 2005. Fisheries-induced adaptive change in reproductive investment in North Sea plaice (Pleuronectes platessa)? Canadian Journal of Fisheries and Aquatic Sciences, 62: 833–843.

Rowell, C. A. 1993. The Effects of Fishing on the Timing of Maturity in North Sea Cod (Gadus Morhua L.). In The Exploitation of Evolving Resources, pp. 44–61. Ed. by K. Stokes, J. M. McGlade, and R. Law. Springer Berlin Heidelberg.

Sampaio, Í., Braga-Henriques, A., Pham, C., Ocaña, O., de Matos, V., Morato, T., and Porteiro, F. M. 2012. Cold-water corals landed by bottom longline fisheries in the Azores (north-eastern Atlantic). Journal of the Marine Biological Association of the United Kingdom, 92: 1547–1555.

Serrano Gordo, L. 2009. Black scabbardfish (*Aphanopus carbo* Lowe, 1839) in the southern Northeast Atlantic: considerations on its fishery. Scientia Marina, 73S2: 11–16.

Silva, M. A., Feio, R., Prieto, R., Gonçalves, J. M., and Santos, R. S. 2002. Interactions between cetaceans and the tuna fishery in the Azores. Marine Mammal Science, 18: 893–901.

Silva, M. A., Machete, M., Reis, D., Santos, M., Prieto, R., Dâmaso, C., Pereira, J. G., et al. 2011. A review of interactions between cetaceans and fisheries in the Azores. Aquatic Conservation: Marine and Freshwater Ecosystems, 21: 17–27.

Smith, C. J., Papadopoulou, K. N., and Diliberto, S. 2000. Impact of otter trawling on an eastern Mediterranean commercial trawl fishing ground. ICES Journal of Marine Science, 57: 1340–1351.

Smith, P. J. 1996. La diversidad genética de los recursos pesqueros marinos: posibles repercusiones de la pesca. FAO Documento Técnico de Pesca. No. 344. Roma, FAO. 1996. 59p.

Stokes, T. K., and Elythe, S. P. 1993. Size-selective harvesting and age-at-maturity II: real populations and management options. In The exploitation of evolving resources, pp. 232–247. Springer.

Tudela, S. 2004. Ecosystem effects of fishing in the Mediterranean: an analysis of the major threats of fishing gear and practices to biodiversity and marine habitats. Studies and Reviews. General Fisheries Commission for the Mediterranean. No. 74. Rome, FAO. 2004. 44p.

Watling, L. 2005. The global destruction of bottom habitats by mobile fishing gear. Marine conservation biology: the science of maintaining the sea's biodiversity. Island Press, Washington, DC: 198–210.

Winemiller, K. O. 2005. Life history strategies, population regulation, and implications for fisheries management. Canadian Journal of Fisheries and Aquatic Sciences, 62: 872–885.

Zhou, S., and Griffiths, S. P. 2008. Sustainability Assessment for Fishing Effects (SAFE): A new quantitative ecological risk assessment method and its application to elasmobranch bycatch in an Australian trawl fishery. Fisheries Research, 91: 56–68.

2 Analysis >> QD1 Pelagic habitats (relating to Descriptor 1)

QD1 Pelagic habitats (relating to Descriptor 1)							
QD	**Criteria (element)**	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
QD1	The condition of the habitat type, including its biotic and abiotic structure and its functions (e.g. its typical species composition and their relative abundance, absence of particularly sensitive or fragile species or species providing a key function, size structure of species), is not adversely affected due to anthropogenic pressures.	D1C6	yes	need further research	need further research	need further research	need further research

2.1 D1C6

As mentioned in the D1C5 criteria, only detailed information of the habitats up to 50 meters deep is available, so the impacts of fishing on this criterion have been developed in the previous sections in cases where information was available, such as for species of fishing interest.

3 Analysis >> QD2 Non-indigenous species

QD2 Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems							
QD	**Criteria (element)**	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
QD2	Newly-introduced non-indigenous species.	D2C1	No data & information, needed further research.				
	Abundance and spatial distribution of established non-indigenous species, contributing significantly to adverse effects on particular species groups or broad habitat types	D2C2 — Secondary	No data & information, needed further research.				
	Proportion of the species group or spatial extent of the broad habitat type which is adversely altered due to non-indigenous species, particularly invasive non-indigenous species.	D2C3 — Secondary	*No data & information, needed further research.*				

The "introduction or transmission of non-indigenous species" is a biological pressure (Marine Strategy Framework Directive, MSFD, 2008). It is evaluated by D2C1, D2C2 y D2C3 criteria from the QD2. These criteria are associated with different concepts, some of these have got problems in their definition, i.e. "invasive species" (Riera et al., 2018). In order not to take part in debates, and in favor of the applicability of related European law, whenever possible it will be take in count the definitions includes in European regulations.

It is understood by ‘alien species' *means any live specimen of a species, subspecies or lower taxon of animals, plants, fungi or microorganisms introduced outside its natural range; it includes any part, gametes, seeds, eggs or propagules of such species, as well as any hybrids, varieties or breeds that might survive and subsequently reproduce* (Regulation 1143/2014). And 'invasive alien species’ (Regulation 1143/2014) *means an* *alien species whose introduction or spread has been found to threaten or adversely impact upon biodiversity and related ecosystem services.*

3.1 D2C1

'Introduction' *means the movement, as a consequence of human intervention, of a species outside its natural range* (Regulation 1143/2014).

Regarding the fishing activity, the ways of introduction of alien species are, on the one hand, transport by means of the organisms embedded or hooked on the hull and other submerged parts (ropes, anchors, gears ...) (Davidson et al., 2012; Lloret-Capote et al., 2012; Williams et al., 2013); which could be underestimated under certain circumstances (Davidson et al., 2012). And on the other hand, the ballast water of ships (Williams et al., 2013) with its associated biota (eg microbes, plankton and fish, Bailey, 2015). In the case of artisanal boats, some have tanks with water (to live bait), and these take and throw water in a similar way to those that use ballast water (eg tuna boats that use live bait, IEO, 2008; *pers.* Dr. Castro-Hernández and the Canarian fishermen).

The process of introducing an alien species will produce an effect based on its establishment and subsequent behaviour (details in Ricciardi et al., 2013; Maçic et al., 2018; Riera et al., 2018). According to these authors, the process by which an introduced species becomes invasive depends, broadly speaking, on the species itself and the receptor ecosystem, all framed in a great variability and uncertainty (Maçic et al., 2018) that difficulty its prediction (Ricciardi et al., 2013). When the species acquires the invasive character (immediately or not, Capdevila-Argüelles et al., 2013), this could cause different socio-economic and ecological impacts, positive or negative, in relation to the perception of the actors related to the specific invasion process (Ricciardi et al., 2013). The invasive species could alter the ecosystem and cause significant economic losses due to direct impacts, management costs, loss of ecosystem services or health problems (Capdevila-Argüelles et al., 2013).

As for the fishing fleets of the Azores (757 boats), the Canary Islands (771 boats) and Madeira (431 boats), it should be noted that between 83-89% is less than 12 m in length (Fleet Register European Union, 01/06/2018), which influences (along with its power) the distance away from their coasts. Thus, the fleets operate mostly locally in the Canary Islands (Martín-Sosa, 2012), until 50 nm in the Azores (Ojamaa, 2015); and in Madeira, 73% of the fleet does not reach 6 m (Vallerani et al., 2017). This determines that they sail normally close to their coasts and within their waters (Martín-Sosa, 2012, Ojamaa, 2015, Shon et al., 2015, Vallerani et al., 2017).

However, these fleets also have a proportion of boats of greater power and length, which operate at greater distances. For example, in the Canary Islands two fishing areas have co-existed, the fleet that works in its coast and a fraction of the fleet that approaches the African coasts (Balguerías-Guerra, 1993), which continues to a lesser extent until the present (Martín-Sosa, 2012; Popescu and Ortega-Gras, 2013). In 2011, the Madeiran fleet over 12 m that works in the eastern central Atlantic and in international waters had 51 vessels; and the Azores, the fleet that operate in international waters had 117 vessels (Iborra-Martín, 2011; Ojamaa, 2015). These three fleets present a connection between the archipelagos and distant areas of very different biological characteristics. And it implies a potential risk of introduction.

How does the potential described risk affect? So far, it is not known any explicit assessment of the role that artisanal fishing can exercise in the introduction and dissemination of alien species in the archipelagos under study. However, the entry of alien species into the analyzed archipelagos caused by other vectors has been verified. In the Canary Islands, aquaculture (Toledo-Guedes et al., 2009), species associated with oil platforms (see Falcón et al., 2015, Triay et al., 2015, Pajuelo et al., 2016, Brito et al., 2017) and possibly ballast water (Riera et al., 2014; Falcón et al., 2015); and in Madeira, aquaculture (Wirtz et al., 2008) and maritime traffic (Canning-Clode et al., 2013), this last vector could also have affected the other two archipelagos too.

Simultaneously with the connections of fishing vessels, there are similar and different ones created by ships from other maritime sectors (eg maritime transport, recreational navigation, fuel extraction and transport). In the Canary Islands, the number of vessels in the Ports of Santa Cruz de Tenerife and Las Palmas has not decreased by 9,000 per year since 1989 (Puertos del Estado, 2017). In the Azores, those who make scale do not fall below 2,500 annually since 2013 (Portos dos the Azores, Relatório e Contas, 2017). And in Madeira, this number is more than 1,300 annually since 2008 (APRAM, 2017).

Another question is about those fishing vessels, as well as factory ships that are not part of the fishing fleet of these archipelagos but they dock in their ports. As an example, between 1994 and 2012, at the Port of Las Palmas, artisanal vessels arrived in 2,231 occasions, factory vessels in 2,758, and fishing vessels considered non-artisanal reached 22,722 records (data from Port Authority of Las Palmas, 2014).

Given these scenarios, it should be noted that in the marine case, the eradication of invasive species is practically impossible, with rare exceptions and in the initial stages of settlement (Maçic et al., 2018). As it has been described, the islands continually receive ships of different origins, something essential due to the external dependence of the local human population. Therefore, the risk assessment of the introduction of species is essential (Zilletti et al., 2013). Because when a strong influx of organisms is combined in a medium with many environmental alterations (eg urbanized or port coastal zones), the opportunity for invasion increases the risk of colonization in the receiving and surrounding communities (Ashton et al., 2006 in Riera et al., 2018). For this reason, ports and marinas are regions of high pressure with respect to this criterion; as observed in the pressure accumulation map, by the introduction of species in the Canary Islands (Lloret-Capote et al., 2012), and in fact it is in these places where the new introduced species are usually detected for the first time (QMEM, 2014b).

For all that has been described, a priori, it is unlikely that the role of the artisanal fleet in the introduction of new alien species will be significant compared to other routes of introduction. Even so, as recommended (Williams et al., 2013), any potential entry vector must be considered in the evaluation of the introduction routes of alien species, following the precautionary principle (Zilletti et al., 2013).

Regarding the marine species introduced in the archipelagos analyzed by various vectors of introduction, the marine strategy of the Madeira Region lists 39 species. Of which 17 (44%) were considered installed and the rest was not sufficiently known, at least in 2014 (DQEM, 2014a). The strategic environmental assessment (DMEM, 2016) exposes 59 introduced species to the Canary Islands, and includes 8 invasive species (details in its annex I, DMEM, 2016). In the Azores, the list shows 45 species; and they stand out for their invading character to 3 algae, 4 ascidians and one bryozoan (DQEM, 2014b).

The European Union (EU) covers the introduction of species through Regulation 1143/2014. However, for outermost regions (eg studied archipelagos), Regulation 1143/2014 describes the adaptation of the measures intended for invasive species at the European level. This implies the elaboration of a list of each region and the possibility of using restrictive and preventive measures for these species. In the Canary Islands, the list of invasive alien species that includes marine species is described in Real Decreto 630/2013, which also includes strategies for management, control and possible eradication. These will be governed by the minimum content established in article 16 and the provisions of Law 42/2007 (article 61.5).

In addition, the risk analysis of each species (contemplated by Regulation 1143/2014) was completed in 2018 through of Regulation 2018/968, which describes its contents and methodologies. Although Regulation 2018/968 excludes the outermost regions from its scope, the Real Decreto 360/2013 establishes that the "Strategies to fight against invasive alien species" (Chapter IV) must include a risk analysis (Article 16b). Therefore, the regional adaptation of the methodology established in Regulation 2018/968 for invasive species in the EU could be very useful in the invasive species of the outermost regions.

3.2 D2C2, D2C3

Both criteria will be considered together, according to the aim of this analysis.

Invasion involves the phases of transport, introduction, establishment and dispersal of the species (Duggan et al., 2006); and the International Union for the Conservation of Nature (IUCN) defines a species as "established" when it is reproduced without human action. In the Canary Islands, only the impact of the parasite *Sphaerospora testicularis* was observed of the 8 marine species considered invasive in 2016 (DMEM, 2016b). However, in 2017 the scope of two species of coral introduced in the main ports of Gran Canaria and Tenerife (*Tubastraea cocinea* and *Oculina patagonica*) via maritime transport was published; and Tubastrea colonies had already been observed outside the port areas (Brito et al., 2017).

Although in D2C1 it has been assumed that most of the artisanal fleets of both archipelagos do not play an important role; In relation to the criteria related to the spatial distribution of alien species already introduced, the following is proposed:

In each archipelago, what role would be played by artisanal fishing (as well as other sectors, eg coastal excursions, passenger or merchandise transport) in the possible transmission between nearby coastal areas, or between islands, of the species already introduced?

The process of invasion takes place in phases, and from the introduction of an alien species until it becomes invasive (if it becomes so) the time can be very variable (Capdevilla et al., 2013). In this process, different routes of propagation could occur in individual archipelagos, such as those sectors whose vessels often sail through different points of the coast and between the islands. Artisan fleets behave in this way between nearby ports, and even between islands (in each archipelago); and for that reason, this possibility should be evaluated in this sector and others that act in a similar way (*per* Dr. Alberto Brito).

As it has been explained and described with much caution, fishing could cause negative or positive impacts, depending on the perspective used, as other activities (Ricciardi et al., 2013). It would be negative, in terms of the possible transmission of non-native species already introduced by other sectors (if this fact were confirmed), as has been described. But in turn, it could also be positive by controlling species already introduced, by incorporating them into their catches, controlling their populations and obtaining economic benefits (Mancinelli et al., 2017). In the Canary Islands, the latter could already be happening. The sea bass (*Dicentrarchus labrax*) and the gilthead seabream (*Sparus aurata*) are caught in all the islands (data from the first fresh sale system, Canary Islands Government, 2017), and these are considered non-native in 5 of their islands (Toledo-Guedes et al., 2009). Another Canarian example is the catches observed by artisanal fishermen of introduced fish species of diverse origin, such as *Acanthurus monrivae* and *Cephalopholis taeniops* (Falcón et al., 2015); that could have already been established (Riera et al., 2014) and that are subject to capture at least on the island of Gran Canaria (*per* artisan fishermen, Oct., 2018). In addition, in the near future, perhaps the *Cronius ruber* crab, introduced in Gran Canaria (González et al., 2017), could become an objective species. The latest information on *C. ruber* shows its presence in many of the coastal areas of this island (Martín et al., 2019).

As for the synergistic effects that could influence the introduction and transmission of alien species, climate change, nitrogen deposition and disturbances related to the use of the territory, could contribute cumulatively to the impacts of these (Ricciardi et al., 2013).

With respect to climate change, a process of tropicalization of fish species has been described in the Canary Islands (Brito et al., 2005). This has also been found in other regions of the Atlantic, such as the Azores, England, the French-Atlantic coast; and in the Mediterranean Sea, thus showing an expansion of the distribution of species towards the North Atlantic (Brito et al., 2005). These facts are related to events of progressive warming of the waters and even to global climate change (Brito et al., 2005). In Madeira, something similar has been described, due to the appearance of certain macroalgae and coastal fauna (Ribeiro et al., 2019). And in the Azores, it has been suggested that the expansion of the fish *Diplodus vulgaris* could have been facilitated by climate change (Steffani et al., 2015). The temporal analysis of 30 years, on the annual change rate of sea surface temperature, in the large marine ecosystem of the Canary current, reveals positive trends in the archipelagos under study, highlighting those of the Azores (see Vélez-Belchí et al. al., 2015). This could facilitate the expansion of warmer water species (Brito et al., 2005, Espino et al., 2015).

Regarding the adoption of management, control and eradication measures; in the Canary Islands, these will be adapted to the programs of measures of marine strategies (Additional Disp. 1ª Real Decreto 630/2013). The program of measures of the Canarian demarcation establishes the need to implement/improve prevention through vector controls, early detection and eradication (vital in the marine environment, Maçic et al., 2018), and surveillance and monitoring, DMEM, 2016b). In the Canary Islands, the summation of pressures was developed as part of the introduction of marine species (Lloret-Capote et al., 2012, Annex I in DMEM, 2016b). Noting some ports are dependent on the Spanish State as areas of higher load. It is not known if a vector control or early detection plan has been established in these places.

The measures to be adopted vary depending on the nature of the species. In relation to their sampling and within the framework of the MSFD (2008), the PLASMAR project has developed a methodology for the evaluation of the introduction of alien species in Macaronesian sports harbors (Álvarez et al., 2018), which allows an evaluation and exhaustive monitoring of the sedentary species that reach these environments. As an example of adaptation to the characteristics of the species, the best type of sampling of the alien crab *Cronius ruber* in the Canary Islands has recently been evaluated, and its invasive potential, impact and possible control measures are being studied (Martín et al., 2019). Other citizen science tools (eg Poseidón or RedPromar, from the Canary Islands) could provide relevant information about changes in their distribution. It is observed that the application of effective measures that prevent the transfer of species can become complicated by their interference with the normal activity of the artisanal fishing sector, or others. But it is also understood that prevention would be an effective way to avoid the costs associated with the eradication of a species or the solvency of its impacts (such as the loss of biodiversity and the environmental services too). For the description of strategies in general, it is sent to Zilletti et al. (2013), and Maçic et al., (2018); and regarding prevention measures in the hulls of ships, to Zabin et al. (2018).

In conclusion, it is proposed to consider the role of artisanal fishing vessels in the transmission of species within an archipelago; and to other regions. However, it is not intended to place the focus solely on this sector; rather, a scenario has been described in which small vessels of different sectors or uses, and larger vessels could be developing an important role, which should also be evaluated in relation to QD 2. As described by the Canary Islands marine strategy, it is necessary to consider studies that help to evaluate the development of possible invasive species in their areas of special conservation (DMEM, 2016b), and it is understood that this can be extrapolated to the rest of the archipelagos analyzed.

3.3 References

Álvarez, S., Gestoso, I., Ramalhosa, P., Canning-Clode, J. (2018). *Protocolo de amostragem de espécies não indígenas (NIS) em marinas da Macaronésia*. Descritor 2 da DQEM. Setting the basis for sustainable maritime spatial planning in Macaronesia (PLASMAR). Programa INTERREG MAC 2014-2020.

APRAM. (2017). Administração dos Portos da Região Autónoma da Madeira. http://www.apram.pt/site/files/estatisticas2/176.pdf. Octubre de 2018.

Ashton G. (2006). Risk assessment of hull fouling as a vector for marine non-natives in Scotland. *Aquat Invasions* 1:214–8. https ://doi.org/10.3391/ai.2006.1.4.4.

Bailey S.A. (2015). An overview of thirty years of research on ballast water as a vector for aquatic invasive species to freshwater and marine environments. *Aquatic Ecosystem Health & Management* 18:1–8.

Balguerías-Guerra, E. (1993). La pesca de los costeros canarios en el banco sahariano: antecedentes históricos y situación actual. La Chopa (Spondyliosoma cantharus Linnaeus, 1758) como ejemplo de aplicación de métodos simples para la evaluación de stocks en la pesquería artesanal canaria. Tesis doctoral. Facultad de Biología. Universidad de La Laguna. 382 pp.

Brito, A., J.M. Falcón & R. Herrera. (2005). Sobre la tropicalización reciente de la ictiofauna litoral de las Islas Canarias y su relación con cambios ambientales y actividades antrópicas. Vieraea 33: 515-525.

Brito, A., López, C., Ocana, O., Herrera, R., Moro, L., Monterroso, O., Rodríguez, A., Clemente, S. & Sánchez, J.J. (2017). Colonización y expansión en Canarias de dos corales potencialmente invasores introducidos por las plataformas petrolíferas. Vieraea 45:65-82.

Canning-Clode J., Fofonoff P., McCann L., Carlton J.T. & Ruiz G. (2013). Marine invasions on a subtropical island: Fouling studies and new records in a recent marina on Madeira Island (Eastern Atlantic Ocean). *Aquatic Invasions* 8 (3): 261-270.

Capdevila-Argüelles, L., Zilletti, B. & Suárez-Álvarez, V.A. (2013). Causas de la pérdida de biodiversidad: Especies Exóticas Invasoras. *Memorias R. Soc. Esp. Hist. Nat., 2ª ép., 10.*

Davidson, I., Ashton, G., Zabin, C. & Ruiz, G. (2012). Aquatic Invasive Species Vector Risk Assessments: Te role of fishing vessel as vectors for marine and estuarine species in California. Final Report July 2012. The Aquatic Bioinvasion Research & Policy Institute. A Partnership between Portland State University and Smithsonian Environmental Research Center. 70 pp.

DMEM. (2016a). Estrategias Marinas de España. Estudio Ambiental Estratégico. Evaluación Ambiental Estratégica de las estrategias marinas. Madrid. Available at: https://www.miteco.gob.es/es/costas/temas/proteccion-medio-marino/estrategias-marinas/em_programas_medidas-lst.aspx

DMEM. (2016b). Estrategias Marinas de España. Estudio Ambiental Estratégico. Programa de Medidas de las estrategias marinas. Madrid. Available at: https://www.miteco.gob.es/es/costas/temas/proteccion-medio-marino/estrategias-marinas/em_programas_medidas-lst.aspx

DMEM. (2008). Directiva 2008/56/CE del Parlamento Europeo y del Consejo en lo que se refiere a las listas indicativas de elementos que deben tomarse en consideración a la hora de elaborar estrategias marinas. Versión consolidada de 07/06/2017.

DQEM (2014a). Secretaria Regional do Ambiente e dos Recursos Naturais. Estratégia Marinha para a subdivisão da Madeira. Diretiva Quadro Estratégia Marinha. Available at:https://www.dgrm.mm.gov.pt/documents/20143/43971/RelatorioInicial_MAD_FINAL__2014.pdf/0f2783be-bf81-5d26-83cd-15830cff998c.

DQEM (2014b). Secretaria Regional dos Recursos Naturais. Estratégia Marinha para a subdivisão dos Açores. Diretiva Quadro Estratégia Marinha. Available at: https://www.dgrm.mm.gov.pt/documents/20143/43971/RelatorioInicial_AZO_FINAL_2014+%282%29.pdf/a0b43add-3ba3-6e52-e660-11189ef15ad7.

Duggan I.C., Rixon C.A.M. & MacIsaac H.J. (2006). Popularity and propagule pressure: determinants of introduction and establishment of aquarium fish. *Biological Invasions* 8: 377–382. DOI 10.1007/s10530-004-2310-2

Espino, F., Tuya, F. & Brito A. (2015). Occurrence of the African Sergeant *Abudefduf hoefleri* (Steindachner, 1881) (Actinopterygii: Pomacentridae) in the Canary Islands waters. *Rev.Acad. Canar. Cienc*., Vol. XXVII: 83-89.

Falcón, J. M., Herrera, R., Ayza, O. & Brito, A. (2015). New species of tropical littoral fish found in Canarian waters. Oil platforms as a central introduction vector. *Rev.Acad. Canar. Cienc*., Vol. XXVII: 67-82.

Fleet Register EU. 2018. 01/06/2018. http://ec.europa.eu/fisheries/fleet/index.cfm. Comisión Europea.

González, J.A., Triay-Portella, R., Escribano, A. & Cuesta, J.A. (2017). Northernmost record of the pantropical portunid crab Cronius ruber in the eastern Atlantic (Canary Islands): natural range extension or human-mediated introduction? *Scientia Marina* 81(1): 81-89.

Iborra-Martín, J. (2011). La Pesca en Portugal. Editado por V. Kelmelyté. Departamento Temático de Políticas Estructurales y de Cohesión. Parlamento Europeo. 64 pp.

IEO (2008). Descripción general del arte y buques de caña y liña. Capítulo 3.1.5. Descripción de las pesquerías con caña y liña. En Manual ICCAT. https://www.iccat.int/es/ICCATManual.asp?mId=5. Consulta realizada 08/06/2018.

Ley 42/2007, de 13 de diciembre, del Patrimonio Natural y de la Biodiversidad. «BOE» núm. 299, de 14 de diciembre de 2007. Versión consolidada a 21.07.2018. https://www.boe.es/eli/es/l/2007/12/13/42/con

Lloret Capote, A., del Barrio Alvarellos, I. & Moreno Aranda, I. M. (2012). Estrategia Marina. Demarcación Marina Canaria. Evaluación inicial. Parte II: Análisis de presiones e impactos. Madrid. 130 pp. Available at: http://www.mapama.gob.es/es/costas/temas/proteccion-medio-marino/II_Analisis_Presiones_Canarias_tcm30-130933.pdf.

Mancinelli, G., Chainho, P., Cilenti, L., Falco, S., Kapiris, K., Katselis, G., & Ribeiro, F. (2017). On the Atlantic blue crab (Callinectes sapidus; Rathbun 1896) in southern European coastal waters: Time to turn a threat into a resource? *Fisheries Research* Volume 194: 1-8. https://doi.org/10.1016/j.fishres.2017.05.002.

Martín, JA, Triay-Portella, R, Pajuelo, JG, Lorenzo, JM & A. Luque. (2019). Sampling techniques evaluation for monitoring and control of a non-native crab population. 5th European Conference on Scientific Diving. At: Gdansk, Polonia. 24-27 april 2019.

Martín-Sosa, P. (2012). Fisheries around Outermost Regions: Canary Islands. Report for the Scientific, Technical and Economic Committee for Fisheries, European Commission. Instituto Español de Oceanografía, Centro Oceanográfico de Canarias: 61 pp.

Maçic, V., P.G. Albano, V. Almpanidou, J. Claudet, X. Corrales, F. Essl, A. Evagelopoulos, I. Giovos, C. Jimenez, S. Kark, O. Markovic, A.D. Mazaris, G. Á. Ólafsdóttir, M. Panayotova, S. Petovic, W. Rabitsch, M. Ramdani, G. Rilov, E. Tricarico, T. Vega-Fernández, M. Sini, V. Trygonis & S. Katsanevakis. (2018). Biological Invasions in Conservation Planning: A Global Systematic Review. Frontiers in Marine Science Volume 5:1-13 (article 178). doi: 10.3389/fmars.2018.00178

Ojamaa, P. (2015). Fisheries in Azores. Edited by V. Kelmelytė. Policy Department on Structural and Cohesion Policies. European Parliament. 50 pp.

Pajuelo, J.G., González, J.A., Triay-Portella, R., Martín, J.A., Ruíz-Díaz, R., Lorenzo, J.M. & Luque, A. (2016). Introduction of non-native marine fish species to the Canary Islands waters through oil platforms as vectors. Journal of Marine Systems 163: 23–30.

Popescu, I. y Ortega-Gras, JJ. (2013). La Pesca en las Islas Canarias. Editado por V. Kelmelyté. Políticas Estructurales y de Cohesión. Parlamento Europeo. 62 pp.

Portos dos Açores, Relatório e Contas, (2017). https://www.portosdosacores.pt/outrosdocs.html. Octubre de 2018.

Puertos del Estado. Ministerio de Fomento. (2017). http://www.puertos.es/es-es/estadisticas/Paginas/estadistica_Historicas.aspx. Septiembre de 2017.

Real Decreto 630/2013. Real Decreto 630/2013 de 2 de agosto, por el que se regula el Catálogo español de especies exóticas invasoras (BOE nº 185, de 03.08.2013, págs.: 56.764-56.786). CONSOLIDADO (30/03/2019). https://www.boe.es/eli/es/rd/2013/08/02/630/con

Regulation 1143/2014. Regulation (EU) Nº 1143/2014 of the European Parliament and of the Council of 22 october 2014 on the prevention and management of the introduction and spread of invasive alien species. CONSOLIDATED (04/NOV/2014). http://data.europa.eu/eli/reg/2014/1143/oj

Regulation 2018/968. Commission Delegated Regulation (EU) 2018/968 of 30 April 2018 supplementing Regulation (EU) No 1143/2014 of the European Parliament and of the Council with regard to risk assessments in relation to invasive alien species. http://data.europa.eu/eli/reg_del/2018/968/oj

Ribeiro. C., Neto, A.I., Moreu, I., Haroun, R. & Neves, P. (2019). A new signal of marine tropicalization in the Macaronesia region: First record of the mesophotic macroalga *Avrainvillea canariensis*. A. Gepp & E.S. Gepp in the Madeira archipelago.

Ricciardi A., Hoope, M.F., Marchetti, M.P. & Lockwood, J.L. (2013). Progress toward understanding the ecological impacts of nonnative species. *Ecological Monographs*, 83(3), pp. 263–282.

Riera, L., P. Ramalhosa, J. Canning-Clode & Ignacio Gestoso. (2018). Variability in the settlement of non-indigenous species in benthic communities from an oceanic island. Helgol Mar Res 72:15. https://doi.org/10.1186/s10152-018-0517-3.

Riera, R, Becerro, MA, Stuart-Smith, RD, Delgado, JD & Edgar, G.J. (2014). Out of sight, out of mind: Threats to the marine biodiversity of the Canary Islands (NE Atlantic Ocean). *Marine Pollution Bulletin* 86: 9–18.

Shon S, Delgado, J.M., Morato, T., Pham, C.K., Zylich, K., Zeller, D. & Pauly, D. (2015). *Reconstruction of marine fisheries catches for Madeira Island, Portugal from 1950-2010.* Fisheries Centre Working Paper #2015-52, University of British Columbia, Vancouver, 13 p. http://www.seaaroundus.org/doc/publications/wp/2015/Shon-et-al-Madeira.pdf

Stefanni, S, Castilho, R, Sala-Bozan, M, Robalo, JI, Francisco, SM, Santos, RS, Marques, N. Brito, A, Almada, VC & Mariani, S. (2015). Establishment of a coastal fish in the Azores: recent colonisation or sudden expansion of an ancient relict population? doi:10.1038/hdy.2015.55

Toledo-Guedes, K., P. Sánchez-Jerez, G. González-Lorenzo & A. Brito-Hernández. (2009). Detecting the degree of establishment of a non-indigenous species in coastal ecosystems: sea bass Dicentrarchus labrax escapes from sea cages in Canary Islands (Northeastern Central Atlantic). *Hydrobiologia* 623:203–212. DOI 10.1007/s10750-008-9658-8.

Triay-Portella, R., Pajuelo, J., Manent, P., Espino, F., Ruíz-Díaz, R., Lorenzo, J.M. & González, J.A. (2015). New records of non-indigenous fishes (Perciformes and Tetraodontiformes) from the Canary Islands (north-eastern Atlantic). *Cybium* 39(3): 163-174.

Vallerani, M., Martí, C.P. & Ojamaa, P. (2017). Fisheries in Madeira. DG IPOL Policy Department for Structural and Cohesion Policies. European Parliament. 15 pp.

Vélez-Belchí, P., González-Carballo, M., Pérez-Hernández, M. D. and Hernández-Guerra, A. (2015). Open ocean temperature and salinity trends in the Canary Current Large Marine Ecosystem. In: Oceanographic and biological features in the Canary Current Large Marine Ecosystem. Valdés, L. and

Déniz-González, I. (eds). IOC-UNESCO, Paris. IOC Technical Series, No. 115, pp. 299-308. URI: http://hdl.handle.net/1834/9196.

Williams, S.L., Davidson, I.C., Pasari, J. R., Ashton, G.V., Carlton, J.T., Crafton, R.E., Fontana, R.E., Grosholz, E. in D., Miller, A. W., Ruiz, G.M. and Zabin, C.J. (2013). Managing Multiple Vectors for Marine Invasions in an Increasingly Connected World. *BioScience* Vol. 63 No. 12. 952-966 pp.

Wirtz, P., Fricke, R. & Biscoito, M. (2008). The coastal fishes of Madeira Island—new records and an annotated check-list. *Zootaxa* 1715: 1–26.

Zabin, C. J., Davidson, I.C., Holzer, K.K., Smith, G., Ashton, G.V., Tamburri, M.N. & Ruiz, G.M. (2018). How will vessels be inspected to meet emerging biofouling regulations for the prevention of marine invasions? *Management of Biological Invasions* Volume 9 (3): 195–208. https://doi.org/10.3391/mbi.2018.9.3.03

Zilletti, B., Capdevila-Argüelles, L. & Suárez-Álvarez, V.A. (2013). La lucha contra las especies exóticas invasoras: una cuestión de estrategia y compromiso. *Memorias R. Soc. Esp. Hist. Nat., 2ª ép., 10.*

4 Analysis >> QD3 Commercially-exploited fish and shellfish

QD3 Populations of all commercially-exploited fish and shellfish are within safe biological limits, exhibiting a population age and size distribution that is indicative of a healthy stock						
Criteria (element)	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
The *Fishing mortality* rate of populations of commercially-exploited species	D3C1	yes	broader	yes	yes	yes
The Spawning Stock Biomass of populations of commercially-exploited species are above biomass levels capable of producing maximum sustainable yield.	D3C2	yes	broader	yes	yes	yes
The age and size distribution of individuals in the populations of commercially-exploited species is indicative of a healthy population. This shall include a high proportion of old/large individuals and limited adverse effects of exploitation on genetic diversity.	D3C3	yes	broader	yes	yes	yes

4.1 D3C1

In the Canary Island, Madeira and Azores area there is no information about indicators of safe biological limits or reference points of populations of commercially-exploited fish and shellfish species. The REPESCAN report (Gonzalez, 2008) put into evidence the lack of biological and fishery information of most of the target species in the Canary Islands, and assume that most of the bentho-demersal fishing resources targeted by the small-scale fishery are currently overexploited. Moreover, this report indicates that no information about the status of the small and middle-size pelagic fish stock is available. In this way, Castro et al. (2015) indicated that catch per unit of effort of the trap fishery have been decreased by the 90% since 1970. In the same way, Morato (2012) made a detailed description about the knowledge of fish stocks and fisheries in Azores and Madeira, concluding that the status of most of the targeted species is unknown, particularly in Madeira fishing grounds. ICES recommended that catches of horse mackerel (*Trachurus trachurus*), anchovy (*Engraulis encrasicolus*) and sardine (*Sardina pilchardus*) should not be allowed to increase in Azores (ICES 2011). Moreover, some indicators suggested that abundances of black scabbardfish (*Aphanopus carbo*) are declining in some areas of the Northeast Atlantic (Lorance &

Dupouy, 2001; ICES 2008), particularly in Madeira (Morato, 2012), which motivated the implementation of TAC system in 2003 (ICES, 2011).

On the other hand, the minimal length of capture of only thirty species has been regulated in the Canary Islands (Real Decreto 560/1995; BOE 84 de 08/04/1995, updated with the Real Decreto 1076/2015; BOE 285 de 28/11/2015), including tuna species that are regulated according minimal wet weight following ICCAT recommendations. However, most of the minimal length of capture are below the length of first reproduction (González et al. 2012). In Azores and Madeira, more restricted minimum length values were implemented for several fish species in 2009.

Environmental impact

The most obvious effect of fishing on target and non-target species is direct mortality (Pope et al. 2000). For stocks having analytical assessments, exploitation rate (fishing intensity) is usually quantified by fishing mortality (F) (ICES, 2008) and corresponds to the proportion of the population or biomass of a fish stock that is removed per year (FAO, 2010). The precautionary approach in fisheries in Europe, introduced in 1997, has led to the adoption of reference points for F and spawning stock biomass (SSB) (ICES, 2008), and the main focus of fisheries management in the NE Atlantic and elsewhere has been the regulation of F to ensure a target B is achieved (Froese et al., 2008; ICES, 2008).

Exploitation pattern (fishing selection) is defined as the distribution of fishing mortality over the different age/size components of a fish population (FAO, 2010) and depends on the selectivity of the gears used in a fishery and on the extent to which particular age/size classes can be selectively targeted. Exploitation pattern is broadly indicative of the proportional exploitation of juveniles because size in fish is linked to their maturity stage. A relevant metric of the proportional exploitation of juveniles (juvenile exploitation index – JEI) that results from a given exploitation pattern is the fishing mortality of immature fish divided by that of mature fish (JEI = Fimm/Fmat; Vasilakopoulos et al., 2011). Vasilakopoulos et al. (2011) analysed data for 38 ICES stocks, where F and JEI (then named ER and EP, respectively) were averaged over a standardised time-period, to show that both F and JEI have independent, negative effects on stock status, providing empirical evidence for the benefits from protecting juveniles. That study also showed that values of F > 0.63 and JEI > 0.50 are associated with a higher probability of individual stock status falling below precautionary limits (Vasilakopoulos et al., 2012).

In the Canary Islands area, F and Exploitation Rate (E) have been estimated for a few seabream species (*Dentex gibbosus*, *Diplodus cervinus*, *D. sargus cadenati*, *Pagellus acarne*, *P. erythrinus*, *Pagrus pagrus*, and *Spondyliosoma cantharus*) and red mullet (*Mullus surmuletus*) (Lorenzo & Pajuelo, 2002; Pajuelo & Lorenzo, 1995, 1996, 1998, 1999, 2000, 2004; Pajuelo et al., 1997, 2003) from more than one hundred that are targeted by the small-scale fishery. However, for Madeira we have only found F estimation for the blue jack mackerel, *Trachurus picturatus* (Vascocelos et al., 2018). Moreover, no data are available of the proportion of juvenile fish of each species in commercial captures in none of the small-scales fisheries of the Macaronensian archiepelagos, because no length sampling programmes are done regularly, and no information of discard composition is provided.

While in the Canary Islands data of commercial captures are only systematically recorded after 2006 (Castro et al., 2015), the Azores and Madeira have an efficient and unique system for fishery data collection dating back to the 1970s (Morato, 2012). With the exception of part of the pelagic longline catch landed outside the Azores, and tuna going directly to processing factories, all catches by Azorean vessels must be landed at the auction houses distributed throughout the islands (Pham et al., 2013). Nevertheless, in a general context of poor-data fishery that could be applied to the

Macaronesian small-scale fisheries, total mortality (P/B) and fishing mortality of target species groups could be approximate from models. In this way, and through the Ecopath with Ecosim model, Couce-Montero et al. (2015) and Morato et al. (2016) provide estimations of P/B coefficients for several heterogeneous groups of species, and single representative target species, including zooplankton, shellfish, fish, turtles, birds and mammals, respectively for Gran Canaria Island and Azores areas. In the model of Gran Canaria, Couce-Montero et al. (2015) concluded that 43% of fished groups were overexploited, particularly benthic sharks, groupers, breams, parrotfishes, and leatherjacket fishes. In this way, Castro et al. (2015) reported that CPUE of the trap fishery of the Canary Island have decreased by 90% from 1970 to 2014, indicating a chronic overfishing of almost all target species in all range of depths (0 to over 200 m), as was previously indicated by García-Cabrera (1970) and González (2008).

A decreasing trend in the length composition of blue jack mackerel in Madeira was observed between 2002 and 2016, and length at first maturity decreased by 2.78 cm TL (Vasconcelos et al., 2018). Similar decreasing tendency in mean length (-1.2 cm/year) has been mentioned for *Epinephelus marginatus* in the Canary Islands (Jiménez-Alvarado et al., 2019), and it is suspected to occurred a similar process also in *Pagrus pagrus*. The black spot seabream (*Pagellus bagaraveo*), in the Azorean commercial handline and bottom longline fisheries, has been found in recent years to be showing signs of reproductive stress (Stockley et al. 2005). According to ICES, 2012), the abundance of the blackspot seabream stock in the Azores has decreased by more than 20% in 2007–2009 and 2010–2011. Additionally, considering that exploitation is unknown, ICES advised that catches should decrease by a further 20% as a precautionary buffer (ICES, 2012). Analysis of the survey abundance data also suggests that some traditional commercially important demersal or deep water species, like the alfonsinos (Silva & Pinho, 2007) or bluemouth rockfish (Perrota & Hernandez, 2005) are intensively exploited.

Ecological risk assessments for priority species of sharks caught in ICCAT fisheries demonstrated that most Atlantic pelagic sharks (i.e. blue shark and shortfin mako) have exceptionally limited biological productivity and, as such, can be overfished even at very low levels of fishing mortality (ICCAT, 2012, 2017). For both North and South Atlantic blue shark stocks biomass is believed to be above the biomass that would support MSY and current harvest levels below F_{MSY} (ICCAT, 2012). Estimates of stock status for the North Atlantic shortfin mako indicated stock depletion to about 50% of biomass estimated for the 1950s. In 2017, the Standing Committee on Research and Statistics (SCRS) reported depletion and ongoing overfishing of North Atlantic shortfin makos, estimated that only a 0t catch (including dead discards) could result in population rebuilding by 2040 (54% probability), and recommended a “complete prohibition on retention” as the most effective immediate measure for achieving this goal (ICCAT, 2017). On the other hand, preliminary 2017 data reveal that North Atlantic blue shark catches could well exceed the threshold established by ICCAT in 2016. This threshold should be transformed into a total allowable catch limit to better prevent overages.

In relation to benthic and deep-water elasmobranchs species, there is no information about the status of species but it is suspected that most of them are overfished, and some are critically endangered or in risk of extinction (i.e. *Squatina squatina*). In this way, Pajuelo et al. (2018) reported that bycatch of benthic sharks in the semi-floating trap fishery accounted by 0.11% in Madeira and 0.7% in Canary Islands. Morato (2012) indicated that deep-water sharks were represented by at least 10 species and accounted for about 135 t/year, the 16% of the discarded organisms in the bottom longline and handline fisheries of Azores. These species are never landed and accounted in average for 135 tonnes of total discarded amount per year. In the drifting longline fishery of Azores, Machete et al. (2011) found by-catch values varying between 3% and 5% of the black scabbardfish catch, but a recent report (seaExpert, 2012) estimated a much higher by-catch of deep water sharks (gulper shark, *C. squamosus* and the Portuguese dogfish, *Centroscymnus* sp., composed the main by catch).

According to Morato (2012), total biomass of marine turtles killed as a result of bycatch of the longline fleet of Azores was estimated to average about 7.0 tons per year, much higher than reported for the other European fleets (Pham et al. 2013). However, not all sea turtles caught die but no estimates of hooked loggerhead mortality after gear removal are available (Ferreira et al., 2001; Lewison et al., 2004). Apart from that, the Mediterranean monk seals (*Monachus monachus*) are among the most endangered marine mammals in the world that have been impacted by fisheries. Moreover, Hale et al. (2011) has reported and increase in the fishery interactions with marine mammals in Madeira.

<u>Environmental impact spatial extent</u>

Piet et al. (2000) argue that annual fishing mortality on benthic fauna should preferably be based on relevant environmental strata, and accuracy of the estimates increases markedly when the resolution of spatial fishing effort data sufficiently reflects the patchiness of the fleet's activities. In this way, there are no information about the impact of fishing mortality, or fishing effort exerted, on different target fish and shellfish species by spatial or depth strata in the Canary, Madeira and Azores archipelagos. Nevertheless, the Macaronesian archipelago is composed by oceanic islands of volcanic nature, independent of the nearby continents (Africa in the case of the Canary Islands and Madeira, and Europe for the Azores). In addition, the insular buildings are separated from each other by depths ranging between 2000 and 3000 m, or more (Vallerani et al., 2017; Carracedo, 2018). The lack of continuity in the shelf between the islands, and with the neighboring continents, has important repercussions in the distribution of the species and in their biological characteristics and, therefore, has to be considered in the management of the fishing exploitation of each island. This fragmentation of the shelves, with discontinuities marked by great depths of the order of several thousand meters, means that the submarine zone located on the insular shelf also acts as an island for the benthic and benthopelagic species, and for some pelagic-coastal ones (i.e. chub mackerel, sardines, etc.). The great depths that surround each island, beyond the insular shelf, can act as an insurmountable physical barrier for many of the adult forms of these species (Huston, 1985), but not for larval drift (Rodríguez et al., 1999, 2009). Therefore, the populations (or stocks) of species of marine flora and fauna, including fish, of the islands are geographically limited to the waters surrounding them (Brito et al., 1996, Sangil et al., 2011). However, the consequences of this phenomenon do not only have implications on the biology of the species, but also strong repercussions on the local fisheries. Most of these oceanic island communities, like a coral reefs (Huston, 1985), are in a dynamic equilibrium and heavily dependent on an uncertain recruitment that makes them highly unstable, never reaching a persistent equilibrium. This increases their vulnerability to adverse climatic phenomena and excessive fishing pressure (Gardner et al., 2005; Caballero-Afonso et al., 2010; Solari et al., 2010).

This "quasi insulation" of the neritic biological communities and the limited carrying capacity of insular marine systems mean that the fishing configuration that is build on insular stocks also has clear peculiarities, which are reflected in the fishermen communities, the fishing systems, as well as fishing intensity. And the oceanographic, geomorphologic and bio-ecological characteristics of each island strongly condition each small-scale fishery system. These differences are not only between islands, but also along the coastline and fishing grounds of the same island. In this way, the greatest fishing pressure is located in the areas sheltered from the prevailing winds (trade winds coming mainly from the north or northeast), that in the Canary Islands coincide in part with grounds where the shelves are wider (Bas et al., 1995), and where the abundance of the exploited stock is greater (Pastor and Delgado de Molina, 1985, Bordes et al., 1987, 1993, 1995, 1998). Therefore, it is not surprising that these fishing grounds are also close to where fishing infrastructures are concentrated, and so the fleets. In addition, it is in these areas where the use of certain fishing gears, such as purse-seine and gill-nets, is more frequent.

In this context of biological "quasi insulation", it is foreseeable that fishing impacts on fish and shellfish benthic-demersal species, and probably also on small pelagic neritic ones (e.g. mackerel-like species), could be limited to each island fishing grounds, because the great depths between these oceanic islands prevent fluxes or displacement of adults and juvenile individual of those species (but not of eggs and pelagic larvae) beyond the insular shelves, working each island as an isolated system for these fish stocks. So, it is possible to find that level of depletion of fish stocks, or ecosystem/habitat/community conservation, could be relatively different between neighbouring islands, as has been reported between Salvage Islands and Madeira or Canary ones (Friedlander et al., 2017). In this way, in the archipelagos of Canaries and Madeira have been reported high biomass of the sea urchin *Diadema africanum*, and the lack of its fish predators (e.g. hogfishes, triggerfishes), as an indicator of ecosystem unbalanced due to overfishing (Tuya et al., 2005; Hernández et al., 2013; Riera et al., 2016; Friedlander et al., 2017). This high abundances of sea urchins have not been observed in Salvage islands (65% less than in Madeira), but the biomass of top predators was more than 10 times larger than in Madeira, and sea urchin predators were also observed in higher abundance that in this last archipelago (Friedlander et al., 2017).

Curiously, in Madeira, the trophic level of landings has been increasing since 1938, but increased more steeply in the transition from the 1980s to the 1990s, remaining relatively stable at a high level since then. Likewise, the Fishing in Balance (FiB) index also increased steeply during the same time period, indicating an expansion of the fishery. Due to its oceanic and oligotrophic environment, fisheries in Madeira, and probably also in the Canary and Azores islands, have become increasingly dependent on a very small number of high trophic level, migratory species (tuna and tuna-like species), a situation which is ecologically unsustainable. In the period studied, Long relative price index increased, indicating that high trophic level species had become more valuable in relation to species feeding at lower trophic levels. It is likely that the persistence of present trends will compromise the sustainability of fisheries (Baeta et al., 2009). According to Garcia et al. (2012), fishing strategies addressed to selective capture of species, sexes, and sizes in proportions that differ from their occurrence in the ecosystem neither maximizes production nor minimizes impacts. A more balanced harvesting would more effectively mitigate adverse ecological effects of fishing while supporting sustainable fisheries. An exploitation pattern focused on a better utilization of locally available bentho-demersal and small-pelagic fish, would increase sustainability of fisheries in the region (Hermida & Delgado, 2016). Balanced fishing across a range of species, stocks, and sizes could mitigate adverse effects and address food security better than increased selectivity (Garcia et al., 2012).

On the other hand, the amount of baitfish used by the pole-and-line fishery is not reported to local authorities, remaining largely unknown and unmanaged, and is composed by juveniles stages of several small-pelagic fish species. In Azores and Canary Islands, in average, the most frequent species of baitfish were sardines (*Sardina pilchardus* and *Sardinella aurita*), makerel-like species (*Scomber colias* and *Trachurus* spp.), bogue (*Boops boops*) and anchovy (Rico et al. 2002; Morato, 2012; Herrera-Perdomo, 2017). Nevertheless, although baitfishes are caught at anywhere and at anytime, because the tuna fishery is carry out along the year, but on different tuna species (depending of the migratory patter of tropical and template species;

Ganzedo-López, 2005; Delgado de Molina et al., 2005, 2012), most of the baitfish is obtained in shallower waters, near the nursery areas (on meadows or seagrass areas), but also in open waters (near the slopes or even close to cyclonic eddies areas). This open-water baitfish is mainly obtained by the larger boats dedicated to fish tuna far from the islands grounds, while the shallow-water bait is more frequently caught by boats that normally fish near their base-port, and mainly in march (for the bluefin tuna fishery) and during the summer (mainly for skypjack tuna fishery) (Herrera-Perdomo, 2017).

Trawling is forbidden in the Macaronesian Archipelagos, and it is well established that it has dramatic impacts on benthic ecosystems, with damages on habitats that impact directly on stock biomass and recovering (Watling and Norse, 1998). This negative impact on ecosistems could have effects during several decades or a century. In a single bottom trawl, between 5 and 25% of the benthic organisms are eliminated, with cumulative effects in the successive trawls (Poiner et al., 1998). In addition, this type of gear is the one that has a higher proportion of discards, with only 20% or less of the total catch obtained remaining on board (Alverson et al., 1994). Other techniques such as longlines have been suggested to be less harmful to the environment (Chuenpagdee et al., 2003), but a typical longline set in the Azores had an expected bycatch of 1.23 sessile organisms or 0.96 coldwater corals (0.48kg ±0.16) much smaller than the expected bycatch of a bottom trawler operating in the Flemish Cap towing over the same area (37-59 kg; Murillo et al., 2011). Bycatch was found to be higher between 200 and 450 m depth and on seamounts when compared to island shelves. Analysis of video footage suggested that additional impacts were found in the sea bed with some cold water corals being seriously impacted by bottom longlines. Longline have a selective impact on mostly 3-dimensional and branched colonies (Sampaio et al., 2012) which may alter benthic community structure (Morato, 2012).

Maritime Activity pressure solution

The effective conservation of species vulnerable to fishing requires predictions of sensitivity and exposure to fishing mortality, but such predictions are hard to make when the population dynamics of most off these species have not been described (Le Quesne & Jennings, 2012). Appropriate targets levels of F for rebuilding will depend on the extent of overexploitation and on the economic impacts of the action.

The current fisheries resource management strategy of the Azores and Madeira is based on the EU Common Fishery Policy, implemented primarily through Total Allowable Catches (TACs) for various species including blackspot seabream (*Pagellus bogaraveo*), alfonsinos (*Beryx splendens* and *B*. *decadactylus*), and deepwater sharks such as *Deania* spp., *Centrophorus* spp., *Etmopterus* spp., *Centroscymnus* spp. and kitefin shark, *Dalatias licha* (EC Reg. 2340/2002; EC Reg. 2270/2004). Apart from fish quotas, the Azores and Madeira Regional Governments have implemented technical measures such as minimum landings sizes or weights, minimum mesh sizes, allowable percentage of bycatch species, area and temporal closures (Morato et al., 2010; Ojamaa, 2015) and ban on the use of specific gear. Moreover, the Azores and Madeira regulation prohibited deep-sea trawling, that according to Morato (2012), recently became an EC regulation (EC 1568/2005) and the Azorean box of 100 miles limiting fishing to vessels registered in the Azores created in 2003 under the CFP (EC Reg. 1954/2003).

On the other hand, in the Canary Islands, the length of first capture have been fixed for about 30 fish species and 9 shellfish species (Decreto 134/1986; Reglamento (CE) Nº 850/98; Real Decreto 1076/2015), list that is expected to be increased to about 60 species in the current year. Moreover, the capture of 15 shellfish species and 9 bony fishes (Decreto 182/2004), 12 pelagic and deepwater sharks (*Carcharadon carcharias*, *Cetorhinus maximus*, *Lamna nasus*, *Isurus oxyrhinchus*, *Prionace glauca*, *Squatina*

squatina, between others), and 11 ray species (*Manta alfredi*, *Manta birostris*, *Rostroraja alba*, *Taeniura grabata*, *Gymnura altavela*, etc.) has been forbidden in waters of the Archipelago (Reg UE 2015/104; Reg UE 2016/72; Reg UE 40/2013 ARM fin/2689/2009). The Canary Islands Government and the Ministerio of Agriculture and Fisheries of Spain have regulated the size and mesh traps for fish and shellfish, purse seine, gillnets and number of hook of the longlines. Trawling, including beach trawl, are forbidden (Decreto 182/2004; Orden AAA/2536/2015).

There are currently 11 designated marine protected areas (MPAs) in the Azores and 6 (Santos et al., 1995; Morato, 2012). In the Canaries there are three MPA, but the management of two of them (Graciosa and Restinga) is shared by the Canary and Spanish Governments, but that of La Palma is managed by General Secretary of Sea.

The impact of longline fishing on sea turtles in the Azores could be diminished through the regulation of the blue shark fishery (Ferreira et al., 2001; Aires-da-Silva et al., 2008). Ferreira et al. (2011) suggested mitigation measures to reduce turtle by-catch in the Azores, including policy that requires vessels to move away from fishing areas after high catch rates of turtles, longline fishing ban in aggregation areas, and selected gear modifications. There are no data of incidental captures of sea turtles in the Canary Islands area. Nevertheless, has been reported the stranding of several individuals of *Caretta caretta* and *Eretmochelys imbrincata*, some of them after interact with fishing gears (Camiñas, 2000). In the Spanish law, *Caretta caretta* is initially registered in the Spanish Catalog of Endangered Species (Real Decreto 439/1990) as "of special interest", and in its subsequent revision, as "vulnerable" (Real Decreto 139/2011). In the Canary Islands catalog of protected species (Law 4/2010), the loggerhead turtle is listed in its annex VI, in the category of "special interest" (OAG, 2018).

Impact mitigation measures

Since stock rebuilding generally requires several years, fishing intensity need to be reduced continuously for the required period. For relatively long-lived species, Rosenberg and Brault (1991) showed that rebuilding over moderate time spans (say 5 years) is less economically destructive than short, sharp reductions in fishing mortality (2-year rebuilding scenarios), but that longer rebuilding periods are likely to be too long to see signs of effective recovery. For many stocks which are currently heavily exploited, larger-than-normal cohorts make up a progressively larger part of the annual yield, but may not occur very frequently (Caddy and Mahon, 1995).

Alves et al. (2018) indicated that: (i) the MPA could be acting as a refuge for local biodiversity, ii) communities from the highly fished area could be suffering an impoverishment of local biodiversity, and iii) communities from the highly urbanized area would be enriched by the establishment of opportunistic species. These findings support that the level of human-pressure likely plays an important role in the composition of benthic communities in this insular ecosystem, although this was more relevant at the shallower stratum where the key grazer *Diadema africanum* explained 65% of the variance of benthic assemblages. It is suggested that MPA of small dimension and proximity to human impacted areas are limiting the survival of predators of the D. *africanum*.

Based on the decline in the abundance of the stock of black scabbardfish in Northern Europe fishing areas, WGDEEP suggested that fishing effort should be reduced significantly (ICES, 2006). This conclusions lead to the introduction in 2003 of management measures, based on fishing licenses and a TAC enforcement. In subarea CECAF 34.1.2, were Madeira is inserted, TAC suffered its first cuts in 2011, with a reduction from 4,285 to 4,071 tonnes. In the same subarea, for the year of 2012, the TAC was established on 3,867 tonnes.

Monitoring methods

Pope et al. (2000) suggest that to quantify vulnerability of species to fishing requires measurement of the current fishing mortality rate and of the tolerance of the species to fishing mortality. But, in non-target species is difficult to estimate fishing mortality due to the little-studied of them. Pope et al (2000) described two potential methods for estimating current fishing mortality rate when data are limited. The extended length cohort analysis, based in the method developed by Jones (1981), offers a practical approach for common non-target species. Given a modest amount of discard sampling data and biological information, the method can be used to estimate current fishing mortality rate, and its inversion can estimate the potential jeopardy level. Also the method can be used to predict the steady-state catch and spawning-stock biomass expected under any new rate and patterns of exploitation. On the other hand, fishing mortality could be also estimate based on swept-area approaches. In this last approach, local F is considered proportional to the fraction of the area swept by fishing gear, so it is necessary to have an estimation of fishing effort for example tracking vessel by satellite (e.g. the blue or green cages). (Pope et al., 2000).

4.2 D3C2

Environmental impact

Spawning-stock Biomass (SSB) is assumed to be a proxy for stock reproductive potential, and due to the dependence of recruitment on the spawning stock size, SSB has been considered as a reference point to ensure that the spawning capacity of the stocks is conserved. The reference points based on recruitment considerations may be derived from stock-recruitment (S-R) relationships, or from an extension of yield per recruit analysis which incorporates age/size at maturity in calculating the spawning biomass per recruit (SPR) at various levels of F. According to Caddy and Mahon (1995), these two types of analysis have been linked to calculate the stock biomass levels associated with various SPR levels. The targets may be stated in terms of a stock biomass or spawning stock biomass (SSB) that is expected to yield the desired recruits, or in terms of the fishing mortality level which is expected to result in these biomass or SPR levels. A major problem with S-R analysis is that a relatively long time-series spanning a range of stock sizes is needed to produce a reliable stock-recruit curve. The calculation of SPR is an extension to yield-per-recruit analysis which can be carried out in the absence of historical data, if information on maturity/fecundity at size/age is available. Moreover, to calculate the SBB it is necessary to estimate of the number of fish by age group, the average weight of the fish in each age group, and the amount of fish in each age group that are mature. Nevertheless, De Lara et al. (2007) indicated that the ICES precautory approach, based on SSB and F indicators is sustainable only when recruits make significant contribution to SSB. In this case, advice based upon SSB, with an appropriate reference point, is sufficient to ensure sustainability, but in all other cases must be complemented with other management indicators. However, Scott et al. (1999) indicated that the use of spawning stock biomass as a direct measure of reproductive potential may not be valid because of age- or size-specific differences in fecundity and the effect of maternal size and condition on offspring viability.

The maritime European area covered by the Macaronesian Archipelagos are distributed in two big FAO fishing areas for statistical purposes in relation to fisheries, the Northeast Atlantic fishing area, or area 27 (including Azores), and the Central-east Atlantic fishing area, or area 34 (including Canary and Madeira islands). The fishing area 27 is under the scientific supervision of the ICES (International Council for the Exploration of the Sea), while the fishing area 34 is under the management of CECAF (Committee for the Central Easten Atlantic Fisheries). In this geographical framework, Jayasinghe et al. (2017) indicated that fisheries of the subareas 27(I + II, V) appear sustainable according the higher Mean Trophic Level (MTL) in landings and higher Spawning-stock Biomass (SSB) after 2008. But, in subareas 27- VIII and 27- IX the

SSB indicated that were overfished. In subarea 27-VI fish stocks are recovering, but fish stocks in subareas 27-III, 27-IV and 27-VII were heavy fishing. These authors reported that other factors such as eutrophication, seafloor disturbances, marine pollution, invasive species etc., influence SSB ecosystem health options and should also be incorporated in the management criteria. However, for most of the stocks of the Macaronesian area, there are no estimate of spawning stock biomass or parameters that permits its estimation. There is no assessment of SSB or SPR for fishing subarea 27-Xa2, corresponding to Azores, neither for subdivision 34.1.2 corresponding to the Canary Islands and Madeira, for none of the target species. In both fishing areas there are no long-time series that permit to produce stock-recruit curves, but there is information about maturity/fecundity at size/age of several target species, particularly sparidae species and other benthic-demersal species, including octopus, in the Canary Islands (Pajuelo & Lorenzo, 2001; Lorenzo et al., 2002; Hernández-García et al., 2002; González et al., 2003; Pajuelo et al., 2008; between others), *Pseudocaranx dentex* and few demersal species in Azores (Estácio et al., 2001; Afonso et al., 2008), and chub mackerel, and black scabbardfish in Madeira (Figueiredo et al., 2003; Vasconcelos et al., 2012; Bordalo et al., 2001).

The fish larvae community in Azores was dominated by mesopelagic and bathypelagic species, a typical oceanic island environment composition (Arkhipov & Mamedov, 2008; Sobrinho-Gonçalves & Isisdro, 2001). Fish larvae showed an abundance minimum in May and a maximum in June, presenting a general negative relationship with zooplankton biomass and indicating a temporal asynchrony between their annual cycles of production. Moreover, Rodríguez (2000), Bécognée et al. (2006), and Moyano and Hernández-León (2009) reported that temporal and spatial variations of the larval fish community off the Canary Islands was mainly composed by myctophids, followed by sparids, clupeids and gonostomatids. These authors suggest that there are two seasonal larval assemblages corresponding to the two main characteristic periods of the water column in these waters: mixing (winter) and stratification (summer).

According to Couce-Montero et al. (Submitted), if the current fisheries management strategy continue in the Canary Islands, the biomass of the main commercial species would in 2030 be diminished by more than 90%, in relation to that estimated in 2005, with a predictable collapse of some target species. At this point, it is important to mention that in 1970, García-Cabrera warned that the fishing grounds of all the islands with depths less than 100 m were overfished. Moreover, González (2008) confirmed García-Cabrera's findings thirty-eight years before, and indicated that this phenomenon had spread to all fishing grounds and the entire range of depths at which the artisanal fleet operate. But, Castro et al. (2015) estimated that the catch per unit of effort (CPUE) for the trap fishery, based on benthic-demersal species, for Gran Canaria island has been reduced by almost 90% relative to what it was in 1970.

<u>Environmental impact spatial extent</u>

There is no information about spatial extent of impacts of fishing on SBB in none of the fishing grounds were operate the Macaronesian fishing fleets.

On the other hand, Scott et al. (1999) quantified how changes in the age composition of the spawning stock, due to a range of fishing pressures and under different stock–recruitment relationships, could influence the reproductive output. Their results suggested that if the effects of the loss of more fecund older/larger individuals in the population are not considered, the number of potential recruits produced by populations under higher levels of fishing mortality could be overestimated by as much as 60%. When age/size-related maternal effects on egg viability are also considered, the amount of potential recruits can be overestimated by a further 10% in the heavily exploited populations. In this way, it is feared that loss of old and large spawners impairs heavily fished fish stocks' reproductive capacity and increases their sensitivity to environmental fluctuations, and has been predicted to drive stocks toward earlier maturation and smaller adult body size (Garcia et al. 2019). So, Stige et al. (2017) reported that there is a link between demographic structure and abundance and distributional extent of eggs but not between egg distribution and recruitment, questioning the benefits of a wide spatiotemporal distribution of spawning are of quantitative importance for recruitment. Also, Castro et al. (2018) assessing the impact of recreational spearfishing in the Canary Islands, indicated that those fishermen address their fishing effort toward large specimens of a few species (e.g. groupers, seabreams and parrotfish), that also are targeted by other recreational and artisanal fishermen. Some of these species are currently classified as overfished or rarefied (Aguilera-Klink et al., 1993; González, 2008), or suffer process of sexual inversion when reach larger sizes (Pajuelo y Lorenzo, 1995, 1996; Pajuelo et al., 2003, 2006a, 2006b). This higher selectivity of spearfishing could produce a more destructive fishing on vulnerable species, as has been indicated by Johannes (1978), Dayton et al. (1998), Pogonoski et al. (2002), Bikeland y Dayton (2005) or Frisch et al. (2012). In this sense, more that 50% of target species of spearfishermen in the Canary Islands show high of very high vulnerability index (Cheung et al., 2005), and high fishing pressure inhibit the recovering of these species.

In relation to spatial distribution of fish larvae, Moyano and Hernández-León (2009) and Moyano et al. (2009) reported that the higher egg and neritic larval abundnace was found in the leeward and windward retention areas of the Canary Islands. However, seasonality showed a stronger influence on the annual larval assemblage than sampling site, as the latter was not significant on a long time scale. Moreover, Sobrinho-Gonçalves & Isisdro (2001), when compared fish larvae abundance with those of other NE Atlantic regions suggested that the zooplankton production in the Azores may be similar to the one found in the Iberian Peninsula continental shelf. Significant spatial density gradients (for fish larvae and zooplankton) were not found with either bathymetry or distance from shore. The species and quantitative composition of ichthyoplankton of the South Azores seamounts is far more numerous than in the North Azores seamounts (Arkhipov & Mamedov, 2008).

Maritime Activity pressure solution

Couce-Montero et al. 2019, using the Ecopath with Ecosim, consider that from all scenarios they tested, only those that significantly reduce the high effort of the recreational fishing would allow the recovery of the most exploited stocks in the marine ecosystem in the short- and medium-term in Gran Canaria (Canary Islands). Moreover, the best management strategy, in terms of impact on abundance, was obtained with a scenario that has a spatial partition of exploitation rights between artisanal and recreational fishermen, and includes no-fishing areas (MPAs).

Impact mitigation measures

Reference points which indicate when a fishery is entering in areas where resources production is in danger, and immediate actions are needed, can be referred to Target Reference Points (TRPs) and Limit Reference Points (LRPs) (Caddy and Mahon, 1995). TRPs are defined as the level of fishing mortality (or of the biomass), which permit a long-term sustainable exploitation of the stocks, with the best possible catch. TRPs indicates to a state of a fishing and/or resource which is considered to be desirable and at which management action, whether during development or stock rebuilding, should aim, and MSY has most often been used in this sense. A LRP (maximum values of fishing mortality or minimum values of the biomass, which must not be exceeded) indicates a state of a fishery and/or a resource which is considered to be undesirable (e.g. a dangerously low spawning biomass, a high rate of decline in stock size, or a high mortality rate) and which management action should avoid (Caddy and Mahon, 1995). In poor-data fisheries, qualitative or semi-quantitative criteria also can be used directly as LRPs. Even when there is adequate information for the definition of sophisticated LRPs, but there are broader ecological concerns about the sustainability of benefits due to the possible impacts of exploitation on the ecosystem, it may be desirable to set LRPs using a precautionary approach (Garcia 1994). The most well known Ftarget is $F_{0.1}$ but are also used Fmax, Fmed, and F_{MSY}. F_{MSY} or length of capture above the length of fist maturity are used as LRP.

Mace (1994) observed that TRPs and LRPs are highly dependent on the degree of density dependence in the S-R relationship. She recommended that when the S-R relationship is unknown, $F_{40\%}$ be adopted as a target fishing mortality, but that it be adjusted to accommodate any known or assumed degree of density dependence in the S-R relationship. This corresponds to a recruitment of about 50% of that expected from a virgin stock. For recruitment-based TRPs where biomass is stated in relation to virgin biomass, the latter is estimated from the intersection of the S-R curve or mean recruitment with the replacement line corresponding to F=0, the unfished condition.

Monitoring methods

Target Reference Points (TRPs) management requires active monitoring and continual readjustment of management measures on an appropriate (usually annual) time-scale. It also requires attention to the effect of a variety of sources of uncertainty on the estimates of the TRP and of the stock status. Data series required to fix TRPs Limit Reference Points (LRPs) (Caddy and Mahon, 1995) are basically capture and fishing effort, but also the monitoring of length (or age) frequency distribution of target species in captures, an biological data (i.e. length at maturity, natural mortality, length of recruitment, etc).

In poor data fisheries, as are the Macaronesian ones, catch and effort data may be sufficient to estimate sustainable biomass levels. Froese et al., 2017 presented a Monte Carlo method (CMSY) for estimating fisheries reference points from catch, resilience and qualitative stock status information on data-limited stocks and a Bayesian state-space implementation of the Schaefer production model (BSM), fitted to catch and biomass or CPUE data. For data-limited fisheries, quantitative data on SSB in relation to reference points are not available, but these methodologies provide a qualitative estimation of biomass against reference points (B/B_{MSY}).

However, and according to Pascoe et al. (2014), model-derived economic target reference points require robust biological models as well as appropriate economic information, both of which are often unavailable. These last authors demonstrated that there is a relationship between economic (maximum economic yield) and biological (maximum sustainable yield) reference points, and that this depends primarily on the cost.

Cadima (2003) gives extensive information about biological reference points and regulation measures.

4.3 D3C3

Environmental impact

The selective fishing process could generate a deterioration of fish populations, because the active selection of greater individuals causes a significant loss of the reproductive potential of target species (Barneche et al., 2018), the genetic load associated with these individuals of great size (Swain et al., 2007; Pandolfi, 2009; Walsh et al., 2006), and also an increase in vulnerability and predation mortality (Audzijonyte et al., 2013; Alós et al., 2014). On of the negative effects of fishing selection toward individuals of larger size the so called recruitment overfishing, that occurs when the mature adult population is depleted to a level where it no longer has the reproductive capacity to replenish itself—there are not enough adults to produce offspring (Pauly, 1983; Guerra-Sierra & Sánchez-Lizaso, 1998; FAO, 2009). However, Guerra-Sierra & Sánchez-Lizaso (1998) also refer to genetic overfishing that is when the genetic characteristics of the population change due to a selection addressed to individuas of an specific characteristic, as could be to have a large size. Fishing would selectively eliminate larger individuals, so that, over time, undesirable genetic traits would be selected from the human point of view, such as those that slow down growth or that cause reproduction to occur at lower sizes and, therefore, that would negatively affect fecundity. This effect of fishing selectivity of large individual on a significant reduction at the first maturity size has been also described by Olsen et al. (2004), de Roos et al. (2006), and FAO (2009), among others. In this sense, in waters of Gran Canaria, Jiménez-Alvarado et al. (2019) has demonstrated a significant decrease, of about -1.22±0.19 cm per year, in the mean total length of dusky grouper (*Epinephelus marginatus*), a bentho-demersal top-predator, and a concurrent change in the composition of catches by recreational anglers, with a progressive major presence of omnivorous fish species. But, and a despite that fish species in the Canary Islands are in chronic overfishing scenario from de decade of 1970 (García-Cabrera, 1970; González, 2008; Castro et al., 2019), in 2015, the Fishery Ministry of Spain has significantly reduced the length of first capture of *Pagrus pagrus* in waters of the Canary Islands, because recent reproductive studies have determined that the length of maturity of this species have decrease (Pajuelo & Lorenzo, 1996) in relation to that estimated in 1986, from 33 to 28 cm of total length. And this is the reason why fishery management should never be oriented to reduce the legal size of first capture without to have into mind the history of the fishery, because this decrease in the mean size of first reproduction, would only further aggravate the problem of overexploitation.

On the other hand, Audzijonyte et al (2013) explored how a slow (less than 0.1% per year) decrease in the length of five harvested species could affect species interactions, biomasses and yields, and thy found that even small decreases in fish sizes are amplified by positive feedback loops in the ecosystem and can lead to major changes in natural mortality. They observed that a total of 4 per cent decrease in length-at-age over 50 years resulted in 50 per cent increase in predation mortality, although the magnitude and direction in predation mortality changes differed among species, but 50 years of gradual decrease in body size resulted in 1–35% decrease in biomasses and catches of all shrinking species. In this sense, in the case of Canary Islands populations of several target fish species are very rarefied, as is the case of the dusky grouper, island grouper (*Myteroperca fusca*), and barred hogfish (*Bodianus scrofa*) (Aguilera-Klink et al., 1993; Riera et al., 2014; Zeller et al., 2009), but that also may have been the basis for deeper changes in these ecosystems, facilitating the succession of species and the population explosion of other opportunistic ones (Ortega et al., 2009). Therefore, fisheries management practices that ignore contemporary life-history changes are likely to overestimate long-term yields and can lead to overfishing.

Environmental impact spatial extent

There is no information that could permits to asses the spatial extent of the impact of fishing on the age and size distribution of individuals in the populations of commercially-exploited species. However, is expected that it could be variable according to the level of fishing pressure of each island. In this way, and according to commercial capture data, the fish species targeted off the Island of El Hierro seem to support a lower fishing pressure that in the islands of Gran Canaria and Tenerife, because species like the barred hogfish and island groupers are more frequent in catches, and individuals caught seem to be larger length than reported in other islands (there are no length data available to sure this). In relation to this, Castro-Hernández et al. (2018), when assessing the impact of spearfishing in different zones of the Canary archipelago, observed that in shallow waters (less than 20 m depth) the mean length of species censed where frequently smaller that the length of first maturity reported for those species.

The high proportion of old/large individuals on waters of islands like El Hierro could also be associate with the presence of a Marine Protected Area in those Islands, that reduce the spatial impact of fishing and preserve part of the spawning population and older fishes not available to fishing.

Maritime Activity pressure solution

There are two main actions that should be carry out simultaneously to preserve the age and size structure of commercially-exploited species populations, as a healthy indicator, and keeping a high proportion of old/large individuals and genetic diversity of each species. The fist one is to regulate the length at fist capture (first harvest) for all species, including those that are not target species for the fishery actually, but could be potentially in a short time interval, for whole fishing ground. Froese & Kesner-Reyes (2002), after analyse the time series of catch published by ICES and FAO, observed that the length-frequency of commercial landings showed that in most target species the mean length was below length at first maturity, and they proposed as alternative management regime to avoid overfished, collapse or closed fisheries, to allow fish to spawn at least one before being caught. In this way, the length at first capture should be larger than the length at fist maturity of each species, and be adequate to the biological characteristics of each stock. So, it is necessary first to define and geographic delimit the stocks distribution. It is also important to have into consideration those species that suffer sex changes with growth, in the way to avoid the overfishing of the sex of larger length. Curiously, in the Canary Islands, the 70% of the 20 demersal and small pelagic fish species (tuna are excluded) have a lower length at first capture than their respective length at first maturity.

According to FAO (2009), mesh size restrictions can be a useful measure to avoid capturing individuals of target species in the immature stages, but they have limitations in multi-species fisheries and to catch older individuals. The main problem occurs when organisms of different shapes and sizes occur on the same fishing ground, and immature individuals of a co-occurring larger species are captured together. Fishing mortality can be modified by restricting fishing activity to certain times or seasons, or by restricting fishing in particular fishing grounds, particularly during spawning seasons and in recruitment areas.

On the other hand, the second strategy to limit the impact of fishing on the size structure of target or not target population is generally reduces both the direct and indirect effects of fishing on the ecosystem. Closures areas may be used to protect critical habitats where fishing activity would otherwise cause damage to the physical structures supporting the ecosystem (FAO, 2009). And one form of closure is that of marine protected areas (MPA)s. MPAs, when are well designated, can produce considerable benefits for fisheries and marine biota (Rising & Heal, 2014). In this way, Taylor and McIlwain (2010) demonstrated that effective implementation of MPAs allows a larger and older population to accrue, thus yielding considerable reproductive benefits. Moreover, MPAs also allow that a proportion of the stock to remain free of the genetic selective effects of fishing, and may act as refuges for the accumulation of spawning biomass from which replenishment of surrounding fished areas can occur (FAO, 2009).

Impact mitigation measures

To achieve the rebuilding of biomass of prey and predator species, and minimize the impact of fishing on commercial species, it is necessary (i) to keep fishing mortality lower than the natural rate of mortality, (ii) to maintain population sizes above half of natural abundance, and (iii) to adjust the size at first capture equals the the length where the biomass of an unexploited cohort would be maximum, would not only

(Froeser et al, 2016). Biomass, density, species richness, and size of organisms generally increase over time when protected by no-take marine reserves (Starr et al., 2015; Fidler et al., 2018). The magnitude and timing of changes vary greatly and depend upon the taxonomic groups protected, size and type of reserve, oceanographic/climatic regime, and time since the reserve was implemented. Nevertheless, when MPA has only partial restrictions to fishing, this management tool not seems to be successful in achieving benefits (in biomass and in mean length) for sedentary and relatively long-lived target species (Buselic et al., 2015). On the contrary, community-managed MPAs can play an important role for local conservation of high-value fish, and benthic ecosystems (Chirico et al., 2017).

On the other hand, Mascia et al. (2010) argued that food security generally remained stable or increased in older and smaller MPAs, and shape the social well-being and political power of fishing communities. The socioeconomic impacts of MPAs are generally perceived as negative for industrial fishing and positive for artisanal fishing (Pascual et al., 2016).

Monitoring methods

Unfortunately, there are not regular official programmes of control and recording of the length distribution of the target species caught in the artisanal and recreational fisheries in the different islands of the Macaronesian archipelagos. The only data of this nature that are mentioned in the literature are those obtained by different researchers during the preparation of specific biological studies of several target species, recorded on very short temporal intervals (lower than 2 years), but that raw data of length distribution used by them are not available (e.g. Lorenzo & Pajuelo, 1995, 1996; Morales-Nin & Sena-Carcalho, 1996; Lorenzo et al., 2002; García-Santamaría et al., 2012, between many others). Only the tuna fisheries seem to have routine programme of length data recording under the supervision of ICCAT (Delgado de Molina & Santana, 1986; Delgado de Molina et al., 2012; Ariz et al., 1994; Gouveia et al., 2017, between others).

In this context, it is necessary to implement a programme of species length distribution data recording of the different artisanal and recreational fisheries, by fishing gear and geographic location of fishing grounds, for all the target and non-target species. Because of the high spatial and temporal variability of fish recruitment patterns, long-term monitoring is needed to identify positive or negative responses to the diverse fishing management strategies that could be implemented along time, including MPAs or any other measure addressed to the protection of species or the different set of habitats.

4.4 References

Afonso, P., Fontes, J., Morato, T., Holland, K.N., Santos, R.S. 2008. Reproduction and spawning habitat of white trevally, Pseudocaranx dentex, in the Azores, central north Atlantic. Scientia Marina, 72(2), 373-381.

Aguilera Klink, F., A. Brito-Hernández, C. Castilla-Gutiérrez, A. Díaz-Hernández, J.M. Fernández-Palacios, A. Rodríguez-Rodríguez, F. Sabaté-Bel y J. Sánchez-García. 1993. Canarias. Economía, ecología y medio Ambiente. San Cristóbal de La Laguna: Francisco Lemus Editor, 361 pp.

Alós, J., Palmer, M., Catalan, I.A., Alonso-Fernández, A., Basterretxea, G., Jordi, A., Buttay, L., Morales-Nin, B. & Arlinghaus, R. (2014) Selective exploitation of spatially structured coastal fish populations by recreational anglers may lead to evolutionary downsizing of adults. *Marine Ecology -Progress Series* 503,219-233.

Alverson, D.L., Freeberg, M.H., Murawski, S.A., Pope, J.G. 1996. A global assessment of fisheries bycatch and discards. FAO Fish. Tech. Pap., 339, FAO, Rome

Alves, F., Canning-Clode, J., Ribeiro, C., Gestoso, I., Kaufman, M. 2018. Local benthic assemblages in shallow rocky reefs find refuge in a marine protected area at Madeira Island. Journal of Coastal Conservation, 1-11, DOI:10.1007/s11852-018-0669-y

Ariz, J. Delgado de Molina, A., Pallares, P., Santana, J.C., Pereiro, J.A. 1994. Composición por edades del rabil capturado por embarcaciones de cebo vivo en el área de las Islas Canarias. Col. Vol. Sci. Pap. ICCAT, 42(2):138-143

Arkhipov, A.G., Mamedov, A.A. 2008. Ichthyoplankton of Azores seamount. Journal of Ichthyology, 48(3):259-267.

Audzijonyte, A., Kuparinen, A., Gorton, R. & Fulton, E.A. (2013) Ecological consequences of bdy size decline in harvested fish species: positive feedback loops in trophic interactions amplify human impact. *Biology Letters* 9(2),2021103. doi: 10.1098/rsbl.2012.1103.

Baeta, F., Costa, J., Cabral, H. 2009. Changes in the trophic level of Portuguese landing and fish market price variation in the last decades. Fisheries Research., 97(3):216-222.

Barneche, D.R., Robertson, D.R., White, C.R. & Mashall, D.J. (2018) Fish reproductive-energy output increases disproportionately with body size. *Science* 360(6389), 642-645.

Bas, C,. J.J. Castro, V. Hernández-García, J.M. Lorenzo, T. Moreno, J.G. Pajuelo y A.G. Ramos. 1995. La pesca en Canarias y área de influencia. Ediciones del Cabildo Insular de Gran Canaria. Madrid. 331 pp.

Bécognée, P., Almeida, C., Barrera, A., Hernández-Guerra, A., Hernández-León, S. 2006. Annual cycle of clupeiform larva around Gran Canaria Island, Canary Islands. Fisheries Oceanography, 15(4):293-300.

Birkeland, C., P.K. Dayton. 2005. The importance in fishery management of leaving the big ones. Trends in Ecology & Evolution, 20:356-358.

Bordalo-Machado, P.; Figuereido, I.; Reis, S.; Sena-Carvalho, D.; Blasdsale, T.; Newton, A. & Gordo, L. (2001). New findings on the reproductive biology of the black scabbardfish (*Aphanopus carbo*) in the NE Atlantic. NAFO SCR doc.01/89.Serial nº N4477: 12 pp.

Bordes, F., A. Barrera, J. Carrillo, J. Gómez, F. Pérez, S. Álvarez y A. Ojeda. 1995. Cartografía y evaluación de los recursos pesqueros en la plataforma y talud de Lanzarote (Islas Canarias). Informe Técnico. Viceconsejería de Agricultura, Pesca y Alimentación. 38 pp.

Bordes, F., A. Barrera, J. Carrillo, R. Castillo, J.J. Castro, J. Gómez, K. Hansen, V. Hernández, T. Moreno, F. Pérez y F. Ublein. 1997. Evaluación acústica de los recursos epipelágicos y estudio de la capa de reflexión profunda en Lanzarote, Fuerteventura y Gran Canaria (Islas Canarias). Informe Técnico. Viceconsejería de Agricultura, Pesca y Alimentación. 63 pp

Bordes, F., A. Barrera, R. Castillo, J. Gómez, A. Ojeda, F. Pérez, S. Álvarez, J.A. Melluso y S. Ramos. 1993. Cartografía y evaluación de los recursos pesqueros de la plataforma y talud de Gran Canaria (Islas Canarias). Informe Técnico. Viceconsejería de Agricultura, Pesca y Alimentación. 32 pp.

Bordes, F., A. Barrera, R. Ramírez, J.A. Gómez, J.I. Santana, S. Hernández-León y J. Arístegui. 1987. Prospección hidroacústica para la evaluación del stock de peces pelágicos costeros de Canarias. Consejería de Agricultura, Ganadería y Pesca, Gobierno de Canarias. Centro de Tecnología Pesquera, Cabildo Insular de Gran Canaria.

Bordes, F., C. Almeida, A. Barrera, J. Carrillo, R. Castillo, J. Coca-Sáez, J.A. Gómez, K.A. Hansen, F. Pérez, A.G. Ramos y F. Uiblein. 1998. Prospección acústica y pesquera de los recursos pelágicos en Lanzarote, Fuerteventura y Gran Canaria (Islas Canarias). Resultados de la Campaña “Bocaina 1197”. Informe técnico. Viceconsejería de Pesca del Gobierno de Canarias. 73 pp.

Brito, A. I.J. lozano, J.M. Falcón, F. M. Rodríguez y J. Mena, 1996. Análisis biogeográfico de la ictiofauna de las Islas Canarias. En: Llinas, O., J.A. González y M.J. rueda. (eds.) 241-270. *Oceanografia y Rercursos Marinos en el Atlántico Centro-oriental*. Instituto Canario de Ciencias Marinas.

Buselic, I., Staglicic, N., Lloret, J., & Matic-Skoko, S. 2015. Can restrictions in Mediterranean artisanal fisheries be beneficial for target species? Acta Ichthyologica et Piscatoria, 45(1):31-38.

Caballero-Alfonso, A.M., U. Ganzedo, A. Trujillo-Santana, J. Polanco, A. Santana del Pino, G. Ibarra-Berastegi y J.J. Castro-Hernández. 2010. The role of climatic variability on the short-term fluctuations of octopus captures at the Canary Islands. Fisheries Research, 102:258-265.

Caddy, J.F., Mahon, R. 1995. Reference points for fisheries management. FAO Fish. Tech. Pap., 347, 83 pp.

Cadima, E.L. 2003. Fish stock assessment manual. FAO Fisheries Technical Paper, 393.

Camiñas, J., 2000. Estatus y conservación de las tortugas marinas en España, pp. 347- 382 in: Pleguezuelos, J.M., Márquez, R. & Lizana, M. (eds.). Atlas y libro rojo de los anfibios y reptiles de España. Madrid: Ministerio de Medio Ambiente, 2ª edición, 587 pp.

Carracedo, J.C. 2018. Geología de Canarias 1. Rueda Ediciones. 400 pp.

Castro, J.J., D. Jiménez-Alvarado, A. Guerra-Marrero y A. Sarmiento-Lezcano. 2018. Estimación del impacto de la pesca submarina de recreo en las zonas acotadas para dicha actividad en el Archipiélago Canario. Universidad de Las Palmas de Gran Canaria (Instituto Universitario EcoAqua) - Dirección General de Pesca. Gobierno de Canarias. Informe Técnico, 133 pp.

Castro, J.J., Divovich, E. Delgado de Molina Acevedo, A. & Barrera-Luján, A. 2015. *Over-looked and under-reported: A catch reconstruction of marine fisheries in the Canary Islands, Spain, 1950–2010*. Fisheries Centre Working Paper# 2015-26. University of British Columbia, Vancouver, Canada, 1-36.

Castro, J.J., E. Divovich, A. Delgado de Molina, A. Barrera-Luján. 2016. Spain (Canary Islands). P. 395. In: D. Pauly and D. Zeller (eds.). Global Atlas of Marine Fisheries: A critical appraisal of catches and ecosystem impacts. Island Press, Washington, D.C.

Castro, J.J., E. Divovich, A. Delgado de Molina-Acevedo, A. Barrera-Luján, R. Riera. 2019. Reconstruction of marine small-scale fisheries captures in the Canary Islands (NE Atlantic Ocean) from 1950 to 2010. Scientia Marina, 83(1). https://doi.org/10.3989/scimar.04837.18A

Castro-Hernández, J.J., D. Jiménez-Alvarado, A. Guerra-Marrero y A. Sarmiento-Lezcano. 2018. Estimación del impacto de la pesca submarina de recreo en las zonas acotadas para dicha actividad en el Archipiélago Canario. Universidad de Las Palmas de Gran Canaria (Instituto Universitario EcoAqua) - Dirección General de Pesca. Gobierno de Canarias. Informe Técnico, 133 pp.

Cheung, W.W.L., T.J. Pitcher, D. Pauly. 2005. A fuzzy logic expert system to estimate intrinsic extinction vulnerabilities of marine fishes to fishing. Biological Conservation., 124(1):97-111.

Chirico, A.A.D., McClanahan, T.R., & Eklöf, J.S. 2017. Community -and goverment-managed marine protected areas increase fish size, biomass and potential value. PLoS One, 12(8). DOI: 10.1371/journal.pne.0182342.

Chuenpagdee R., L.E. Morgan, S.M. Maxwell, E.A. Norse, D. Pauly, 2003. Shifting gears: assessing collateral impacts of fishing methods in US waters. Frontiers in Ecology and the Environment 1: 517–524.

Couce-Montero, L., Christensen, V., Bilbao-Sieyro, A., Pérez-González, Y., Castro, J.J. 2019. Temporal and spatial predictions of impacts of alternative fishing policies for the Gran Canaria marine ecosystem. Journal of Fish Biology. DOI: 10.1111/jfb.14012.

Dayton, P.K. 1998. Reversal of the burden of proof in fisheries management, Science, 279:821-2.

De Lara, M., Doyen, L., Guilbaud, Th., Rochet, M. 2007. Is a management framework based on spawning-stock biomass indicators sustainable? Aviability approach. ICES Journal of Marine Science, 64, 761-767.

de Roos, A.M., Boukal, D.S. & Persson, L. (2006) Evolutionary regime shifts in age and size at maturation of exploited fish stocks. *Proceeding of the Royal Society of London B - Biological Sciences* 273(1596),1873-8*0.*

Delgado de Molina, A., Ariz, J., Delgado de Molina, R., Santana, J.C, y Pallarés, P. 2005. Análisis de los datos de marcado de patudo en las islas Canarias. *Col. Vol. Sci. Pap. ICCAT., 57*(1), 100-115.

Delgado de Molina, A., Delgado de Molina, R., Santana, J.C. y Ariz. J. 2012. Datos estadísticos de la pesquería de túnidos de las islas canarias durante el periodo 1975 a 2010. *Col. Vol. Sci. Pap. ICCAT, 68*(3), 1221-1230.

Delgado de Molina, A., Santana, J.C. 1986. Estimación de la edad y crecimiento del patudo (Thunnus obesus, Lowe, 1939) capturado en las Islas Canarias. Col. Vol. Sci. Pap. ICCAT, 25:130-137.

Estácio, S., Mendonça, A., Krug, H., Menezes, G.M., Pinho, M.R. . 2001. Aspects of the reproduction of six exploited demersal fish species in the Azores archipelago. Arquiptlago. Life and Marine Sciences. Supplement 2 (Part B): 83- 94. Ponta Delgada.

FAO (2009). Fisheries management. 2. The ecosystem approach to fisheries. 2.2 The human dimensions of the ecosystem approach to fisheries. *FAO Technical Guidelines for Responsible Fisheries*. No. 4, Suppl. 2, Add. 2. Rome, 88p.

Ferreira, R. L., Martins, H. M., Da Silva, A. A., Bolten, A. B., 2001. Impact of swordfish fisheries on sea turtles in the Azores. *Arquipelago Life and Marine Sciences,* 18: 75-79.

Fidler, R.Y., Carroll, J., Rynerson, K.W., Matthews, D.F., Turingan, R.G. 2018. Coral reed fishes exhibit beneficial phenotypes inside marine protected areas. PLoS One, 13(2). DOI: 10.1371/journal.pone.0193426.

Figuereido, I., Bordalo-Machado, P., Reis, S., Sena-Carvalho, D., Blasdale, T.; Newton, A. & Gordo, L. 2003. Observations on the reproductive cycle of the black scabbardfish (*Aphanopus carbo*) in the NE Atlantic. ICES Journal of Marine Science, 60: 774-779.

Frielander, A.M., Ballesteros, E., Clemente, S., Gonçalves, E., Estep, A., Rose, P., Sala, E. 2017. Contrasts in the marine ecosystem of two Macaronesian islands: A comparison between the remote Salvages Reserve and Madeira. PloS one, 12(11), DOI:10.1371/journal.pne.0187935

Frisch, A.J., A.J. Cole, J-P.A. Hobbs, J.R. Rizzari, K.P. Munkres. 2012. Effects of spearfishing on reef fish populations in a multi-use conservation Area. PLoSONE 7(12): e51938. doi:10.1371/journal.pone.0051938.

Froese, R., & Kesner-Reyes, K. 2002. Impact of fishing on abundance of marine species. ICES CM 2002/L:12

Froese, R., Demirel, N., Coro, G., Kleisner, K. M., & Winker, H. 2017. Estimating fisheries reference points from catch and resilience. Fish and Fisheries, 18(3), 506-526.

Froese, R., Winker, H., Gascuel, D., Sumaila, U.R., & Pauly, D. 2016. Minimizing the impact of fishing. Fish and Fisheries, 17(3):785-802.

Ganzedo-López, U. 2005. Efecto de las variaciones climáticas en la distribución espacio-temporal de *Thunnus thynnus thynnus* (Linnaeus, 1758) y *Thunnus alalunga* (Bonnaterre, 1788) en el Océano Atlántico. *Tesis Doctoral, Universidad de Las Palmas de Gran Canaria*

García, S. M.,, Kolding, J., Rice, J., Rochet, M.J., Zhou, S., Arimoto, T., Beyer, J.E., Borges, L., Bundy, A., Dunn, D., Fulton, E.A., Hall, M., Heino, M., Law, R., Makino, M., Rijnsdorp, A.D., Simard, F., Smith A.D.M. 2019. Reconsidering the consequences of selective fisheries. Science, 335, 1045-1047.

Garcia, S.M. (1994) Precautionary principle: its implications in capture fisheries management. Ocean and Coastal Management, 22: 99–125.

García, S.M., Kolding, J., Rice, J., Rochet, M.J., Zhou, S., Arimoto, T., Beyer, J.E., Borges, L., Bundy, A., Dunn, D., Fulton, E.A., Hall, M., Heino, M., Law, R., Makino, M., Rijnsdorp, A.D., Simard, F., Smith, A.D.M. 2012. Recosindering the consequences of selective fisheries. Science, 335:1045-1047

García-Cabrera, C. 1970. *La pesca en Canarias y Banco Sahariano*. Consejo Económico Sindical Interprovincial de Canarias. 176 pp.

García-Cabrera, C., 1970. La pesca en Canarias y Banco Sahariano. Consejo Económico Sindical Interprovincial de Canarias. 176 pp.

García-Santamaría, M.T., Balguerías, E., González, J.F., Pascual, P., Díaz, J.A., González, E., Suarez, M., Fernández, A., González, M.A., 2001. A pilot study for estimation of data from local fisheries in Tenerife (Canary Islands) (Study Contract 00/022). Final Report. European Commision.

Gardner, T.A., Côté, I.M., Gill, J.A., Grant, A., Watkinson, A.R. 2005. Hurricanes and Carinnean coral reefs: Impacts recovery patterns, and role in long-term decline. Ecology, 86(1):174-184.

González, J.A. (2008) *Memoria científico-técnica final sobre el Estado de los Recursos Pesqueros de Canarias (REPESCAN)*. Instituto Canario de Ciencias Marinas, Agencia Canaria de Investigación, Innovación y Sociedad de la Información, Gobierno de Canarias, Las Palmas de Gran Canaria, 210 pp.

González, J.A., J.G. Pajuelo, J.M. Lorenzo, J.I. Santana, V.M. Tuset, S. Jiménez, C. Perales-Raya, G. González-Lorenzo, P. Martín-Sosa & I.J. Lozano. 2012. Talla Mínima de Captura de peces, crustáceos y moluscos de interés pesquero en Canarias. Una propuesta científica para su conservación. González, J.A., J.G.

Pajuelo & J.M. Lorenzo (eds.), Viceconsejería de Pesca del Gobierno de Canarias. Las Palmas de Gran Canaria: 252 p.

González, J.A., Rico, V., Lorenzo, J.M., Reis, S., Pajuelo, J.G., Afonso-Dias, M., Mendoça, A., Krug, H.M., Pinho, M.R. 2003. Sex and reproduction of the alfonsino *Beryx splendens* (Pisces, Berycidae) from the Macaronesian archipelagos. Journal of Applied Ichthyology, 19(2): doi.org/10.1046/j.1439-0426.2003.00358.x

Gouveia, L., Amorim, A., Alves, A, Hermida, M. 2017. Updated fishery statistics for bigeye, skipjack and albcore tunas from Madeira Archipelago. Collect. Vol. Sci. Pap. ICCAT, 73(4):1547-1560.

Guerra-Sierra, A., & Sánchez-Lizaso, J.L. 1998. Fundamentos de explotación de recursos vivos marinos. Editorial Acribia. Zaragoza, 249 pp.

Hale, R., Pires, R., Santos, P., Karamanlidis, A.A. 2011. Mediterranean monk seal (*Monachus monachus*): Fishery interactions in the Achipelago of Madeira. Aquatic Mammals, 37(3), 298-304.

Hermida, M., Delgado, J. 2016. High trophic level and low diversity: Would Madeira benefit from fishing down? Marine Policy, 73, 130-137.

Hernández, J.C., Clemente, S., Tuya, F., Pérez-Ruzafa, A., Sangil, C., Moro-Abad, L., Bacallado-Arénega, J.J. 2013. Echinoderms of Canary Islands, Spain. In: Alvarado, J.J., Solís-Marín, F.A. (eds.), Echinoderms Research and Diversity in Latin Amerina. 471-510 pp. Springer-Verlag.

Hernández-García, V., Hernández-López. J.L., Castro-Hernández, J.J. 2002. On the reproduction of *Octopus vulgaris* off the coast of the Canary Islands. Fisheries Research, 57(2), 197-203.

Herrera-Perdomo, J. 2017. Caracterización del cebo vivo utilizado e impacto sobre el mismo en la pesquería de túnidos en Canarias. Mem. Tesis Máster en Gestión Sostenible de Recursos Pesqueros. Universidad de Las Palmas de Gran Canaria.

Huston, M.A. 1985. Patterns of species diversity on Coral Reefs. Annual Review of Ecology and Systematics, 16:149-177.

ICCAT, 2012. Report for Biennial Period, 2010-2011, Part II, Vol. 2. Standing Committee on Research and Statistics (SCRS). Madrid, Spain.

ICCAT. 2017. Report of the 2017 ICCAT Shortfin Mako Assessment meeting (Madrid, Spain 12-16 June 2017). SMA Assessment Meeting - Madrid 2017. www.iccat.int/Documents/SCRS/DetRep/SMA_SA_ENG.pdf

ICES. 2008. Report of the working group on the biology and assessment of deep-sea fisheries resources. ICES CM 2008/ACOM 14, 531 pp.

ICES. 2011. Report of the Working Group on Anchovy and Sardine (WGANSA), 24–28 June 2011, Vigo, Spain. ICES CM 2011/ACOM:16.

ICES. 2012. Report of the Working Group on the Biology and Assessment of Deep-Sea Fisheries Resources (WGDEEP), 29 March–5 April. 2012, ICES Headquarters, Copenhagen. ICES CM 2012/ACOM:17.

Jayasinghe, R.P.P.K., Amarasinghe, U.S., Newton, A. 2017. Evaluation of status of commercial fish stocks in European marine subareas using mean trophic levels of fish landing and spawning stock biomass. Ocean & Coastal Management, 143:154-163.

Jiménez-Alvardo, D., Sarmiento-Lezcano, A., Guerra-Marrero, A., Tuya, F., Santana del Pino, A, Sealey, M.J., Castro, J.J. 2019. Historical photographs of captures of recreational fishers indicate overexploitation of nearshore resources at an oceanic island. Journal of Fish Biology, 2019; 1-8 DOI: 10.1111/jfb.13969

Johannes, R.E. 1978. 'Traditional marine conservation methods in Oceania and their demise', Annual Review of Ecology and Systematics, 9: 349-64.

Le Quesne, W.J.F., Jennings, S. 2012. Predicting species vulnerability with minima data to support rapid risk assessment of fishing impacts on biodivesity. J. Appl. Ecol., 49(1):20-28.

Lewison, R., Freeman, S.A., Crowder, L. B. 2004. Quantifying the effects of fisheries on threatened species: the impact of pelagic longlines on loggerhead and leatherback sea turtles. *Ecological Letters*, 7:221-231

Lorance, P., H. Dupouy, 2001. CPUE abundance indices of the main target species of the French deep-water fishery in ICES Sub-areas V–VII. Fisheries Research, 51: 137-149.

Lorenzo, J. M., Pajuelo, J.G., Méndez-Villamil, M., Coca, J. y Ramos, A.G., 2002. Age, growth, reproduction and mortality of the striped seabream, *Lithognathus mormyrus* (Pisces, Sparidae), off the Canary Islands (Central-east Atlantic). J. Appl. Ichthyol., 18: 204–209.

Lorenzo, J.M. y Pajuelo, J.G., 1995. Biological parameters of the roudi escolar *Promethichthys prometheus* (Pisces: Gempylidae) off the Canary Islands. Fish. Res. 24: 65-71.

Lorenzo, J.M. y Pajuelo, J.G., 1996. Growth and reproductive biology of chub mackerel *Scomber japonicus* off the Canary Islands. S. Afr. J. mar. Sci., 17: 275-280.

Lorenzo, J.M., J.G. Pajuelo, M. Méndez-Villamil, J. Coca & A.G. Ramos. 2002. Age, growth, reproduction and mortality of the striped seabream, *Lithognathus mormyrus* (Pisces, Sparidae), off the Canary Islands (Central-east Atlantic). *J. Appl. Ichthyol*., 18: 204-209.

Mace, P.M. 1994. Relationships between common biological reference points used as thresholds and targets of fisheries management strategies. Can. J. Fish. Aquat. Sci., 51: 110–122.

Machete, M., Morato, T., Menezes, G., 2011. Experimental black scabbardfish (*Aphanopus carbo*) fishery in the Azores, NE Atlantic. ICES Journal of Marine Science 68(2): 302-308.

Mascia, M., Claus, C.A., & Naido, R. 2011. Impacts of marine protected areas on fishing communities. Conservation Bioloogy, 24(5):1424-1429.

Méndez-Villamil, M., J.M. Lorenzo, J.G. Pajuelo, A.G. Ramos & J. Coca. 2002. Aspects of the life history of the salema, *Sarpa salpa* (Pisces, Sparidae), off the Canarian Archipelago (central-east Atlantic). *Environ. Biol. Fish*., 63: 183-192.

Morales-Nin, B., Sean-Carvalho, D. 1996. Age and growth of the black scabbard fish (*Aphanopus carbo*) off Madeira. Fisheries Research, 25(3):239-251.

Morato, T. 2012. Description of environmental issues, fish stocks and fisheries in the EEZs around the Azores and Madeira. Resquest for Services Commitment no. SI2.615631. Commission Decision 2005/626/EC

Morato, T., Lemey, E., Menezes, G., Pham, C.K., Brito, J., Soszynski, A., Pitcher, T., Heymans, J.J. 2016. Food-web and ecosystem structure of the open-ocean and deep-sea environments of the Azores, NE Altlantic. Frontiers in Marine Sciece. https://doi.org/10.3389/fmars.2016.00245.

Moyano, M., Hernández-León, S. 2009. Temporal and along-shelf distribution of the larval fish assemblage at Gran Canaria, Canary Islands. Scientia Marina, 73(S1). DOI: 10.3989/scimar.2009.73s1085.

Moyano, M., Rodríguez, J.M. Hernández-León, S. 2009. Larval fish abundance and distribution during the bloom off Gran Canaria Island, Canary Islands. Fisheries Oceanography. Fish. Oceanogr., 18(1):51-61.

OAG. 2018. Estado de conservación de la tortuga boba (*Caretta caretta*) en las islas Canarias, 2012-2017. Santa Cruz de Tenerife: Observatorio Ambiental Granadilla, 31 páginas. Informe OAG_PSTB/2018.1 [unpublished]

Ojamaa, P. 2015. Fisheries in Azores. Directorate-General for Internal Policies of the Union. Policy Department B. Structural and Cohesion Policies. IP/B/PECH/INT/2015-02. PE 540.355.

Olsen, E.M., Heino, M., Lilly, G.R., Morgan, M.J., Brattey, J., Ernande, B. & Dieckmann, U. (2004) Maturation trends indicative of rapid evolution preceded the collapse of northern cod. *Nature* 428(6986),932-935.

Pajuelo J.G. & Lorenzo, J.M. (1996) Life history of the red porgy *Pagrus pagrus* (Teleostei: Sparidae) off the Canary Islands, central east Atlantic. *Fishery Research* 28,163- 177.

Pajuelo, J.G. & J.M. Lorenzo. 1995. Biological parameters reflecting the current state of the exploited pink dentex *Dentex gibbosus* (Pisces: Sparidae) population off the Canary Islands. *S. Afr. J. mar. Sci*., 16: 311-319.

Pajuelo, J.G. & J.M. Lorenzo. 1996. Life history of the red porgy *Pagrus pagrus* (Teleostei: Sparidae) off the Canary Islands, central east Atlantic. *Fish. Res.*, 28: 163-177.

Pajuelo, J.G. & J.M. Lorenzo. 1998. Population biology of the common pandora *Pagellus erythrinus* (Pisces: Sparidae) off the Canary Islands. *Fish. Res.*, 36: 75-86.

Pajuelo, J.G. & J.M. Lorenzo. 1999. Life history of black seabream, *Spondyliosoma cantharus*, off the Canary Islands, Central-east Atlantic. *Environ. Biol. Fish.*, 54: 325-336.

Pajuelo, J.G. & J.M. Lorenzo. 2000. Reproduction, age, growth and mortality of axillary seabream, *Pagellus acarne* (Sparidae), from the Canarian archipelago. *J. Appl. Ichthyol*., 16: 41-47.

Pajuelo, J.G. & J.M. Lorenzo. 2000. Reproduction, age, growth and mortality of axillary seabream, *Pagellus acarne* (Sparidae), from the Canarian archipelago. *J. Appl. Ichthyol*., 16: 41-47.

Pajuelo, J.G. & J.M. Lorenzo. 2004. Basic characteristics of the population dynamic and state of exploitation of Moroccan white seabream *Diplodus sargus cadenati* (Sparidae) in the Canarian archipelago. *J. Appl. Ichthyol*., 20: 15-21.

Pajuelo, J.G., J. Socorro, J.A. González, J.M. Lorenzo, J.A. Pérez-Peñalvo, I. Martínez & C.M. Hernández-Cruz. 2006. Life history of the red-banded seabream *Pagrus auriga* (Sparidae) from the coasts of the Canarian archipelago. *J. Appl. Ichthyol.*, 22: 430-436.

Pajuelo, J.G., J.M. Lorenzo, A.G. Ramos & M. Méndez-Villamil. 1997. Biology of the red mullet *Mullus surmuletus* (Mullidae) off the Canary Islands, Central-east Atlantic. *S. Afr. J. mar. Sci*., 18: 265-272.

Pajuelo, J.G., J.M. Lorenzo, R. Domínguez, A.G. Ramos & M. Gregoire. 2003. On the population ecology of the zebra seabream *Diplodus cervinus cervinus* (Lowe

1838) from the coasts of the Canarian archipelago, North West Africa. *Environ. Biol. Fish*., 67: 407-416.

Pajuelo, J.G., Triay-Portella, R., Delgado, J., Gois, A.R., Correia, S., Martins, A., González, J.A. 2018. Changes in catch and bycatch composition and in species diversity of a semi-floating shrimp-trap fishery in the eastern Atlantic island ecosystems with different degrees of human alteration. Scientia Marina, 82S1, 107-114.

Pajuelo, J.G.; Santana, J.I.; Lorenzo, J.M.; García-Mederos, A. & Tuset, V. (2008). Biological parameters of the bathyal fish black scabbardfish 33 (*Aphanopus carbo* Lowe, 1839) off the Canary Islands, Central-east Atlantic. Fisheries Research 92 (2008): 140-147.

Pandolfi, J.M. (2009) Evolutionary impacts of fishing: overfishing's "Darwinian debt". *F1000 Biology Reports* 2009, 1:43 (http://f1000researchdata.s3.amazonaws.com/f1000reports/files/9002/1/43/article.pdf).

Pascoe, S., Thebaud, C., Vieira, S. 2014. Estimating proxy economic target reference points in data-poor single species fisheries. Marine and Coastal Fisheries, Dynamics, Management, and Ecosystem Science, 6. doi.org/10.1080/19425120.2014.966215

Pascual, M., Rossetto, M., Ojeda, E., Milchakova, N., Giakoumi, S., Kark, S., Korolesova, D. & Melia, P. (2016). Socieconomic impacts of mariune prpotected areas in the Mediterranean and Black Seas. Ocean & Coastal Management, 133:1-10.

Pauly, D.l (1983). *Some simple methods for the assessment of tropical fish stocks*. *FAO* Fisheries technical paper 234. *ISBN 978-92-5-101333-5. Retrieved 2012-05-01*.

Perrotta, R.G., D.R. Hernandez, 2005. Stock assessment of the bluemouth (*Helicolenus dactylopterus*) in Azorean waters during the 1990-2002 period, applying a biomass dynamic model. Revista de Investigación y Desarrollo Pesquero 17:81-93.

Pham, C.K., Canha, A., Diogo, H., Pereira, J. G. Prieto, R., Morato, T. 2013. Total marine fishery catch for the Azores (1950–2010). – ICES Journal of Marine Science, 70: 564–577.

Piet, G.J., Rijnsdorp, A.D., Bergman, M.J., Van santbrink, J.W., Craeymeersch, J., Buijs, J. 2000. A quantitative evaluation of the impact of beam trawling on benthic fauna in the southern North Sea. ICES J. Mar Sci., 57(5):1332-1339.

Pogonoski, J.J., D.A. Pollard, J.R. Paxton. 2002. Conservation overview and action plan for Australian threatened and potentially threatened marine and estuarine fishes, Australian Museum, Sydney.

Poiner, I., Glaister, J., Burridge, J. Pitcher, T., Wassenberg, T., Gribble, N., Hill, B., Blaber, S., Brewer, D., & Ellis, N. 1998. Environmental Effects of Prawn Trawling on the Far Northern Sections of the Great Barrier Reef. Final report the Great Barrier Reef Marine Park Authority and the Fisheries Research and Development Corporation. CSIRO Publication Vols. 1 and 2, Cleveland, 500 pp.

Pope, J.G., Macdonald, D.S., Daan, N, Reynbold, J.D. Jennings, S. 2000. Gauging the impact of fishing mortality on non-target species. ICES J. Mar. Sci., 57(3):689-696.

Rico, V., Santana, J.I. y González, J.A. (2002). Técnicas de pesca artesanal en la isla de Gran Canaria. *Monogr. Inst. Canario Cienc. Mar.*, *3*, 1-318.

Riera, R., Becerro, M. A., Stuart-Smith, R. D., Delgado, J. D., & Edgar, G. J. (2014). Out of sight, out of mind: Threats to the marine biodiversity of the Canary Islands (NE Atlantic Ocean). Marine Pollution Bulletin, 86 (1–2), 9–18.

Riera, R., Menci, C., Sanabria-Fernández, J.A., Becerro, M.A. 2016. Do recreational activities affect coastal biodiversity? Estuarine, Coastal and Shelf Science, 178, 129-136.

Rising, J., & Heal, G. (2014). Global benefits of Marine Protected Areas. National Bureau of Economic Research. Working Paper 19982. https://www.nber.org/papers/w19982.pdf

Rodríguez, J.M. 2000. Fish larvae from the Canary region in autumn. Scientia Marina, 64(1):79-85.

Rodríguez, J.M., Moyano, M. y Hernández-León, S., 2009. The ichthyoplankton assemblage of the Canaries–African Coastal Transition Zone: A review. Prog. Oceanogr., 83: 314–321.

Rodríguez, J.M., S. Hernández-León & E. D. Barton, 1999. Mesoscale distribution of fish larvae in relation to an upwelling filament off Northest Africa. Deep Sea Research Part-1, *Oceanographic Research Papers*, 46 (11): 1969-1984.

Rosenberg, A.A, Brault, S. 1991. Stock rebuilding strategies over different time scales. NAFO Sci. Coun. Studies, 16:171–181.

Sampaio, I., A. Braga-Henriques, C. Pham, O. Ocaña, V. de Matos, T. Morato, F.M. Porteiro. 2012. Cold-water corals landed by bottom longline fisheries in the Azores (north-eastern Atlantic). Journal of the Marine Biological Association of the United Kingdom.

Sangil, C., M. Sansón y J. Afonso-Carrillo. 2011. Spatial variation patterns of subtidal seaweed assemblages along a subtropical oceanic archipelago: Thermal gradient vs herbivore pressure. Estuarine, Coastal and Shelf Science, 94:322-333.

Scott, B., Marteinsdottir, G, Wright, P. 1999. Potential effects of maternal factors on spawning stock-recruitment relationships under varying fishing pressure. Can. J. Fish. Aquat. Sci., 56, 1882-1890.

Silva, H.M., Pinho, M.R., 2007. Exploitation, management and conservation: Small-scale fishing on seamounts. In Pitcher, T.J., Morato T., Paul J.B., Clark, M.R., Haggan, N., and Santos R.S., (eds). Seamounts: Ecology, Fisheries & Conservation. Blackwell Publishing, UK, pp. 333-399.

Sobrihno-Gonçalves, L., Isisdro, E. 2001. Fish larvae and zooplankton biomass around Faial Island (Azores archipelago). A preliminary study of species occurrence and relative abundance. Arquipélago. Life and Marine Sciences 18A: 35-52.

Solari, A.P., M.T.G. Santamaría, M.F. Borges, A.M.P. Santos, H. Mendes, E. Balguerías, J.A. Díaz-Cordero, J.J. Castro & C. Bas. 2010. On the dynamics of *Sardina pilchardus*: orbits of stability and environmental forcing. ICES J. Mar. Sci., 67: 1565-1573.

Starr, R.M., Wendt, D.E., Barner, C.L., Marks, C., Malone, D., Waltz, G., Schmidt, K.S., Chiu, J., Launer, A.L., Hall, N.C., & Yochum, N. 2015. Variation in responses of fish across multipel reserves within a network of Marine Protected Areas in Temperate Waters. PLoS One, 10(3). DOI:10.1371/journal.pone.0118502

Stige, L.C., Yaragina, N.A., Langangen, O., Bogstad, B., Stenseth, N.C., Ottersen, G. 2017. Effect of fish stock's demographic structure on offspring survival and senitivity to climate. Proc. Natl. Acad. Sci., USA, 114(6), 1347-1352.

Swain, D.P., Sinclair, A.F. & Hanson, J.M. (2007) Evolutionary response to size-selective mortality in an exploited fish population. *Proceeding of the Royal Society of London B - Biological Sciences* 274(1613),1015-22.

Taylor, B.M., & McIlwain, J.L. (2010). Beyond abundance and biomass: effects of marine protected areas on the demography of a highly exploited reef fish. Marine Ecology Progress Series, 411:243-258.

Tuya, F., Sánchez-Jerez, P., Haroun, R. 2005. Influence of fishing and functional group of algae on sea urchin control of algal communities in the Central Atlantic. Marine Ecology Progress Series, 287, 255-260.

Vallerani, M., Martí, C-P., Ojamaa, P. 2017. In-depth analysis- Research for PECH Committee-Fisheries in Madeira. European Parliament.

Vasconcelos, J., M. Afonso-Dias & G. Faria 2012. Atlantic chub mackerel (Scomber colias) spawning season, size and age at first maturity in Madeira waters. Arquipelago. Life and Marine Sciences 29: 43-51.

Vasconcelos, J., Sousa, R., Riera, R., Delgado, J., Graça, F., Serrano-Gordo, L. 2018. Stock assessment of the blue jack mackerel, Trachurus picturatus, in the North-eastern Atlantic. Fisheries Management and Ecology, 25(4):233-239.

Vasilakopoulos, P., O'Neill, F.G., Marshall, C.T. 2012. Differential impacts of exploitation rate and juvenile exploitation on NE Atlantic fish stock dynamics over the past half century. Fish. Res., 134-136:21-28.

Zeller, D., Cheung, W., Close, C., & Pauly, D. (2009). Trends in global marine fisheries - a critical view. In Fisheries, sustainability and development (pp. 87–107). Stockholm, Sweden: Royal Swedish Forestry.

5 Analysis >> QD1 & QD4 Ecosystems, + food webs (relating to Descriptor 1 & Descriptor 4)

QD1&QD4 Ecosystems, including food webs (relating to Descriptors 1 and 4)							
QD	**Criteria (element)**	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
QD1&QD4	The diversity (species composition and their relative abundance) of the trophic guild is not adversely affected due to anthropogenic pressures.	D4C1	yes	broader	yes	yes	yes
	The balance of total abundance between the trophic guilds is not adversely affected due to anthropogenic pressures.	D4C2	yes	broader	yes	yes	yes
	The size distribution of individuals across the trophic guild is not adversely affected due to anthropogenic pressures.	D4C3 — Secondary	yes	need further research	need further research	need further research	need further research
	Productivity of the trophic guild is not adversely affected due to anthropogenic pressures.	D4C4 — Secondary	yes	need further research	need further research	need further research	need further research

An ecosystem is defined as “any entity or natural unit that includes living and nonliving parts interacting to produce a stable system in which the exchange of materials between the living and nonliving parts follows circular paths in an ecological system or ecosystem. The ecosystem is the largest functional unit in ecology, since it includes both organisms (biotic communities) and abiotic environment, each influencing the properties of the other and both necessary for maintenance of life as we have it on the earth” (Odum, 1953). That is why this is one of the most complex descriptors; since it takes into account the structure of the ecosystem, the interactions between species and the energy transfer flows that take place.

Commission Decision (EU) 2017/848 on criteria and methodological standards to evaluate the good environmental status of marine waters establishes, in the case of Descriptor 4, the use of four criteria: diversity of the trophic guild, balance of total abundance between trophic levels, size distribution of individuals and productivity of the trophic guilds understood as production per unit of biomass of individuals.

To correctly evaluate this descriptor, the three types of control that can occur in the ecosystem must be considered: bottom-up, top-down and wasp-waist. In ecology, bottom-up control refers to when primary producers control the ecosystem structure; top-down refers to ecosystems in which the predators exert control over the prey and wasp-waist control occurs when one or a very few species have a substantial influence on the flow of energy through the mid-trophic levels (Pauly et al., 1998; Cury, 2000; Cury et al., 2003, 2005; Bakun, 2006; Hunt and McKinnell, 2006). This is the only way to ensure that all possible keystone species in the ecosystem be taken into account.

An important thing to keep in mind when evaluating Descriptor 4 is that this descriptor is related to all the descriptors contemplated by Commission Decision (EU) 2017/848. The relationships with Descriptor 1 (biodiversity) and Descriptor 3 (fishing) are the most obvious with a direct impact. However, Descriptor 2 (non-indigenous species) can modify the structure and functioning of the ecosystem, Descriptor 5 (eutrophication) can affect abundance of the primary producers, which would have an impact on the entire trophic chain, Descriptor 6 could cause changes in the biotic and abiotic structure of the ecosystem and its functions and the remaining descriptors would also be related though in an indirect way. Since the previous sections have already addressed the impacts of fishing on the other descriptors, the discussion of the following criteria may be redundant in some aspects.

5.1 D4C1, D4C2

The species composition of food webs and their relative abundance vary according to region and habitat, so a great deal of information is required to establish an assessment of the state of the ecosystem although in most instances it is not available. In some cases, when target or representative species cannot be evaluated due to lack of data, functional groups can be considered, but it must never be forgotten that any impact that involves changes in the abundance or distribution of species will directly or indirectly affect other species and therefore the trophic network. To select the groups or species the following criteria must be met: (a) include at least three trophic guilds; (b) two shall be non-fish trophic guilds; (c) at least one shall be a primary producer trophic guild; (d) preferably represent at least the top, middle and bottom of the food chain.

The impacts of fisheries on the ecosystem have been abundantly described and reviewed (Bianchi, 2000; Froese and Kesner-Reyes, 2002; Costello et al., 2010; Smith et al., 2011; Branch, 2015) and include severe reductions in upper trophic level predators, removal of keystone species, changes in prey species biodiversity and cascading responses through the food web, involving changes in ecosystem functioning (Pauly et al., 1998; Christensen et al., 2003; Essington et al., 2006; Morato et al., 2006).

Many of the fishing impacts on the marine food-webs in the three archipelagos have been discussed in the previous sections and could be summarized as follows: decrease in the abundance of target species and decrease in the sizes of the specimens, bycatch of species without commercial interest but important in the trophic network such as marine mammals, turtles, seabirds and sharks and, finally, fishing directed to high-trophic level species, which may compromise the stability of the ecosystems (Castro et al., 2015; Couce-Montero et al., 2015; Friedlander et al., 2017;

González, 2008; Morato et al., 2006, 2016; Pham et al., 2013; Sangil et al., 2013a, 2013b). In Canarias and Madeira high biomass of the sea urchin Diadema africanum have been reported, and this is an indicator of ecosystem unbalanced due to overfishing. Removal of top predators has been linked to hyperabundances of this sea urchin, with the subsequent creation of urchin barrens (Hernández et al., 2008a, 2008b; Friedlander et al., 2017).

There is not enough data about abundance and stocks status of commercially-exploited fish and shellfish species in the three archipelagos, except for tuna whose management is regulated by the International Commission for the Conservation of Atlantic Tunas (ICCAT). The REPESCAN report (González, 2008) put into evidence the lack of information regarding the stocks status of the main target species in the Canary Islands, but scientifics assume that most of the benthic and demersal species targeted by this small-scale fishery are currently overexploited and they also conclude that the current status of medium-sized coastal pelagics is unknown. Morato (2012) provided a detailed description about the fisheries and the stocks in Azores and Madeira grounds, concluding that the status of the main target species is also unknown.

Evaluating fisheries such as those included in this study is very complicated, because they are characterized by their complexity and the versatility of the fleets, with the additional problem of geographical dispersion of the extractive units. Despite this, there are indicators and models that can be used to assess the state of the populations and the impact of fishing when there are no real or reliable data available, although they must always be interpreted in the appropriate context and bearing in mind that other factors may be conditioning the results. Within the overall concept of ecosystem-based management, ecosystem models can describe the structure and function of an ecosystem and can be used to evaluate the effect of fishing. The most widely used approach to model ecosystems is Ecopath with Ecosim (EwE) due to the user friendly interface and on-going improvements to the software (Plagányi, 2007) and in the Macaronesian region there are some published models that have used this approach (Stobberup et al., 2004; Couce-Montero et al., 2015; Morato et al., 2016; Couce Montero et al., 2019; Couce-Montero et al., 2019). The largest part of the ecosystem biomass for Azores model (Morato et al., 2016) was composed by zooplankton and invertebrate groups and models developed in Canary Islands also showed higher biomass concentration at trophic levels II and III, mainly composed of planktonic and benthic organisms (Couce-Montero et al., 2015, 2019). In these ecosystems, fishing is focused on higher trophic levels in both regions and the total catches suggest an intensive rate of exploitation by the fleets. Excluding tuna fishing, deep-water bottom longline is the fishing technique with higher capture yields in Azores (Morato et al., 2016), which could compromise the structure of the ecosystem since bathypelagic fishes are considered keystone species within the ecosystem due to their important role as prey in the food web and also because deep-water species are highly vulnerable to overfishing and potentially have little resilience to over-exploitation. In Gran Canaria, the exploitation rates estimated by the model suggest that the groups of benthic sharks, serranids, wreckfish, sparids, parrotfishes, morays and leatherjacket fishes are overexploited (Couce-Montero et al., 2015), which is in line with the conclusions of the REPESCAN report (González, 2008); however only tunas and benthic sharks present fishing exploitation rates above the desirable values in Tenerife and La Gomera islands (Couce-Montero et al., 2019). The impacts related to the structure of the ecosystem and the abundance of species due to fishing mortality and bycatch have been extensively analyzed in Descriptor 1 and 3, so they will not be discussed in this section to avoid redundancies.

The relative abundance of the fish of each trophic group is useful for identifying overexploited areas. (Friedlander and DeMartini, 2002) noted that in non-overfished and well preserved areas, top predators constitute at least 50% of the total fish biomass; however fisheries tend to first remove these species that are characterized by being large, slow growing and long-lived predatory fish. The problem with this fishing strategy is that once these top predators are depleted, the fishing pressure will gradually move towards the smaller species (Pauly et al., 1998). This tendency has been observed worldwide and can be detected by the decline in the mean trophic level (TLc) of catches over time. Fisheries in Azores and Madeira are characterized by a higher mean trophic level (Baeta et al., 2009; Hermida and Delgado, 2016) and a similar trend is observed in Canary Islands (Couce-Montero et al., 2015, 2018); nevertheless, these results should be interpreted with caution, as they can give a distorted view of reality due to the peculiarities of these archipelagos. The Fishing in Balance Index (FiB) (Pauly et al., 2000) is often used in conjunction with the TLc to assess the effect of fishing on the ecosystem. FiB index for Azores and Madeira show a high variability with periods where a decreasing trend is observed reaching negative values, which are followed by periods with increasing trend (Baeta et al., 2009). This pattern suggests an expansion in fisheries, geographically and bathymetrically, beyond its traditional fishing area or ecosystem; but ,excluding large pelagic species (e.g. tunas), Azores and Madeira fleets are focused on deep-water species such as blackspot seabream (Pagellus bogaraveo), alfonsinos (Beryx splendens and B. decadactylus), and deepwater sharks in the Azores (Pham et al., 2013), and scabbardfishes (Aphanopus spp.) in Madeira (Shon et al., 2015). The TLc in Canary Islands shows a marked seasonality due to tuna fishing, which in conjunction with relatively short time series of catches, make it difficult to adequately describe the current state of the fishery as a whole, since the overfishing of benthic-demersal species can be masked (Couce-Montero et al., 2018).

Fisheries management carried out to date have a series of constraints that have hindered the correct management of resources (e.g., TACs established without enough information or fisheries management from a monospecific procedure). Proof of this is that stock assessment published by FAO (2018), found that 59.9% of the worldwide populations are exploited at a maximum level of sustainability, 33.1% are at biologically unsustainable levels and only 7% of the stocks are underexploited. On this background, it becomes evident the negative impact that fishing activities exert on marine ecosystems, highlighting the need to manage the fisheries through an ecosystem approach (García et al., 2003). Under the EU Regulation 1380/2013, the Common Fisheries Policy (CFP) recognized not only that an ecosystem-based approach to fisheries management needs to be implemented but also that recreational fisheries can have a significant impact on fish resources and Member States should ensure that they are conducted in a compatible manner with the objectives of the CFP.

All regional, national and European regulations as well as management strategies regarding the preservation of species included in Descriptors 1 and 3 are also applicable to Descriptor 4. The main measures adopted to ensure the ecosystem sustainability include limitations on the volume of catches and fishing effort, prohibition of fishing gears or authorization of their use in specific grounds, minimum sizes of some target species, creation of marine protected areas and regulation of recreational fishing.

The following are the main recommendations for monitoring the impact in abundance and species composition within the food webs from fishing activities:

- Conducting campaigns to determine the abundance of resources, or at least provide information on the status of apex predators and primary producers to consider the two main types of control that occur within the ecosystem.
- It is necessary to analyze the diets for target species if there is not enough information available, since they can be keystone species for the functioning and structure of the ecosystem.

- Creating time series that collect landings and effort data, including fishing grounds and gears employed and to improve the selectivity of the fishing gears to avoid discards as well as the capture of small specimens.
- Performing periodic controls to avoid illegal, unregulated and undeclared fishing and complement these controls with studies on the impact that recreational fishermen have on the ecosystem

An overall assessment of the relative abundance of the species and the ecosystem structure requires multiple indicators because marine ecosystems are generally complex and the available tools cannot capture all of this complexity (ICES, 2018), so one possible approach to monitor progress toward good environmental status is ecosystem modeling. Several methodologies have been developed to evaluate the marine resources through the analysis of food-webs and the choice of one or another will be conditioned by several factors, including the study area and available data. In general terms, ecosystem models can be classified in four groups (Plagányi, 2007):

1. **Whole ecosystem models**: attempt to take into account all trophic levels in the ecosystem
2. **Minimum Realistic Models (MRM)**: limited number of species most likely to have important interactions with a target species of interest.
3. **Dynamic System Models (Biophysical)**: represent both bottom-up (physical) and top-down (biological) forces interacting in an ecosystem.
4. **Extensions of single-species assessment models (ESAM)**: expand on current single-species assessment models taking into account a few additional inter-specific interactions.

5.2 D4C3

Fishing is a size-selective activity mainly focuses on large-bodied fish, resulting in declines in these target species size, density, and biomass, while smaller-sized organisms increase in the ecosystem (Jennings and Kaiser, 1998).

Decrease in adult body sizes can lead to life history changes of target species. Fish can mature at smaller sizes and younger ages (Sharpe and Hendry, 2009; Audzijonyte et al., 2013a) and this might have an effect on recruitment (Longhurst, 2002; Birkeland and Dayton, 2005) and increase natural mortality by investing more energy in reproduction (Jørgensen and Fiksen, 2010). Reductions in body size can also alter predator–prey dynamics through changes in prey vulnerability (Audzijonyte et al., 2013b; Jørgensen and Holt, 2013). These impacts are directly related with Descriptor 3, so only some examples have been cited and since maritime activity pressure solutions, impact mitigation measures and monitoring methods are the same as those included in D3C3, a detailed review of the bibliography can be found in the analysis of that section.

Large fish are usually considered as a key attribute of a healthy marine ecosystem (Greenstreet et al., 2011) and these species generally occupy the top trophic levels, so given that body size is strongly correlated with trophic level in the community (Jennings et al., 2001), it is reasonable to expect that size-based indicators (SBI) will reflect the impact of fishing on the ecosystem (Jennings and Kaiser, 1998; Jennings et al., 1999; Bianchi, 2000; Daan et al., 2005; Rochet and Rice, 2005; Shin et al., 2005; Travers et al., 2006). These indicators include, inter alia, mean length in a population, mean length in the community, mean maximum length in the community and the slope and intercept of size spectra. The latter are often used in fisheries to understand the structure of marine ecosystems and establish abundance baselines of marine communities and their responses to the potential effects of fishing (Bianchi, 2000; Benoît and Rochet, 2004; Shin et al., 2005; Travers et al., 2006; Blanchard et al., 2009; Law et al., 2009, 2012; Jacobsen et al., 2013). Several authors have hypothesized that exploitation should decrease the slope of a fish size spectrum, and reported decreasing trends of this slope in exploited systems, although this pattern is not consistent across systems (Bianchi, 2000). For example, deep-sea communities have slower vital rates and body size relationships vary with depth so the use of these indicators may not be too successful to detect changes through time (Mindel et al., 2018). In addition, fish recruitment can also affect size structure of the ecosystem; a good recruitment can result in an overall decrease in the average size while a poor recruitment may lead to an opposite effect (Greenstreet and Rogers, 2006).

Availability of species-size-abundance data collected through monitoring programs favors the use of these indicators (Rochet and Trenkel, 2003); however, the current monitoring programs in the archipelagos, such as IEO sampling and information network in the Canary Islands or the POPA and ARQDAÇO programs in the Azores, have important information gaps since, usually, only fishery target species are considered. And this, coupled with the fact that changes in size distributions can result for several reasons, including predator-prey relationships, life history traits, spatial scale and habitats, environmental effects, ecosystem maturity, food availability and environment-induced or genetic variability in life history, further complicates the problem to quantify the extent of the impact of fisheries.

The International Council for the Exploration of the Sea (ICES) was requested to evaluate methods to analyze the size distribution of the stocks, concluding that it is not possible to set biologically meaningful threshold or reference values of SBI using time-series methods. These reference values would allow comparison of current indicator values with past values, but they could not account for interactions within and between stocks that might occur when fishing consistently at F_{MSY}; therefore ICES only recommends the use of these indicators for surveillance purposes (ICES, 2016, 2017). For all these reasons, we believe that in order to evaluate the impact that fishing has on the sizes distribution within the ecosystem, it is necessary to combine monitoring methods described for Descriptors 1, 3 and those proposed for criteria D4C1 and D4C2 with campaigns to obtain data about age, growth and reproduction parameters.

5.3 D4C4

The performance of the species, as measured by their productivity (production per unit of biomass; P/B), summarizes the main predator-prey processes in the ecosystem that they inhabit. In order to correctly evaluate this criterion it would be necessary to know in detail the abundance of the different species or functional groups within the ecosystem as well as their production, and this is a difficult task due to lack of accurate information. However, approximate values of the P/B ratios can be estimated from indirect methods such as those mentioned below.

Under steady-state conditions, P/B for fishery target species or groups is equal to the total mortality Z (Allen, 1971), where Z is the sum of natural (M) and fishing (F) mortality rates; F can be calculated as the ratio between catches and biomass and M can be estimated using this empirical equation (Pauly, 1980):

$$LogM = -0.0066 - 0.279 \cdot LogL_{\infty} + 0.6543 \cdot LogK + 0.4634 \cdot LogT$$

L_{∞} represents the asymptotic length and K the curvature parameter of the Von Bertalanffy growth function and T is the mean water temperature in the study area, expressed in oC.

For macroinvertebrate species, P/B ratios can be estimated with Brey´s models (Brey, 2001, 2012; Brey et al., 2010) and ratio for primary producers can be calculated with information provided by satellite data (e.g. SeaWiFS)

For those species for which there is not enough information and are important within the ecosystem, trophic network models can be used to estimate this parameter, as it has been done in the ecosystem models of the Azores and the Canary Islands developed to date (Guénette and Morato, 2001; Morato et al., 2009, 2016; Couce-Montero et al., 2015, 2019).

As previously mentioned, fishing influences total mortality of species and interferes in predator-prey relationships, and hence in food consumption (Christensen et al., 2008), so it is clear that this activity has an impact on productivity of the trophic network; but to quantify the magnitude of this impact further research is needed. The monitoring methods for this criterion are the same as those detailed for the three criteria discussed above.

5.4 References

Allen, K. R. 1971. Relation between production and biomass. Journal of the Fisheries Board of Canada, 28: 1573–1581.

Audzijonyte, A., Kuparinen, A., and Fulton, E. A. 2013a. How fast is fisheries-induced evolution? Quantitative analysis of modelling and empirical studies. Evolutionary Applications, 6: 585–595.

Audzijonyte, A., Kuparinen, A., Gorton, R., and Fulton, E. A. 2013b. Ecological consequences of body size decline in harvested fish species: positive feedback loops in trophic interactions amplify human impact. Biology Letters, 9: 20121103.

Baeta, F., José Costa, M., and Cabral, H. 2009. Changes in the trophic level of Portuguese landings and fish market price variation in the last decades. Fisheries Research, 97: 216–222.

Bakun, A. 2006. Wasp-waist populations and marine ecosystem dynamics: Navigating the “predator pit” topographies. Progress in Oceanography, 68: 271–288.

Benoît, E., and Rochet, M.-J. 2004. A continuous model of biomass size spectra governed by predation and the effects of fishing on them. Journal of Theoretical Biology, 226: 9–21.

Bianchi, G. 2000. Impact of fishing on size composition and diversity of demersal fish communities. ICES Journal of Marine Science, 57: 558–571.

Birkeland, C., and Dayton, P. 2005. The importance in fishery management of leaving the big ones. Trends in Ecology & Evolution, 20: 356–358.

Blanchard, J. L., Jennings, S., Law, R., Castle, M. D., McCloghrie, P., Rochet, M.-J., and Benoît, E. 2009. How does abundance scale with body size in coupled size-structured food webs? Journal of Animal Ecology, 78: 270–280.

Branch, T. A. 2015. Fishing Impacts on Food Webs: Multiple Working Hypotheses. Fisheries, 40: 373–375.

Brey, T. 2001. Population dynamics in benthic invertebrates. A virtual handbook. http://www.thomas-brey.de/science/virtualhandbook/.

Brey, T., Müller-Wiegmann, C., Zittier, Z. M. C., and Hagen, W. 2010. Body composition in aquatic organisms — A global data bank of relationships between mass, elemental composition and energy content. Journal of Sea Research, 64: 334–340.

Brey, T. 2012. A multi-parameter artificial neural network model to estimate macrobenthic invertebrate productivity and production: Macrobenthic production model. Limnology and Oceanography: Methods, 10: 581–589.

Castro, J. J., Divovich, E., Delgado de Molina Acevedo, A., and Barrera-Luján, A. 2015. Over-looked and under-reported: A catch reconstruction of marine fisheries in the

Canary Islands, Spain, 1950-2010. Working Paper 26. Fisheries Centre: The University of British Columbia.

Christensen, V., Guenette, S., Heymans, J. J., Walters, C. J., Watson, R., Zeller, D., and Pauly, D. 2003. Hundred-year decline of North Atlantic predatory fishes. Fish and fisheries, 4: 1–24.

Christensen, V., Walters, C. J., Pauly, D., and Forrest, R. 2008. Ecopath with Ecosim version 6 User guide. Lenfest Ocean Futures Project, 235.

Costello, M. J., Coll, M., Danovaro, R., Halpin, P., Ojaveer, H., and Miloslavich, P. 2010. A Census of Marine Biodiversity Knowledge, Resources, and Future Challenges. PLoS ONE, 5: e12110.

Couce Montero, L., Christensen, V., Bilbao Sieyro, A., Pérez González, Y., Jiménez-Alvarado, D., and Castro, J. J. 2019. Temporal and spatial predictions of effect of alternative fishing policies for the Gran Canaria marine ecosystem. Journal of Fish Biology.

Couce-Montero, L., Christensen, V., and Castro, J. J. 2015. Effects of small-scale and recreational fisheries on the Gran Canaria ecosystem. Ecological Modelling, 312: 61–76.

Couce-Montero, L., Bilbao-Sieyro, A., Pérez-González, Y., and Castro, J. J. 2018. Temporal variations in the mean trophic level of the catches reported by the small-scale fishery of the Canary Islands (Central-East Atlantic). ISMS, VI International Symposium on Marine Sciences, Vigo, Spain.

Couce-Montero, L., Christensen, V., and Castro, J. J. 2019. Simulating trophic impacts of recreational fishing scenarios on two oceanic islands using Ecopath with Ecosim. Article submitted for publication.

Cury, P. 2000. Small pelagics in upwelling systems: patterns of interaction and structural changes in “wasp-waist” ecosystems. ICES Journal of Marine Science, 57: 603–618.

Cury, P., Shannon, L., and Shin, Y.-J. 2003. The functioning of marine ecosystems: a fisheries perspective. Responsible fisheries in the marine ecosystem: 103–123.

Cury, P., Fréon, P., Moloney, C. L., Shannon, L., and Shin, Y.-J. 2005. Processes and patterns of interactions in marine fish populations: an ecosystem perspective. The Sea, 13: 475–553.

Daan, N., Gislason, H., Gpope, J., and Crice, J. 2005. Changes in the North Sea fish community: evidence of indirect effects of fishing? ICES Journal of Marine Science, 62: 177–188.

Essington, T. E., Beaudreau, A. H., and Wiedenmann, J. 2006. Fishing through marine food webs. Proceedings of the National Academy of Sciences, 103: 3171–3175.

FAO (Ed). 2018. The state of world fisheries and aquaculture - Meeting the sustainable development goals. Rome. 210 pp.

Friedlander, A. M., and DeMartini, E. E. 2002. Contrasts in density, size, and biomass of reef fishes between the northwestern and the main Hawaiian islands: the effects of fishing down apex predators. Marine Ecology Progress Series, 230: 253–264.

Friedlander, A. M., Ballesteros, E., Clemente, S., Gonçalves, E. J., Estep, A., Rose, P., and Sala, E. 2017. Contrasts in the marine ecosystem of two Macaronesian islands: A comparison between the remote Selvagens Reserve and Madeira Island. PLoS ONE, 12: e0187935.

Froese, R., and Kesner-Reyes, K. 2002. Impact of fishing on the abundance of marine species. ICES Council Meeting Report CM.

García, S. M., Zerbi, A., Aliaume, C., Do Chi, T., and Laserre, G. 2003. The ecosystem approach to fisheries. Issues, terminology, principles, institutional foundations, implementation and outlook. FAO Fisheries Technical Paper. No. 443. Rome, FAO, 71 p. 71 pp.

González, J. A. 2008. Memoria científico-técnica final sobre el Estado de los Recursos Pesqueros de Canarias (REPESCAN). Instituto Canario de Ciencias Marinas, ACIISI, Gobierno de Canarias. Telde, Spain.

Greenstreet, S. P. R., and Rogers, S. I. 2006. Indicators of the health of the North Sea fish community: identifying reference levels for an ecosystem approach to management. ICES Journal of Marine Science, 63: 573–593.

Greenstreet, S. P. R., Rogers, S. I., Rice, J. C., Piet, G. J., Guirey, E. J., Fraser, H. M., and Fryer, R. J. 2011. Development of the EcoQO for the North Sea fish community. ICES Journal of Marine Science, 68: 1–11.

Guénette, S., and Morato, T. 2001. The Azores Archipelago, 1997. Fish. Cent. Res. Rep., 9: 241–270.

Hermida, M., and Delgado, J. 2016. High trophic level and low diversity: Would Madeira benefit from fishing down? Marine Policy, 73: 130–137.

Hernández, J. C., Clemente, S., Sangil, C., and Brito, A. 2008a. The key role of the sea urchin Diadema aff. antillarum in controlling macroalgae assemblages throughout the Canary Islands (eastern subtropical Atlantic): An spatio-temporal approach. Marine Environmental Research, 66: 259–270.

Hernández, J. C., Clemente, S., Sangil, C., and Brito, A. 2008b. Actual status of the sea urchin Diadema aff. antillarum populations and macroalgal cover in marine protected areas compared to a highly fished area (Canary Islands-eastern Atlantic Ocean). Aquatic Conservation: Marine and Freshwater Ecosystems, 18: 1091–1108.

Hunt, G. L., and McKinnell, S. 2006. Interplay between top-down, bottom-up, and wasp-waist control in marine ecosystems. Progress in Oceanography, 68: 115–124.

ICES. 2016. EU request to provide guidance on operational methods for the evaluation of the MSFD criterion D3C3. ICES Special Request Advice Northeast Atlantic Ecoregion.

ICES. 2017. EU request to provide guidance on operational methods for the evaluation of the MSFD criterion D3C3 (Second Stage 2017). ICES Special Request Advice Northeast Atlantic Ecoregion sr.2017.07.

Jacobsen, N. S., Gislason, H., and Andersen, K. H. 2013. The consequences of balanced harvesting of fish communities. Proceedings of the Royal Society B: Biological Sciences, 281: 20132701.

Jennings, S., and Kaiser, M. J. 1998. The effects of fishing on marine ecosystems. In Advances in marine biology, pp. 201–352. Elsevier.

Jennings, S., Greenstreet, S. P. R., and Reynolds, J. D. 1999. Structural change in an exploited fish community: a consequence of differential fishing effects on species with contrasting life histories. Journal of Animal Ecology, 68: 617–627.

Jennings, S., Pinnegar, J. K., Polunin, N. V., and Boon, T. W. 2001. Weak cross-species relationships between body size and trophic level belie powerful size-based trophic structuring in fish communities. Journal of Animal Ecology, 70: 934–944.

Jørgensen, C., and Fiksen, Ø. 2010. Modelling fishing-induced adaptations and consequences for natural mortality. Canadian Journal of Fisheries and Aquatic Sciences, 67: 1086–1097.

Jørgensen, C., and Holt, R. E. 2013. Natural mortality: Its ecology, how it shapes fish life histories, and why it may be increased by fishing. Journal of Sea Research, 75: 8–18.

Law, R., Plank, M. J., James, A., and Blanchard, J. L. 2009. Size-spectra dynamics from stochastic predation and growth of individuals. Ecology, 90: 802–811.

Law, R., Plank, M. J., and Kolding, J. 2012. On balanced exploitation of marine ecosystems: results from dynamic size spectra. ICES Journal of Marine Science, 69: 602–614.

Longhurst, A. 2002. Murphy's law revisited: longevity as a factor in recruitment to fish populations. Fisheries Research, 56: 125–131.

Mindel, B. L., Neat, F. C., Webb, T. J., Blanchard, J. L., and Handling editor: James Watson. 2018. Size-based indicators show depth-dependent change over time in the deep sea. ICES Journal of Marine Science, 75: 113–121.

Morato, T., Watson, R., Pitcher, T. J., and Pauly, D. 2006. Fishing down the deep. Fish and Fisheries, 7: 24–34.

Morato, T., Bulman, C., and Pitcher, T. J. 2009. Modelled effects of primary and secondary production enhancement by seamounts on local fish stocks. Deep Sea Research Part II: Topical Studies in Oceanography, 56: 2713–2719.

Morato, T. 2012. Description of environmental issues, fish stocks and fisheries in the EEZs around the Azores and Madeira. Report Prepared for the STECF Plenary Meeting, 12.

Morato, T., Lemey, E., Menezes, G., Pham, C. K., Brito, J., Soszynski, A., Pitcher, T. J., et al. 2016. Food-Web and Ecosystem Structure of the Open-Ocean and Deep-Sea Environments of the Azores, NE Atlantic. Frontiers in Marine Science, 3: 245.

Odum, E. P. 1953. Fundamentals of ecology. Saunders, Philadelphia. 384 pp.

Pauly, D. 1980. On the interrelationships between natural mortality, growth parameters, and mean environmental temperature in 175 fish stocks. ICES Journal of Marine Science, 39: 175–192.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., and Torres, F. 1998. Fishing down marine food webs. Science, 279: 860–863.

Pauly, D., Christensen, V., and Walters, C. 2000. Ecopath, Ecosim, and Ecospace as tools for evaluating ecosystem impact of fisheries. ICES Journal of Marine Science, 57: 697–706.

Pham, C. K., Canha, A., Diogo, H., Pereira, J. G., Prieto, R., and Morato, T. 2013. Total marine fishery catch for the Azores (1950-2010). ICES Journal of Marine Science, 70: 564–577.

Plagányi, É. E. 2007. Models for an ecosystem approach to fisheries. FAO Fisheries Technical Paper. No. 477. Rome, FAO, 108 pp.

Rochet, M., and Rice, J. 2005. Do explicit criteria help in selecting indicators for ecosystem-based fisheries management? ICES Journal of Marine Science, 62: 528–539.

Rochet, M.-J., and Trenkel, V. M. 2003. Which community indicators can measure the impact of fishing? A review and proposals. Canadian Journal of Fisheries and Aquatic Sciences, 60: 86–99.

Sangil, C., Martín-García, L., Hernández, J. C., Concepción, L., Fernández, R., and Clemente, S. 2013a. Impacts of fishing and environmental factors driving changes on littoral fish assemblages in a subtropical oceanic island. Estuarine, Coastal and Shelf Science, 128: 22–32.

Sangil, C., Martín-García, L., and Clemente, S. 2013b. Assessing the impact of fishing in shallow rocky reefs: A multivariate approach to ecosystem management. Marine Pollution Bulletin, 76: 203–213.

Sharpe, D. M., and Hendry, A. P. 2009. Life history change in commercially exploited fish stocks: an analysis of trends across studies. Evolutionary Applications, 2: 260–275.

Shin, Y. J., Rochet, M. J., Jennings, S., Field, J. G., and Gislason, H. 2005. Using size-based indicators to evaluate the ecosystem effects of fishing. ICES Journal of Marine Science, 62: 384–396.

Shon, S., Delgado, J. M., Morato, T., Pham, C. K., Zylich, K., Zeller, D., and Pauly, D. 2015. Reconstruction of marine fisheries catches for Madeira Island, Portugal from 1950-2010. Fisheries Centre, The University of British Columbia, Working Paper Series, Working Paper #2015-52.

Smith, A. D. M., Brown, C. J., Bulman, C. M., Fulton, E. A., Johnson, P., Kaplan, I. C., Lozano-Montes, H., et al. 2011. Impacts of Fishing Low-Trophic Level Species on Marine Ecosystems. Science, 333: 1147–1150.

Stobberup, K. A., Ramos, V. D. M., and Coelho, M. L. 2004. Ecopath model of the Cape Verde coastal ecosystem. In Palomares, M.L.D., Pauly, D. (eds.) West African marine ecosystems: models and fisheries impacts. Fisheries Centre Research Reports 12(7): 39-56. Fisheries Centre, UBC, Vancouver.

Travers, M., Shin, Y.-J., Shannon, L., and Cury, P. 2006. Simulating and testing the sensitivity of ecosystem-based indicators to fishing in the southern Benguela ecosystem. Canadian Journal of Fisheries and Aquatic Sciences, 63: 943–956.

6 Analysis >> QD5 Eutrophication

QD5 Human-induced eutrophication is minimised, especially adverse effects thereof, such as losses in biodiversity, ecosystem degradation, harmful algae blooms and oxygen deficiency in bottom waters							
QD	**Criteria (element)**	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
QD5	Nutrient concentrations (Dissolved Inorganic Nitrogen (DIN), Total Nitrogen (TN), Dissolved Inorganic Phosphorus (DIP), Total Phosphorus (TP)) are not at levels that indicate adverse eutrophication effects.	D5C1	No				
	Chlorophyll a concentrations are not at levels that indicate adverse effects of nutrient enrichment.	D5C2	No				
	The number, spatial extent and duration of harmful algal bloom (e.g. cyanobacteria) events are not at levels that indicate adverse effects of nutrient enrichment.	D5C3 — Secondary					
	The photic limit (transparency) of the water column is not reduced, due to increases in suspended algae, to a level that indicates adverse effects of nutrient enrichment.	D5C4 — Secondary					
	The concentration of dissolved oxygen is not reduced, due to nutrient enrichment, to levels that indicate adverse effects on benthic habitats (including on associated biota and mobile species) or other eutrophication effects.	D5C5	No				
	The abundance of	D5C6 —					

	opportunistic macroalgae is not at levels that indicate adverse effects of nutrient enrichment.	Secondary					
	The species composition and relative abundance or depth distribution of macrophyte communities achieve values that indicate there is no adverse effect due to nutrient enrichment including via a decrease in water transparency	D5C7 — Secondary					
	The species composition and relative abundance of macrofaunal communities, achieve values that indicate that there is no adverse effect due to nutrient and organic enrichment	D5C8 — Secondary					

6.1 D5C1, D5C2, D5C5

Eutrophication can be understood as an increase in the entry or production of organic matter (for example, through wastewater, fertilizers ...). The effects of eutrophication range from small changes in the production and composition of species in areas exposed to incipient enrichment, through a significant simplification of the environment, to a complete elimination of organisms (Valiela, 2006).

Those responsible for carrying out the Marine Strategies in Madeira and the Azores do not indicate any influence of the fishing on the processes of eutrophication in these regions (Regional Secretariat for Environment and Two Naturais Resources, 2014, Regional Secretary for Naturais Resources, 2014). However, in the Canary Islands they do indicate the existence of a pressure called 'incidental bycatch' that can generate the 'organic matter input' as an impact (Lloret Capote, del Barrio Alvarellos and Moreno Aranda, 2012; Perez Puyol et al., 2012), although they do not explain how that entry is made. It would be possible to assume that it refers to the discharging into the sea of discards from fishing or even the dumping of offal after the completion of some type of processing of catches on board. On the other hand, in the Canary Islands, when the 'analyzes and impacts' are discussed, they report that artificial reefs ('related to fisheries management') can produce an increase in organic matter and its subsequent alteration in water properties, although they point out that significant negative impacts are not to be expected (Lloret Capote, del Barrio Alvarellos and Moreno Aranda, 2012).

Eutrophication is commonly considered as a synonym for fertilization, with phosphorus (above all) and nitrogen being the limiting elements that are usually the source of domestic, agricultural or industrial discharges (Lee, Jones and Jones, 1991). On a global scale, the oxygen content of the open ocean and coastal waters has decreased since the middle of the last century largely due to human activity that has increased the global temperature and the discharge of nutrients to coastal waters (Breitburg et al. ., 2018). These authors point out areas of hypoxia on a global scale and none is Azores, Madeira or Canary Islands. Among the causes of oxygen depletion are global warming and nutrient enrichment of coastal waters (eutrophication) due to human, agricultural waste as well as the deposition of N from fossil fuels. In no case fishing appears as a cause. However, as will be seen in the following paragraphs, more or less direct relationships between eutrophication and fish or fishing have been described, albeit at smaller scales.

An increase in primary production (derived from eutrophication) could increase the quantity of fish although its quality (from a trophic point of view) could be reduced in favor of fish with less interest. In general, the oceans have little nutrient load so their fisheries are usually limited by primary production. However there are situations where an excess of nutrients can have harmful effects on fishing and water quality, for example by blooms of dinoflagellates that can cause specific events of anoxia by the decomposition of these organisms, which can lead to death of fish in special circumstances both geographical (coastal waters, bays) and environmental - very stratified zones, high temperatures, etc. - (Lee, Jones and Jones, 1991).

On the other hand it has been described in lakes, bays or seas with little renovation, how fish and fishing can influence the dynamics of nutrients through excretion, through feeding on the bottom (mobilizing nutrients on the sediment), by zooplantivorous feeding (decreasing the regeneration of nutrients) or by permanently removing nutrients from the biomass of the fish caught (Hjerne and Hansson, 2002, Iho et al., 2017). In the Baltic Sea where eutrophication is considered a problem, the elimination of biomass and therefore nutrients, through fisheries (catches) especially in those that are balanced, can influence the marine dynamics of nutrients and should be considered in terms of of management (Hjerne and Hansson, 2002). In this sense, in addition to considering the positive effect of fishing (in the MSY) on nutrient reductions Nielsen et al. (2019) point out that it can be achieved at a low cost compared with other measures that combat eutrophication (for example the use of algae or mussels).

Madeira, Azores and the Canaries are oceanic islands with insular platforms where, at a short distance from the coast, considerable depths are reached. That is to say, the exchanges of nutrients between the bottom and the water column are practically negligible, with certain exceptions in intertidal zones, shallow zones or others where there is a certain platform. In any case, in general, the oceanic waters that surround the islands have an oligotrophic character (Hernández et al., 2012) and the renewal of the waters is high.

In short, problems of eutrophication are more common in water bodies that receive abundant nutrients and that, in addition, have a low renewal. In general terms, as indicated, the limiting nutrients of the primary producers are nitrogen and phosphorus that can reach the marine environment from land (eg urban and industrial spills, surface runoff, etc.), from the sea (eg ships, platforms, aquaculture) or from the air (atmospheric depositions) (Martín Partida, Arrieta Algarra, Martínez García-Denche, et al., 2019). However, in the Canary Islands ships (any) are not considered as the origin of activities that contribute to eutrophication (Cortés et al., 2012, Martín Partida, Arrieta Algarra, Martínez García-Denche, et al., 2019). In addition, according to Cortés et al. (2012) 'there are no problems of eutrophication in the Demarcation (Canary)'.

In any case, perhaps anecdotally - in the context of the Life Cycle Assessment (LCA) - another of the possible sources of N and P that fishing can contribute towards eutrophication is through the consumption of fuel and the production of same (indirectly) (Abdou et al., 2018), although this approach is beyond the scope of the analysis of this work.

However, previously we talked about the possibility of enrichment in the form of organic matter through fishing discards. The contributions of organic matter (decomposition), although they do not affect D5 'Eutrofization' do affect D4 'Trophic networks' (Martín Partida, Arrieta Algarra, Martínez García-Denche, et al., 2019). These contributions in abundance can cause a decrease in oxygen levels and, in the most extreme cases, anoxia. However, these contributions of organic matter are defined as that matter that arrives at the (marine) system from the outside (Martín Partida, Arrieta Algarra, Martínez García-Denche, et al., 2019). Therefore, discards in fishing seem to be outside this influence on the marine environment (regardless of whether it is towards D5 or D4). Anyway, in the following paragraphs this possibility is also discussed.

Discards (which returns to the sea) are consumed mainly by birds, fish and scavenger species in the bottom (Sánchez and Olaso, 2004). Factory trawlers can discharge to the sea waste from fishing processing (casings, heads, pieces), as well as whole fish that are lost or discarded on the surface. In certain circumstances they can cause a reduction in oxygenation levels and modify the composition of benthic communities (Ramirez-llodra et al., 2011). However, the boats that develop this type of fishing are outside the scope of this work that focuses, fundamentally, on the small-scale fishing fleet such as the one developed by the Canary Islands, Madeira and Azores. On a global scale, industrial fishing produces the majority of discards, while small-scale fisheries generate only 7% (Fauconnet et al., 2019). As an approximation to the scale of the problem in a fishery close to Macaronesia, such as the one developed in the coastal ecosystem of the Cantabrian Sea, it was estimated that discards were 20%. For this case, the importance of discards as food in ecosystems turned out to be low, in comparison with detritus, primary producers and other low trophic levels. Discards represented 0.07% of the total food intake in this ecosystem (Sánchez and Olaso, 2004). In the Azores, it has been estimated that around 5% of the total catch (1950-2014) is discarded (Fauconnet et al., 2019). In the Canary Islands, there were no studies on discards on a regional scale, although it is true that a large number of species are fished and most are commercialized, which means that the existence of discards, theoretically, is low (Rico, Santana and González, 1999). As a curiosity according to Celi et al. (2018) the measure proposed by the EU to counteract discards - known as Landing Obligation, which requires the discharge (not for human consumption) in port of discards of species subject to quotas or to 'minimum conservation reference size' (MCRS) - can cause negative effects both ecologically and economically in multi-species fisheries not regulated by quotas, due to the reduction of biomass in ecosystems, the reduction of catches of commercial species, the increase in the workload of fishermen and the reduction of economic benefits..

In general, it can be assumed that the production of discards could be insignificant in general terms in the areas under analysis, above all because of the prohibition of trawling gear and the fishing techniques typical of these regions. However, it is necessary to highlight the scarcity of quality data on discards. In this sense, the best tool for monitoring discards (commercial, non-commercial and vulnerable species) would be through on-board observers, which, although an expensive option, provides more reliable and complete information than official statistics (Fauconnet et al. ., 2019). Due to the importance of recreational fishing in Macaronesia and the overlap of catches with professional fishing (Jiménez Alvarado, 2015, Fauconnet et al., 2019), it is suggested the need to provide data in this regard for both fisheries. Finally, 'the transformation of fish and shellfish is an activity carried out in facilities on land, so it does not have a direct impact on the marine environment' (Martín Partida, Arrieta Algarra, Martínez García-Dencha, et al., 2019).

In short, within the framework of the Marine Strategy Framework Directive, eutrophication is analyzed by means of limiting nutrients (phosphorus and nitrogen) that tend to have domestic, agricultural or industrial discharges as a source. Although relationships between fishing and eutrophication have been described, these have been rather towards the effects of eutrophication (of terrestrial origin) on fishing or on the possible use of fisheries in the management of eutrophication problems in certain conditions. No references have been found that suggest impacts of small-scale fishing on D5 within the framework of the MSFD. However, on a wide scale, some indirect effects of fishing have been found through discards to the marine environment that in certain circumstances may cause changes in the composition of the benthic communities (perhaps in relation to the D4 'trophic relations'). In any case, due to the scale and fishing techniques that are developed in Macaronesia, significant amounts of discards are not to be expected. However, this does not prevent to highlight the scarcity of quality data on discards. In that sense, the best tool for monitoring discards would be through on-board observers. Due to the importance of recreational fishing in Macaronesia and the overlap of catches with professional fishing, the need to provide data in this sense for both fisheries is suggested.

6.2 References

Abdou, K. et al. (2018) ‘Environmental life cycle assessment of seafood production: A case study of trawler catches in Tunisia’, Science of the Total Environment. Elsevier B.V., 610–611(November 2017), pp. 298–307. doi: 10.1016/j.scitotenv.2017.08.067.

Breitburg, D. et al. (2018) ‘Declining oxygen in the global ocean and coastal waters’, Science, 359(6371). doi: 10.1126/science.aam7240.

Celi, I. et al. (2018) ‘Original Article Ecological and economic effects of the landing obligation evaluated using a quantitative ecosystem approach : a Mediterranean case study’, ICES Journal of Marine Science, 75, pp. 1992–2003. doi: 10.1093/icesjms/fsy069.

Cortés, D. et al. (2012) Estrategia Marina. Demarcación Marina Canaria Parte IV. Descriptores del buen estado ambiental. Descriptor 5: eutrofización. Evaluación inicial y buen estado ambiental. Madrid. Available at: http://www.mapama.gob.es/es/costas/temas/proteccion-medio-marino/IV_D5_Canarias_tcm30-130939.pdf.

Fauconnet, L. et al. (2019) ‘An overview of fisheries discards in the Azores’, Fisheries Research. Elsevier, 209(4), pp. 230–241. doi: 10.1016/j.fishres.2018.10.001.

Hernández, C. L. et al. (2012) Estrategia Marina. Demarcación Marina Canaria Parte I. Marco general. Evaluación inicial y buen estado ambiental. Madrid. Available at: http://www.mapama.gob.es/es/costas/temas/proteccion-medio-marino/I_Marco General_Canarias_tcm30-130932.pdf.

Hjerne, O. and Hansson, S. (2002) ‘The role of fish and fisheries in Baltic Sea nutrient dynamics’, Limnol. Oceanogr., 47(4), pp. 1023–1032. Available at: https://doi.org/10.4319/lo.2002.47.4.1023.

Iho, A. et al. (2017) ‘The Role of Fisheries in Optimal Eutrophication Management’, Water Economics and Policy, 3(2), pp. 1–27. doi: 10.1142/s2382624x16500314.

Jiménez Alvarado, D. L. (2015) Pesca recreativa en Canarias : aspectos principales y evolución. Universidad de Las Palmas de Gran Canaria. Available at: https://opac.ulpgc.es/cgi-bin/abnetopac?TITN=728041%0A%0A.

Lee, G. F., Jones, P. E. and Jones, R. A. (1991) ‘Effects of Eutrophication on Fisheries’, Reviews in Aquatic Sciences, 5(3), pp. 287–305. Available at: www.gfredlee.com/Nutrients/fisheu.html.

Lloret Capote, A., del Barrio Alvarellos, I. and Moreno Aranda, I. M. (2012) Estrategia Marina. Demarcación Marina Canaria. Evaluación inicial. Parte II: Análisis de presiones e impactos. Madrid. Available at:

http://www.mapama.gob.es/es/costas/temas/proteccion-medio-marino/II_Analisis_Presiones_Canarias_tcm30-130933.pdf.

Martín Partida, I., Arrieta Algarra, S., Martínez García-Denche, L., et al. (2019) Estrategias marinas de España. Estrategia marina de la demarcación Canaria. Parte II. Análisis de presiones e impactos. Anexo Parte II. Fichas del análisis de presiones e impactos (En Fase de exposición Pública: 12.03.19-12.04.19). Madrid. Available at: https://www.miteco.gob.es/es/costas/participacion-publica/anexoparteiianalisispresioneseimpactosdmcan_tcm30-487968.pdf.

Martín Partida, I., Arrieta Algarra, S., Martínez García-Dencha, L., et al. (2019) Estrategias marinas de España. Estrategia marina de la demarcación Canaria. Parte III. Análisis económico y social (En Fase de exposición Pública: 12.03.19-12.04.19). Madrid. Available at: https://www.miteco.gob.es/es/costas/participacion-publica/parteiiianalisiseconomicoysocialdmcan_tcm30-487965.pdf.

Nielsen, R. et al. (2019) 'Fishing for nutrients – economic effects of fisheries management targeting eutrophication in the Baltic Sea', Ecological Economics. Elsevier, 160(June 2018), pp. 156–167. doi: 10.1016/j.ecolecon.2019.02.013.

Perez Puyol, A. et al. (2012) Estrategias Marinas. Documento marco. Evaluación inicial, buen estado ambiental y objetivos ambientales. Madrid. Available at: http://www.mapama.gob.es/es/costas/temas/proteccion-medio-marino/0_Documento marco estrategias marinas_tcm30-130950.pdf.

Ramirez-llodra, E. et al. (2011) 'Man and the Last Great Wilderness : Human Impact on the Deep Sea', PLoS ONE, 6(7). doi: 10.1371/journal.pone.0022588.

Rico, V., Santana, J. I. and González, J. A. (1999) Técnicas de pesca artesanal en la isla de Gran Canaria. Monografías del Instituto Canario de Ciencias Marinas. No 3. Available at: https://mdc.ulpgc.es/utils/getfile/collection/MDC/id/112680/filename/150124.pdf.

Sánchez, F. and Olaso, I. (2004) 'Effects of fisheries on the Cantabrian Sea shelf ecosystem', Ecological Modelling, 172, pp. 151–174. doi: 10.1016/j.ecolmodel.2003.09.005.

Secretaria Regional do Ambiente e dos Recursos Naturais (2014) Estratégia Marinha para a subdivisão da Madeira. Diretiva Quadro Estratégia Marinha. Available at: https://www.dgrm.mm.gov.pt/documents/20143/43971/RelatorioInicial_MAD_FINAL__2014.pdf/0f2783be-bf81-5d26-83cd-15830cff998c

Secretaria Regional dos Recursos Naturais (2014) Estratégia Marinha para a subdivisão dos Açores. Diretiva Quadro Estratégia Marinha. Available at: https://www.dgrm.mm.gov.pt/documents/20143/43971/RelatorioInicial_AZO_FINAL_2014+%282%29.pdf/a0b43add-3ba3-6e52-e660-11189ef15ad7.

Valiela, I. (2006) Global Coastal Change. Edited by O. B. Publishing.

7 Analysis >> QD6 Sea-floor integrity

QD6 Sea-floor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected						
Criteria (element)	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
Spatial extent and distribution of physical loss (permanent change) of the natural seabed	D6C1	yes	need further research	yes	yes	need further research
Spatial extent and distribution of physical disturbance (including intertidal areas) pressures on the seabed.	D6C2	yes	need further research	yes	yes	yes
Spatial extent of each habitat type which is adversely affected, through change in its biotic and abiotic structure and its functions (e.g. through changes in species composition and their relative abundance, absence of particularly sensitive or fragile species or species providing a key function, size structure of species), by physical disturbance.	D6C3	yes	need further research	yes	yes	yes

7.1 D6C1

Environmental impact

Physical loss shall be understood as a change to the seabed which could be restored if the activity causing the disturbance pressure ceases, but recovery time leading are larger than 12 years (Commission Decision on Good Environmental Status criteria, for which the Marine Strategy Regulatory Committee). In this way, time intervals needed to restoring sea bottom or benthic communities impacted by fishing could be highly variable, depending of species and biological fluxes affected, and could need from years or decades to centuries (Blackwood, et al., 2012; Lambert et al., 2014; Hiddink et a., 2017).

The main impact of fishing on natural seabed, producing changes that could be permanent in a time lapse of decades or centuries, is generated through it effects on benthic communities due to alteration of trophic fluxes, because overfishing of one or several key species, reducing the resilience of the ecological system in front of climatic change by loss of genetic variability and subsequent ecological/behavioural response of species (Pikitch, 2012; Neubauer et al., 2013; Sangil et al., 2013; Coll et al., 2014; Couce-Montero et al., 2015).

The abrupt topography of seabed of the of Canary, Madeira and Azores islands, with very narrow island shelves make difficult bottom trawling and restricting the use of other gear by the small-scale fleet to specific fishing grounds, mainly to the leeward areas of the islands (Bas et al., 1995; Ojamaa, 2015). In this way, but principally for the dramatic impacts that these fishing gears have on the singular marine seabed communities of the islands, all modalities of bottom trawling has been prohibited in all fishing areas of the three archipelagos (Morato, 2012; MAGRAMA, 2015), including beach trawl. Nevertheless, there is no studies on the impact that other fishing gears have on natural seabed, although it thought that bottom longlines and handlines are less harmful or have no impact on sessile organisms (Chuenpagdee et al., 2003; Pham et al., 2014). However, Morato (2012) indicate that bottom longlines set in the Azores impact on sessile organism, particularly cold-water corals that are seriously damaged, which may alter benthic community structure (Sampaio et al., 2012), but Pham et al. (2014) estimated that this impact is low and these slow-growing vulnerable species are still common in the traditional deep-water fishing grounds of Azores.

On the other hand, other important impact of fishing on seabed is that produced by ghost fishing due to gear lost (NOAA, 2015). Although ghost fishing has not been evaluated in the region, Castro and Hernández-García (2012) estimated that about 10% of fish-traps deployed in deep-water areas (deeper than 200 m) were lost each fishing campaign, mainly due to deep-water current or foul on rocky-beds. Pham et al. (2013) reported that lost fishing line was the dominant litter item encountered on Condor seamount, all being entirely or partly entangled in the sessile fauna, like gorgonians, but also lost weights and anchors. Probably the presence of litter related to fishing activities, particularly pieces of lines and gill-nets like, are common in other fishing grounds of the Macaronesian archipelagos, but Pham et al. (2013) considered that abundance of litter on the Condor seamount was much lower than that reported from other locations closer to populated areas.

In a similar context, marine litter associate to fishing activities, is the use of scrap (old card, refrigerators, etc.) by fishermen like artificial reef to aggregate fish resources in specific fishing grounds under their own control and knowledge. In this way, in the Canary Islands some fishermen have been using scrap for build these artificial reef, since the decade of 1980, without any administrative licence.

Pole and line tuna fishing are considered extremely selective a fishing method (Silva et al., 2011), but baitfish used by the tuna vessels are frequently caught near the shore with purse seines, in waters shallower than 5-15 m depth, and most of the times on seagrass beds. It is frequent that the weight of the purse seine reach the bottom, and works as a trawl while closing the net. During this closing manoeuvre, part of the seagrass are cut and produce clears or gaps in the meadows. Anchoring and mooing have a more dramatic negative effect on seagrass meadows (Unsworth et al., 2017), but also in natural reef due to foul and soft sediment bottoms. Collins et al. (2013) reported that when comparing the undisturbed seagrass sediment with the bare, impacted areas, the latter sediments are less cohesive, contain less organic material and have a lower silt fraction, infaunal organism number and taxa.

Probably one of the most visible impact of fishing activity on natural seabed is related with the shellfish gathering on foot on intertidal areas, where during harvesting of resource the intertidal ground is heavily walked, many rocks are turned over regularly (Brey, 1991), and overfished (Riera et al., 2016). This activity of turning the rocks to catch crabs (e.g. *Xantho* spp. as bait to fish parrotfish), sea urchins or octopus, produce the exposition to light of the cryptic or scyaphila fauna, provoking its death, and changing deeply the benthic community structure and composition (Jones, 1992).

Also, and in a similar way that indicated by DeGroot (1984) in relation to trawling, intertidal fishing of gastropods and other animal that burrow in the sand can change to the physical integrity of the sediment system, due to scraping, digging or ploughing of intertidal sediments, destruction or disturbance of bedforms, and damage to the benthos.

Environmental impact spatial extent

There are no data about the spatial distribution of impacts due to the use of bottom longlines or handlines on sessile fauna, or the impact of ghost fishing, but it could be assumed that it may be produced in all fished areas around the islands and, in the case of ghost fishing (and other marine litter associated to fishing), beyond, due to drifting and transport of lost gears by deep-water marine currents.

On the other hand, the impact of shellfish gathering on natural seabeds can be mainly circumscribed to intertidal areas of low-lying shores, accessible by foot to shellfish-gatherers.

One of the most visible effects of fishing on benthic communities due to alteration of trophic fluxes, is the seabeds barren grounds called "blanquizales" (white bottoms) in the Canary Islands. Blanquizales are areas with no erect macroalgal cover as a result of overgrazing produced by the high density of the sea urchin (*Diadema africanum*) that generate the subsequent formation of ‘‘urchin barren grounds’’ (Tuya et al., 2004a; Ortega et al., 2009; Hernández et al. 2013). The influence of echinoids grazing activity on rocky reefs communities of the eastern Atlantic oceanic islands, particularly in the Canary Islands, have been well studied (Alves et al. 2001, Tuya et al. 2004b; Hernández et al. 2008a). Nevertheless, the progression of these urchin barren grounds lay over many factor, as water warming, but mainly due to overfishing of natural predator of this sea urchin specie (Hernández et al., 2008b; Clemente et al., 2009), like seabreams (*Diplodus* spp., *Dentex* spp.) or barred hogfish (*Bodianus scrofa*) (Aguilera-Klink et al., 1993).

Maritime Activity pressure solution

In the Canary, Madeira and the Azores islands, trawling is prohibited according to Regulation (EC) 1568/2005. In the Canary Islands, since 1986, any form of trawling has been prohibited (MAGRAMA, 2015).

In relation to the use of scrap for artificial reef, and that also could be applicable to lost fishing gears, the OSPAR Convention has established a series of guidelines for the installation of artificial reefs, which Spain and Portugal as part contractor of the agreement, are forced to comply. The OSPAR Guidelines on Artificial Reefs in relation to Living Marine Resources, resulted of the working group created for this purpose, being approved at the 1999 meeting of the OSPAR Commission. The contracting parties to the OSPAR Convention undertake to take all measures possible in order to prevent and eliminate pollution and protect the maritime area of the harmful effects of human activities. It is intended to protect human health, preserve marine ecosystems and, if possible, recover marine areas that have been harmed by anthropogenic activity (MIMEA, 2008). Within Annex 2 of the OSPAR Convention, the dumping of all waste at sea is prohibited other materials, except the following: (i) Dredged material, (ii) Inert materials of natural origin (i.e., not elaborated chemically whose chemical components are not likely to be released in the marine environment), (iii) Waste water sludges until December 31, 1998, (iv) Fish waste from fishing industries, and (v) Ships or aircraft until, at the latest, on December 31, 2004 (i.e. artificial reefs). Also in the same Annex it is established that "No materials will be placed in the area maritime for purposes other than those for which they were designed or constructed originally without authorization or regulation from the competent authority of the Corresponding Authority. That authorization or regulation shall be in accordance with the applicable criteria, guidelines and procedures adopted by the Commission".

Impact mitigation measures

There is no legal enforceability to report the lost of a fishing gear, and we think that should be highly recommended to have a official data series of lost gears by type, year and fishing area, in order to estimate the potential ghost fishing and to develop a plan of recovering when possible.

Moreover, it will be very useful to prepare a guide of best practices for shellfish gathering in order to minimize the impact of fishermen on seabed and scyaphila fauna. This guide should be also complemented by a more extensive plan of catch recording, including species and sizes of individuals caught.

Monitoring methods

Unfortunately, there are not official programmes of data recording of the fishing gear loss by type, year and fishing ground for artisanal and recreational fisheries in the different islands of the Macaronesian archipelagos. Moreover, in the Canary Islands captures obtained during shellfish gathering on intertidal shore are not recorded or not separated from data recorded of the whole artisanal fishery.

7.2 D6C2

Environmental impact

There are no data about the spatial extent and distribution of physical disturbance pressure on the seabed associated with fishing activities. There are no information about fishing pressure due to intertidal gathering or associated to different fishing method on the insular shelves.

On the other hand, the high demographic growth of *Diadema africanum* population, due to overfishing of its natural predator (Hernández et al., 2013), has been widely documented and frequently associated to the acute impoverishment of coastal rocky substrates in all Canarian islands, with the exception of El Hierro island, where fishing pressure has been lower and more strictly regulated in recent decades (Tuya et al., 2004a; Riera et al., 2014). In this way, Sangil et al. (2012) recorded declines in *D. africanum* populations and a recovery of algal assemblages in a Marine Protected Area (MPA) in La Palma island after 4 years, concurrent with increases in densities of predatory fishes (e.g. hogfishes, snappers and groupers). Also, Hernández et al. (2006) point out that the current fishing pressure and overfishing of sea urchin predators, facilitate that a high rate of turnover of *D. africanum* population to maintain the barren habitat.

Environmental impact spatial extent

In the case of small-scales fishing of the Macaronesian archipelago, it is possible to assume that spatial extent of physical alteration of seabed, due to directly or indirectly to fishing, is limited to the insular shelf of each island, but some effects could be felt beyond, in deeper areas. For example, in the case of litter due to fishing gear lost, it could be transported by currents and accumulates in troughs, canyons, and local depressions, rather than reflecting the fisheries footprints directly, and this transport could produce a trawling effect due to its dragging movements on the bottom by deepwater currents.

Maritime Activity pressure solution

The solution of physical disturbance on seabed due to biological marine communities perturbations because overfishing depends on the implementation of actions, or strategies, addressed to reduce fishing pressure on target species and marine ecosystems. There are several measures to reach a fishing effort reduction, that obviously should be taken on gradual way, in order to not generate collapses in the economical system associate to fishing (artisanal and recreational) or negative effects on non regulated areas or resources (Hiddink et al., 2006). Nevertheless, the actual overfishing situation of fish stocks in the Canary Islands (Castro et al., 2019), and probably in the other two Macaronensian archipelagos (Pham et al., 2013; Shon et al., 2015), pass necessarily through a reduction of fishing effort (number of fishermen/boats, number of fishing hours, implementation of TACs for each target species, and for the whole fishery, and/or introducing use rights) (Bjordal, 2002; Pope, 2002; Charles, 2002; Tserpes et al., 2016), and the establishment of marine protected areas (Taylor & McIlwain, 2010; Rising & Heal, 2014). These last would help to restore the biological balances and to rebuild the biomass of those species that have been depressed by the different fishing actors or fleets (FAO, 2011; Menildrey et al., 2013).

Impact mitigation measures:

All mitigation measures, as spatial closures and fishing effort restrictions, addressed to reduce the physical disturbances on seabed would have positive and direct effects on habitat structure and in rebuilding target species biomass (Botsford et al., 1997; Hilborn et al., 2004; Zeller & Reinert, 2004; Worm et al., 2009; Couce-Montero et al., 2019).

Fishing gears should be equipped with sensor that allow its location in case of loss, doing possible its recovery.

Monitoring methods

Regular programmes of monitoring of species richness of benthic macrofauna in areas under fishing should be implemented, and compare the variations recorded with non-fishing areas used as control. The number of species and density in the benthic invertebrate community as well as the average individual weight, should be recorded. Biodiversity, density and mean weight are rather strong indicators of impacts of fishery on the benthic invertebrate community. It is evident that the fishing pressure has different impacts on the biodiversity and density in different habitats dependent on the season of the year. Nevertheless, variations on species distribution and species life history the approach may provide useful impact estimates that could be produced by other stressors or pressures different than fishery only (Korpinen et al., 2018).

On the other hand, the physical disturbance pressure has spatial extent which is not regularly monitored. Environmental monitoring programmes rarely have spatial components that cover effect distances from activities, and it is necessary to develop estimates of the spatial extent of different pressures.

7.3 D6C3

Environmental impact

There are no information about the spatial extent of each habitat type that is adversely affected, through changes in its biotic and abiotic structure and its function due to fishing, and particularly because modification of physical characteristics of seabed, loss of fishing gear that could damage the sessile organism, or deep changes in the ecological structure of benthic communities that could produce permanent changes in bottom nature. Nevertheless, in 1970, García-Cabrera warned that the fishing grounds of all the Canary Islands with depths less than 100 m were overfished. Moreover, the REPESCAN report (González, 2008) confirmed García-Cabrera's findings 38 years before and indicated that this phenomenon had spread to all fishing grounds and the entire range of depths at which the artisanal fleet operate. However, the fishing impact could be spatially variable, most likely due spatial differences in habitat patchiness (Stobart et al., 2012) according to terrain and slope variations along the island shelf as well as differences in the fishing potential of each fraction of the artisanal fleet and the on-shore infrastructures available to them. On the other hand, the impact of intertidal gathering could be extended to all tidal rocky habitat type present in accessible beaches, particularly in the most western islands of the Canary Islands.

Environmental impact spatial extent

There are no data about the environmental impact spatial extent of physical disturbance pressure on the seabed associated with fishing activities, because there are no information available about fishing pressure due to intertidal gathering or associated to different fishing method on the insular shelf of each island (Ramirez et al., 2008).

Hernández et al. (2013) indicate that *Diadema africanum* is the most abundant sea urchin species in the Canary Islands region, reaching densities up to 240 individuals per 100 m^2 covering large extensions of unvegetated rock. It can reach depths up to 100 m. The species has been also described in Madeira (Alves et al., 2001). Nevertheless, Tuya et al. (2007) reported that *D. africanum* is less adapted to support intense water movement, and due to this its density increase with depth. The result is that areas with higher hydrodynamics show greater urchin density in a deep band (15–20 m) just below the algal stand, whereas along sheltered coasts algal beds the urchins occur only at the first meters depth where more food is available (Hernández et al. 2008).

According to Aguilera-Klink et al. (1994), Brito et al. (2004) and Tuya et al. (2004c), barren grounds resulting from the intense grazing activity of *Diadema africanum* are commonly spread throughout the entire region, reaching up to 50 m depth and covering about 75 % of the total littoral rocky bottoms of the Canary Islands (Barquín et al. 2004). This intense sea urchin predation on macroalgae results in areas denuded of all but encrusting algae (Lawrence 1975; Sangil et al. 2006a, 2006b).

Maritime Activity pressure solution:

The protection of each habitat type which is adversely affected by fishing could be done through the creation of marine protected areas (MPAs), that would contain significative representation of these benthic and pelagic habitats in neritic waters, in each island (Hoagland et al., 2001). Currently, in the Canary Islands there are three MPA, located in waters of the islands of Lanzarote, La Palma and El Hierro, that cover about 74.500 hectares. But for protecting the different habitat types of this archipelago it is necessary to equipped it with a larger number of MPA, deploying at least one per island. On the other hand, the Madeira archipelago have five MPAs (The Savage Islands, Garajay, Desert Islands, Porto Santo and Rocha do Navio) covering 1.466 hectares of marine area, while Azores has other eleven MPAs (Terceira, Pico, Santa María, Sao Jorge, Corvo, Formigas, Graciosa, Faial, Flores, and Sao Miguel Islands) that protect about 101.100 hectares.

Impact mitigation measures

MPAs alone are not a guarantee to obtain a real protection of habitats and ecological marine communities, and must be seen as one of the tools to be considered in the overall goal of achieving sustainable use of oceans (FAO, 2011). It is necessary a more conservative management of the whole fishing ground affected for the small-scales and recreational fleets to achieve the optimal and sustainable utilization of the fishery resources.

Monitoring methods

Regular programmes of monitoring of species richness of benthic macrofauna in areas under fishing should be implemented, and compare the variations recorded with non-fishing areas used as control. Biodiversity, density and mean weight are rather strong indicators of impacts of fishery on the benthic invertebrate community, and consequently on habitats structure (Korpinen et al., 2018).

7.4 References

Aguilera Klink, F., A. Brito-Hernández, C. Castilla-Gutiérrez, A. Díaz-Hernández, J.M. Fernández-Palacios, A. Rodríguez-Rodríguez, F. Sabaté-Bel y J. Sánchez-García. 1993. Canarias. Economía, ecología y medio Ambiente. San Cristóbal de La Laguna: Francisco Lemus Editor, 361 pp.

Aguilera Klink, F., A. Brito-Hernández, C. Castilla-Gutiérrez, A. Díaz-Hernández, J.M. Fernández-Palacios, A. Rodríguez-Rodríguez, F. Sabaté-Bel y J. Sánchez-García. 1993. Canarias. Economía, ecología y medio Ambiente. San Cristóbal de La Laguna: Francisco Lemus Editor, 361 pp.

Alves, F.M.A., Chícharo, L.M., Serrao, E., & Abreu, A.D. 2001. Algal cover and sea urchin spatial distribution at Madeira Island (NE Atlantic). Sci Mar 65:383–392.

Alves, F.M.A., Chícharo, L.M., Serrao, E., & Abreu, A.D. 2001. Algal cover and sea urchin spatial distribution at Madeira Island (NE Atlantic). Sci Mar 65:383–392.

Barquín, J., Gil-Rodríguez, M.C., Del Arco Aguilar, M.J., González, G., Domínguez-Álvarez, S., González, R., Aldanondo-Ariztizabal, N., Rodríguez, M., Cruz-Reyes, M.A., Herrera-López, G., Sancho, A,, Cruz, T., O'dwyer, J., Martin, L., & Cansado, S. 2004. La Cartografía bionómica del litoral de Tenerife: resultados preliminares. XIII Simposio Ibérico de Estudio del Bentos Marino. Libro de resúmenes.

Bas, C,. J.J. Castro, V. Hernández-García, J.M. Lorenzo, T. Moreno, J.G. Pajuelo y A.G. Ramos. 1995. La pesca en Canarias y área de influencia. Ediciones del Cabildo Insular de Gran Canaria. Madrid. 331 pp.

Bjordal, A. 2002. The use of technical measures in responsible fisheries: Regulation of fishing gear. In: Cochrane, K.L. (ed.), A fishery manager's guidebook - Management measures and their application; Chapter 2. FAO Fisheries Technical Paper, 424.

Blackwood, J.C., Hastings, A., & Mumby, P.J. 2012. The effect of fishing on hysteresis in Caribbean coral reefs. Theoretical Ecology, 5(!):105-114.

Botsford, L.W., Castilla, J.C. & Peterson, C.H. 1997. The management of fisheries and marine ecosystems. Science, 277(5325):509-515.

Brey, T. (1991) The relative significance of biological and physical disturbance: an example from intertidal and subtidal sandy bottom communities. Estuarine and Coastal Shelf Science 33, 339-60.

Brito, A., Hernández, J.C., Falcón, J.M., García, N., González-Lorenzo, G., Gil-Rodríguez, M.C., Cruz, A., Herrera, G., Sancho, A., Clemente, S., Cubero, E., Girad, D., & Barquín, J. 2004. El erizo de Lima (*Diadema antillarum*): una especie clave en los fondos litorales rocosos de Canarias. Makaronesia, 6:68–86.

Castro, J.J., & Hernández-García, V. 2012. Caracterización del poder de pesca de la flota artesanal canaria, con especial referencia a las fracción con eslora superior a 12 m, y análisis del estado de los recursos que explota. Informe Técnico, Viceconsejería de Pesca, Gobierno de Canarias.

Castro, J.J., E. Divovich, A. Delgado de Molina-Acevedo, A. Barrera-Luján, R. Riera. 2019. Reconstruction of marine small-scale fisheries captures in the Canary Islands (NE Atlantic Ocean) from 1950 to 2010. Scientia Marina, 83(1). https://doi.org/10.3989/scimar.04837.18A

Charles, A.T. 2002. Use rights and responsible fisheries: Limiting access and harvesting through rights-based management. In: Cochrane, K.L. (ed.), A fishery manager's guidebook - Management measures and their application; Chapter 6. FAO Fisheries Technical Paper, 424.

Clemente, S., Hernández, J.C., & Brito, A. 2009. Evidence of the top–down role of predators in structuring sublittoral rocky-reef communities in a marine protected area and nearby areas of the Canary Islands. ICES J Mar Sci 66:64–71.

Coll, M., Carreras, M., Ciércoles, C., Cornax, M.-J., Gorelli, G., Morote, E., & Sáez, R. (2014). Assessing Fishing and Marine Biodiversity Changes Using Fishers' Perceptions: The Spanish Mediterranean and Gulf of Cadiz Case Study. PLoS ONE 9(1): e85670. doi:10.1371/journal.pone.0085670.

Collins, K.J., Suonpää, A.M., & Mallinson, J.J. 2010. The impacts of anchoring and mooring in seagrass, Studland Bay, Dorset, UK . Underwater Technology The International Journal of the Society for Underwater 29(3):117-123.

Couce-Montero, L., Christensen, V., Bilbao-Sieyro, A., Pérez-González, Y., Jiménez-Alvarado, D., Castro, J.J. 2019. Article Temporal and spatial predictions of effect of alternative fishing policies for the Gran Canaria marine ecosystem. Journal of Fish Biology. DOI: 10.1111/jfb.14012

DeGroot, S.J. (1984). The impact of bottom trawling on benthic fauna of the North Sea. Ocean Manage 9: 177-190.

FAO, 2011. Fisheries management. 4. Marine protected areas and fisheries. FAO Technical Guidelines for Responsible Fisheries, 4 (Suppl. 4). 210 pp.

García-Cabrera, C. 1970. La pesca en Canarias y Banco Sahariano. Consejo Económico Sindical Interprovincial de Canarias. 176 pp.

González, J.A. 2008. Memoria científico-técnica sobre el Estado de los recursos pesqueros de Canarias (REPESCAN). Instituto Canario de Ciencias Marinas. Telde (Las Palmas), 210 pp.

Hernández, J.C., Brito, A., Cubero, E., García, N., Girard, D., González-Lorenzo, G., & Falcón, J.M. 2006. Temporal patterns of larval settlement of *Diadema antillarum* (Echinodermata: Echinoidea) in the Canary Islands using an experimental larval collector. Bull. Mar. Sci., 78:271–279.

Hernández, J.C., Clemente, S., Sangil, C., & Brito, A. 2008a The key role of the sea urchin Diadema aff. antillarum throughout the Canary Islands (eastern subtropical Atlantic) in controlling macroalgae assemblages: an spatio-temporal approach. Mar Environ Res 66:259–270

Hernández, J.C., Clemente, S., Sangil, C., & Brito, A. 2008a. Actual status of the sea urchin Diadema aff. antillarum populations and macroalgal cover in marine protected areas compared to a highly fished area (Canary Islands-eastern Atlantic Ocean). Aquat Conser: Mar Freshw Ecosyst 18:1091–1108.

Hernández, J.C., Clemente, S., Sangil, C., Brito, A. 2008. The key role of the sea urchin *Diadema* aff. *antillarum* throughout the Canary Islands (eastern subtropical Atlantic) in controlling macroalgae assemblages: an spatio-temporal approach. Mar Environ Res, 66:259–270

Hernández, J.C., Clemente, S., Tuya, F., Pérez-Ruzafa, A., Sangil C., Moro-Abad, L., & Bacallado-Arenaga, J.J. 2013. Echinoderms of the Canart Islands, Spain. In: J.J. Alvarado and F.A. Solís-María (eds.), Echinoderm Research and Diversity in Latin America, Chapter 15, 471-510 pp. Springer-Verlag, Berlin, Heidelberg.

Hernández, J.C., Clemente, S., Tuya, F., Pérez-Ruzafa, A., Sangil C., Moro-Abad, L., & Bacallado-Arenaga, J.J. 2013. Echinoderms of the Canart Islands, Spain. In:

J.J. Alvarado and F.A. Solís-María (eds.), Echinoderm Research and Diversity in Latin America, Chapter 15, 471-510 pp. Springer-Verlag, Berlin, Heidelberg.

Hiddink, J.G., Hutton, T., Jennings, S., & M.J. Kaiser. 2006. Predicting the effects of area closures and fishing effort restrictions on the production, biomass, and species richness of benthic invertebrate communities. ICES Journal of Marine Science, 63(5):822-830.

Hiddink, J.G., Jennings, S., Sciberras, M., Szostel, C.L., Hughes, K.M., Ellis, N., Rijnsdorp, A.D., McConnaughey, Mazor, T., Hilborn, R., Collie, J.S., Pitcher, C.R., Amoroso, R.O., Parma, A.M., Suuronen, P., & Kaiser, M.J. 2017. Global analysis of depletion and recovery of seabed biota after bottom trawling disturbance. Proc. Natl. Acad. Sci. USA, 114(31):8301-8306.

Hilborn, R., Stokes, K., Maguire, J-J., Smith, T., Botsford, L., Mangel, M., Orensanz, J., Parma, A., Rice, J., Bell, J., Cochrane, K., Garcia, S., Hall, S.J., Kirkwood, K., Sainsbury, K., Stefansson, G., & Walters, C. 2004. When can marine reserves improve fisheries management? Ocean & Coastal Management, 47(3-4):197-205.

Hoagland, P., Sumaila, U.R., & Farrow, S. 2001. Marine protected areas, Vol. 2. In. Encyclopedia of ocean sciences, 2nd ed. Elsevier, 672-677 pp.

Korpinen, S., Zweifel, U.L., Bastardie, F., van Denderen, D., Hoppe, K., Jonsson, P., Kauppila, P., Milardi, M., Nielsen, R., Nilsson, H., Norén, K., Nygård, H., Sköld, M., Valanko, S., & Zettler, M. 2018. "Estimating physical disturbance on seabed – Baltic Sea Environment Proceedings No. 164. http://www.helcom.fi/Lists/Publications/BSEP164_Disturbance_on_seabed.pdf

Lambert, G.I., Jennings, S., Kaise, M.J., Davies, T.W., & Hiddink, J.G. 2014. Quatifying recovery rates and resilience of seabed habitat impacted by bottom fishing. Journal of Apllied Ecology, 51:1326-1336.

Lawrence JM (1975) On the relationships between marine plants and sea urchins. Oceanogr Mar Biol Ann Rev 13:213–286.

MAGRAMA, 2015. Orden AAA/2536/2015, de 30 de noviembre, por la que se regulan las artes y modalidades de pesca marítima y se establece un plan de gestión para los buques de los censos del Caladero Nacional Canario. BOE 257, de 1 de diciembre de 2015. Sec. I.: 113420-113435.

Mesnildrey, L., Gacuel, D., & Le Pape, O. 2013. Integrating marine protectedareas in fisheries management systems: some criteria for ecological effciency. Aquatic Living Resources, 26:159-170.

MIMEA (Ministerio de Medio Ambiente). 2008. Guía metodológica para la instalación de arrecifes artificiales. Ministerio de Medio Ambiente. Gobierno de España.

Morato, T. 2012. Description of environmental issues, fish stocks and fisheries in the EEZs around the Azores and Madeira. Resquest for Services Commitment no. SI2.615631. Commission Decision 2005/626/EC.

Neubauer, P., Jensen, O.P., Hutchings, J.A., Baum, J.K. 2013. Resilience and recovery of overexploited marine populations. Science, 340:347-349.

NOAA (Marine Debris Program). 2015. Report on the impacts of "ghost fishing" via derelict fishing gear. Silver Spring, MD. 25 pp.

Ojamaa, P. 2015. Fisheries in Azores. Directorate General for International Policies. Policy Department B- Structural and Cohesion Policies- Fisheries. European Parliament. IP/B/PECH/INT/2015-02- PE540.355

Ortega, L., Tuya, F., & Haroun, R.J. 2009. El erizo de mar Diadema antillarum Phillipi, 1845 influye sobre la diversidad y composición de la comunidad de mega-invertebrados vágiles en fondos rocosos del Archipiélago Canario. Revista de Biología Marina y Oceanografía, 44(2):489-495.

Pham, C.K., Canha, A., Diogo, H., Pereira, J.G., Prieto, R., & Morato, T. 2013. Total marine fishery catch for the Azores (1950-2010). ICES Journal of Marine Science, 70(3):564-577.

Pham, C.K., Diogo, H., Menezes, G., Porteiro, F., Braga-Henriques, A., Vandeperre, F., & Morato, T. 2014. Deep-water longline fishing has reduced impact on vulnerable marine ecosystems. Scientif Reports, 4(4837)

Pham, C.K., Gomes-Pereira, J.N., Isidro, E.J., Santos, R.S., & Morato, T. 2013. Abundance of litter on Condor seamount (Azores, Portugal, Northeast Atlantic). Deep Sea Research Part II: Topical Studies in Oceanography, 98:204-208.

Pikitch, E.K. 2012. The risks of overfishing. Science, 338:474-475.

Pope, J. 2002. Imput and output controls: The practice of fishing effort and catch management in responsible fisheries. In: Cochrane, K.L. (ed.), A fishery manager's guidebook - Management measures and their application; Chapter 4. FAO Fisheries Technical Paper, 424.

Ramírez, R., Tuya, F., & Haroun, R. 2008. El intermareal canario. Poblaciones de lapas, burgados y cañadillas. BIOGES, Universidad de Las Palmas de Gran Canaria.

Riera, R., Becerro, M.A., Stuart-Smith, R.D., Delgado, J.D., & Edgar, G.J. 2014. Out of sight, out of mind: Threats to the marine biodiversity of the Canary Islands (NE Atlantic Ocean). Marine Pollution Bulletin. 86(1-2):9-18.

Riera, R., Pérez. O., Álvarez, O., Simón, D., Díaz, D., Monterroso, O., & Núñez, J. 2016. Clear regression of harvested intertidal mollusks. A 20-year (1994-2014) comparative study. Marine Environmental Research, 113:56-61.

Rising, J., & Heal, G. (2014). Global benefits of Marine Protected Areas. National Bureau of Economic Research. Working Paper 19982. https://www.nber.org/papers/w19982.pdf

Sampaio, I., A. Braga-Henriques, C. Pham, O. Ocaña, V. de Matos, T. Morato, F.M. Porteiro. 2012. Cold-water corals landed by bottom longline fisheries in the Azores (north-eastern Atlantic). Journal of the Marine Biological Association of the United Kingdom.

Sangil C, Afonso-Carrillo J, Sansón M (2006b) Estructura y composición de las comunidades algales sublitorales de los fondos rocosos de la isla de El Hierro (islas Canarias). Proc XIV Simp Ibérico de Estudios en Biol Mar, Barcelona.

Sangil C, Sansón M, Afonso-Carrillo J (2006a) Distribución de las comunidades algales de sustratos rocosos en la isla de La Palma (islas Canarias). Proc XIV Simp Ibérico de Estudios en Biol Mar, Barcelona.

Sangil, C., Martín-García, L., Hernández, J.C., Concepción, L., Fernández, R., & Clemente, S. 2013. Impacts of fishing and environmental factors driving changes on littoral fish assemblages in a subtropical oceanic island. Estuarine, Coastal and Shelf Science, 128:22-32.

Sangil, C., Sansón, M., Afonso-Carrillo, J., Herrera, R., Rodríguez, A., Martín-García, L., Díaz-Villa, T., 2012. Changes in subtidal assemblages in a scenario of warming: proliferations of ephemeral benthic algae in the Canary Islands (eastern Atlantic Ocean). Mar. Environ. Res. 77, 120–128.

Shon, S., Delgado, J.M., Morato, T., Pham, C.K., Zylich, K., Zeller, D., & D. Pauly. 2015. Reconstruction of marine fisheries catches for Madeira Island, Portugal, from 1950-2010. Fisheries Centre. The University of British Columbia. Working Paper Series, WP 2015-52.

Silva, M.A., M. Machete, D. Reis, M. Santos, R. Prieto, C. Dâmaso, J.G. Pereira, R.S. Santos, 2011. A review of interactions between cetaceans and fisheries in the Azores. Aquatic Conservation: Marine and Freshwater Ecosystems 21: 17–27.

Stobart, B., Alvarez-Barastegui, D., & Goñi, R. 2013. Effect of habitat patchiness on the catch rates of a Mediterranean coastal bottom long-line fishery. Fisheries Research, 129-130:110-118.

Taylor, B.M., & McIlwain, J.L. (2010). Beyond abundance and biomass: effects of marine protected areas on the demography of a highly exploited reef fish. Marine Ecology Progress Series, 411:243-258.

Tserpes, G., Nikolioudakis, N., Maravelias, C., Carvalho, N., & Merino, G. 2016. Viability and management targets of Mediterranean demersal fisheries. The case of the Aegean Sea. PLoS One, 11(12):e0168694

Tuya, F., Boyra, A., & Haroun, R. 2004a. Blanquizales en Canarias. La explosión demográfica del erizo *Diadema antillarum* en los fondos rocosos de Canaria. BIOGES: Centro de Investigación en Biodiversidad y Gestión Ambiental. 34 pp.

Tuya, F., Boyra, A., Sánchez-Jerez, P., Barbera, C., & Haroun, R.J. 2004b. Can one species determine the structure of the benthic community on a temperate rocky reef? The case of the long-spined sea-urchin *Diadema antillarum* (echinodermata: echinoidea) in the Easter Atlantic. Hydrobiology, 519:211–214

Tuya, F., Cisneros-Aguirre, J., Ortega, L., Haroun, R.J. 2007 Bathymetric segregation of sea urchins on reefs of the Canarian Archipelago: role of flow-induced forces. Estuar. Coast. Shelf Sci., 73:481–488.

Unsworth, R.F., Williams, B., Jones, B.L., & Cullen-Unsworth, L.C. 2017. Rocking the boat: damage to eelgrass by swinging boat moorings. Frontiers in Plant Science, 8:1309 - doi: 10.3389/fpls.2017.01309.

Worm, B., Hilborn, R., Baum, J.K., Branch, T.A., Collie, J.S., Costello, C., Fogarty, M.J., Fulton, E.A., Hutchings, J.A., Jennigs, S., Jensen, O.P., Lotze, H.K., Mace, P.M., McClanahan, T.R., Minto, C., Palumbi, S.R., Parma, A.M., Ricard, D., Rosenberg, A.A., Watson, R., & Zeller, D. 2009. Rebuilding global fisheries. Science, 325(5940):578-585.

Zeller, D., & Reinert, J. 2004. Modelling spatial closures and fishing effort restrictions in the Faroe Islands marine ecosystem. Ecological Modelling, 172(2-4):403-420.

8 Analysis >> QD1 & QD6 Benthic habitats (relating to Descriptors 1 & Descriptor 6)

QD1& QD6 Benthic habitats (relating to Descriptors 1 and 6)							
QD	**Criteria (element)**	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
QD1&QD6	The extent of loss of the habitat type, resulting from anthropogenic pressures, does not exceed a specified proportion of the natural extent of the habitat type in the assessment area.	D6C4	yes	need further research	need further research	need further research	need further research
	The extent of adverse effects from anthropogenic pressures on the condition of the habitat type, including alteration to its biotic and abiotic structure and its functions (e.g. its typical species composition and their relative abundance, absence of particularly sensitive or fragile species or species providing a key function, size structure of species), does not exceed a specified proportion of the natural extent of the habitat type in the assessment area.	D6C5	yes	need further research	need further research	need further research	need further research

8.1 D6C4, D6C5

As previously mentioned in the document, information on the types of habitats in the Macaronesian region is scarce because only information of the habitats included in the first 50 meters of depth is available, so it is impossible for us to evaluate the D6C4 criterion. Regarding criterion D6C5, alterations on biotic and abiotic structure and functions of the habitat types have mainly been discussed in descriptors 1 and 6, however it is not possible to know the spatial extent of the aforementioned impacts. Therefore, our suggestion is that more information is needed to properly assess these criteria.

9 Analysis >> DC7 Hydrographical conditions

DC7 Permanent alteration of hydrographical conditions does not adversely affect marine ecosystems							
QD	**Criteria (element)**	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
QD7	Hydrographical changes to the seabed and water column (including intertidal areas).Spatial extent and distribution of permanent alteration of hydrographical conditions (e.g. changes in wave action, currents, salinity, temperature) to the seabed and water column, associated in particular with physical loss (7) of the natural seabed.	D7C1 — Secondary	yes	equal	yes	no	not applicable
	Spatial extent of each benthic habitat type adversely affected (physical and hydrographical characteristics and associated biological communities) due to permanent alteration of hydrographical conditions.	D7C2 — Secondary	yes	equal	yes	no	not applicable

9.1 D7C1, D7C2

The OSPAR and Barcelona conventions define artificial reefs as: 'an artificial reef is a submerged structure placed deliberately on the ocean floor to mimic some of the characteristics of a natural reef. They may be partially exposed in some tidal states'.

Those responsible for the implementation of the Marine Strategies in Madeira and Azores have not described the possible influence of fishing on the hydrographic conditions in these regions (Regional Secretariat for the Environment and Two Naturais Resources, 2014, Regional Secretary for Naturais Resources, 2014). In the Azores the sinking of wrecks is explained (which could be extended to artificial reefs), however the extent of this influence is very small in the coastal context of the Azores (Secretaria Regional dos Recursos Naturais, 2014). In the Canary Islands, in the framework of the reports on the DMEM, according to Lloret Capote, del Barrio Alvarellos and Moreno Aranda (2012) 'the artificial reefs and wrecks sunk for this purpose constitute obstacles that, depending on the place where they are located and their distribution density, can cause modifications in the local system of currents, altering, therefore, the hydrodynamic conditions of the medium'. However, when they deal in detail with D7, they point out that artificial reefs cause very little hydrodynamic distortion (González-Pola, Vélez-Belchí and Izquierdo, 2012).

There are many designs of artificial reefs that, depending on their destination of use, can be categorized as those destined to: (i) act on the physical environment (coastal protection, tourism and leisure, creation of anchoring zones, protection of marine infrastructures), (ii) act on biota, such as fisheries management (protection, production / concentration), biofilters, mariculture, etc. and (iii) other uses, such as recreational diving, scientific research, etc. According to the Ministerio de Medio Ambiente (2008) 'in the case of artificial reefs destined mainly for the protection of the physical environment by dissipating the energy of the waves, it will be necessary to carry out a detailed study of wave propagation since this type of reefs affect directly to the hydrodynamics of the area'. However, in the reefs destined to the fishery management it is only necessary to carry out a previous basic study (analysis of currents) to demonstrate the non-alteration of the hydrographic conditions. In this sense, except for the case of the reef-dikes, the anchoring depth must be sufficient to allow navigation. It is recommended to leave at least one layer of water 15 meters above the reef at the equinoctial low tide. In places where the possibility of passage of vessels with drafts exceeding 10 meters, the minimum water layer is established in 25 meters and so on' (Ministerio de Medio Ambiente, 2008). Regarding the specific content on the scope of the environmental impact study, it is necessary to take into account the legislation that exists in this matter for each region, but which we consider to be outside the scope of this report. As an example, it is recommended to review the 'Guía metodológica para la instalación de arrecifes artificiales' (Ministerio de Medio Ambiente, 2008)'.

9.2 References

González-Pola, C., Vélez-Belchí, P. and Izquierdo, P. (2012) *Estrategia Marina. Demarcación Marina Canaria Parte IV. Descriptores del buen estado ambiental. Descriptor 7: condiciones hidrográficas. Evaluación inicial y buen estado ambiental*. Madrid. Available at: https://www.miteco.gob.es/es/costas/temas/proteccion-medio-marino/IV_D7_Canarias_tcm30-130941.pdf.

Lloret Capote, A., del Barrio Alvarellos, I. and Moreno Aranda, I. M. (2012) *Estrategia Marina. Demarcación Marina Canaria. Evaluación inicial. Parte II: Análisis de presiones e impactos*. Madrid. Available at: http://www.mapama.gob.es/es/costas/temas/proteccion-medio-marino/II_Analisis_Presiones_Canarias_tcm30-130933.pdf.

Ministerio de Medio Ambiente (2008) *Guía metodológica para la instalación de arrecifes artifi ciales*. Madrid. Available at: https://www.miteco.gob.es/es/costas/temas/proteccion-medio-marino/Guía metodológica para la instalación de arrecifes artificiales_tcm30-157012.pdf.

Secretaria Regional do Ambiente e dos Recursos Naturais (2014) *Estratégia Marinha para a subdivisão da Madeira. Diretiva Quadro Estratégia Marinha*. Available at: https://www.dgrm.mm.gov.pt/documents/20143/43971/RelatorioInicial_MAD_FINAL__2014.pdf/0f2783be-bf81-5d26-83cd-15830cff998c.

Secretaria Regional dos Recursos Naturais (2014) *Estratégia Marinha para a subdivisão dos Açores. Diretiva Quadro Estratégia Marinha*. Available at: https://www.dgrm.mm.gov.pt/documents/20143/43971/RelatorioInicial_AZO_FINAL_2014+%282%29.pdf/a0b43add-3ba3-6e52-e660-11189ef15ad7.

10 Analysis >> DC8 Contaminants

DC8 Concentrations of contaminants are at levels not giving rise to pollution effects							
QD	**Criteria (element)**	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
QD8	Concentrations of contaminants (ubiquitous persistent, bioaccumulative and toxic substances - Article 8a(1)(a) of Directive 2008/105/EC) do not exceed the established (WFD) threshold values in water, sediment or biota.	D8C1	No data & information, needed further research.				
	The health of species and the condition of habitats (such as their species composition and relative abundance at locations of chronic pollution) are not adversely affected due to contaminants including cumulative and synergetic effects.	D8C2	No data & information, needed further research.				
	The spatial extent and duration of significant acute pollution events (Dicharging oil and noxious liquid substances - MARPOL 73/78Article 2(2) of Directive 2005/35/EC) are minimised.	D8C3	No data & information, needed further research.				
	The adverse effects of significant acute pollution events on the health of species and on the condition of habitats (such as their species composition and relative abundance).	D8C4	No data & information, needed further research.				

10.1 D8C1, D8C2

The D8C1 and D8C2 criteria of QD 8 are dedicated to prolonged or chronic processes; while the D8C3 and D8C4 are related to punctual or acute events of contamination, such as the accidental sinking of a ship or runoff. In addition, for both cases (chronic or punctual), one of the criteria is dedicated to the evaluation of pollutants in the environment; and the other, how these substances affect the marine environment that contains them (see Decision 2017/848). The following sections will deal with the criteria associated with prolonged or chronic processes, due to the characteristics of the sector evaluated (artisanal fishing).

The marine strategy (Decision 2017/848) determines that in coastal and territorial waters, the concentrations of the substances in Annex IA of the directive on pollutants (Directive 2008/105/EC) and Annex VIII (with 12 types of substances) of the directive of water (Directive 2000/60/CE), will not exceed the established thresholds. When the concentration for which these thresholds have been established is measured in another matrix (compartment of the aquatic environment, eg, water, sediment or biota), the member states will establish the corresponding values through cooperation between regions and subregions, if necessary (Decision 2017/848). The same shall apply for the relevant contaminants not included in these annexes, in the types of water cited, or beyond, for any substance that produces important effects (Decision 2017/848).

The Directive 2000/60/CE establishes as conditioner of the "Very Good State" physicochemical of the coastal waters that the concentrations of specific synthetic contaminants are close to 0 or less than the limits of detection; and for specific non-synthetic products, the concentrations must be within the range considered as usual or unaltered conditions (see definitions in its Annex V).

At this point, could artisanal vessels be considered sources of pollutants? A priori, it can be expected that these fleets are not a significative source to the total pollution contributed to the marine environment in each of the three archipelagos, due to the low number of craft boats in comparison with other sectors or larger vessels associated with others sectors (maritime transport, pleasure craft, ships engaged in coastal passenger movements, etc. see section of quality descriptor 2); and also, at least in the Canary Islands, due to the greater load of pollutants associated with terrestrial sources (GESPLAN, 2012) (eg, discharges of wastewater, hydrocarbons, pesticides, etc.); this circumstance also seems important in Madeira (see DQEM, 2014). In fact, in Spain, the marine strategy does not include artisanal fishing as a relevant source of pollution (Lloret-Capote et al., 2012, DMEM, 2016). And in the case of the Canary Islands, the cumulative analysis of the pressures carried out within the framework of the marine strategy refers to the fact that in the Canary Islands there was only available information at the time about liquid controlled discharges (DMEM, 2016). In relation to this, the updated hydrological plans of the Canary Islands could improve these lagoons (Directive 2000/60/CE).

In addition, there are few studies in fisheries that include other impacts of this activity, on different aspects that are not those of the species captured or their ecosystem; such as the materials and energy of manufacturing and maintenance, the provision of gear (and ghost fishing), the consumption of fuel, ice, other substances; and the cost of the catches and their commercialization (Abdou et al., 2018).

Even so, artisanal vessels can produce different types of pollutants due to fuel losses, greenhouse gas emissions, heavy metals contained in batteries (when thrown into the sea), marine debris, antifouling paints, etc. Perhaps the antifouling paints have the most relevant role in terms of sources of pollution of these ships due to the continued contribution of their particles to the marine environment (associated with the maintenance of the hull of the vessels); and in addition, because they are recognized as sources of heavy metals and biocides (Soroldoni et al., 2018). Up to 2017, some 23 compounds were used as biocides in antifouling paints (eg, Irgarol 1051, Diuron, Chlorothalonil, Dichlofuanido, TCMTB and DCOIT) (Soroldoni et al., 2017). The biocides are in group 9 of Annex VIII of the Directive 2000/60/CE. And the diuron is included in Annex AI of the Directive 2008/105/EC, this compound is also used in agriculture and wood preservation (Sánchez-Rodríguez, 2011).

In a study carried out in an estuary in southeastern Brazil, it has been suggested that fishing ports could be potential sources of anti-fouling paint particles due to their high consumption of these materials and their higher frequency of maintenance operations compared to marine or relatively close shipyards (Soroldoni et al., 2018). However, the frequency and manner of developing maintenance operations, the hydrographic conditions and the fact that the fishermen themselves visit other types of ports, such as sports harbours and shipyards, for these operations, determine the great difficulty involved in confirming these sources (Soroldoni et al., 2018). To all this, the contributions of terrestrial polluting sources are confused; a circumstance that further complicates the determination of sources of heavy metals or biocides (Soroldoni et al., 2018). Even so, in coastal systems, a biocide has been observed that is closely associated with anti-fouling paints, the DCOIT; which has become an emerging biocide due to its global use in antifouling paints (Chen & Lam, 2017). Although this is recognized as a biocide of rapid degradation, its continuous contribution at the local level implies that it acts as a pseudo-persistent substance (Soroldoni et al., 2018). And the systematic evaluation of the risk of the impact that this compound could have on the marine ecosystem has already been recommended (Chen & Lam, 2017, Soroldoni et al., 2018) due to its relatively high toxicity, the dependence on the environmental conditions of its supposed rapidity of degradation, and the accumulation of its particles in the sediments (Chen & Lam, 2017).

In the Canary Islands, it has been observed the use of an anti-fouling paint by some artisanal fishermen that contains DCOIT (this is an active principle of the antifouling agent called SeaNine 211, Chen & Lam, 2017). In relation to its impact, significant effects on biota have been obtained by contact with DCOIT, with relatively low concentrations (Chen & Lam, 2017). These effects could be associated with specific periods, such as the summer maintenance of large yachts (eg, Catalan ports during the summer of 1999, Martínez et al., 2001).

In this way, this type of paintings could produce important effects at the local level (for example in fishing ports, sports docks or important ports that have shipyards). Sánchez-Rodríguez (2011) performed an evaluation of Diuron and 3 more biocides: Irgarol 1051, TCMTB and dichlofluanido; in the water of 5 ports (fishing, merchant and sports) of the island of Gran Canaria (Canary Islands). One of the conclusions obtained is that marinas have higher concentrations of some biocides than commercial and fishing ports, due to their lower water exchange and higher vessel density; as well as the uncertainty associated with the sources of these pollutants (Sánchez-Rodríguez, 2011). However, the evaluation of the ecological risk carried out by these authors on diuron and irgarol 1051 (it was not necessary for the others biocides), determined that these risks were acceptable; although again, the uncertainty associated with this evaluation is described and caution is advised (Sánchez-Rodríguez, 2011). As an example (in terms of natural sources of pollutants and uncertainty), one issue to be taken into account in the Azores would be the possible natural enrichment of heavy metals by volcanic contributions in hydrothermal systems (Raimundo et al., 2013).

In Amara et al (2018), they review the effects that certain biocides produce in different organisms and the concentrations on different magnitudes related to the toxicity produced. Irgarol 1051 is very effective against algae and produces deadly effects on marine life, the DCOIT produces effects on a wide spectrum of marine organisms although its effects are not chronic (these effects could be it, if its contribution were continuous; Soroldoni et al., 2018), copper-based compounds affect molluscs and algae, and diuron is effective on microalgae species and certain bacteria (Amara et al., 2018).

To conclude, it is not possible to assume that the artisanal fleet does not affect the pollution in some insular port, in this way, the sector could also affect water, sediments and the different organisms that are part of the local biological community. But it is important to note that an improvement in the maintenance processes of these vessels (in case these improvements are not already being applied) can prevent the continuous contribution of paint particles to marine systems while at the same time clarifying the gaps on the concentrations and sources of pollutants (Soroldoni et al., 2018). For this, it is advisable to systematically evaluate the risk that contaminants could exert on marine ecosystems due to their indiscriminate use (Soroldoni et al., 2018). On the other hand, it is not considered probable that the artisanal fleet causes acute effects of pollution in the environment. However, if other sources of pollutants (point or chronic) produce some effect (a priori negative) on the marine ecosystem; this effect will be able to affect to local fisheries (eg, by loss of more suitable areas for fishing).

10.2 References

Abdou, K., Gascuel, D., Aubin, J., Salah Romdhane, M., Rais-Lasram, F.B. & Le Loc'h, F. (2018). Environmental life cycle assessment of seafood production: A case study of trawler catches in Tunisia. *Science of the Total Environment* 610–611: 298–307.

Amara, I., Miled, W., Slama, R.B. & Ladhari, N. (2018). Antifouling processes and toxicity effects of antifouling paints on marine environment. A review. *Environmental Toxicology and Pharmacology* 57: 115–130.

Chen, L. & Lam, J.C.W. (2017). SeaNine 211 as antifouling biocide: A coastal pollutant of emerging concern. *Journal of Environmental Science* 61: 68 – 79.

Directive 2000/60/CE. (2000). Directive 2000/60/CE of the European Parliament and the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. DO L 327 de 22.12.2000. CONSOLIDATED (20/November/2014).

Directive 2008/105/CE. (2008). Directive 2008/105/CE of the European Parliament and the Council of of 16 December 2008 on environmental quality standards in the field of water policy, amending and subsequently repealing Council Directives 82/176/EEC, 83/513/EEC, 84/156/EEC, 84/491/EEC, 86/280/EEC and amending Directive 2000/60/EC of the European Parliament and of the Council. CONSOLIDATED (13/March/2013).

Decision 2017/848. (2017). Commission Decision (UE) 2017/848 of 17 May 2017 laying down criteria and methodological standards on good environmental status of marine waters and specifications and standardised methods for monitoring and assessment, and repealing Decision 2010/477/EU (Text with EEA relevance).

DMEM. (2016). Estrategias Marinas de España. Estudio Ambiental Estratégico. Evaluación Ambiental Estratégica de las estrategias marinas. Madrid. Available at: https://www.miteco.gob.es/es/costas/temas/proteccion-medio-marino/estrategias-marinas/em_programas_medidas-lst.aspx

DQEM (2014). Secretaria Regional do Ambiente e dos Recursos Naturais. Estratégia Marinha para a subdivisão da Madeira. Diretiva Quadro Estratégia Marinha. Available at:https://www.dgrm.mm.gov.pt/documents/20143/43971/RelatorioInicial_MAD_FINAL__2014.pdf/0f2783be-bf81-5d26-83cd-15830cff998c.

GESPLAN. (2012). Estudio ambiental de caracterización, diagnóstico y pronóstico de la orla litoral. Proyecto LITOMAC. PCT-MAC 2007-2013. Gobierno de Canarias y Unión Europea (FEDER). 59 pp.

Lloret-Capote, A., del Barrio Alvarellos, I. & Moreno Aranda, I. M. (2012). Estrategia Marina. Demarcación Marina Canaria. Evaluación inicial. Parte II: Análisis de presiones e impactos. Madrid. 130 pp. Available at: http://www.mapama.gob.es/es/costas/temas/proteccion-medio-marino/II_Analisis_Presiones_Canarias_tcm30-130933.pdf.

Martínez, K., Ferrer, I., Fernández-Alba, A.R., Marcé, R.M., Borrull, F., Barceló, D. (2001). Occurrence of antifouling biocides in the Spanish Mediterranean marine environment. *Environ. Toxico.*22: 543–552.

Raimundo, J., Vale, C., Caetano, M., Giacomello, E. & Menezes, G.M. (2013). Natural trace element enrichment in fishes from a volcanic tectonically active region (Azores archipelago). *Deep-Sea Research II*, 98 (A): 137-147.

Sánchez-Rodríguez, A., Sosa-Ferrera, Z., Santana-del Pino, A. & Santana-Rodríguez, J.J. (2011). Probabilistic risk assessment of common booster biocides in surface waters of the harbours of Gran Canaria (Spain). *Marine Pollution Bulletin* 62: 985–991.

Soroldoni, S., Abreu, F., Braga Castro, I., Andrei-Duarte, F. & Leães Pinho, G.L. (2017). Are antifouling paint particles a continuous source of toxic chemicals to the marine environment? *Journal of Hazardous Materials* 330: 76-82.

Soroldoni, S., Braga-Castro, I., Abreu, F., Andrei-Duarte, F., Brasil-Choueri, R., Olinto Moller O. Jr., Fillmann, G., Leaes-Pinho, G.L. (2018). Antifouling paint particles: Sources, occurrence, composition and dynamics. *Water Research* 137: 47-56.

11 Analysis >> QD9 Contaminants in fish and other seafood

QD9 Contaminants in fish and other seafood for human consumption do not exceed levels established by Union legislation or other relevant standards							
QD	**Criteria (element)**	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
QD9	The level of contaminants in edible tissues (muscle, liver, roe, flesh or other soft parts, as appropriate) of seafood (including fish, crustaceans, molluscs, echinoderms, seaweed and other marine plants) caught or harvested in the wild (excluding fin-fish from mariculture) does not exceed Regulation (EC) No 1881/2006	D9C1	No data & information, needed further research.				

11.1 D9C1

Currently, the European Union set in the Regulation 1881/2006 the máximum contents in pollutants in the edible products (It has been modified by Regulation 629/2008). To the QD9, maximun metal limits in fish meat, crustaceans, bivalve molluscs and cephalopods have set by Regulation 1881/2006 and its modifications.

As in QD8, in first, the artisanal fishing sector should not play a significant role as a source of contaminants that affect to the target species. Because both, the number and size of artisanal fishing boats, are less than in ships of other economic sectors. However, as in QD8, in QD9 it is necessary to evaluate this statement (i.e. in the assesment of the role that the biocide DCOIT, used in antifouling paints, it could play locally; Chen & Lam, 2017; Soroldoni et al., 2018).

In the last 20-30 years, in the Canary Islands have proposed various species as bioindicators of pollutants from the marine environment (and methods of analysis too), some of these species are usually consumed. For example, high concentrations of cadmium and lead have been found in the liver (but not in muscle) of *Sarpa salpa* (local common name: Salema) or *Trachurus picturatus* (local common name: Chicharro), and therefore only the consumption of their livers is discouraged (Afonso et al., 2017), and n-alkanes and hydrocarbons in marine gastropods (ie *Phorcus* spp.; previously, genus *Osilinus* spp.)(Peña et al., 1996). Heavy metals in the genus *Patella* spp. (gastropods, limpets) (Collado et al. 2006; Bergasa et al., 2007), or *Phorcus* spp. (Ramírez-Cañada, 2009; 2013). In other species that are not usually consumed, or to a lesser extent, too. For example, *Diadema africanum* to heavy metals (Hernández et al., 2010), *Arbacia* *lixula* to n-alkanes and polynuclear aromatic hydrocarbons (Peña-Méndez et al., 2000). These studies describe possible sources of pollution and the associated uncertainty to their confirmation.

On the other hand, in Madeira, the DQEM (2014) states that the frequency of exceeding mercury above regulatory levels varies between 2% in *Plesionika edwardsii* (Prawn of Madeira) and 78% in the deep-water fish *Epigonus telescopus* (common name in Madeira: Robaldo-preto and the Canary Islands: Candil). So the results provided in its marine strategy may suggest different measures of caution to propose.

Another important issue would be to evaluate the pollution caused by persistent chemical compounds, such as tributyltin (TBT), which due to its harmful effects on marine biota was restricted since the beginning of the 1990s (Amara et al., 2018). Studies carried out in the Canary Islands (Ramírez-Cañada, 2009) suggest that the mollusc locally known as Carñadilla (*Stramonita haemastoma*, little consumed) could be used as a bioindicator of organic tin (Sn) compounds, such as TBT used in the manufacture of antifouling paints. For a priori, its concentration should be higher in specimens that live near important maritime transit areas, such as port areas. For this, the imposition of the male sex in female individuals of this species for contaminating effects would be estimated (fact caused by the effect of these pollutants), which is known as the IMPOSEX index (Ramírez-Cañada, 2009).

One issue to consider in the Azores is the possible natural enrichment of heavy metals by volcanic contributions in hydrothermal systems (Raimundo et al., 2013). However, in the Canary Islands, the sources proposed for this contamination, could be agriculture, maritime transport, urban spills and losses of refineries or their facilities according to references described above. It is important denote the diffcult to determine the source of the pollution (Soroldoni et al., 2018) in shellfish and fish.

On the other hand, what is transcendental is that if other sources of pollutants are locally important, and influence both the marine benthic regional communities and the target species in the artisanal fisheries (i.e. *Phorcus* spp., *Patella* spp.); these pollutants could have a negative impact on the current and future developments of the sector (see annexe II, DMEM, 2016).

To conclude, it is not possible to assume that the artisanal fleet does not affect (or has affected, i.e. TBT) the pollution on some island port, mainly in a prolonged manner. In this way, the sector could also affect water, sediments and the different organisms of the local biological community. However, no study has been found that proves that any artisanal fishing port has a contaminating load that affects marine foods, and it will be a risk to human health. Although, due to the precautionary principle, and knowing the uncertainty of these analyzes (Sánchez-Rodríguez et al., 2011, Soroldoni et al., 2018) it would be necessary to establish a risk assessment for contamination in ports so that they are also included to those of a fishing nature.

On the other hand, the artisanal fleet is not considered likely to cause significant acute effects of pollution on the environment, compared to those that could be caused by vessels from other sectors (eg transport of chemical substances, including oil).

11.2 References

Afonso, A., Gutiérrez, A.J., Lozano, G., González-Weller, D., Rubio, C., Caballero, J.M., Hardisson, A. & Revert, C. (2017). Determination of toxic metals, trace and essentials, and macronutrients in Sarpa salpa and Chelon labrosus: risk assessment for the consumers. *Environ Sci Pollut Res* 24: 10557–10569. DOI 10.1007/s11356-017-8741-y

Amara, I., Miled, W., Slama, R.B. & Ladhari, N. (2018). Antifouling processes and toxicity effects of antifouling paints on marine environment. A review. *Environmental Toxicology and Pharmacology* 57: 115–130.

Bergasa, O., Ramíırez, R., Collado, C., Hernández-Brito,J.J., Gelado-Caballero, M.D., Rodríguez-Somozas, M. & Haroun, R.J. (2007). Study of metals concentration levels in Patella piperata throughout the Canary Islands, Spain. *Environ Monit Assess* 127:127–133.

Chen, L. & Lam, J.C.W. (2017). SeaNine 211 as antifouling biocide: A coastal pollutant of emerging concern. *Journal of Environmental Science* 61: 68 – 79.

Collado C.,Ramírez, R., Bergasa, O., Hernández-Brito, J.J., Gelado-Caballero, M.D. & Haroun, R.J. (2006). Heavy metals (Cd, Cu, Pb and Zn) in two species of limpets (Patella rustica and Patella candei crenata) in the Canary Islands, Spain. *In: Brebbia CA, Antunes do Carmo JS (eds)* Water Pollution VIII. Modelling, monitoring and management. WIT Press, Southampton, UK, pp 45-53.

DMEM. (2016). Estrategias Marinas de España. Estudio Ambiental Estratégico. Evaluación Ambiental Estratégica de las estrategias marinas. Madrid. Available at: https://www.miteco.gob.es/es/costas/temas/proteccion-medio-marino/estrategias-marinas/em_programas_medidas-lst.aspx

DQEM (2014). Secretaria Regional do Ambiente e dos Recursos Naturais. Estratégia Marinha para a subdivisão da Madeira. Diretiva Quadro Estratégia Marinha. Available at:https://www.dgrm.mm.gov.pt/documents/20143/43971/RelatorioInicial_MAD_FINAL__2014.pdf/0f2783be-bf81-5d26-83cd-15830cff998c.

Hernández, O.D., Gutiérrez, A.J., González-Welter, D., Lozano, G., Melón, E.G., Rubio, C. & Hardisson, A. (2010). Accumulation of Toxic Metals (Pb and Cd) in the Sea Urchin Diadema aff. antillarum Philippi, 1845, in an Oceanic Island (Tenerife, Canary Islands). *Enviromental Toxixology* 25(3): 227-233.

Peña, E., Conde, J.E. & Garcia-Montelongo, F.J. (1996). Evaluation of *Osilinus attratus* as a Bioindicator Organism to Monitor Oil Pollution in the Canary Islands. Arch. Environ. Contam. Toxicol. 31: 444-452--452 (1996).

Peña-Méndez, E.M., Astorga-Espafia, M.S., & Garcia-Montelongo, F.J. (2000). Interpretation of analytical data on n-alkanes and polynuclear aromatic hydrocarbons in *Arbacia lixula* from the coasts of Tenerife (Canary Islands, Spain) by multivariate data analysis. *Chemosphere*, Vol. 39, No. 13, pp. 2259-2270.

Regulation 1881/2006 (2006). Commission Regulation (EC) No 1881/2006 of 19 December 2006 setting maximum levels for certain contaminants in foodstuffs (Text with EEA relevance). DO L 364 de 20.12.2006. CONSOLIDATED (19/March/2018).

Raimundo, J., Vale, C., Caetano, M., Giacomello, E. & Menezes, G.M. (2013). Natural trace element enrichment in fishes from a volcanic tectonically active region (Azores archipelago). *Deep-Sea Research II*, 98 (A): 137-147.

Ramírez-Cañada, R. (2009). Moluscos gasterópodos como bioindicadores en el Archipiélago Canario: de procesos naturales a causas antropogénicas. Instituto Universitario de Sanidad Animal y Seguridad Alimentaria. Tesis doctoral de la Universidad de Las Palmas de Gran Canaria. Las Palmas de Gran Canaria, España. 303 pp.

Ramírez-Cañada, R. (2013). The gastropod Osilinus atrata as a bioindicator of Cd, Cu, Pb and Zn contamination in the coastal waters of the Canary Islands. *Chemistry and Ecology*. Vol. 29, No. 3, 208-220.

Sánchez-Rodríguez, A., Sosa-Ferrera, Z., Santana-del Pino, A. & Santana-Rodríguez, J.J. (2011). Probabilistic risk assessment of common booster biocides in surface waters of the harbours of Gran Canaria (Spain). *Marine Pollution Bulletin* 62: 985–991.

Soroldoni, S., Braga-Castro, I., Abreu, F., Andrei-Duarte, F., Brasil-Choueri, R., Olinto Moller O. Jr., Fillmann, G., Leaes-Pinho, G.L. (2018). Antifouling paint particles: Sources, occurrence, composition and dynamics. *Water Research* 137: 47-56.

12 Analysis >> QD10 Marine litter

QD10 Properties and quantities of marine litter do not cause harm to the coastal and marine environment							
QD	**Criteria (element)**	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
QD10	The composition, amount and spatial distribution of litter (excluding micro-litter, classified in the following categories: artificial polymer materials, rubber, cloth/textile, paper/cardboard, processed/worked wood, metal, glass/ceramics, chemicals, undefined, and food waste) on the coastline, in the surface layer of the water column, and on the seabed, are at levels that do not cause harm to the coastal and marine environment.	D10C1	Yes	Broader	Yes	Yes	Yes
	The composition, amount and spatial distribution of micro-litter (particles < 5mm) on the coastline, in the surface layer of the water column, and in seabed sediment, are at levels that do not cause harm to the coastal and marine environment.	D10C2	Yes	Broader	Yes	Yes	Yes
	The amount of litter and micro-litter classified in the categories 'artificial polymer materials' and 'other' ingested by marine animals (birds, mammals, reptiles, fish or invertebrates.) is at a level that does not adversely affect the health of the species concerned.	D10C3 — Secondary	Yes	Broader	Need further research	Need further research	Need further research
	The number of individuals of each species which are adversely affected due to litter, such as by entanglement, other types of injury or mortality, or health effects.	D10C4 — Secondary	Yes	Broader	Need further research	Need further research	Need further research

Marine Strategy Framework Directive (MSFD) includes marine litter as one of the eleven qualitative descriptors that have to be considered towards achieving Good Environmental Status (GES). Marine litter can be found floating on the sea surface, on the coastline, in the water column or on the seabed in the form of macro litter or small pieces of plastics called microplastics which are directly introduced into the marine environment as plastic microparticles. Approximately 20% of marine debris is originated from at-sea activities (Smith et al., 2018), including merchant shipping, cruise liners, fishing vessels, military fleets, and offshore installations.

12.1 D10C1

Several studies have been conducted to evaluate the density and composition of anthropogenic marine litter in European waters noting the presence of fishing debris (Cau et al., 2017; Consoli et al., 2018; Galgani et al., 2000; García-Rivera et al., 2017; Ioakeimidis et al., 2014; Pasquini et al., 2016; Pham et al., 2014; Veiga et al., 2016). However, marine litter studies in the Macaronesian region are scarce even though it is assumed that islands have a higher plastic debris accumulation rate when compared to continental sites (Perez-Venegas et al., 2017). Recent studies have been published to evaluate marine debris sources in the Azores (Chambault et al., 2018; Pham et al., 2013; Pieper et al., 2015, 2019; Ríos et al., 2018; Rodríguez and Pham, 2017) and Madeira arquipelagos (Chambault et al., 2018). In the Canarian archipelago, the information referring to marine litter (excluding microplastics) comes from the evaluations realized during the OMARCOST Project (González-Lorenzo et al., 2013a, 2013b, 2013c). From these studies it is found that on the coast and beaches, plastics are the main type of marine litter, while in studies carried out on the ocean floor (e.g. Condor seamount and Faial-Pico Passage), what predominates is derelict fishing gear. Floating macro litter off the Azores archipelago and Madeira are mainly composed by general plastic user items followed by plastic packaging and derelict fishing gears.

Abandoned, lost or otherwise discarded fishing gears (ALDFG) sink to the seabed and do not degrade and represent a problem that is increasingly of concern due to the negative impacts on marine ecosystems, benthic habitats, wildlife and fisheries resources. These impacts include ghostfishing and damage to non-target species (Barreiros and Raykov, 2014; Orós et al., 2005), physical damage on benthic species and the seabed, navigational hazards (Macfadyen et al., 2009) and ALDFG is also a source of microplastics when it disintegrates over time (FAO, 2018a).

FAO and UNEP estimated a decade ago that ALDFG in the oceans represented approximately 10% of all marine debris, but studies suggest that there are large differences in the proportion of ALDFG found among all marine litter in different regions (Macfadyen et al., 2009). Fishing waste also includes food waste and other materials such as dolly ropes (strands of plastic rope used to protect trawled gear from rocky sea floors), buoys, Styrofoam fish boxes or rubber gloves (Belin et al., 2017; Veiga et al., 2016). However, many litter items cannot be directly connected to a particular source, way of release or pathway since they can have a number of potential sources and pathways of entry as well as geographic origins; so establishing what proportion belongs to the fishing sector is not an easy task. To this must be added that the studies carried out to date in the Macaronesian region are scarce and focused on specific areas, so it is difficult to extrapolate the results within each archipelago due to their oceanographic and geographic peculiarities.

Marine litter composition and density is also different depending on habitats since its distribution is conditioned not only for anthropogenic activities but also for oceanographic conditions as well as climate factors, and degradation times are different on the surface or deep-water (Pham et al., 2014; Tubau et al., 2015). Moreover, there are notable temporal variations, with accumulation and concentration trends of marine litter in particular geographic areas, although interpretation of these trends is, difficult because the ageing of plastics at depth is unknown and the accumulation of debris on the seafloor predated the scientific investigations which began in the 1990s (Galgani et al., 2015). Recently, EMODnet Chemistry released the fist preliminary partial maps for seafloor litter and for beach litter at European scale (www.emodnet-chemistry.eu).

Fishermen have the obligation to retrieve or report lost gear (Council Regulation (EC) No 1224/2009) and MARPOL Annex V requires signatory nations to provide adequate port reception facilities for accepting garbage generated by ships, including ALDFG; however, many port reception facilities did not accept fishing gears. The proposal for a revision to the new Directive on Port Reception Facilities (COM (2018)33 final) goes a long way to ensuring that ship generated waste is delivered to adequate port reception facilities, instead of being discharged at sea. In addition, a proposal for a Directive of the European Parliament and of the Council on the reduction of the impact of certain plastic products on the environment has been submitted for adoption. This proposal pretends to provide tools and incentives to facilitate recovery, re-use and recycling of the plastic material in fishing gears by adding Extended Producer Responsibility (EPR) schemes to the existing measures.

Some impact prevent and mitigation measures to control ALDFG and ghost fishing include gear marking, port state measures to combat illegal, unreported and unregulated (IUU) fishing, reduction of fishing effort, establishment of spatial management measures, use of biodegradable materials, reducing the ghost fishing of non-target species by applying the same by-catch prevention measures as in active fishery or reuse and recycling ALDFG (Macfadyen et al., 2009). For example, Healthy Seas recovers fishing nets and regenerates them to develop new products including socks, carpets and swimwear (https://healthyseas.org/). FAO has developed a guideline for the application of a system for the marking of fishing gears, including small-scale fisheries, to contribute to sustainable fisheries and to improve the marine environment by combating, minimising and eliminating ALDFG (FAO, 2018b); although some of these proposals are difficult to enforce and to monitor.

Regional intergovernmental bodies and agreements with the competence to establish binding measures and data collection protocols on monitoring and controlling ALDFG and ghost fishing can be found in the article developed by Gilman (2015).

12.2 D10C2

Plastics are by far the most abundant litter and their degradation generate microplastics which, when ingested by organisms, can deliver contaminants across the food-web (Andrady, 2011; Cole et al., 2013; Farrell and Nelson, 2013). Microplastics represent an increasing proportion of marine debris and may enter aquatic environments from land-based and maritime sources and follow diverse pathways (Lusher et al., 2017). Nevertheless, over large spatial scales average microplastic concentrations decrease with distance from land due to dilution, implying that the major input comes from land (Enders et al., 2015).

In the fisheries and aquaculture sector, the construction, use, maintenance and disposal of fishing gear, cages, buoys, boats and product packages are sources of secondary microplastics (FAO, 2018a). It is pertinent, however, to reiterate that it is difficult to estimate the percentage of marine litter originated from maritime activities, and only a proportion of them will result from the fisheries and aquaculture industry; so it is even more complicated to estimate the proportion of microplastics originated as a result of these activities. Also, the sources of microplastics in offshore fishing grounds may be harder to interpret because of the influence of oceanic distribution (Lusher et al., 2017).

Different studies have been carried out to evaluate the presence of microplastics as well as their abundance and composition in the Azores (Enders et al., 2015), Madeira (Lots et al., 2017) and Canarian (Baztan et al., 2014; Herrera et al., 2019, 2018) Archipelagos. But, it can be extremely complicated to directly source microplastics to fisheries activities unless microplastics have the same visual (Herrera et al., 2019) or chemical composition (Andrady, 2011) as the gears employed.

The distribution of microplastics in the ocean is influenced by the nature and location of the source of entry, as well as the complex interaction of physical, chemical and biological processes (GESAMP, 2015). So, we assume that the spatial extent of the environmental impact is broader than fishing grounds, although the studies previously mentioned have been conducted in specific areas of the three archipelagos, in some cases the samples have been collected on beaches and others in the open ocean, so the results are not comparable to each other and there is no global view that reflects the abundance and distribution of microplastics both in the coastal areas and in the open ocean for none of the archipelagos.

Due to their small size and abundance, microplastics are potentially consumed by a wide range of organisms (Lusher et al., 2017; Rezania et al., 2018) and, once ingested, they may be egested or retained. But microplastics also have other indirect impacts that can affect the environment. For example, the latent impact of micro-plastics as transport vectors of pathogenic bacteria must be also considered as a grave threat to coastal zones and the marine environment (Baztan et al., 2014) as well as risks associated with sorbed pollutants since plastic debris readily accumulates harmful pollutants from the surrounding environment increasing their concentration and it is suspected that these pollutants can be accumulated in the tissues of organisms (Wang et al., 2016).

The Plastics Strategy (COM(2018)28) included in the Circular Economy Package was adopted on 16 January 2018. It contains a wide range of legislative and non-legislative measures some of which are new and others which are either already being developed or in the process of review/ revision. Some of these measures are aimed directly at microplastics, but others focus on the reduction of macroplastics whose degradation is also responsible for the introduction of microplastics into the environment. Actions to tackle fishing sources of marine litter include the adoption of a legislative proposal on port reception facilities for the delivery of waste from ships and the development of measures to reduce ALDFG. To reduce discharges of waste by ships, the Commission presented a legislative proposal on port reception facilities (COM (2018)33 final).

Monitoring and assessment are essential steps towards addressing specific questions about marine litter, including microplastics. That is why GESAMP (2019) has been developed aguidelines including monitoring methods for shorelines, sea surface and water column, seafloor and marine biota.

12.3 D10C3

At the Canary Islands, fledgling cory shearwaters (*Calonectris diomedea*) contained plastics (nylon threads) related to fisheries activities in their guts, and it is assumed that these items were regurgitated during parental feeding since these chicks never feed in the marine environment (Rodríguez et al., 2012). Similarly to what has been reported for Cory's shearwater in the Canaries, in the Azores, parents transfer plastic debris to fledglings (Rodríguez et al., 2016).

Turtles examined in Azorean waters showed debris in their stomachs (Pham et al., 2017; Rodríguez et al., 2016), while no plastic debris was found in fish, but a posible explanaition to this is that a large number of deep-sea fish regurgitate their stomach contents when brought to the surface (Rodríguez et al., 2016). In Canary Islands, a recent study revealed the presence of microplastic particles in the digestive tract of Atlantic chub mackerel (*Scomber colias*) (Herrera et al., 2019).

Due to the high biodiversity present in the three archipelagos, studies on the stomach content of the species are scarce. To this, we must add that the studies focused on the ingestion of litter by marine animals are merely anecdotal and the contribution of fishing in the generation of this garbage in some cases is not completely conclusive, so more research is needed to evaluate this criterion in a proper way..

12.4 D10C4

Ocean-based anthropogenic litter causes harm to a wide range of marine biota. The impacts of ocean-based litter on the marine organisms have been mentioned in the previous sections; seabirds, fish, turtles and marine mammals suffer from entanglement with and ingestion of marine litter ítems. However, we do not have enough information to be able to evaluate this criterion because the studies that exist have been carried out in specific areas of the archipelagos, at specific moments, with different methodologies and in different habitats without continuity over time and where the number of species for which information is available is very limited.

12.5 References

Andrady, A.L., 2011. Microplastics in the marine environment. Mar. Pollut. Bull. 62, 1596–1605.

Barreiros, J.P., Raykov, V.S., 2014. Lethal lesions and amputation caused by plastic debris and fishing gear on the loggerhead turtle *Caretta caretta* (Linnaeus, 1758). Three case reports from Terceira Island, Azores (NE Atlantic). Mar. Pollut. Bull. 86, 518–522. https://doi.org/10.1016/j.marpolbul.2014.07.020

Baztan, J., Carrasco, A., Chouinard, O., Cleaud, M., Gabaldon, J.E., Huck, T., Jaffrès, L., Jorgensen, B., Miguelez, A., Paillard, C., Vanderlinden, J.-P., 2014. Protected areas in the Atlantic facing the hazards of micro-plastic pollution: First diagnosis of three islands in the Canary Current. Mar. Pollut. Bull. 80, 302–311. https://doi.org/10.1016/j.marpolbul.2013.12.052

Belin, A., Dom, A., Stockhausen, B., Priestland, E., Moore, M., 2017. Tackling overfishing and marine litter. An analysis of Member States measures under the Marine Directive. Seas Risk Tech. Rep. 60pp.

Cau, A., Alvito, A., Moccia, D., Canese, S., Pusceddu, A., Rita, C., Angiolillo, M., Follesa, M.C., 2017. Submarine canyons along the upper Sardinian slope (Central Western Mediterranean) as repositories for derelict fishing gears. Mar. Pollut. Bull. 123, 357–364. https://doi.org/10.1016/j.marpolbul.2017.09.010

Chambault, P., Vandeperre, F., Machete, M., Lagoa, J.C., Pham, C.K., 2018. Distribution and composition of floating macro litter off the Azores archipelago and Madeira (NE Atlantic) using opportunistic surveys. Mar. Environ. Res. 141, 225–232. https://doi.org/10.1016/j.marenvres.2018.09.015

Cole, M., Lindeque, P., Fileman, E., Halsband, C., Goodhead, R., Moger, J., Galloway, T.S., 2013. Microplastic ingestion by zooplankton. Environ. Sci. Technol. 47, 6646–6655.

Consoli, P., Falautano, M., Sinopoli, M., Perzia, P., Canese, S., Esposito, V., Battaglia, P., Romeo, T., Andaloro, F., Galgani, F., Castriota, L., 2018. Composition and abundance of benthic marine litter in a coastal area of the central

Mediterranean Sea. Mar. Pollut. Bull. 136, 243–247. https://doi.org/10.1016/j.marpolbul.2018.09.033

Enders, K., Lenz, R., Stedmon, C.A., Nielsen, T.G., 2015. Abundance, size and polymer composition of marine microplastics ≥ 10 µm in the Atlantic Ocean and their modelled vertical distribution. Mar. Pollut. Bull. 100, 70–81. https://doi.org/10.1016/j.marpolbul.2015.09.027

FAO, 2018a. The state of world fisheries and aquaculture - Meeting the sustainable development goals. Rome. Licence: CC BY-NC-SA 3.0 IGO.

FAO, 2018b. Report of the Technical Consultation on Marking of Fishing Gear. Rome, Italy, 5–9 February 2018. FAO Fisheries and Aquaculture Report No.1236. Rome, FAO.

Farrell, P., Nelson, K., 2013. Trophic level transfer of microplastic: *Mytilus edulis* (L.) to *Carcinus maenas* (L.). Environ. Pollut. 177, 1–3.

Galgani, F., Hanke, G., Maes, T., 2015. Global distribution, composition and abundance of marine litter. In: M. Bergmann, L. Gutow, M. Klages (Eds.), Marine anthropogenic litter (pp. 29–56). Berlin: Springer.

Galgani, F., Leaute, J.P., Moguedet, P., Souplet, A., Verin, Y., Carpentier, A., Goraguer, H., Latrouite, D., Andral, B., Cadiou, Y., Mahe, J.C., Poulard, J.C., Nerisson, P., 2000. Litter on the Sea Floor Along European Coasts. Mar. Pollut. Bull. 40, 516–527. https://doi.org/10.1016/S0025-326X(99)00234-9

García-Rivera, S., Lizaso, J.L.S., Millán, J.M.B., 2017. Composition, spatial distribution and sources of macro-marine litter on the Gulf of Alicante seafloor (Spanish Mediterranean). Mar. Pollut. Bull. 121, 249–259. https://doi.org/10.1016/j.marpolbul.2017.06.022

GESAMP, 2019. Guidelines or the monitoring and assessment of plastic litter and microplastics in the ocean. Kershaw P.J., Turra A. and Galgani F. editors), (IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP/ISA Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection). Rep. Stud. GESAMP No. 99, 130p.

GESAMP, 2015. Sources, fate and effects of microplastics in the marine environment: a global assessment. (Kershaw, P. J., ed.). (IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection). Rep. Stud. GESAMP No. 90, 96 p.

Gilman, E., 2015. Status of international monitoring and management of abandoned, lost and discarded fishing gear and ghost fishing. Mar. Policy 60, 225–239. https://doi.org/10.1016/j.marpol.2015.06.016

González-Lorenzo, G., Jiménez, S., González, J.F., Boza, C., Bartolomé, A., Hernández, C.L., Martín-Sosa, P., Rodríguez, .J E., Santamaría, M.T.G., 2013a. Informe de Campaña “FUERTEVENTURA 1305”. Proyecto OMARCOST. Inst. Esp. Oceanogr. Cent. Ocean. Canar. Mem. Científico-Téc. 26pp.

González-Lorenzo, G., Jiménez, S., González, J.F., González-Gil, S., Rodríguez, J.E., Hernández, C.L., Martín-Sosa, P., Santamaría, M.T.G., 2013b. Informe de Campaña "FUERTEVENTURA 1312". Proyecto OMARCOST. Inst. Esp. Oceanogr. Cent. Ocean. Canar. Mem. Científico-Téc. 32pp.

González-Lorenzo, G., Jiménez, S., Rodríguez, J.E., González, J.F., Boza, C., Bartolomé, A., Hernández, C.L., Martín-Sosa, P., González, Y., González, E.L., Santamaría, M.T.G., 2013c. Informe de Campaña "HIERRO 1310". Proyecto OMARCOST. Inst. Esp. Oceanogr. Cent. Ocean. Canar. Mem. Científico-Téc. 28pp.

Herrera, A., Asensio, M., Martínez, I., Santana, A., Packard, T., Gómez, M., 2018. Microplastic and tar pollution on three Canary Islands beaches: An annual study. Mar. Pollut. Bull. 129, 494–502. https://doi.org/10.1016/j.marpolbul.2017.10.020

Herrera, A., Ŝtindlová, A., Martínez, I., Rapp, J., Romero-Kutzner, V., Samper, M.D., Montoto, T., Aguiar-González, B., Packard, T., Gómez, M., 2019. Microplastic ingestion by Atlantic chub mackerel (*Scomber colias*) in the Canary Islands coast. Mar. Pollut. Bull. 139, 127–135. https://doi.org/10.1016/j.marpolbul.2018.12.022

Loakeimidis, C., Zeri, C., Kaberi, H., Galatchi, M., Antoniadis, K., Streftaris, N., Galgani, F., Papathanassiou, E., Papatheodorou, G., 2014. A comparative study of marine litter on the seafloor of coastal areas in the Eastern Mediterranean and Black Seas. Mar. Pollut. Bull. 89, 296–304. https://doi.org/10.1016/j.marpolbul.2014.09.044

Lots, F.A.E., Behrens, P., Vijver, M.G., Horton, A.A., Bosker, T., 2017. A large-scale investigation of microplastic contamination: Abundance and characteristics of microplastics in European beach sediment. Mar. Pollut. Bull. 123, 219–226. https://doi.org/10.1016/j.marpolbul.2017.08.057

Lusher, A.L., Hollman, P.C.H., Mendoza-Hill, J.J., 2017. Microplastics in fisheries and aquaculture: status of knowledge on their occurrence and implications for aquatic organisms and food safety, FAO fisheries and aquaculture technical paper. Food and Agriculture Organization of the United Nations, Rome.

Macfadyen, G., Huntington, T., Cappell, R., 2009. Abandoned, lost or ortherwise discarded fishing gear, FAO fisheries and aquaculture technical paper. UNEP Regional Seas Reports and Studies, No. 185; FAO Fisheries and Aquaculture Technical Paper, No. 523. Rome, UNEP/FAO. 2009. 115p.

Orós, J., Torrent, A., Calabuig, P., Déniz, S., 2005. Diseases and causes of mortality among sea turtles stranded in the Canary Islands, Spain (1998-2001). Dis. Aquat. Organ. 63, 13–24. https://doi.org/10.3354/dao063013

Pasquini, G., Ronchi, F., Strafella, P., Scarcella, G., Fortibuoni, T., 2016. Seabed litter composition, distribution and sources in the Northern and Central Adriatic Sea (Mediterranean). Waste Manag. 58, 41–51. https://doi.org/10.1016/j.wasman.2016.08.038

Perez-Venegas, D., Pavés, H., Pulgar, J., Ahrendt, C., Seguel, M., Galbán-Malagón, C.J., 2017. Coastal debris survey in a Remote Island of the Chilean Northern

Patagonia. Mar. Pollut. Bull. 125, 530–534. https://doi.org/10.1016/j.marpolbul.2017.09.026

Pham, C.K., Gomes-Pereira, J.N., Isidro, E.J., Santos, R.S., Morato, T., 2013. Abundance of litter on Condor seamount (Azores, Portugal, Northeast Atlantic). Deep Sea Res. Part II Top. Stud. Oceanogr. 98, 204–208. https://doi.org/10.1016/j.dsr2.2013.01.011

Pham, C.K., Ramirez-Llodra, E., Alt, C.H.S., Amaro, T., Bergmann, M., Canals, M., Company, J.B., Davies, J., Duineveld, G., Galgani, F., Howell, K.L., Huvenne, V.A.I., Isidro, E., Jones, D.O.B., Lastras, G., Morato, T., Gomes-Pereira, J.N., Purser, A., Stewart, H., Tojeira, I., Tubau, X., Van Rooij, D., Tyler, P.A., 2014. Marine Litter Distribution and Density in European Seas, from the Shelves to Deep Basins. PLoS ONE 9, e95839. https://doi.org/10.1371/journal.pone.0095839

Pham, C.K., Rodríguez, Y., Dauphin, A., Carriço, R., Frias, J.P.G.L., Vandeperre, F., Otero, V., Santos, M.R., Martins, H.R., Bolten, A.B., Bjorndal, K.A., 2017. Plastic ingestion in oceanic-stage loggerhead sea turtles (*Caretta caretta*) off the North Atlantic subtropical gyre. Mar. Pollut. Bull. 121, 222–229. https://doi.org/10.1016/j.marpolbul.2017.06.008

Pieper, C., Amaral-Zettler, L., Law, K.L., Loureiro, C.M., Martins, A., 2019. Application of Matrix Scoring Techniques to evaluate marine debris sources in the remote islands of the Azores Archipelago. Environ. Pollut. 249, 666–675. https://doi.org/10.1016/j.envpol.2019.03.084

Pieper, C., Ventura, M.A., Martins, A., Cunha, R.T., 2015. Beach debris in the Azores (NE Atlantic): Faial Island as a first case study. Mar. Pollut. Bull. 101, 575–582. https://doi.org/10.1016/j.marpolbul.2015.10.056

Rezania, S., Park, J., Md Din, M.F., Mat Taib, S., Talaiekhozani, A., Kumar Yadav, K., Kamyab, H., 2018. Microplastics pollution in different aquatic environments and biota: A review of recent studies. Mar. Pollut. Bull. 133, 191–208. https://doi.org/10.1016/j.marpolbul.2018.05.022

Ríos, N., Frias, J.P.G.L., Rodríguez, Y., Carriço, R., Garcia, S.M., Juliano, M., Pham, C.K., 2018. Spatio-temporal variability of beached macro-litter on remote islands of the North Atlantic. Mar. Pollut. Bull. 133, 304–311. https://doi.org/10.1016/j.marpolbul.2018.05.038

Rodríguez, A., Rodríguez, B., Nazaret Carrasco, M., 2012. High prevalence of parental delivery of plastic debris in Cory's shearwaters (*Calonectris diomedea*). Mar. Pollut. Bull. 64, 2219–2223. https://doi.org/10.1016/j.marpolbul.2012.06.011

Rodríguez, Y., Frias, J., Carriço, R., Neves, V., Bried, J., Martins, H.R., Vandeperre, F., Santos, M.R., van Franeker, J.A., Bolten, A.B., 2016. Monitoring plastic ingestion in selected Azorean marine organisms. MICRO 2016 Fate Impact Microplastics Mar. Ecosyst. 150.

Rodríguez, Y., Pham, C.K., 2017. Marine litter on the seafloor of the Faial-Pico Passage, Azores Archipelago. Mar. Pollut. Bull. 116, 448–453. https://doi.org/10.1016/j.marpolbul.2017.01.018

Smith, S.D.A., Banister, K., Fraser, N., Edgar, R.J., 2018. Tracing the source of marine debris on the beaches of northern New South Wales, Australia: The Bottles on Beaches program. Mar. Pollut. Bull. 126, 304–307. https://doi.org/10.1016/j.marpolbul.2017.11.022

Tubau, X., Canals, M., Lastras, G., Rayo, X., Rivera, J., Amblas, D., 2015. Marine litter on the floor of deep submarine canyons of the Northwestern Mediterranean Sea: The role of hydrodynamic processes. Prog. Oceanogr. 134, 379–403. https://doi.org/10.1016/j.pocean.2015.03.013

Veiga, J.M., Fleet, D., Kinsey, S., Nilsson, P., Vlachogianni, T., Werner, S., Galgani, F., Thompson, R.C., Dagevos, J., Gago, J., 2016. Identifying sources of marine litter. MSFD GES TG Marine Litter Thematic Report. JRC Tech. Rep. EUR28309. https://doi.org/10.2788/018068

Wang, J., Tan, Z., Peng, J., Qiu, Q., Li, M., 2016. The behaviors of microplastics in the marine environment. Mar. Environ. Res. 113, 7–17. https://doi.org/10.1016/j.marenvres.2015.10.014

13 Analysis >> QD11 Introduction of energy, including underwater noise.

QD11 Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment							
QD	**Criteria (element)**	**CODE Criteria**	**Env. Impact**	**Env. impact spatial extent**	**MA pressure solutions**	**Impact mitigation measures**	**Monitoring method**
QD11	The spatial distribution, temporal extent, and levels of anthropogenic impulsive sound sources do not exceed levels that adversely affect populations of marine animals.	D11C1	No				
	The spatial distribution, temporal extent and levels of anthropogenic continuous low-frequency sound do not exceed levels that adversely affect populations of marine animals.	D11C2	Yes	Broader	Needed further research	Needed further research	Needed further research

The anthropogenic contributions of energy with a potential impact on marine ecosystems are diverse (Perez Puyol, Lloret Capote, et al., 2012, Carbonel et al., 2019). According to Aguilar and Tejedor (2012) 'current knowledge about the effects of noise on fauna, including humans, are incomplete, inconclusive and sometimes contradictory'.

With the information available to date, the main pressure related to this descriptor is the contribution of anthropogenic sound (impulsive, continuous) (Perez Puyol, Lloret Capote, et al., 2012, Carbonel et al., 2019). The sources of underwater noise can be of short duration (impulsive, such as seismic campaigns, or piloting of platforms and wind farms) as well as long-term (dredging, navigation and energy installations) (Regional Secretariat for Environment and Two Naturais Resources, 2014 Regional Secretary for Naturais Resources, 2014, Martín Partida et al., 2019a). According to Aguilar and Tejedor (2012) 'the potential acute (impulsive) effects include the risks of immediate damage or injuries to the auditory or body, due to an intense sound source, while the possible chronic effects (continuous) involve the risk of degradation of habitat or the exclusion of important areas for long periods of time, even at moderate levels of sound pressure. Both types of impact can affect individuals, sub-populations or even populations, at unpredictable levels at present. With respect to the spatial scale, different sound sources will have radius of affection that can vary between a few meters to hundreds of kilometers. These effects depend not only on the characteristics of the sound source, but also on the sensitivity of the affected species'.

13.1 D11C1, D11C2

The main contribution of continuous anthropogenic sound in the marine environment is associated with the activity of navigation and maritime transport, whose most representative indicator is the density of maritime traffic, with which it is directly correlated (Martín Partida et al., 2019a). According to Lloret Capote, del Barrio Alvarellos and Moreno Aranda (2012) 'in general, the acoustic energy produced by a ship increases in proportion to its size, displacement, speed and age. Oil tankers and solid bulk vessels are among the main producers of noise. The noise generated by large vessels in rapid movement is quite intense and is concentrated in the low frequency ranges (5-500 Hz). These sources of noise are the most frequent near the large ports and along the most used navigation routes and can spread over very large distances due to their low frequency. This leads to an increase in seafloor noise even away from the hot spots of emission. On the other hand, high noise levels produced by modern freighters or fast-ferries, which emit at higher frequencies (up to 600 Hz) from 16 knots have also been described. The problem with these emissions is that they have the potential to interfere with the vocalizations of many odontocete cetacean species. As for small-medium-length and recreational boats, they tend to create sound at higher frequencies, due to the higher rotational speeds of the propeller'. According to Aguilar and Tejedor (2012), small recreational and fishing boats, cetacean observation boats, and passenger transport boats, such as ferries / high-speed ferries, generate noise, whose characteristics depend on the type of engines , the size of the vessel and its speed, with considerable individual variation between ships of comparable classes. The cavitation of the propeller is generally the predominant origin of the sound in all the boats. In small and fast boats tend to create sound at higher frequencies, due to the higher speeds of rotation of the propeller'.

Among the activities related to navigation are the maritime traffic of goods, passengers, sports and recreational boating, and commercial fishing (Perez Puyol, Arrieta Algarra, et al., 2012). In the Canarian demarcation, proposals have been made on the location of boat routes to contribute to the elaboration of noise maps. Thus, the AIS (Automatic Identification System) for commercial vessels and the VMS (Vessel Monitoring System) have been considered for the fishing vessels older than 15 m in length (in the inferiors to this length it is not obligatory). In any case, 'the traffic of fishing boats is much smaller in number than that of merchant ships' (Perez Puyol, Lloret Capote, et al., 2012). In short, noise from ships is mainly associated with freight traffic, passenger traffic and, to a lesser extent, fishing activity. Thus, for the Canary Islands, it is indicated that the human activity generated by the pressure in D11 with respect to noise is maritime transport (Martín Partida, Arrieta Algarra, Martínez García-Denche, et al., 2019a).

With respect to navigation echo sounders in the fishing sector, according to Aguilar and Tejedor (2012) 'contribute as part of the noise generated by maritime transport, to produce sound by means of transducers to locate the depth of the seabed, or schools of fish. They are used on almost all large vessels and on many smaller vessels, including most fishing vessels and a large number of recreational vessels. Its frequency range coincides with that of many odontocetes, and the significant number of navigational echo sounders in use means that this sound source contributes considerably to the total amount of anthropogenic underwater energy noise. The frequency range of the echo sounder depends on the depth of the bottom. Water absorbs high frequencies rather than low frequencies, therefore low frequency transducers provide the deepest readings (frequencies less than 50 kHz in waters of more than 100-200 m depth). However, high frequencies provide better resolution and detail on the screen. If 50 kHz and 200 kHz screens are examined simultaneously, the highest detail is observed in the high frequency screen. This detail could show schools of fish, bait, structures, etc. Many dual-frequency fishing sonars combine low and high frequency operation on a single screen / transducer. This gives the advantages of both professional fishermen but produces a greater noise pollution, unnecessary if operated at depths of less than 100 m, as is most of the cases'. According to Aguilar and Tejedor (2012) 'it is advisable to restrict the use of navigation echo sounders in shallow waters to instruments that emit at high frequencies, preferably higher than 150 kHz (including 200 kHz), in areas of possible distribution of porpoises, such as coasts Atlantic). This measure reduces both the spatial range of sounds (by absorbing high frequencies very quickly) and their impact on nearby fauna, given that there are few species sensitive to more than 120 kHz and only one group of species (porpoises, pygmy sperm whales / dwarf) vocalize up to about 180 kHz'. According to Aguilar and Tejedor (2012), echo sounders are sometimes used without interruption, without there being a real need. Most recreational boats only need them for approach maneuvers to the anchoring. Sport and professional fishing boats in shallow waters also use echo sounders to locate fishing areas such as shallows or rocky areas. This indicates that an awareness campaign, focused on reducing the use of echo sounders when they are not necessary, could have a direct effect on the control of this source of marine pollution'. In any case, in the Canarian region at least for the time being, the sound emission by the fishing activity is not considered to be relevant to the margin generated by the navigation of these vessels (Martín Partida et al., 2019b).

According to statistics provided by the European ORFISH project (www.orfish.eu), in the Canary Islands, Madeira and Azores there are 773, 431 and 757 fishing vessels (March 2018), respectively. In the Canary Islands, according to data provided (in 2019) by the 'Dirección General de la Marina Mercante Española (Ministerio de Fomento)' there are 13,178 vessels with a base port in the Canary Islands in provisional or definitive service, of which only 7.46% are dedicated to professional fishing (list 3ª). Of these almost 90% of the boats are registered for recreational boating. Thus, it highlights the scarce contribution of fishing to the number of total vessels and that, in addition, 90.17% 91.65% 91.94% (www.orfish.eu) of these fishing vessels are below 15 meters in length, respectively, for Canary Islands, Madeira and Azores. Finally, within the framework of the Canary Islands Marine Strategies, it is indicated that there are no international standards for the monitoring of ambient noise and suggest, initially, the realization of a submarine noise map of the region obtained from a sound propagation model. The model would be validated and calibrated by real measurements obtained from observation stations. So far no proposal has been made for the reduction of ambient noise (Alonso Rodriguez et al., 2012), and less in the framework of the fishing sector.

In short, the anthropogenic contributions of energy with a potential impact on marine ecosystems are very varied, but, to date, the main pressure related to this descriptor is the contribution of sound. The main contribution of continuous sound in the marine environment is associated with the activity of navigation whose most representative indicator is the density of maritime traffic. In general, sound emission for fishing activity is not considered relevant because its contribution both in number of boats and the size of them is not significant if we compare it with recreational boating and with maritime transport (goods and passengers), the latter being the main activity that contributes to the noise coming from the boats in the Macaronesia region.

13.2 References

Aguilar, N. and Tejedor, A. (2012) Documento técnico sobre impactos y mitigación de la contaminación acústica marina. Madrid. Available at: https://www.miteco.gob.es/gl/costas/temas/proteccion-medio-marino/doc-tecnico-impactos-mitigacion-contaminacion-acustica-marina_tcm37-157028.pdf.

Alonso Rodriguez, J. et al. (2012) Estrategias Marina. VI. Programas de Seguimiento. Madrid. Available at: https://www.miteco.gob.es/es/costas/temas/proteccion-medio-marino/vii_pdm_eemm_memoria_tcm30-130995.pdf.

Carbonel, A. et al. (2019) Estrategias marinas de España. Estrategia marina de la demarcación Canaria. Parte IV. Evaluación del estado del medio marino y definición del buen estado ambiental en la demarcación marina canaria (En Fase de exposición Pública:12.03.19-12.04.19). Madrid. Available at: https://www.miteco.gob.es/es/costas/participacion-publica/parteivevaluaciondelestadoydefinicionbeadmcan_tcm30-487966.pdf.

Lloret Capote, A., del Barrio Alvarellos, I. and Moreno Aranda, I. M. (2012) Estrategia Marina. Demarcación Marina Canaria. Evaluación inicial. Parte II: Análisis de presiones e impactos. Madrid. Available at: http://www.mapama.gob.es/es/costas/temas/proteccion-medio-marino/II_Analisis_Presiones_Canarias_tcm30-130933.pdf.

Martín Partida, I. et al. (2019a) Estrategias marinas de España. Estrategia marina de la demarcación Canaria. Parte II. Análisis de presiones e impactos. Anexo Parte II. Fichas del análisis de presiones e impactos (En Fase de exposición Pública: 12.03.19-12.04.19). Madrid. Available at: https://www.miteco.gob.es/es/costas/participacion-publica/anexoparteiianalisispresioneseimpactosdmcan_tcm30-487968.pdf.

Martín Partida, I. et al. (2019b) Estrategias marinas de España. Estrategia marina de la demarcación Canaria. Parte II. Análisis de presiones e impactos (En Fase de exposición Pública: 12.03.19-12.04.19). Madrid, España. Available at: https://www.miteco.gob.es/es/costas/participacion-publica/parteiianalisispresioneseimpactosdmcan_tcm30-487964.pdf.

Perez Puyol, A., Lloret Capote, A., et al. (2012) Estrategia Marina. Demarcación Marina Canaria Parte IV. Descriptores del buen estado ambiental. Descriptor 11: ruido submarino. Evaluación inicial y buen estado ambiental. Madrid. Available at: https://www.miteco.gob.es/es/costas/temas/proteccion-medio-marino/IV_D11_Canarias_tcm30-130945.pdf.

Perez Puyol, A., Arrieta Algarra, S., et al. (2012) Estrategias Marinas. Documento marco. Evaluación inicial, buen estado ambiental y objetivos ambientales. Madrid. Available at: http://www.mapama.gob.es/es/costas/temas/proteccion-medio-marino/0_Documento marco estrategias marinas_tcm30-130950.pdf.

Secretaria Regional do Ambiente e dos Recursos Naturais (2014) Estratégia Marinha para a subdivisão da Madeira. Diretiva Quadro Estratégia Marinha. Available at: https://www.dgrm.mm.gov.pt/documents/20143/43971/RelatorioInicial_MAD_FINAL__2014.pdf/0f2783be-bf81-5d26-83cd-15830cff998c.

Secretaria Regional dos Recursos Naturais (2014) Estratégia Marinha para a subdivisão dos Açores. Diretiva Quadro Estratégia Marinha. Available at: https://www.dgrm.mm.gov.pt/documents/20143/43971/RelatorioInicial_AZO_FINAL_2014+%282%29.pdf/a0b43add-3ba3-6e52-e660-11189ef15ad7.

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