Universidad de La Laguna

Facultad de Ciencias: Sección Biología

A worldwide review of fish trap fisheries at small scale: main trends and features.

Una revisión mundial de la pesca de nasas en pequeña escala: principales tendencias y características.

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Máster en Biología Marina:
Biodiversidad y Conservación

Junio 2018

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DECLARAN:

Que la memoria presentada por la Graduada en Biología Maryia Vadziutsina, titulada "A worldwide review of fish trap fisheries at small scale: main trends and features" ha sido realizada bajo su dirección y consideran que reúne todas las condiciones de calidad y rigor científico requeridas para optar a su presentación como Trabajo de Fin de Máster, en el Máster Oficial de Postgrado de Biología Marina: Biodiversidad y Conservación de la Universidad de La Laguna, curso 2017/2018.

Y para que así conste y surta los efectos oportunos, firman el presente informe favorable en San Cristóbal de La Laguna a 28 de Mayo de 2018.

Fdo. Dr. Rodrigo Riera Elena — Codirector

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ABSTRACT

Fish traps are an important gear for artisanal fishers. They are extensively used in some reef fisheries, mainly in tropical regions. Despite that, generally there is a significant lack of knowledge concerning basic aspects of this fishing art such as, caught species composition, CPUE or management approaches. We herein compiled information from fish traps fisheries worldwide to carry out a comparative analysis based on its main characteristics, i.e. mesh size, target species, CPUE and catch composition, ghost fishing and management. A wide range of mesh sizes from 12.5 to 64 mm was used; however, 30-32 mm, 37-40 mm and 50-51 mm wide remain the most common. Fishers tend to target larger-bodied predatory and herbivorous fish species. However, in some trap fisheries their proportion in catches has significantly declined, which resulted in fishing down the web and targeting a variety of reef fish species that are still available. Most fish stocks demonstrated declining trends, with only few species being capable of withstanding high fishing pressure due to heavy management and particular life history characteristics. CPUE greatly varied from 0.16 to 23.9 kg per trap per day, with lowest and highest catch rates reported from the Canaries and Australia, respectively, but in other areas averaged between 1.99 and 5.64 kg per trap. Well-managed fisheries tend to increase minimum mesh sizes, however it is far from being a common trend, especially in areas facing severe overfishing, and in other regions minimum mesh sizes remain unregulated. Across fisheries from 9 to 100% of fish traps is lost annually, mostly due to poaching and vandalism, gear conflict and collisions with vessels. Despite wide range of estimated fish mortality in ghost fishing traps with closed or without escape panels, presence of an escape mechanism results in almost complete elimination of mortality. Other management tools such as, minimum legal length of target fish species (MLL), temporal and spatial closures are also widely used, however they are effective only if well enforced and interconnected.

Keywords: coral reef fishes, fish trap impact, ghost fishing, management, small-scale fisheries.

RESUMEN

Las nasas son un arte utilizado de forma extensive por los pescadores artesanales, siendo utilizadas en muchas pesquerías, principalmente en regiones tropicales. A pesar de ello, existe poca información sobre diferentes aspectos de este arte de pesca, como la composición de las capturas, el rendimiento (CPUE) y acciones de gestión. En este trabajo se realiza una recopilación de información sobre la pesquería de nasas a nivel mundial con el fin de llevar a cabo un análisis comparativo de sus características principales, como el tamaño de la luz de malla, especies objetivo, CPUE, composición de las capturas, pesca fantasma y medidas de gestión. Existe un rango amplio de luces de malla, desde un mínimo de 12.5 mm hasta un máximo de 64 mm, sin embargo, los diámetros de malla 30-32 mm, 37-40 mm y 50-51 mm son los utilizados comúnmente en esta pesquería. Las especies objetivo corresponden a depredadores de gran tamaño y peces de alimentación herbívora, sin embargo, en algunas pesquerías las capturas han disminuido de forma alarmante y en la actualidad capturan una gran variedad de peces costeros disponibles. La mayoría de los stocks pesqueros muestran una tendencia regresiva, con únicamente unas pocas especies capaces de soportar una presión pesquera elevada, gracias a las acciones de gestión costera implementadas y a las particularidades de su ciclo de vida. La CPUE varió bastante, entre un mínimo de 0,16 kg por nasa por día hasta un máximo de 23,9 kg por nasa por día en Canarias y Australia, respectivamente. Los rendimientos medios se situaron principalmente entre 1.99 y 5.64 kg por nasa por día en otras áreas geográficas analizadas. Las medidas de gestión de las pesquería de nasas han sido variadas, destacando el incremento de las luces de malla, en especial en aquellas áreas caracterizadas por una sobrepesca acusada, aunque cabe destacar que en otras regiones las luces de malla siguen sin estar reguladas. En cuanto a la pérdida de nasas, se estima que entre el 9 y 100% de las nasas se pierden debido a vandalismo, furtivismo, colisiones con embarcaciones y problemas en el izado y/o calado de las nasas. A pesar de esta gran variación en los porcentajes de pesca fantasma, la colocación de un mecanismo de escape para los peces resulta en la práctica ausencia total de pesca fantasma. Otras medidas de gestión que han sido implementadas en ciertas áreas y que no son específicas de esta pesquería, son el establecimiento de talla mínima de captura, vedas temporales y espaciales, que han sido utilizadas de forma extensiva en muchas pesquerías. Estas medidas son muy efectivas si se cumplen y se encuentran integradas en un modelo de gestión sostenible de los recursos pesqueros.

Palabras clave: gestión, impacto de nasas, pesca fantasma, peces costeros, pesquerías artesanales.

INTRODUCTION

Marine fisheries have a very significant contribution to the well-being and economy of coastal communities, as they provide food security, job opportunities, source of income and livelihood for millions of people (FAO, 2016a). In 2014, global marine fisheries produced over 81.5 million of tonnes of fish, which constitutes 48.7% of the overall world fish production (167.2 millions of tonnes) (FAO, 2016a) and this number could largely been underreported (Pauly & Zeller, 2016). More than eightfold dramatic increase in total fish production since 1950s (FAO, 2011) followed by a drastic expand of human population and growing demand of high-quality protein, has led to a state, where around 31.4% of commercial fish stocks worldwide are either overexploited, depleted or recovering from depletion (FAO, 2016a).

Of 4.6 million of fishing vessels registered in the world in 2014, only 15% belong to the large industrial fleet (FAO, 2016a). In comparison with large-scale fishery, artisanal or Small-Scale Fisheries (hereafter SSF) were often misinterpreted as "inefficient" and until the last decades did not receive the adequate amount of attention both from science and policymakers (Misund et al., 2008; Kolding et al., 2014; Hutubessy et al., 2017). However, SSF are estimated to contribute to around 70% of the total world catches and engage around nearly 90% of all fishers worldwide (Kolding et al., 2014). Artisanal SSF are by definition multi-gear and multi-species, with multiple landings sites. Anglers operate on small boats, within limited fishing grounds, and tend to use passive gears and methods (FAO, 2011). Among its advantages are lower running costs, flexibility in fishing methods applied throughout the year and relatively low environmental impact, compared with industrial fisheries (Misund et al., 2008; Kolding et al., 2014). The optimization of human and economic resources is usually achieved by the use of more passive fishing techniques and gears, such as fish traps, long-lining, hand-lining and gillnets, as well as, by the use of Fish Aggregating Devices (FADs) and low-intensity light attraction methods (FAO, 2016*b*).

In some areas, especially associated with reef environments, fish traps tend to be the principle gear used historically or due to difficulties or restrictions put on using trawls, purse seines or other nets (Munro, 1983; Williams et al., 1997; Chen et al., 2012). For instance, in the Caribbean trap fisheries accounted for decades for a significant portion of the total landings of finfish and shellfish (Munro, 2007). Popularity of fish traps is due to simple design and construction, little specialized training required for fishermen; they don´t require special assistance during the fishing process and can be left at sea for days or weeks (Chen et al., 2012; Hubert et al., 2012; Gomes et al., 2014). Furthermore, fish traps and pots are considered to be low-impact and fuel-efficient gear, suitable for most bottom types, can be deployed regardless of weather conditions (Miller & Hunte, 1987; Williams et al., 1997; García-Mederos et al., 2015), and particular designs can be highly species selective (Hedgärde et al., 2016). Fish retained aboard are usually alive, which facilitates selective harvesting and by-catch can be released back into the sea (He & Inoue, 2010). Moreover, in the last decades there was an increasing trend in using traps as a tool to assess changes in relative abundance, biomass and diversity of reef fish communities in response to growing fishing pressure, infections and changing environmental conditions (Koslow et al., 1988; Recksiek et al., 1991; Barlaup et al., 2013; Bacheler et al., 2017).

Fish traps have been used for catching fish in a wide range of environments from the Mesolithic era to present (Montgomery et al., 2015). The variation of trapping devices is highly dependent on the availability of fish species, environment, building materials present in the area and cultural background of people (Greene et al. 2015). Intertidal fishing weirs were a fundamental gear used worldwide before the spreading of large-scale fishing in the middle of the 20s century (Al-Abdulrazzak & Pauly, 2014). Stone-walled fish traps and wood-stake fishing structures of considerable antiquity have been reported from South Africa, Australia, North America and Europe (O'Sullivan, 2003; Gribble, 2005; Kemp et al., 2009; Greene et al., 2015; Montgomery et al., 2015). Stone-walled tidal traps were geared towards the exploitation of shoaling species and could be described as natural enclosures along the coast, submerged during the high or spring tide. Fish trapped behind the wall was then collected on the ebb tide, with spear, netting, or bare hands (Kemp et al., 2009; Hine, 2010). Nowadays, most of these stone-walled traps are neglected, however, some of them are still in use (Gribble, 2005; Kemp et al., 2009). Moreover, wood-stake intertidal traps are still widely used in the Arabian Gulf (Al-Abdulrazzak & Pauly, 2014). Large-scale aboriginal fish trap complexes operated in British Columbia 1300 - 100 years B.P. were considerably more technologically sophisticated. They had "winged heart" and "winged chevron" shape, which highly resembled modern salmon or whitefish traps (He & Inoue, 2010), and targeted salmon and herring in their spawning grounds (Gribble, 2005). Medieval trapping weirs highly varied in size, shape and operation mode. According to O'Sullivan (2003), during the Medieval Ages in Great Britain the use of stone-built and wooden V-shaped ebb-weirs was very common;

however, woven baskets, box-like pounds, woven wicker fish traps and others were also in use.

Traditionally traps and pots were constructed of natural materials available at a particular location, and although nowadays some small-scale subsistence fisheries construct gear from bamboo and mangrove wood (Wolff, 1999; Samoilys et al., 2011), in past few decades most fishermen switched to utilization of more durable metal or synthetic materials (Gobert et al., 2000).

The variety of fishing traps and trap modifications is overwhelming. Despite the initial differences in interpretation, the terms "fish trap" and "fish pot" are currently used in an interchangeable manner. Modern fish traps implemented by professional and artisanal fishermen can be of various forms and designs, depending on the area fished, environmental conditions and finfish and shellfish species they target. The most popular trap designs include: Chevron (arrowhead), Z-type and S-type traps in coral reef fisheries throughout the Caribbean (Munro, 1983); cylindrical (O-type) traps along the Australian coast (Williams et al., 1997); gargoor traps in the Arabian Gulf (Lee & Al-Baz, 1989), cod traps of various design in North-West Atlantic; set nets in Japan; large-scale floating whitefish/salmon traps used in Baltic Sea, Scandinavia and Alaska (He & Inoue, 2010), among others.

Table 1. Main characteristics of fish traps and operation mode that influence on catch rate and species composition (based on Mahon & Hunte, 2001)

Physical characteristics	Operation mode	Fish biology and ecology	Environment
Mesh size	Soak time	Body depth and shape	Moon phase
Trap size	Location	Fish behaviour	Structural complexity
Trap design	Depth	Migration and mobility	Visual image and shelter
Escape gaps and escape panel	Presence and characteristics of bait	Conspecific and interspecific attraction	

However, regardless the variety of fish trap designs available for fishers, the main concept is the same in all cases: to lure the fish into the trap and impede leaving it until trap retrieval. Effectiveness of traps depends on multiple factors. Catch rate and composition are determined by an interplay of physical characteristics of the trap itself, i.e. trap size and design, mesh size, presence of escape gaps, particularities of the fishing methods applied, i.e. soak time, characteristics of bait if used, etc., fish biology and ecology, i.e. fish size and body shape, behaviour, mobility and migrations, and environmental conditions (Munro, 1974; Fogarty & Addison, 1997; He & Inoue, 2010). In the past decades, numerous researches have been conducted in order to determine how fluctuations of different parameters influence on the effectiveness of fish traps.

Traditionally traps have been fished within shallow reef areas and targeted multiple species. As fish traps are highly unselective, catch composition can be very diverse: Munro et al. (2003) noted that although catchability varies, almost all species of Caribbean fish might enter the trap; Jiménez & Sadovy (1997) reported a high diversity of catches in Puerto Rico, i.e. 90 species, belonging to 35 families. High effectiveness of this gear combined with low selectivity has led to drastic decline in overall fish abundance (Bacheler & Smart, 2016). Poor and untimely placed management, inability of fishers to fish at a sustainable level and overall trend of intensification of fishing effort in response to declining catches resulted in commercial stock depletions and collapses in many coral reef environments. Severe decline in key target species in Bermuda has led to complete ban of use of fish traps in 1990 (Burnett-Herkes & Barnes, 1996). Similar situation occurred with the Nassau grouper (*Epinephelus striatus*), whose spawning aggregation were extensively targeted within the Caribbean region for several decades until a drastic decline in abundance and complete disappearance of some spawning aggregations by early 1990s (Sadovy & Eklund, 1999). Numerous studies stressed that species of major commercial importance such as, groupers (Serranidae) and snappers (Lutjanidae), which comprised the bulk of trap catches in 1970s and 1980s now may constitute as little as 1% or less of the total catch and could be considered mostly as by-catch in some Caribbean reef fisheries (Garrison et al., 2004; Hawkins & Roberts, 2004). Thus, intensification of fishing effort in response to reduction of stocks in most cases leads to fishing "down the web", where second- and third-class herbivore fish species such as, parrotfish, surgeonfish, butterflyfish, etc. which earlier were considered as "trash" (Koslow et al., 1988) are now the ones comprising most of the catch (Hawkins & Roberts, 2004; Marshak et al., 2007). As a result of decrease in total landings, scarcity of first-class species and habitat degradation, a tendency has been observed to move offshore, away from coral reefs to deeper habitats and areas with lower structural complexity, such as algal plains, sandy and hard gorgonian bottoms (Garrison et al., 2004; Sheridan et al., 2006; Marshak et al., 2007). This could be considered as mutually favourable strategy, as redirection of fishing effort has greater economic benefits for anglers and at the same time reduces direct anthropogenic pressure on reef environment, giving an ecosystem an opportunity to recover (Marshak et al., 2007).

One of the criticisms of traps is their role in habitat degradation. According to Marshak et al. (2007) almost half of traps are deployed within colonized hard bottoms dominated by reef or soft corals, and continuous trap fishing contributes to the direct distraction and degradation of coral reef and threatens sessile benthic flora and fauna (Sheridan et al., 2006).

Ghost fishing or the ability of lost, abandoned and discarded gear to continue catching target and non-target fish species for a particular period of time, causing significant economic and ecological impact is another threat, strongly associated fish traps (Clark et al., 2012; Arthur et al., 2014). Sutherland (1983) and Smolowitz (1978) defined derelict traps as gear that cannot be located and retrieved, but which is incapable of fishing due to structural damage or deterioration, while ghost traps could continue catching fish. However, according to modern interpretation derelict fishing gear (hereafter referred as to DFG) include any lost, abandoned or discarded fishing gear regardless of its ability to ghost fish (NOAA Marine Debris Program, 2015).

Fish traps often become derelict as a result of unintentional loss, due to human mistake, entanglement of buoy lines with protruding parts of the coral reef, storms and collisions with vessels, which may result in dragging the trap into the deeper water or simply detaching it from the buoy (Breen, 1990; Macfadyen, 2009). Additionally, although in many areas fishers are obliged to use individually buoyed traps (Munro, 1983), they tend to neglect these regulations, as buoys make traps and pots a simple target, and vandalism and/or theft of gear itself or of the actual catch in some areas is not uncommon (Sutherland et al., 1983; Al-Masroori et al., 2009; Clark et al., 2012). The lack of buoys facilitates the loss of traps, as they can be moved from tens of meters to several kilometres by strong winds and storms (Clark et al., 2012). However, old or damaged gear can be and often is discarded at sea deliberately (Al-Masroori et al., 2009; Renchen et al., 2014). But if lost or discarded trawls nets and purse seines have relatively little potential to continuously catch fish (Breen, 1990; NOAA Marine Debris Program, 2015) lost traps stay robust and may continue ghost fishing for years, before fouled or broken (Clark et al., 2012; Artur et al., 2014). Given the wide distribution of trap fisheries worldwide and the estimated numbers of traps used, the reported 20-30% of fish traps being abandoned or lost annually (Smolowitz, 1978) would result in striking numbers. A high number of programs have been developed concerning this issue, aiming at implementation of prevention measures or at removal and recycling of already existing DFG (Macfadyen et al., 2009; NOAA Marine Debris Program, 2015; Scheld, 2016).

During the last decades marine fishery resources have undergone through an overall decline in landings due to environment degradation, overexploitation and destructive fishing practices (Hughes, 1994; Bohnsack & Ault, 1996). Probably the most important problem associated with overexploitation of resources is a lack of good governance, which includes lack of coordination and conflicts between interested sides, insufficient and ineffective monitoring, poor enforcement and lack of control (Béné, 2006). Although SSF usually have a significant contribution into the well-being of local communities, the needs of this industry in proper management and governance are often neglected. In such conditions high demands for limited marine resources are usually considered as a factor of unsustainability (Béné, 2006; FAO, 2016*b*).

The purpose of this study was to analyse and summarize the features of small-scale trap fisheries worldwide based on bibliographic compilation. The specific objectives are the following:

(i) To compile the available information on the influence of specific fish trap parameters, i.e. trap size and design, mesh size, presence of escape gaps and escape panels, and operation mode variations, e.g. soak time, location of deployment, on catch rate and composition;

(ii) To compare the state of marine fishery resources throughout the main trap fisheries worldwide;

(iii) To examine the main trends in management approaches towards sustainable harvesting of marine resources, considering the trade-offs between fisheries and conservation.

MATERIAL AND METHODS

The research was focused on data, published in peer-reviewed journals since 1980 until 2018. Nevertheless, some scientific papers published prior to that date were also included into the compilation, as providing important and fundamental data on the topic. Additionally, grey literature, such as relevant publications, fishery reports from FAO and other data-bases (ReefBase; Scopus) as well as government reports were analyzed. Comparative characteristics of fish in catch compositions reported in different papers, focused on multispecies reef fisheries, were obtained through the FishBase.org. The number of papers directly related to fish trap fisheries was 186. Moreover, the total number of papers, reports and documents reviewed for the study concerning general description of coral reef fisheries, overfishing, main management tools and strategies, biology and ecology of vulnerable species, among others, was 294.

It should be noted, that regardless the great variety of gear, that could be referred to as “fish trap” or “fish pot”, from floating salmon traps and crab pots to set nets and stack traps fixed within the tidal zone, the emphasis here was put on the portative traps, which are deployed to the bottom and then hauled after a particular soak time and target marine coastal fish species.

Additionally, only the ecoregions, where fish trap fishing grounds were gaining greater or successive attention from the scientific community were used for the further analysis. Therefore, sporadic research dedicated to other areas was omitted. The data from the Mediterranean coast of Turkey was compiled with the information provided by other Middle East countries for Arabian Gulf, despite representing the other marine ecoregion.

Differences in mesh sizes used and analysed across fisheries

Of 186 papers included into the fish trap compilation, only nearly 12.4% (23 articles) specifically aimed at comparative analysis of various mesh sizes both used and proposed for introduction into trap fishery. However, as the information given in these 23 articles was insufficient and often outdated, the data on mesh sizes currently used were sourced from grey literature and other available documents.

Target species

Many authors pointed the main species targeted by fishers, however sometimes the designation of target species was lacking. In this case the species within the catch specified as being the "first-class" or "of higher commercial importance" were accepted as "targets", united and classified to subfamily, e.g. Epinephelinae, or to Family where representatives of several subfamilies were given.

CPUE and Catch composition

Due to the differences in the amount and quality of available data provided from different sources and the difficulty of its standardizing, catch per unit effort (CPUE) was chosen as the most reliable index of fish trap efficiency despite the seasonal and spatial variations in values in local fisheries. However, some authors reported CPUE only in terms of abundance while others – only in terms of biomass. As the information on the average fish weight (kg) per trap haul was relatively more frequent, CPUE in terms of biomass was selected for further analysis.

The data from the papers dedicated to assessing various mesh sizes and/or other fish parameters were used only for the conventional/commercial traps if the CPUE and catch composition were reported separately. CPUE and catch composition of the alternative trap designs were omitted. In articles where authors did not stressed on the conventional trap used in trials, mean CPUE was calculated for several trap types which were in use in the region according to the text; catch rates provided for the trials of different operation modes were calculated as a mean of all given values.

Another problem associated with the comparative CPUE analysis was that in some SSF, e.g. in Kenya, the data was provided pooled for all types of gear used. Such efficiency indices, e.g. kg/fisher per trip, kg/fisher per hour, were omitted even if trap fishery was representing the significant part of the overall landings.

Total annual landings for local trap fisheries were reported only sporadically and thus could not be included in the analysis. Captured species if given were summarized and classified to family level.

RESULTS

Mesh size

The total estimated number of different mesh sizes used or tested in different fisheries worldwide since 1980 was 41, ranking from 12.5 mm square mesh tested by Newman & Williams (1995) on the Great Barrier Reef, Australia to 102 x 102 mm square and 76 x 152 mm rectangular mesh tested in Florida, US (Bohnsack et al., 1989). Nevertheless, of these 41 different mesh sizes, only 14 (34%) were actually used by fishers throughout the survey period. Other 27 were used for evaluation of fish stocks, and for determination of the optimum mesh size for the target species.

The most widely used mesh sizes across fisheries according to the frequency of mentions in papers was defined as 37-40 mm, followed by 50-51 mm and 30-32 mm for square and hexagonal and 25*51 mm for rectangular meshes.

Given the long history of trap fishing in the Caribbean and the heterogeneity of the region, predictably it has the highest diversity of the trap sizes in use (Fig. 3). Despite the variation in mesh sizes from 12.5 to 64 mm, 30-32 and 38 mm wires still remain the most widely used, followed by 25 mm. Traps with mesh opening of 50 - 51 mm and larger were reported only from Puerto Rico (Marshak et al., 2008), US Virgin Islands (Sheridan et al., 2006) and from Bermuda before the closure of the trap fishery (Luckhurst & Ward, 1987). In Lesser Antilles minimum mesh sizes vary from 30 mm in Martinique to 45 mm in Guadeloupe (Chakalall, 1995). Should be noted, that mesh sizes referred to as “commonly used” by some authors were lower, than the ones, legally accepted, e.g. in Barbados (Robichaud et al. 1999); the use of traps with mesh size below legal was also reported from US Virgin Islands (Garrison et al. 1998).

In the Canary Islands, the minimum mesh size depends on the size of the trap: 32 mm for the small traps and 51 mm for the large ones (García-Mederos et al., 2015; Barrera-Luján, 2016).

Similar requirements are in Tanzania: 30 and 50 mm for small and large traps respectively (Kamukuru, 2009). In Kenya, which had the same minimum mesh sizes as Tanzania, lately has been banned any gear including traps, with mesh size less than 63 mm (Hicks & McClanahan, 2012).

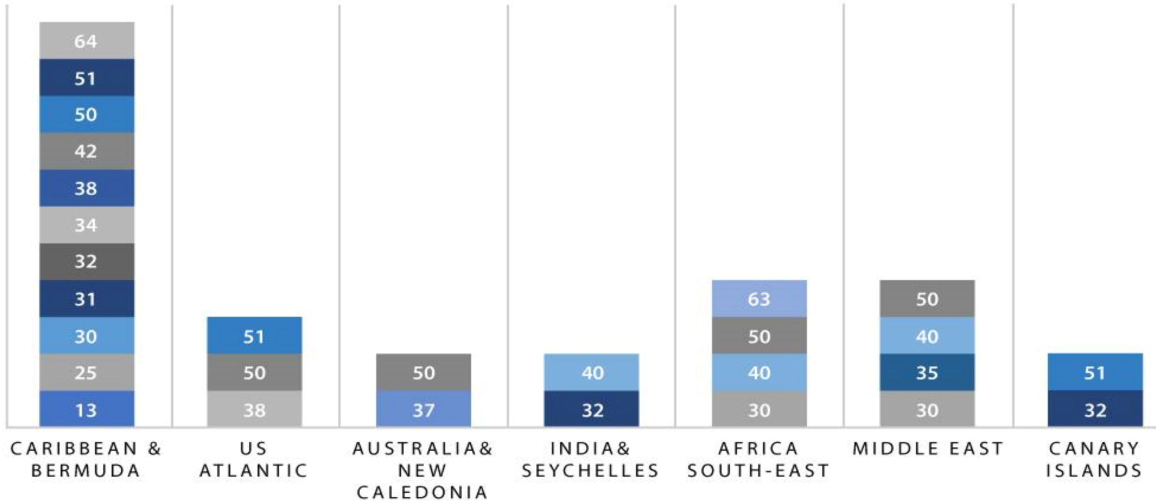

Figure 1. The most commonly used or the minimum legal meshes, mm (square and hexagonal) used in local trap fisheries.

Traditional gargoor traps used in the Arabian Gulf do not have a minimum mesh size, thus, the reported meshes, ranging from 30 to 50 mm are the most commonly used.

Although in the 1980s and 90s a lot of research on mesh size assessment was conducted in the Caribbean region (Fig. 2) and a shift to larger mesh have been repeatedly recommended, a clear trend in switching to increased mesh size was not detected. Additionally, recent information on the Caribbean trap fisheries is scarce, with only few papers published in the last 10 years.

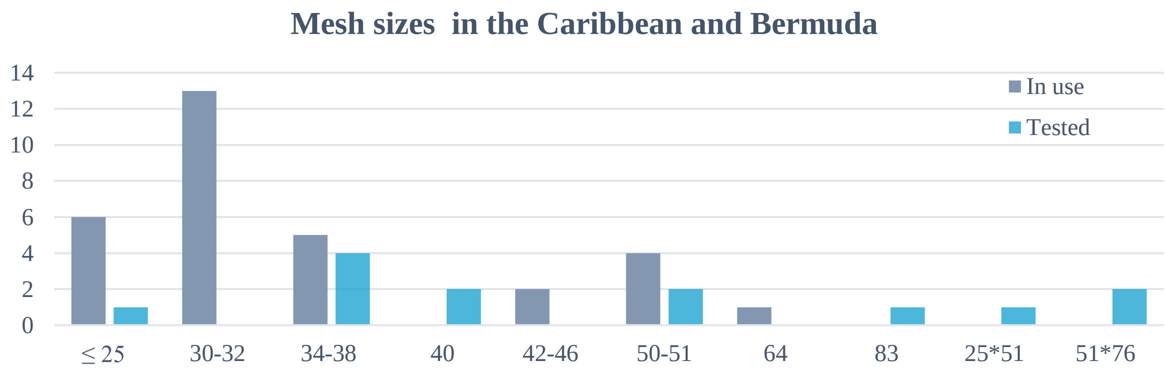

Figure 2. Mesh sizes (mm) "In use" are the ones reported from the Caribbean Region as minimum legal or most commonly used. Mesh sizes used in trials presented here DO NOT include the ones used in the local trap fishery during trials. Overlap in values occurred due to heterogeneity of fisheries within the region and temporal changes in legislation. The height of bars corresponds to the number of mentioning for the used mesh and the number of trials for the tested.

In US Mid-Atlantic, black sea bass *Centropristis striata* is the main target species in trap fisheries, so catches can be easily optimized through trap mesh size modifications. Current minimum mesh size is 38 mm (Fig. 3); many fishers voluntarily use uniform 51 mm mesh traps though the presence of at least one panel of 51 mm mesh is obligatory (Rudershausen et al., 2016).

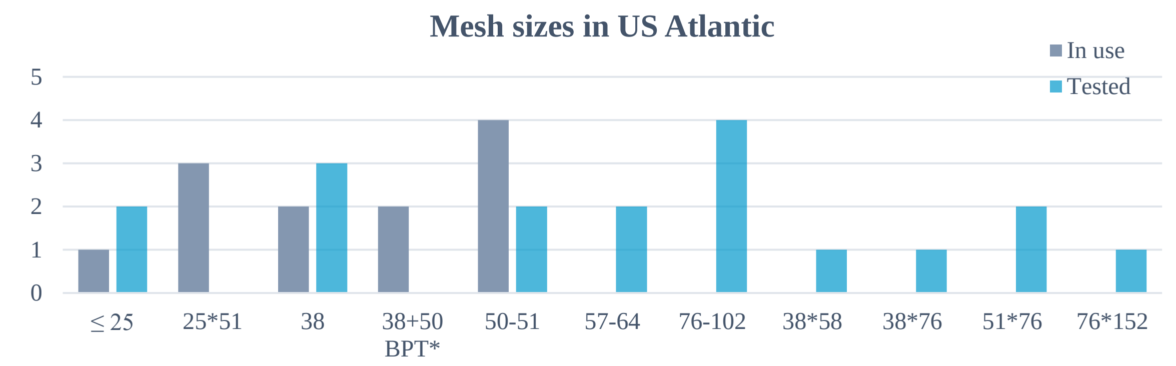

Figure 3. Mesh sizes "In use" are the ones reported from the US East and Southeast as minimum legal or most commonly used. Mesh sizes used in trials presented DO NOT include the ones used in the local trap fishery. Overlap in values occurred due to heterogeneity of fisheries within the region. The height of bars corresponds to the number of mentioning for the used mesh and the number of trials for the tested.
** Refers to back panel traps, with the back panel covered with 50 mm wire mesh, while other sides are covered with 38 mm wire.*

In Australia, the minimum and the most commonly used mesh is 50 mm hexagonal (Fig. 4). However, as it does not provide the proper size selection around one third of fishers incorporate 50 x 75 mm panels in their traps (Stewart & Ferrell, 2003).

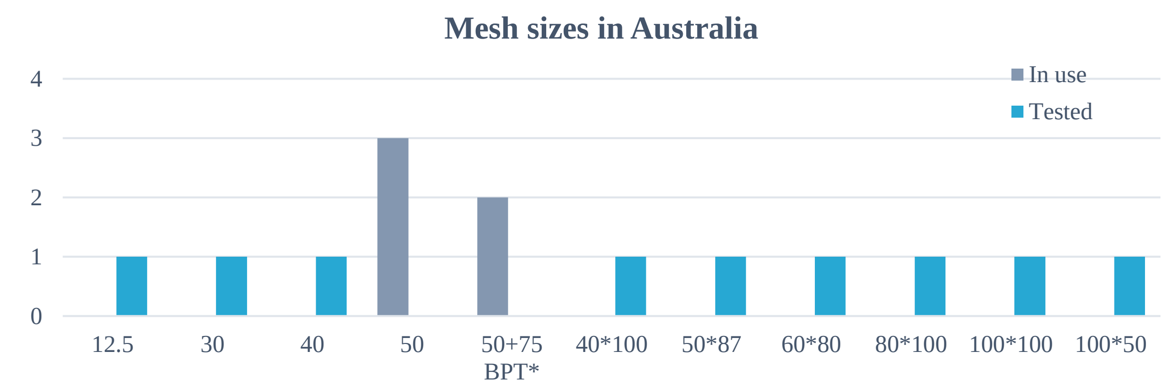

Figure 4. Mesh sizes used and tested in Australia. Mesh sizes "In use" are the ones reported as minimum legal or most commonly used. Mesh sizes used in trials presented DO NOT include the ones used in the local trap fishery.

The variety of mesh sizes and configurations tested reflects continuous effect to reach equilibrium between fisheries and fish stock conservation. In most cases research was focused on comparison of larger mesh with the ones commonly used in the fishery. Although experiments on very small mesh (0.5 inches or 12.5-13 mm) were conducted almost ubiquitously, it was mostly used in ecological studies for monitoring fish abundance and overall comparison of mesh size effectiveness in fish retention (Newman & Williams, 1995). Except for Jamaica, Haiti and western Florida in the 1980s, where that small mesh was still available for the fishers (Taylor & McMichael Jr., 1983; Ferry & Kohler, 1987; Sary et al., 1997) it was not incorporated directly in trap fisheries elsewhere even according to the data provided in earliest papers.

In India, trials of different mesh sizes were conducted in order to introduce gargoor traps from the Arabian Gulf to the local fisheries. However, of 4 mesh sizes tested the smallest one (40 mm) demonstrated better results, followed by 50 mm mesh, while catches in traps with two larger mesh sizes (80 mm and 100 mm square mesh) were null (Prajith et al., 2015). It should be noted, that as fisheries regulations do not restrict the mesh size, it has become a trend in eastern Indian wetlands to cover fish traps with mosquito netting (mesh size 1 mm), hence fishers catch everything including small fish fry, prawns and fish eggs (Manna & Bhattacharjya, 2009).

There was a tendency in using rectangular meshes during trials as they provide better size and species selectivity for fish with greater body depth.

Another trend is the use of back panel traps reported in the articles published since 2000 in Australia, United States and Mexico (Stewart &Ferrell, 2001, 2003; Nevárez-Martínez et al., 2008; Rudershausen et al., 2016, 2008). In the Southeast US regulations specifically required at least one outside panel to be composed by larger mesh (50.8 mm mesh on the back panel compared to 38.1 mm in other trap walls). This mesh configuration remains legal even after the increase of the minimum legal length of the target species (Rudershausen et al., 2016). In other regions, fishers elect to use back panel traps voluntarily as it reduces by-catch.

The smallest mesh sizes examined were in the Caribbean, where due to severe overexploitation of fish stocks incorporation of larger meshes would lead to drastic decrease in yields and landings. Both in Australia and in the US, where current mesh size regulations are insufficient for culling unwanted or undersized fish, the research was based on development of novel meshes and mesh configurations (Stewart & Ferrell, 2003;

Rudershousen et al., 2008, 2016). However, implementation of much larger mesh is unlikely due to reduced fishing power of traps and low catches.

Additionally, in multispecies trap fishery of Australia or the Caribbean, changes in mesh size would affect the entire catch composition and would lead to considerable loss in ingress of several target species (Newman & Williams, 1995; Jiménez & Sadovy, 1996). In US trap fisheries which mainly target black sea bass (*Centropistris striata)* its minimum legal size changes frequently (Rudershousen et al., 2016), thus the corresponding increase of minimum mesh size supposedly should occur more willingly. Except for Munro (1983), which back in 1980s proposed 6.6*5.1 mm mesh for Jamaican fisheries, given the reluctance of fishers to increase the mesh sizes of their traps, most researchers provide discrete or conservative recommendations over the optimum mesh size based on the results of mesh trials.

Munro (2007) mentioned progressive reductions in used mesh sizes across the Caribbean fisheries, however, it was not recognized by any other study. Except for the use of mosquito netting in traps in Indian wetlands (Manna & Bhattacharjya, 2009), the trend of switching to smaller meshes was not reported elsewhere. Moreover, sometimes fishers tend to utilize significantly larger mesh (64 mm instead of currently established minimum 38 mm reported from Puerto Rico (Sheridan et al., 2006), especially if they target particular species and it provides higher revenue withal reduces culling time. Additionally, long term experiment carried out by Sary et al. (1996, 1997) in an overexploited fishery of Jamaica demonstrated, that 3 years after implementation of larger mesh (38 mm instead of 32 mm) catches increased above the initial level both in terms of biomass and abundance.

Target species

The determination of a target species within multispecies fisheries can be challenging as anglers to some extend are able to modify fishing efforts and emphasize on species that would provide the higher economic opportunities. The priority of one species over others is defined by the actual availability of fish resources, its economic value and regulations (Asche et al., 2015).

Except for some trials for catching cod (*Gadus morhua*) in the Baltic Sea (Hedgärde et al., 2016), in absolute majority of cases demersal reef species are the ones targeted with fish traps (Figs. 5, and 6). Despite the fact that most of the target fish species belong to relatively few families, their current number is quite significant and varies greatly among the examined regions. The analysis revealed that although fishers tend to target large piscivorous fishes, the overall share of omnivorous and herbivorous fishes reaches around 22% globally, but varies from region to region.

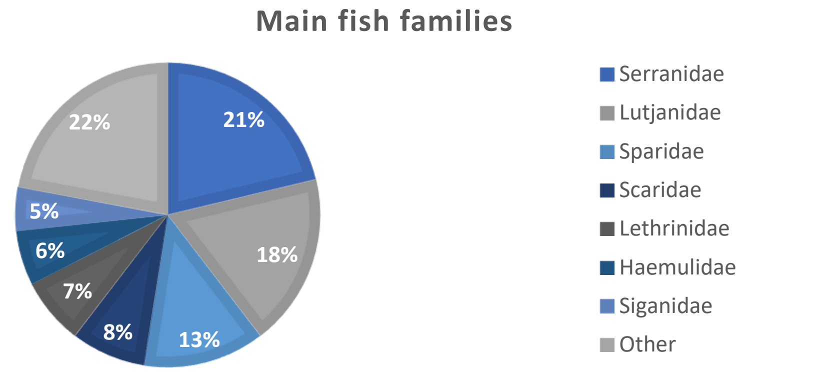

Figure 5. Main fish families targeted worldwide by fish traps. All species given were classified to family level.

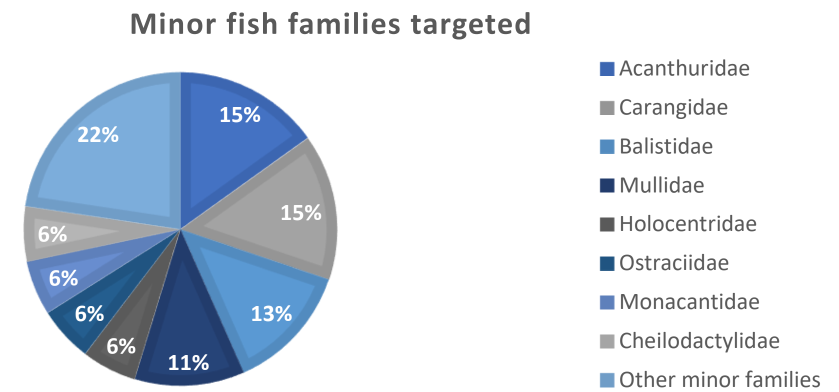

Figure 6. Minor fish families targeted worldwide. All species given were classified to family level.

Caribbean Region

Caribbean demersal trap fishery is the most diverse of all examined, with fish species belonging to 14 families being targeted (Fig. 7). Although large predatory fish such as snappers and groupers (Lutjanidae and Serranidae) are among the preferred species, due to multidecadal decrease in its landings they have been replaced with second- and third-class species (Marshak et al., 2007). Changes in species composition over the Caribbean resulted in ubiquitous shift to targeting previously less desirable herbivorous and omnivorous species, such as grunts (Haemulidae), large parrotfishes (*Sparisoma* spp. and *Scarus* spp.), surgeonfishes (Acanthuridae), goatfishes (Mullidae), etc. In some regions, fish belonging to such families as Balistidae, Holocentridae, Ostraciidae, Chaetodontidae, Sparidae, Labridae, Carangidae and Pomacanthidae are also among targeted.

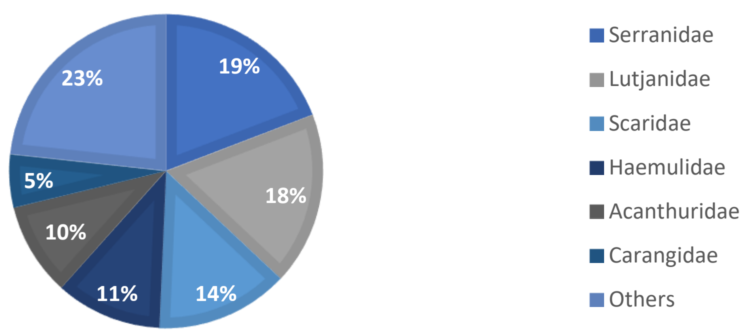

Figure 7. Main fish families targeted in the Caribbean.

US Mid & South Atlantic

In demersal fisheries of Mid-Atlantic Bight black sea bass (*Centropristis striata*) is the main target species, whose proportion in the total catch may reach 96% (Eklund & Targett, 1991). Commercial fisheries in South Atlantic are more diverse and in addition to black sea bass target snapper-grouper complex and other species (Fig. 8), of which vemillion snapper (*Rhomboplites aurorubens*), northern red snapper (*Lutjanus campechanus*), red porgy (*Pagrus pagrus*), grey triggerfish (*Balistes capriscus*), scamp (*Mycteroperca phenax*), etc. are among the most important (Bacheler et al., 2014; Bacheler & Smart, 2016).

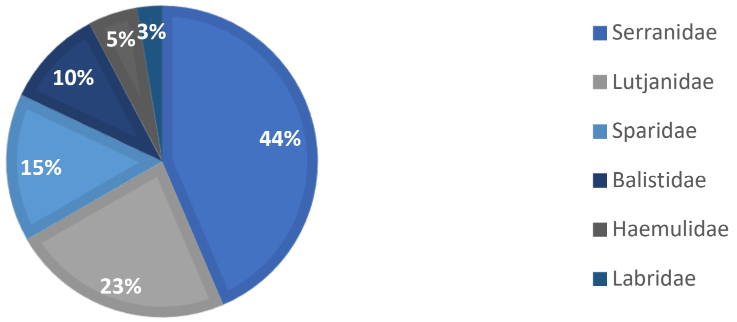

Figure 8. Main fish families targeted along the Eastern US Coast.

Indian Ocean

Several authors reported Serranidae, Siganidae, Lethrinidae, Scaridae as the most important families in the catch of artisanal fishers in the Indian coast as well as in Seychelles (Fig. 9). Within these families, orange-spotted grouper (*Epinephelus coioides*), spangled emperor (*Lethrinus nebulosus*), white-spotted spinefoot (*Siganus canaliculatus*), shoemaker and streaked spinefoots (*Siganus sutor and S. javus*) blue-barred parrotfish (*Scarus ghobban*) are considered to be of higher importance (Lal Mohan, 1985; Mariappan et al., 2016; Varghese et al., 2017), followed by the snappers *Lutjanus malabaricus*, *L. argentimaculatus*, *Pristipomoides filamentosus*, etc. (Murugan et al., 2014).

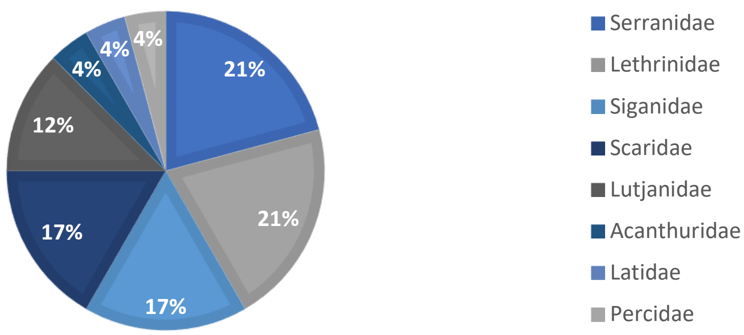

Figure 9. Main fish families targeted along the Indian coast and on Seychelles.

Middle East

In the Arabian Gulf, fishers target mostly large predatory fishes. Orange-spotted grouper (*Epinephelus coioides*) and painted sweetlips (*Diagramma pictum*) have been reported as key species in the United Arab Emirates (Grandcourt et al., 2011), followed by emperors (*Lethrinus lentjan* and *L. nebulosus*), porgies (*Argyrops spinifer* and *Acanthopagrus bifasciatus*) and jacks (*Gnathanodon speciosus* and *Carangoides bajad*) (Fig. 10). Similar species were reported from Kuwait (Chen et al., 2012). In the Mediterranean coast of Turkey along with species belonging to Epinephelinae (*Mycteroperca rubra*, *Epinephelus marginatus*, etc.) and Haemulidae (*Pomadasys incisus*), fishers target Sparidae species such as, common dentex (*Dentex dentex*), seabreams (*Diplodus vulgaris* and *Pagrus coeruleostictus)* and common pandora (*Pagellus erythrinus*) (Çekiç et al., 2005).

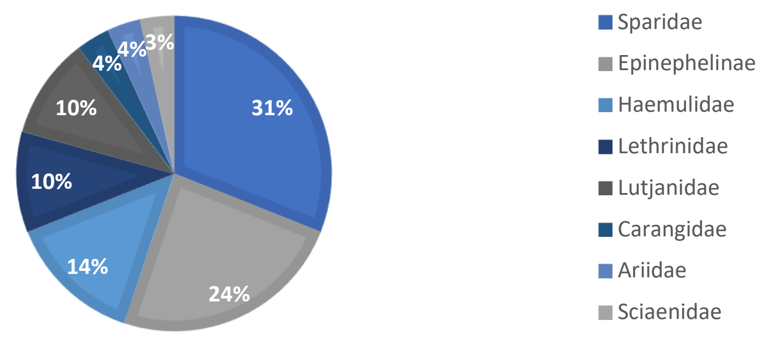

Figure 10. Main fish families and subfamilies targeted in the Middle East.

African East Coast

One of the most important species along Eastern Africa is *Siganus sutor* (Siganidae). It comprises around 85% of total trap catches in Tanzania (Kamukuru, 2009) and one of the most abundant species landed by Kenyan trap fishery (Hicks & McClanahan, 2012; Tuda et al., 2016). Among the other important species listed were *Lethrinus harak*, *L. lentjan*, *L. mahsena, L. miniatus (Lethrinidae), Leptoscarus vaigiensis (Scaridae), Parupeneus macronemus (Mullidae*), and other reef fish species (Fig. 11). It was noted that while landings of most commercially important families (Lethrinidae, Lutjanidae, etc.) decline, other families (Scaridae, Acanthuridae, etc.) demonstrate increase in landings, followed by increase in their market value (Kaunda-Arara et al., 2003).

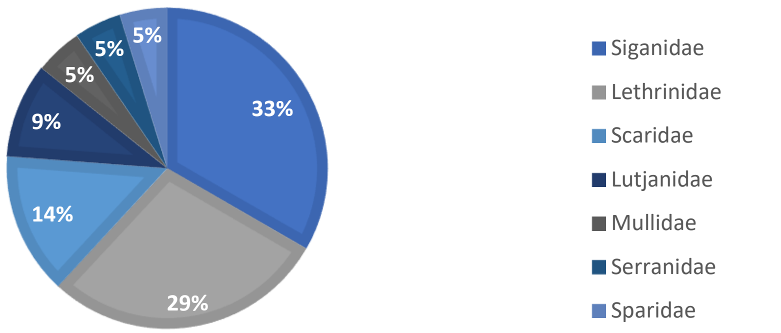

Figure 11. Main fish families targeted along the Eastern African Coast.

Australia

There are two main trap fisheries in Australia: Pilbara Trap Managed Fishery and New South Wales (NSW) demersal fishery. Snappers and emperors are the main target species of the Pilbara region, in particular: *Pristipomoides multidens*, *Lethrinus hutchini*, *L. nebulosus*, *Lutjanus erythropterus*, *L. malabaricus* and *L. sebae*, followed by *Epinephelus multinolatus*, and *E. bilobatus* (Serranidae) (Department of the Environment and Heritage Assessment report, 2004; Newman et al., 2008; Langlois et al., 2015).

Completely different species composition is targeted in NSW, which is comprised by rubberlip morwong (*Nemadactylus douglasii*, Cheilidactylidae), sparids tarwine (*Rhabdosargus sarba*), silver seabream (*Pagrus aurata*), yellowfin bream (*Acanthopagrus australis*), silver trevally (*Pseudocaranx dentex*, Carangidae), ocean leatherjackets (*Nelusetta ayraudi*, Monacanthidae), pearl perch (*Glaucosoma scapulare*, Glaucosomatidae), among others (Stewart & Farrell, 2002; Stewart & Hughes, 2008).

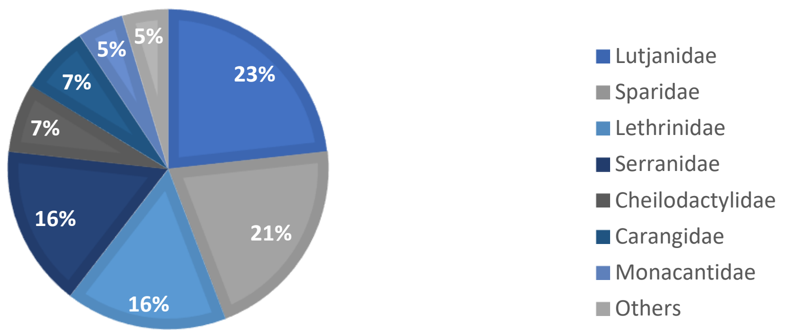

Figure 12. Main fish families targeted along the Australian Coast.

Canary Islands

Fish stocks around the Canary islands are considered to be overexploited, with scarcity of highly valued predatory fish species (Serranidae and Haemulidae). Thus, only the limited number of fish species belonging to Scaridae, Sparidae, Monacanthidae, Pomacentridae and Mullidae can provide relatively high revenue and are targeted by local fishers (Fig. 13). Of them, *Sparisoma cretense* (Scaridae), *Mullus surmuletus* (Mullidae), *Dentex gibbosus*, *Pagellus* spp., *Diplodus* spp. (Sparidae) and some others are of pivotal importance (García-Mederos et al., 2015).

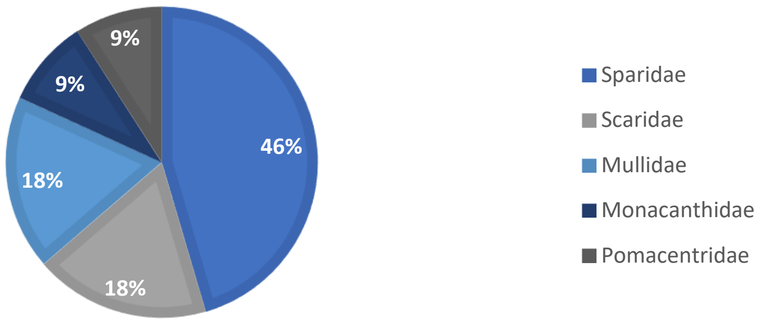

Figure 13. Main fish families targeted in the Canary Islands.

CPUE and Catch composition

Catch rates varied significantly both spatially and temporally as the article compilation covers a period of more than 35 years. However, differences in catch rates may not only reflect relative fish abundance and point overfishing, but also occur due to the variations in operation mode, trap design, area fished, etc. In general, mean CPUE comprised a wide range of values, with minimum of an average of 0.16 kg per trap in the Canary Islands and maximum of 23.9 kg per trap per day in Australia (Fig. 14).

<u>US South and Central Atlantic</u>

The average CPUE from the US Atlantic coast was 5.64 kg per trap. However, there were significant differences in catch rates. In US Middle Atlantic, the average catch was 2.2 kg/trap in 2002 (Shephert et al., 2002) and two years later was reported to be 13.6 kg/trap (Fisher et al., 2004). Rudershausen (2008) informed that the average number of 21.6 fish individuals to be retained per trap, which is within the range recorded for the fish abundance caught in two previous studies (8.1 *vs* 32.2 fish per trap haul in 2002 and 2004, respectively) (Shephert et al., 2002; Fisher et al., 2004).

In the US South Atlantic, the mean catches fluctuated temporally from 4.22 kg (Taylor & McMichael Jr., 1983) to 1.97 kg/trap (Sutherland, 1991) and then increased again to 6.3 kg/trap (Harper et al., 1994).

Black sea bass (*C. striata*) is dominating the catches in US Mid Atlantic. Extreme dominance of this species was reported by Eklund & Targett (1991), reaching 96% of the entire catch composition and later proved in trials, i.e. 94-95% of the catch, held by Cullen & Stevens (2017). Fish traps account for 38-45% of the total black sea bass landed in the Mid. Atlantic (Shepherd et al., 2002; Fisher et al., 2004), thus although such high catch rates are seldom reported, dominance of this species in traps in the region is doubtless (Rudershausen et al., 2008; Bacheler et al., 2013*b*).

In US Southeast during 1980s and 1990s primary commercial species, belonging to Serranidae, Lutjanidae, Labridae, etc. dominated the catch, making up 66-77%; Serranidae were the most abundant (49.8% of total weight), mainly with the species *Epinephelus morio* and *Mycteroperca bonaci* (Taylor & McMichael Jr., 1983). In the period 1990-2011, the most commonly caught target species were black sea bass (*C. striata*), red porgy (*Pagrus pagrus*), gray triggerfish (*Balistes capriscus*), and vermilion snapper (*Rhomboplites aurorubens*), while tomtate (*Haemulon aurolineatum*), bank sea bass (*Centropristis ocyurus*), sand perch (*Diplectrum formosum*) and *Stenotomus* spp. were among the most abundant non-targeted species. However, none of the species accounted for more than 20% of the catch (Bacheler et al., 2013*a*). Moreover, Bacheler & Smart (2016) reported the decline in the total number of species and in number of individuals over the three decades. Abovementioned studies noted that non-targeted fish species declined to the larger extend than the targeted ones.

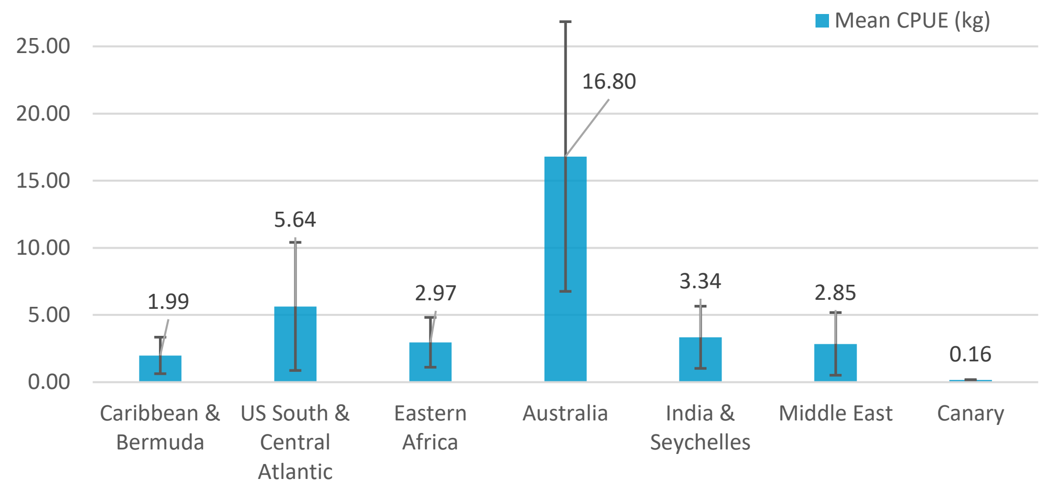

Figure 14. Mean catch rates (kg) of fish traps per trap haul reported from the main trap fisheries worldwide. Error bars represent standard deviation.

Eastern Africa

The mean CPUE value of 2.97 corresponds to South Africa, Kenya and Tanzania (Table 1). In Kenya, fishers often combine several gears, so in most papers the given catch rates were given pooled for all gear types. Thus, although total catch rates in Kenya varied from 2.0 to 5.4 kg per fisher per trip (Tuda et al., 2016; Samoilys et al., 2017), for fish traps CPUE was reported once and was 1.0-1.4 kg per haul (Gomes et al., 2014). In South Africa, Gray et al. (2007) reported catch rate of 4.9kg/trap (data only for the main target species *Pterogymnus laniarius*); CPUE of 2.8kg/trap was reported from Tanzania (Kamukuru, 2009).

The families Siganidae, Scaridae and Lethrinidae dominated the catch composition in Kenya. Although species making the majority of the catch as well as the reported percentages varied among studies, 3 – 6 species accounted for 57-75% of the total catch in most papers. Among them, *Siganus sutor, Leptoscarus vaigiensis, Lethrinus lentjan L. sanguineus, L. mahsenoides, Leiognathus equulus, Acanthurus tennenti, Scarus ghobban, Parupeneus barberinus*, etc. (McClanahan & Mangi, 2004; Hicks & McClanahan, 2012; Gomes et al., 2014; Tuda et al., 2016). The first species (*S. sutor*) accounts 65-75% of the total catch in Kenya (Samoilys et al. 2017), and even reaches 85% of the total catch in Tanzania (Kamukuru, 2009).

The catch composition was represented by 19 species in South Africa (Gray et al., 2007), and the main target species, panga (*Pterogymnus laniarius*) accounted for over 55% of the total catch. Only two other Sparidae species *Argyrozona argyrozona* and *Chrysoblephus laticeps* were of commercial importance and comprised 10% and 4% of overall abundance, respectively. The most abundant by-catch species was *Spondyliosoma emarginatum* that comprised 17% of the total catch. However, Gray et al. (2007) pointed out that catch composition was highly depth-dependent, with the percentage of target species caught increased from 15 to 78% with the increase of depth of trap deployment.

Australia

The catch weight reported from Australia is considerably larger than the ones reported elsewhere and seem to be relatively constant in time. Newman et al. (2008) reported around 23.9 kg per trap per day. Earlier Whitelaw et al. (1991) noted that the average catch rate of fish traps in North West Australia is about 10 kg per trap for 1 hour soaks. However, Whitelaw et al. (1991) claimed that the CPUE highly depend on the soak time and the calculated average of catch rate for different soak times resulted in mean CPUE of 9.7 kg per trap per day.

In North Australian fisheries 2 primary target species, red emperor (*Lutjanus sebae*) and goldband snapper (*Pristipomoides multidens*), dominate the catch. When pooled, catches for both species represented 58% of the total landings (Newman et al., 2008) Other Lutjanidae, Serranidae and Serranidae are considered as by-product species and account for 34% of the total catch. According to another study, 7 primary target species belonging to the families Lutjanidae, Lethrinidae and Epinephelinae accounted for 79% of the catch (Department of the Environment and Heritage, Australian Government, 2004). In both studies by-catch was not specified but considered to be low.

Lutjanidae accounted to 63% of the catch on the Great Barrier Reef, with *Lutjanus adetii* and *L. quinquelineatus* as the most abundant species; Lethrinidae and Serranidae made up about 16% and 3.6%, respectively (Newman, 1995). According to Sheaves et al. (1995) *L. russelli* (Lutjanidae) contributed to 31.4% of the catch, followed by the Sparidae *Acanthopargus berda* (30.4%) and *A. australis* (18.7%).

Pagrus auratus (Sparidae) comprised 48% of landings in NSW demersal fishery in the end of 1990s (Stewart & Ferrell, 2001). *Acanthopargus australis* (Sparidae) may account to 25-78% of landings depending on the location. The remaining catch is represented by *Nemadactylus douglasii* (Cheilodactylidae), *Nelusetta ayraud* (Monacanthide), *Pseudocaranx dentex* (Carangidae), among others.

India & Seychelles

Although the mean CPUE in India is about 2.6 kg/trap, catch weights were susceptible to temporal fluctuations: 0.34-0.48 kg/trap reported by Lal Mohan (1985), increased to 5.9 kg/trap according to Murugan et al. (2014), and subsequently dropped to 1.59 kg/trap (Varghese et al., 2017). However, Varghese et al. (2017) pointed out that improvement of CPUE, as well as, increase in catches per month since 1950s (0.14 kg/trap) according to Prabhu (1954) and 1980s (0.41 kg/trap) (Lal Mohan, 1985) is evident.

Catch composition is represented by almost a hundred of species; Scaridae (21-34%), Siganidae (34-44%) and Lutjanidae (22%), dominated the catch (Varghese et al., 2017; Beenamol et al., 2017). Lethrinidae comprised 11% of the catch, while Serranidae and Mullidae composed less than 10% each (Varghese et al., 2017). Beenamol et al. (2017) also pointed out that seasonality of catches is very pronounced, e.g. Siganidae comprising 83% of catches in November and zero catches in February-April.

In Seychelles, average catch rates decreased from 5.1 kg in 1988 (Sanders et al., 1988) to 3.7 kg per haul in 2011 (Robinson et al. 2011). Recently, Robinson et al. (2017) provided the CPUE of 1.7-7.4 fish individuals per trap, but did not state the average catch biomass.

The entire catch composition for Seychelles was not provided by former studies. Nevertheless, *Siganus sutor* (Siganidae) made up to 60% of the total annual landings of local fisheries in 2003-2012 (Robinson et al., 2017).

Middle East

Mean CPUE were available only for the local fisheries of Iran (Shabani et al., 2010) and Kuwait (Chen et al., 2012) and constituted 1.2 kg/trap and 4.5 kg per trap, respectively. However, in Kuwait the catch rate is subjected to seasonal fluctuations and changes from 2-5 kg to 8 kg in spring, when fishers target spawning aggregations.

Of 70 fish species caught in Kuwait, 60% were belonging to the first and second class commercial species and 91.5% of the catch was marketable. Of the total biomass almost 57% were primary commercial species (Sparidae, Serranidae, Haemulidae, Lethrinidae, Carangidae). *Argyrops spinifer* (Sparidae) and *Epinephelus coioides* (Serranidae) (both first class species) dominated the catch in terms of biomass in the overall catch, 13% and 12.2% respectively; discards comprised 2.56% of the total biomass (Chen et al., 2012).

In United Arab Emirates 65 fish species were caught, 10 of them made up 70% of the total weight. *Epinephelus coioides* (Serranidae) and *Diagramma pictum* (Haemulidae), the two most important commercial species, represented 68% of the total weight of marketable fish (Grandcourt et al., 2011). For both species occurrence of growth and recruitment overfishing was reported. However, there was no information provided for the rest of the catch, and by-catch represented 4.3% of the catch by weight and 6.0% by abundance.

Canary Islands

Coastal fishery resources are severely over-exploited, with over 21 fish species being under the fishing ban by the Canarian Law (Tuya et al., 2004; Riera et al., 2014). Mean catch rate is only 0.14-0.18 kg per trap per day (García-Mederos et al., 2015) which is lower than reported elsewhere worldwide. According to García-Mederos et al. (2015) the average CPUE in the 1970s was 2.2 kg per trap but then dropped, followed by overall decrease in fished traps and 65% reduction in size of operating fleet. The main fish species landed are Sparidae (21-60% depending on the location), *Sparisoma cretense* (10-35%) and *Mullus surmuletus* (13%). Highly commercial Serranidae (*Mycteroperca fusca* and *Epinephelus marginatus*) were caught only occasionally (García-Mederos et al., 2015). Additionally, although only 9 species classifies as "discard" and 91-99% of the catch was marketable, most of it was represented by juveniles and sub-adults (García-Mederos et al., 2015). Substantial amount of *Octopus vulgaris* were also landed, albeit this species is not directly targeted by fishers (Hernandez-Garcia et al., 1998).

Caribbean region

Although the mean catch rate in the Caribbean is 1.99 kg/trap haul, due to the heterogeneity of local fisheries within the region this value does not truly reflects the real situation. Notably, almost all the reported catch rates within the Caribbean region since 1980 were much lower than the ones reported from Bermuda (2.5-3.6 kg per trap haul) in 1975-

1987 (Butler, 1993), which however were low enough to implement a complete ban of trap fishing in 1990.

The minimum values of around 500 g per trap haul were reported from Guadeloupe (Beliaeff et al., 1992). The maximum CPUE of 4.18 kg per trap was reported from Puerto Rico (Stevenson & Stewart, 1980) but referred to the data obtained from the fisheries in 1973-1974 and subsequent studies demonstrate a sharp decline to 0.75 kg/trap in 1999 (Posada & Appeldorn, 1999). Recently, Marshak et al. (2008) although does not provide the weight of the catch, reports the average of 1.9 fish individuals per trap from Puerto Rico.

Similar trend has been observed in overexploited Jamaican fisheries: after relatively high catch rates of 2.2 kg/trap and 5.1 kg/trap (Munro, 1973, 1974) fish trap efficiency dropped and fluctuated between 0.5 kg (Koslow et al., 1988) and 1.06 kg per trap haul (Koslow et al., 1994). The trap exchange program conducted in one local trap fishery, when small mesh traps were replaced with larger mesh traps, resulted in increase of mean catch rates from 0.6 kg/trap in 1991 to 1.24 kg/trap in 1994 (Sary et al., 1997).

In US Virgin Islands the reported catch rates were stable, 2.0 kg per trap since 1998 (Garrison et al., 1998, 2004). Beets (2005) although does not provide the weight, reports the average of 4.35 fish individuals per trap, which corresponds with the average number of fish (4.7 individuals per trap) reported by Garrison (1998, 2004).

Fisheries of Barbados are the only one who have relatively high catch rates. Willoughby et al. (1999) and Robichaud et al. (1999) provided different catch rates for the conventional traps, 3.9 kg per trap and 0.9 kg per trap, respectively. Later, further research reported average CPUE of 4.0 kg per trap (Selliah et al., 2001) and 3.0 kg per trap (Baldwin, 2003). The pooled CPUE data for other Lesser Antilles islands such as, Martinique, Dominica, Guadeloupe and Saint-Lucia was less than a kilogram per haul (Gobert, 2000).

In general, catch composition throughout the Caribbean region varies depending on the exploitation status and fishing pressure applied within the location (Hawkins & Roberts, 2004; Hawkins et al., 2007). In heavily fished areas, e.g. Jamaica, Dominica, St. Lucia small-bodied fish species dominate the catch, while on lightly fished Bonaire coral reef support a broad variety of grouper species. The mean and overall biomass of Lutjanidae, Scaridae and Acanthuridae reduced significantly with the increase of fishing pressure. Moreover, according to Hawkins et al. (2007) all fish species trapped in Jamaica, were also caught in St. Lucia, but 38% of species caught in St. Lucia were lacking in traps set in Jamaican waters.

Low value species of Scaridae, Sparidae, Labridae, Mullidae, Holocentridae, and Acanthuridae accounted for 62% of catches in Jamaica, while Serranidae and Lutjanidae made up 2% and 23%, respectively (Koslow et al., 1994). Scarcity of large piscivores and herbivores was detected back in the 1980s (Koslow et al., 1988) and although later Munro (2003) stated the presence of Nassau grouper (*Epinephelus striatus*) and red hind (*Epinephelus guttatus*), juveniles of these species were not observed, which might eventually lead to their local extinction. Less desirable or "trash" fishes of Holocentridae, Pomacentridae, Chaetodontidae and Tetraodontiformae were the only ones that demonstrated increase in catches (Koslow et al., 1994, 1988).

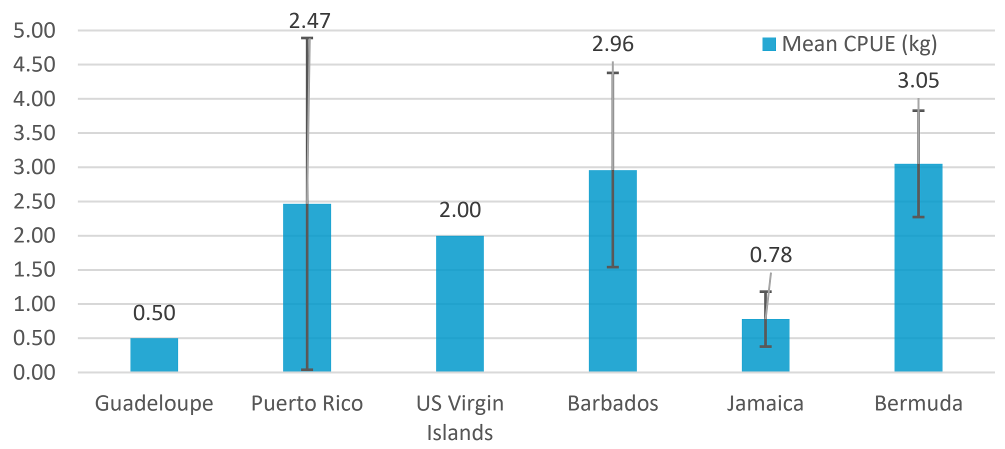

Figure 27. Mean catch rates (kg) of fish traps per trap haul reported from the Caribbean region. Error bars represent standard deviation.

Similarly in Puerto Rico, Scaridae, Acanthuridae, Ostraciidae, Haemulidae and Chaetodontidae comprised the bulk of the catch; snappers and groupers composed only 7% (Marshak et al., 2007). In response to overfishing fishers tend to move from coral reefs to algal plains and gorgonian hard bottoms (Schärer et al., 2004). Generally the contribution of fish trap fishery to total landings decreased from 71% in 1982 to 21-22% in early 2000s (Agar et al., 2005; Shärer et al., 2004).

Scaridae were also dominant in catches in Haiti (40% of the total number), followed by Pomacentridae 12.4%. The remaining 13 families comprised less than 10% of the catch each (Ferry & Kohler, 1987).

In US Virgin Islands species composition was dominated with Acanthuridae and Scaridae, which represented 61-71% of the catch, with *Acanthurus coeruleus, A. bahianus and Sparisoma aurofrenatum* being the most abundant (Wolff et al., 1999; Rogers & Beets, 2001; Garrison et al., 2004). Other families commonly caught are Ballistidae, Pomacanthidae, Haemulidae, and Lutjanidae; Serranidae were relatively rare and accounted for 6.8-7.9% of total biomass (Garrison et al., 2004; Sheridan et al., 2006). Nassau grouper (*Epinephelus striatus*) comprised 1% of the catch or less (Wolff et al., 1999; Rogers & Beets, 2001). Other Epinephelinae species, such as *E. adscencionis*, *E. afer* and *E. morio* were captured in very low numbers (Garrison et al., 2004) or not captured at all (Beets, 2005).

In Guadeloupe of 68 species caught, 19 represented 90% of the total abundance (Beliaeff et al., 1992). Herbivores (Acanthuridae, Scaridae) dominated the catch with *Sparisoma viride, S. aurofrenatum* and *S. chrysopterum* as the most abundant species. Highly valued species (Lutjanidae, Serranidae and Pomadasyidae) comprised 17% of the biomass. By-catch accounted for more than 35% of the fish caught and included fishes from the families Diodontidae, Aulostomidae, Chaetodontidae, and undersized individuals of marketable fish species.

According to Baldwin (2003), 9 out of 37 fish species caught in Barbados accounted for 86% of the catch by number and belonged to Chaetodontidae, Acanthuridae, Haemulidae, Lutjanidae and Scaridae. However, of them more than a half were juveniles, with the number of immature fish caught reaching 97% for ocean surgeon (*Acanthurus bahianus*). In trials set by Selliah et al. (2001) parrotfishes (*Scarus taenopterus* and *Sparisoma aurofrenatum*), and ocean surgeon (*Acanthurus bahianus*) were the most abundant, each accounting for 12% of the total number.

Derelict Fish Traps (DFT) and ghost fishing

Fish traps and pots are considered to be one of the most common derelict gear worldwide (Macfadyen et al., 2009; Jeffrey et al., 2016). The research was mainly focused on the assessment of derelict crab and lobster pots, which are known to have a significant impact on shellfish populations. Surprisingly, after the separation of the data presented for shellfish trap fisheries, the lack of research published for derelict finfish traps was evident. Despite the concerning number of gear being lost annually only a few studies were conducted in order to quantify the catch rate and fish mortality, and associated economic and ecological impacts for fisheries and ecosystem.

The estimated number of fish traps lost or discarded vary considerably, e.g. in US from an average of 10% of gear being lost in Virgin Islands according to Clark et al. (2012) to 85% reported by Sutherland (1991) in Florida. In general, according to Taylor & McMichael Jr. (1983) in Florida trap loss annually can accounted for 4 – 200% of the total number fished. In fact, the main reason for the fish trap ban in Florida state waters in 1980 was the concerns about the amount of gear ghost fishing and high by-catch rates (Newman et al., 2011). Similarly, long-term ghost fishing, non-functioning escape-panels and lack of compliance with fish trap regulations were one of the main reasons that stipulated a complete ban of fish traps in the U.S. federal waters of the Gulf of Mexico in 2007 (Newman et al., 2011).

Al-Masroori et al. (2004) reported the average annual trap loss of around 20% of fish traps in Sultanate of Oman. However, several years later the loss rate increased to as many as 88% (Al-Masroori et al. 2009). Field experiments conducted by Chen et al. (2012) in Kuwait waters resulted in loss of 60% of gargoor traps tested from March to September. Although it is unknown to what extend this data could be applied for the entire trap fishery, still it provides an overview of the gear loss.

For the Caribbean region the information available only for Guadeloupe, where the average trap loss accounts for about 20,000 per year (Clark et al., 2012). Some authors, such as Marshak et al. (2008) and Breen (1990) indicate a high number of ghost-fishing traps, however it remains unquantified and their current impact unexamined.

In South Korea almost 43% of fish traps are lost annually, which in terms of weight makes up to 11,436 tonnes per year (Kim et al., 2014).

The amount of traps lost at sea in Australia is unknown but considered low due to short soak times (Department of the Environment and Heritage, Australian Government. 2004; Neman et al., 2008). Fishers tended to leave their traps at sea between the fishing trips, however the gear was left with doors open and thus believed to cause very little or no mortality even if lost (Newman et al., 2008).

Table 2. Estimated number of fish traps lost annually		
Location	**DFT lost (%)**	**Reference**
Florida, US	4 – 200% annually; 63% in average	Taylor & McMichael Jr., 1983
Florida, US	85%	Sutherland et al., 1991
Florida, US	20 – 100%	Sutherland & Harper, 1983
Virgin Islands, US	10%	Clark et al., 2012
Portugal	24%	Erzini et al., 2008
New Caledonia	50%	Anon., 1985
South Korea	43%	Kim et al., 2014
Oman	20%	Al-Masroori et al., 2004
Oman	35 – 88%	Al-Masroori et al., 2009
Kuwait	60%	Chen et al., 2012
Turkey	9%	Yildiz & Karakulak, 2016

Based on the survey made by Al-Masroori et al. (2009) in Gulf of Oman, 6 main reasons for fish trap loss were stated. Interference with other gear and theft or/and vandalism were the main reasons according to the interviews and accounted for 33.3% and 32.9% respectively. The other reasons in descending order of importance were collision with boats (21.9%), and environmental condition: strong currents (5.9%), bad weather (4.2%) and bottom snag (1.7%). Gear conflict was also the main cause (41% of all cases) for trap loss indicated in Portugal (Erzini et al., 2008). Theft and vandalism were identified as the major problem in the US Virgin Islands (Clark et al., 2012). Notably, 100% of trap loss reported from Turkey were caused by bottom structures (Yildiz & Karakulak, 2016). One of the important reasons of gear loss is the reluctance of fishers to use buoys, as it makes the traps more conspicuous for thieves and vandals (Schärer et al., 2004; Clark et al., 2012). According to Kojis & Quinn (2006), in US Virgin Islands fishers set their strings of traps unbuoyed also in order to minimise losses due to collision with boats and entanglement with vessel propellers. Chen et al. (2012) noted that it is a common practice in Kuwait to deploy traps without a buoy and record the trap position using a GPS. Similarly, Al-Masroori et al. (2004) reported that while some fishers mark gear location with buoys, others tend to rely on their experience to relocate the deployed traps.

The dependence of trap loss on the depth was also variable: in New Caledonia as well as in Sultanate of Oman the gear was mainly lost in shallow water (Anon., 1985; Al-Masroori et al., 2009), probably due to increased trap movement with higher wave action. Taylor & McMichael Jr. (1983) indicated the opposite tendency. Positive correlation between soak time and trap loss was indicated in several studies (Taylor & McMichael Jr., 1983; Chen et al., 2012). In Kuwait the increase in soak time from 10 to 60 days resulted in 7-fold increase in the number of lost traps. Moreover, increase in soak time up to 100 days would result in loss of 88% deployed fish traps (Chen et al., 2012). Low estimated number of DFG in Australia is believed to be the reason of short soaks (Newman et al., 2008).

Deliberately discarded old or damaged fish traps contribute for about 9% in US Virgin Islands (Clark et al., 2012) and almost 45% in South Korea (Kim et al., 2014). Interviews organised by Al-Masroori et al. (2009) in Oman revealed that although 70.7% of the fishers are aware of the negative impact of DFT on the environment and fisheries resources, 64% of them still continue dumping their gear into the sea.

Catch composition in lost traps were similar to the one observed in active gear, with 81-94% of the total biomass comprised with commercial species (Al-Masroori et al., 2004; Clark et al., 2012). Based on field surveys conducted in Australia Newman et al. (2011) stated, that changes in catch composition in DFT over time were insignificant, with the increase of non-target species; additionally, in contrast with previous studies, the catch rate did not decline with the increase of soak time and ingress of the main target species after 12 days soaks was the same as for a few hour soaks.

Despite the considerable amount of the DFT, in most cases the associated ghost-fishing mortality was unexpectedly low. Arthur et al. (2014) estimates only 5% of the DFT being capable of ghost fishing, with 76 fish individuals caught per DFT per year. Most of the DFT observed with the remotely operated vehicle (ROV) in US Virgin Islands were either damaged and degraded or had their escape panels open (Clark et al., 2012). However, almost 22% (5 out of 23) of lost traps found in field surveys conducted by Sutherland et al. (1983) were undamaged and continued fishing. Renchen et al. (2014) provides the approximate mortality of 2% of fish in traps with closed escape panels and zero mortality for the traps with open escape panels. Similarly, Clark et al. (2012) reported that 95% of the fish were able to escape and only 5% died. Several authors observed fish entering and exiting from the funnel of lost traps on a regular basis (Luckhurst & Ward, 1987; Renchen et al., 2012). Munro (1974) documented about 50% escape rate for fish traps soaked for 14 days. However, extrapolation of this egress rate for longer soaks would not result in zero ingress as traps still act as fish aggregating devices, providing shelter and food in a result of self-rebaiting (Bullimore et al., 2001; Newman et al., 2011). Moreover, the estimated egress for sablefish (*Anoplopoma fimbria*) was considered null for traps without any escape mechanism incorporated into the trap, which resulted in estimated loss of 73-1,331 tonnes of fish to ghost fishing in 1977-1983 (in contrast with 1,085-4,409 tonnes of sablefish as annual commercial catch) (Scarsbrook et al., 1988). According to trials, incorporation of time-release escape devices of various designs allowed to reduce would allow 56-99% of fish escape, which would have resulted in almost complete elimination of ghost fishing (Scarsbrook et al., 1988). Extremely high ghost fishing mortality reaching 95% or 1.34 kg per trap per day was assumed in Oman (Al-Masroori et al., 2004). Nevertheless, the accurate escape rate from traps was not examined, and based on personal observations of Al-Masroori et al. (2004) was estimated as 10%.

Fish mortality in traps may occur due to predation, starvation and injuries. Signs of physical deterioration, such as skin abrasions, bruising and skin removal from the snout and nape, are common and associated with the attempt of fish to escape through the mesh (Taylor & McMichael Jr., 1983; Newman et al., 2011; Renchen et al., 2012). According to Clark et al. (2012), around 5% of fish observed in traps had skin wounds or abrasions, and 20% of dead fish had abrasions on their snouts. 47% of dead fishes observed by Renchen et al. (2014) had wounds due to repeated collisions with the mesh. Taylor & McMichael Jr. (1983) stated that 17% of trapped fish were dead or injured after 2 months soak, compared to 27% of dead or injured fish after 20-day soaks. Regardless of fish escapement, cryptic fishing mortality still might occur due to stress or wounds obtained in traps and associated infections.

The estimated time a fish trap can continue ghost fishing depended on durability of materials and ranged from 3 months to 2 years (Al-Masroori et al., 2009; Arthur et al., 2014; Jeffrey et al., 2016). However, Arthur et al. (2014) stated, that these numbers might be conservative, as after conducted 1–2 year surveys were finished, some traps were still capable of fishing. Clark et al. (2012) reported that after one-year study, the traps did not demonstrated the signs of degradation despite being covered by biofilm and various fouling communities.

Deep water traps due to lower fouling rate and reduced wave action were deteriorating more slowly than the ones in shallow zones and thus continued ghost fishing for longer time periods; thereby the overall decline in ghost fishing with time was proven to be very slow (Matsuoka et al., 2005; MacFadyen et al., 2009).

Measures to address DFG include prevention of gear loss, mitigation of its impact and removal of already lost fish traps. Among the preventive measures that could be applied are appropriate gear marking, use of buoys, onshore collection of old/damaged/retrieved gear, reducing fishing effort and introduction of specific zones for each type of gear (Al-Masroori et al., 2009; MacFadyen et al., 2009; Clark et al., 2012). Numerous initiatives have been developed to provide appropriate facilities in ports and harbours for unwanted gear reception. The buyback program launched in South Korea to encourage fishers to bring back to the port DFG for a small incentive fee resulted in collection of 29,472 tonnes of marine litter from 2004 to 2008 (Morishige, 2010).

The mandatory use of biodegradable escape mechanism as a main tool for reduction of ghost fishing impact was continuously recommended in a number of fisheries (Scarsbrook et al., 1988; Al-Masroori et al., 2009 Newman et al., 2011; Clark et al., 2012). Such trap modifications are very effective, inexpensive and easy to install and service (Breen, 1990), however, sometimes fishers do not easily accept them (Kumpf, 1994). MacFadyen et al. (2009) stated that 40% of commercial traps lifted from the ocean floor during the trap recovery programme in Washington State did not have the required “rot cord”. The rot cord or biodegradable fasteners keep the escape panel shut and are expected to deteriorate in a reasonable period of time and reduce the impact of ghost fishing. According to Clark et al. (2012), it took around 3.59 months for the jute twine to break and for the escape panel to open in new traps with new rot cords, which represents the worst case scenario.

Clean-up or gear recovery projects specifically aim at removal of DFG, however, they are costly and funding for such programs is not consistently available (NOAA Marine Debris Program, 2015). According to Macfadyen et al. (2009), voluntary short-term closures and clean-ups in trap fisheries in a number of states within the Gulf of Mexico proved to be successful and resulted in retrieval of about 16,000 of traps in 2004-2007. DFT recovery could be facilitated with the improvement of DFG detection mechanisms (Clark et al., 2012; NOAA Marine Debris Program, 2015) as well as better reporting of the gear loss, as such about 80% of losses are reported in Norway, where this procedure is obligatory (Macfadyen et al., 2009).

Nevertheless, Clark et al. (2012) pointed out that traps fouled with different invertebrate communities acted as a surrogate reef habitat, attracted various finfish and shellfish species and thus enhanced biodiversity. Thereby, the assessment of the actual contribution of the DFT to the habitat should be made prior to removal actions undertaken.

Management

As the effective area fished by a single trap ranges from 25 to 348m^2 (Miller & Hunte, 1987; Acosta et al., 1994), big number of utilized traps coupled with poor management has led to severe overexploitation of marine resources in the Caribbean coral reefs, which undergone a complete reordering of local fish communities (Garrison et al., 2004; Hawkins et al., 2007). Additionally, in absolute majority of cases fishery regulations do not restrict the number of traps fished. Fisheries regulations and management are often difficult to enforce in areas facing overfishing, as anglers tend to ask for easing of the current regulations and at the same time want to increase not only gross captures, but also the mean lengths of the caught fish (Munro, 2007; García-Mederos et al., 2015). Successful implementation of any management option highly depends on the compliance of fishers, as well as, on effective surveillance and law enforcement (Hicks & McClanahan, 2012).

The vital long-standing purpose of fisheries management is to reduce the catch of juvenile fish of the target species, which currently may reach more than 95% for some fish species caught (Stewart & Ferrell, 2001; Grandcourt et al., 2011; Hicks & McClanahan, 2012). Despite local variations, the regulation of fish trap mesh size and application of escape gaps and escape panels are the most common and feasible approaches. In some countries, such as US Virgin islands, mesh size is set by regulations (Beets, 2005), while in others, as United Arab Emirates fishers are not restricted with the minimum mesh size, but the presence of an escape panel of a particular size is required (Grandcourt et al., 2011). Notably, probably due to apprehension of short-term decrease in fishing yields, sometimes the adopted minimum mesh sizes were below optimal ones and reflected the most commonly used meshes in the fishery (Mahon & Hunte, 2001). Increasing the mesh size of fish traps is expected to improve size and species selectivity, reduce by-catch, including juveniles of the target species, prevent or reduce growth and recruitment overfishing and thus result in long-term improvement of catches and yields after the recovery of fish populations (Sary et al., 1997; Mahon & Hunte, 2001).

In US South Atlantic fish traps, with the exception for small fish traps for capturing black sea bass (*C. striata*) were outlawed in federal waters 3 miles off the coast (NOAA Marine Debris Program, 2015), so gear requirements are well described and include minimum mesh size (38 mm), presence of at least 2 escape vents and ¨ghost panel¨ with biodegradable fasteners. Additionally, at least one outside panel of the trap should be composed of a larger mesh (51mm) (Rudershausen et al., 2016). Along with the abovementioned gear restrictions, each trap and buoy should have an identification number, and minimum legal length for the black sea bass, as well as annual commercial quotas were also set by regulations (SAFMC, 2012; SAFMC, 2013).

In the Caribbean area, gear restrictions include minimum mesh size and presence of an escape panel of approved size and design. Despite local differences in mesh sizes, this approach was followed by trap fisheries of Barbados (McConney, 2011), US Virgin Islands (Beets et al., 2005) and Puerto Rico (NOAA-CFMC, 2015). Additionally traps in Barbados and U.S. Caribbean EEZ must be deployed with a buoy, and have an identification mark (FAO, 2011; NOAA-CFMC, 2015). The Fishing Industry Act, approved in Jamaica back in the 1976 probably remains the main document regulating fishing activity; however, it does not contain any restrictions for the fishing gear (The Fishing Industry Act, 1976). Furthermore, while local conch and lobster fisheries in Jamaica were getting more attention from managers (Kong, 2003), troubles in implementing management and conservation tools for finfish fisheries were continuously reported (Munro, 1983; Haughton, 1987; Bellwood et al., 2004).

In Seychelles, mesh size regulation is the only management option applied (40mm), yet fisheries are open access and otherwise unmanaged (Robinson et al., 2011). For Indian fisheries published information regarding current fishery legislation and applied management was very scarce if present. According to Mathew (2009), Indian fisheries remain highly unregulated: the fishery is open-access, besides, there is no licencing system of artisanal crafts and in general the concept of responsible fishing is totally lacking.

In Kenya, current fishery laws prohibited the use of mesh size smaller than 6.35 cm in all types of gear used. However, field survey conducted by Hicks & McClanahan (2012) observed that these regulations were not properly enforced, and the actual mesh size used in the fishery was about 5 cm. The fisheries were open-access, which caused uncontrollable fishing effort (Kenya Fisheries Governance, 2011).

In United Arab Emirates only the presence of an escape panel is required, while the minimum mesh size generally is unregulated (Grandcourt et al., 2011). However, the trials, organised by Grandcourt et al. (2011) observed that escape panels of currently accepted size and design provided only marginal benefits in terms of the originally set objectives, as retention of juveniles of the key species was similar to the fish traps set without an escape panel. In Kuwait fisheries, neither mesh size limitations nor escape panels were adopted in the gargoor fishery. However, it is believed that the commonly used mesh size (50 mm) allows sustainable harvesting of local resources (Chen et al., 2012).

Nevertheless, it was proven that only gear regulations (mesh size regulations, incorporation of escape panels and escape gabs) are insufficient for the preventing and/or mitigating the adverse effects of fishing, improving the state of fish stocks and reducing fish mortality (Grandcourt et al., 2011; Tuda et al., 2016).

There are only few countries which restrict the amount of fish traps that can be deployed. In the Canary Islands, the maximum number of authorized fish traps allowed per boat is 30, however, temporarily until 2019 it was allowed to deploy 75 fish traps per boat in Gran Canaria and 40 in Lanzarote (BOE, 2015). For the Australian Pilbara Trap Managed Fishery the primary tool for effort limitation is the system of annual quotas; the number of traps per boat is not restricted, but regulations curtail the number of trap deployment per year. Other management arrangements include limited entry, establishment of defined fishing areas and mandatory use of Vessel monitoring System (VMS) (Department of the Environment and Heritage, Australian Government, 2004). Similar system management approach was reported from the Australian Northern Demersal Scalefish Fishery (Newman et al., 2008, 2011): "each license is allocated an annual effort quota in 'standard fishing days' that is based on the use of 20 traps or less, when the number of traps being fished increases, the number of allowable standard fishing days declines". Newman et al. (2008) also reported that in 2002 most fishers used 20-40 fish traps per boat. These numbers did not differ much from the ones reported by Agar et al. (2008) from the US Caribbean: 25.7 – 68.1 traps hauled per trip depending on the location. Despite the fact that the estimated amount of traps owned per fishermen is much larger and reaches 300 (Scharer et al., 2006), fishers do not operate all their traps simultaneously (Scheridan et al., 2006). Nevertheless, according to Kojis & Quinn (2006) in US Virgin Islands there was over a 500% increase in the total number of traps used since 1930 (1,600 in 1930 *vs* 8,642 in 2003), while the number of fishers remained stable, which indicates a substantial increase in fishing pressure. Beets (2005) stated that the current fishing effort was too high, and resulted in declines in catches, decline in sizes of captured fish and near complete loss of some species. At the same time in Australia, the introduction of management control has led to an increase in catch rates due to limited fishing effort and improved efficiency (Newman et al., 2008).

Another common management tool applied to prevent growth and recruitment overfishing is establishment of minimum legal lengths (MLL) for commercial species. One of the limitations of this management option is that size limitations usually are established only for the key target species and ignore others of smaller commercial value. So, in the Canary Islands, the planehead filefish (*Stephanolepis hispidus*) is the most abundant species caught in traps, yet its minimum size at capture has not been already set (R. Riera, pers. comm.). Additionally, some of the sizes were established long ago and do not correspond with the current knowledge of fish reproductive biology and should be reviewed (Barrera-Luján, 2011; García-Mederos et al., 2015). Similar situation in Australia: in NSW only 6 of 13 most important species landed have MLL, while such ´growth overfished´ species as silver trevally (*Pseudocaranx dentex*) and pearl perch (*Glaucosoma scapulare*) remain unregulated (Stewart & Hughes, 2008); in Pilbara Region, only 3 main target species have stated MLL (red emperor *Lutjanus erythropterus*, spangled emperor *Lethrinus nebulosus*, and blue-spot emperor *Lethrinus hutchinsi*) (Department of the Environment and Heritage, Australian Government, 2004).

However, it should be noted, that for MLLs restrictions to be effective even if applied, large quantity of undersized fish should either escape trapping or survive after release (Rudershausen & Buckel, 2007). Additionally, the estimated post-release fish mortality is positively correlated with the depth of capture and culling time, so deeper water species (around 40 m depth and below) regardless of minimum size limits and species prohibition remain highly unprotected, as often are hauled to the surface already being dead (Rudershausen & Buckel, 2007; Coleman et al., 2013). Another complication is that for complete establishment of MLLs, it should be linked to the corresponding selectivity of fishing gear ensured by mesh size and the presence of escape gaps and escape panels, otherwise the conflict of different management approaches might compromise the results of MLL implementation (Stewart & Hughes, 2008). As in multi-species fisheries there is no one mesh size perfect for all species harvested, there will be clear trade-offs between resource protection and sustainability of the fishery. Implementation of a series of staged increases in MLL, as well as, in mesh sizes would allow reaching a desired outcome with minimum immediate short-term losses (Stewart & Hughes, 2008).

Protection of vulnerable species is implemented via restrictions in their capture as well as via time and space closures. For instance, according to Fishery Management Plan for Puerto Rico and US Virgin Islands implemented in 1985, restrictions were placed on the harvest or possession of Nassau and Goliath groupers (*Epinephelus striatus* and *E. itajara*) and certain species used in the aquarium trade; area closures were organized for red hind *Epinephelus guttatus*, mutton snapper *Lutjanus analis* and other grouper and snapper species (Salas et al., 2011; NOAA, CFMC, 2015). Targeting grouper spawning aggregations during their mating season as well as capture of some other species, such as jack, *Seriola rivoliana*, was prohibited in the Dominican Republic (Mateo, 2004). Unfortunately, in some countries, e.g. in Belize spawning aggregations of *E. striatus* were still open for fishing despite the collapse of local grouper populations (Sala et al., 2001). In Western Australian waters, potato cod (*Epinephelus tukula*) is a fully protected species and has to be released if trapped. Despite the fact that fish caught from depths more than 40 m were unlikely to survive after release, the capture of this species was accidental, and impact from trap fishing was considered to be negligible (Newman et al., 2008). At the same time, the removal of fishing pressure does not guarantee the reversal of overfished populations to the prior state. The state of cod (*Gadus morhua*) fishery in the Canadian coast is a perfect example, when 10 years after the establishment of moratorium in 1992, the recovery of fish populations remain elusive (Campbell et al., 2009). Additionally, high market value of the species may contribute to neglect of fishing restrictions: according to Luckhurst & Trott (2015), despite "one fish per boat per day" bag limit for the black grouper (*Mycteroperca bonaci*), profit for the sold fish significantly exceeds the fine, so fishers go beyond the limits.

In situations where management options listed above fail to secure fish stocks from further declines, establishment of marine protected areas (MPAs) and no-take areas (NTAs) is often seen as a solution with a great potential for improvement of the state of the ecosystem in terms of abundance of exploited marine resources and overall biodiversity (Beets, 2005; Leslie, 2005; Eklöf et al., 2009). Among the primary goals set for the MPAs and NTAs are biodiversity conservation, fisheries enhancement, creation of areas for scientific research, tourism and education (Leslie, 2005; Edgar et al., 2009). Moreover, particularly the establishment of NTAs and NTA networks have been seen as a prevalent management approach towards increasing coral reef resilience (Bellwood et al., 2004). Clearly, the conservation efforts should focus on protecting the highly fished areas and key biodiversity areas (KBAs), which cover sites inhabited by species at high risk of extinction and other species that meet vulnerability and irreplaceability criteria (Eken et al., 2004; Edgar et al., 2008, 2009; Ambal et al., 2012). Nevertheless, Edgar et al. (2009) reported, that in Australia concessions to the fisheries stakeholders resulted in selective declaring MPAs in areas with little fishery resources. This might compromise the main biodiversity goals, as capability for recovery in lightly fished areas are much less compared to the ones heavily fished, and changes in response to protection are unlikely to be significant (Edgar et al., 2009).

Another main obstacle to overcome is the lack of no-take areas within MPAs. In India, the Gulf of Mannar Marine National Park was established in 1986 and was the first of its kind in South Asia, however the NTAs within the park yet was not demarked, as “the fishing communities are not ready to go out of the no-take-zone” (Murugan et al., 2014). Similar situation occurs within the MPAs of US Virgin Islands: while the commercial fishing is prohibited, artisanal fishing with rod, lines, fish traps, and small seine nets is allowed (Beets, 2005), and although no-take areas were established they have minimal reef cover, suffer from heavy recreational use and illegal fishing (Rogers & Beets, 2001). Although the Virgin Islands National Park was gazetted back in 1961, due to the lack of compliance and poor enforcement around 50 years later there still was little difference in reef fish assemblages and relative fish abundance within and outside park boundaries (Beets, 2005; Friedlander & Beets, 2008). Gill et al. (2017) stated that although MPAs with low management capacity still may provide some positive ecological impact, its magnitude is positively correlated with the availability of human and financial resources for management implementation. Similar situation stands within the Dar es Salaam Marine Reserves (DMRs) in Tanzania: Kamukuru (2009) reported, that DMRs are no-take zones, however fishing has not been prohibited and trap and beach seine fisheries were operating within its boundaries. Moreover, basket traps were considered the preferred gear over beach seines, as they catch target fish species (*Siganus sutor*) at size greater than the size of first maturity and cause less damage to the adjacent shallow coral reef areas (Kamukuru, 2009).

According to the extensive research, conducted by Edgar et al. (2014) the efficiency of MPAs was highly dependent on 5 planning and management features, so called “NEOLI”: No-take, Enforced, Old, Large and Isolated, with isolation probably being the most important. Across the 87 MPAs investigated, the ones with 4 or 5 of the abovementioned features (10% of all MPAs examined) demonstrated the best overall performance in terms of recovery of fish biomass, while MPAs with 1 or 2 features (84%) did not differ significantly from the fished sites (Edgar et al., 2014).

DISCUSSION

Fish traps, as all artisanal gear, are traditionally considered to be a relatively low impact gear (Quetglas et al., 2016), however, still it can be responsible for severe overexploitation of local marine resources (Hawkins & Roberts, 2004). In developing countries the situation aggravates with the fact that fishers usually operate within rapidly degrading environments, with no adequate management mechanisms in use and restrictions been neglected even if placed (FAO, 2016*b*). Moreover, Hawkins et al. (2007) stated that "the perception that trap fishing is a benign fishing method has perhaps permitted these fisheries to operate with little regulation or control". Surprisingly, despite the fact that fish traps are extensively used in many places, there were no compilation works on this fishing technique, to our knowledge. Several attempts were previously made to compile the information concerning particular aspects of trap fishing (Mahon & Hunte, 2001; Matsuoka et al., 2005); however, none previously tried to unite all the available information in one review.

The comparative analysis of fish trap fisheries worldwide revealed that despite the basic affinity in their functioning, significant dissimilarities do occur due to the distinctions in historically established fishing practices, exploitation status of fishery resources and management approaches.

Mesh size

It is hardly possible to establish one particular mesh size suitable for harvesting all targeted fish species in multispecies reef fisheries. Thus, currently used mesh sizes to a large extend reflect the state of local fishery resources. The fact that generally smaller mesh sizes (with 32 mm mesh as the most common) are reported from the Caribbean trap fisheries could probably be considered as a reason as well as a consequence of severe exploitation of reef fish communities. In some fisheries, e.g. in Australia, Kenya, US Atlantic and Virgin Islands, there was a tendency to increase mesh size, i.e. to 50, 63, 64 mm, which occurred both voluntarily and by law, or to oblige fishers to use escape panels with larger mesh, as it provides long-term benefits towards sustainability (Sheridan et al. 2006; Hicks & McClanahan, 2012; Rudershausen et al., 2016). While the improved selectivity of traps with larger mesh for reduction of juvenile fishes in catches is widely recognized among managers, as well as among fishers, its implementation still rarely receives full community support and compliance, especially in areas facing overfishing. It is usually caused by fishers´ concerns of inevitable short-term losses in revenue, despite the fact that fishes are not lost to the fishery and would be retained later of larger size and weight (Mahon & Hunte, 2001). In fact, while many trap fisheries, e.g. in India, Kuwait and United Arab Emirates, have no or little gear restrictions in place (Mathew, 2009; Grandcourt et al., 2011), fishers use mesh sizes grounded in traditions, and switch to novel mesh sizes cannot be fast and easy (McClanahan & Mangi, 2004). At the same time, only restrictions in minimum mesh sizes cannot prevent from the overharvesting of fishing resources, as could be seen in overexploited Canary Islands (Tuya et al., 2004; Garcia-Mederos et al., 2015), while in Kuwait traditionally used mesh sizes allow fishing within the sustainable limits (Chen et al., 2012).

Target species and Catch composition

Commonly trap fisheries target larger predatory and herbivorous species belonging Serranidae, Sparidae, Lutjanidae, Lethrinidae, among others. However, a mismatch was observed in terms of species targeted and landed, as presence and abundance of a particular species in the catch was determined by the actual state of fish stocks. Despite the great variety of species landed, several families if not species generally dominated the catches. Catches rarely were dominated by a single target species, such as, the shoemaker spinefoot *S. sutor* from East African and Seychelles´ fisheries (Kamukuru, 2009; Robinson et al., 2017; Samoilys et al., 2017), black sea bass *C. striata* from US Mid Atlantic (Cullen & Stevens, 2017) and for panga *P. laniarius* from South Africa (Gray et al., 2007). In general, catches were represented by multiple primary and secondary species, where proportion of one family usually did not exceeded 35%. Moreover, due to fishing down the web, in some areas with high fishing pressure, such as the Caribbean, after near extirpation of primary target species (large Serranidae, Lutjanidae and Scaridae) fishers switched to catching miscellaneous reef fishes, mostly herbivores, omnivores and invertivores, formerly of little commercial interest (Koslow et al., 1988; Marshak et al., 2008). In Jamaica, the rates of overfishing are so high that fishers catch whatever is available and literally, every fish larger than 10 cm is a target (Hawkins & Roberts, 2004). Additionally, different authors reported decrease either in overall landings or in proportions of primary commercial species and the total number of fish species landed almost across all trap fisheries (Hawkins et al., 2007; Chen et al., 2012; Garcia-Mederos et al., 2015; Bacheler & Smart, 2016; Samoilys et al., 2017). Bacheler & Smart (2016) stated that in Southeastern US some target species, such as red snapper (*L. campechanus*) and black sea bass (*C. striata*) did increase in abundance, however it was noted, that these species “are under heavy management and have life history characteristics such as early maturity that make them more likely to respond to management measures”. Similarly, the main target species in

Seychelles, *S. sutor* is capable of withstanding the high fishing pressure due to its short life span, fast growth and high rates of natural mortality (Grandcourt, 2002; Robinson et al., 2017). Nevertheless, in Western Africa this species was reported to demonstrate signs of overexploitation and growth overfishing (Hicks & McClanahan, 2012; Gomes et al., 2014).

CPUE

While the highest reported CPUE is almost 150 times larger than the reported lowest one for Australia and the Canaries, respectively (Newman et al., 2008; García-Mederos et al., 2015), average catch rates within other trap fisheries were within 1.99 – 3.34 kg of fish per trap and exceeded 5 kg only in US Atlantic (see Fig. 14). CPUE generally is assumed to be proportional to relative fish abundance (Bishop et al., 2008) and to a large extend reflects the exploitation status of local fishery resources. Nevertheless, it is a complex variable and highly depends on a wide range of factors, such as the efficiency of the fishing fleet, changes in fishing practices and species targeted, environmental conditions (such as changes in sea surface temperature and salinity, El Niño events), natural fluctuations of fish abundance and different catchability of species, among others (Maunder et al., 2006; Bishop et al., 2008).

Temporal and spatial differences in CPUE within regions to a large extend could be explained by changes in fishing tactics or effort applied, seasonal shifts to targeting migratory stocks or spawning aggregations. At the same time, broad-scale differences in catch rates among trap fisheries worldwide could be seen as a result of an interplay of biological and environmental factors and overfishing (both inherent to fish trap use and caused by other gears).

Differences in catch rates could be partly explained by trophodynamic factors, i.e. by the distinctions in primary production among regions (Stock et al., 2017), with eutrophic ecosystems, such as Australia and the Caribbean, being able to support a higher number of species of potential fishery interest. On the contrary, in oligotrophic locations, e.g. in the Canaries with its narrow shoreline, extensive use of fish traps coupled with a substantial area fished by a single trap has led to a rapid depletion of fishery resources, while primary production required to support local fisheries is too high to correspond with sustainable use of fish stocks (Pauly & Christensen, 1995; Chassot et al., 2010). Additionally, fish species significantly differ in their response to the presence of fish traps: ones, e.g. highly mobile or schooling species are highly attracted by traps, others, e.g. solitary species are more cautious and thus rarely seen trapped (Robichaud et al., 2000). While differences in catchability of different species clearly affects the catch composition, the general proportion of species entering or ignoring fish traps may vary among regions, which also would affect the CPUE.

The state of fish stocks clearly influence on the CPUE, with catches rates markedly lower in areas facing overfishing. Nevertheless, there is a non-linear relationship between CPUE and stock abundance, as well as, between CPUE and fishing effort (Maury & Gascuel, 2001), thus solely this index cannot be representative of the impact of fish trapping or overall state of fishery resources.

Ghost fishing

The information published for derelict finfish traps was scarce. Published data strongly supports only the influence of long soaks on the probability of fish trap loss (Taylor & McMichael Jr., 1983; Newman et al., 2008; Chen et al., 2012), while data regarding the influence of depth, structural complexity and absence/presence of buoys was not univocal. The reasons causing the trap loss across fisheries diverged significantly, with one factor being the major cause in some fisheries and insignificant in others, e.g. complex bottom structure was reported to cause 100% of trap loss in Turkey (Yildiz & Karakulak, 2016), 18-42.4% in Algarve, Portugal (Erzini et al., 2008) and only 1.7% in Gulf of Oman (Al-Masroori et al., 2009). The importance of presence of a buoy was hardly possible to assess as the percentages of its use among fishers was not given and even in fisheries where buoys are obligatory, neglect of this rules is common (Clark et al., 2012). Nevertheless, despite the fact that lack of buoys clearly facilitates trap loss (Chen et al., 2012; Clark et al., 2012) reluctance of buoy use could be considered as an indirect cause of fish trap loss. It occurs due to intentions of fishers to avoid poaching/vandalism, gear conflict and collision with vessels, which were reported as main reasons of unintentional gear loss (Kojis & Quinn, 2006; Erzini et al., 2008; Al-Masroori et al., 2009; Chen et al., 2012). Despite the great number of DFT and estimated long-term capability of ghost fishing, their actual impact on fish populations was surprisingly low. Although the rates of fishing mortality given by different authors were widely divergent for traps without escape panels (2 *vs* 100%), incorporation of escape panels or other escape mechanism allowed almost complete elimination of ghost fishing mortality (Scarsbrook et al., 1988; Renchen et al., 2014). Although escape panels are mandatory only in some areas, e.g. in U.S. Caribbean EEZ, US Atlantic, Barbados, their use must be encouraged and adopted in other trap fisheries worldwide (FAO, 2011; Clark et al., 2012). Time-release mechanisms are relatively cheap, could be easily adapted by fishers and clearly represent a conservation tool beneficial for fisheries as well as for the environment (Al-Masroori et al., 2004; Clark et al., 2012). Together with provision of appropriate facilities for old gear reception in ports and harbours, post clean-ups and gear-recovery programmes it will allow to mitigate impact for the environment as well as to reduce the contribution of fish traps to marine litter (Macfadyen et al., 2009).

Number of traps and fishing effort

There were no a clear trend within the fisheries regarding the intensification or decrease of fishing pressure with traps. Reduction in number of fish traps per boat or its substitution with other gear, such as gill and trammel nets and especially SCUBA was reported from the Canary Islands (García-Mederos et al., 2015) and Puerto Rico (Schärer et al., 2004; Agar et al., 2008), respectively. Abovementioned authors stated that changes in gear preferences happened due to inability to compete with other gears, trap loss, high material costs for trap construction and repairing which were exceeding the decreasing catch value. Increasing modernization of gear was also reported from Kenyan SSF, where younger fishers elect spears and "ring nets" over the passive gear as traps (Tuda et al., 2016; Samoilys et al., 2017). Nevertheless, in fisheries that do not restrict the number of traps per vessel, it is a common practice to increase the fishing pressure in response to decreasing catches (Hawkins & Roberts, 2004) and to growing demand for food for the increasing population (Kaunda-Arara et al., 2003); Kojis & Quinn (2006) reported over a 500% increase in number of traps used by nearly the same number of fishers from the US Virgin Islands between 1930 and 2003. Besides, in multispecies reef fisheries where fishers utilize a wide range of gears interchangeably it is naive to believe that reduction in the number of fish traps would result in a decrease of fishing pressure.

Management

If trap fisheries intend to sustainably harvest reef fish species, both resilient and vulnerable for fishing pressure, they have to be well regulated (Marshak et al., 2007). Fishery management tends to be focused on a limited amount of possible solutions, which usually include temporal and spatial closures, restrictions in gear, effort, species, fish size and sex (Munro, 2007; McClanahan & Cinner, 2008; McClanahan, 2011; Hicks & McClanahan, 2012). These restrictions rarely find support among fishers, and could be seen as unethical, especially in the areas with high dependence on marine resources and limited livelihood options (Hicks & McClanahan, 2012). Nevertheless, despite the fact that sometimes gear restrictions could be difficult to enforce, according to Cinner et al., (2009), fishers still prefer gear-based regulations to complete fishery closures, as usually they have alternative gears in use. Notably, the observed fisheries differed considerably, with some of them applying most if not all the abovementioned management tools, e.g. in Australia, the majority of trap fisheries, e.g. in Jamaica, India, Seychelles or Kuwait, remain highly unregulated, with only few restrictions placed. Gear restrictions are among the most popular ones among managers worldwide (Misund et al., 2008), but even minimum mesh size and presence of escape panels are not established and enforced ubiquitously (Grandcourt et al., 2011; Robinson et al., 2011; Chen et al., 2012). Nevertheless, even in well-managed trap fisheries management covers only the main target species, while others, and especially those comprising the "by-catch" remain highly unprotected (Stewart & Hughes, 2008; Bacheler & Smart, 2016). Implementation of long-term networks of marine protected areas (MPAs) and no-take areas (NTAs) is nowadays seen as a primary tool for conservation of the ecosystem (Brown et al., 2017), which also provides benefits to neighbouring fisheries (McClanahan & Mangi, 2000). Nevertheless, while its advantages for sedentary fish species are well recognised, conservation results for heavily exploited migratory or mobile fish species are much more elusive (West et al., 2009). Additionally, lack of NTAs within marine parks, or even permission to fish with traps and other artisanal gear, as it was reported from US Virgin Islands, India and Tanzania, as well as, lack of financial resources for enforcement, monitoring and administration highly compromise the conservation purposes of MPAs and NTAs (Friedlander & Beets, 2008; Kamukuru, 2009; Brown et al., 2017). Timely and appropriate management is a foundation stone for the security of the conservation outcomes, and it requires sufficient funding as well support and engagement of the local community (Brown et al., 2017).

The alternative approach for areas with high coral mortality and climate disturbances was proposed by Cinner et al. (2009) and included selective banning of gears, such as, traps, spear guns and nets that preferentially catch larger predatory and herbivorous species, crucial for supporting resilience and prevention on phase shifts on reefs (Hughes et al., 2007). This represents an extreme measure of fishery management, however, it was implemented for trap fishing in Bermuda, Gulf of Mexico and state waters of Florida (Burnet-Herkes & Barnes, 1996; Newman et al., 2011). In Bermuda it resulted in fast recovery of previously skewed sex ratios of local parrotfish populations (O'Farrell et al., 2015). Less categorical approach suggests the minimization of an overlap in gear selectivity (McClanahan & Cinner, 2008): establishment of specific zones for each type of gear would reduce the competition among gears and potentially may reduce the fishing effort.

Conclusions

The current study has revealed the insufficiency of current data available for the fish trap fisheries. In the Caribbean, for decades it was gaining significant attention from the scientific community; for other regions, such as the Canary Islands or India, scarce published data were available, while in the Pacific and western Indian oceans the published research was scarce if present, although traps are extensively used in several coastal areas (Mahon & Hunte, 2001). While overfishing and habitat degradation strike the ecosystems worldwide (Pauly & Watson, 2003; Hawkins & Roberts, 2004), stochastic and sporadic research does not allow to reveal the trends within the fisheries and most likely would lead to the shifting baseline syndrome for further research (Pauly et al., 1995; Duarte et al., 2009).

Mesh sizes varied within and between the ecoregions. However, despite the fact that no single mesh size is perfect for targeting the variety of fish species, authors consider that establishment of at least 50 mm minimum mesh size would be optimal for the majority of trap fisheries. Generally fishers target larger predatory fishes (Serranidae, Sparidae, Lutjanidae, Lethrinidae). However, while in some areas the primary target species comprise the bulk of the catch, in other regions its proportion within catches is from moderate to insufficient, and due to fishing down the web fishers target miscellaneous reef fish species.

In most regions CPUE averaged from 2 to 3.3 kg of fish per trap, nevertheless extremely high as well as low values were also detected (0.16 *vs* 23.9 kg per trap per day for the Canary Islands and Australia, respectively). However, catch rates were subjected to considerable fluctuations both spatially and temporally, which probably occurred due to the actual state of fish stocks, changes in fishing practices, fishing fleet efficiency, as well as environmental and biological factors affecting fish behaviour, among others.

The current impact of derelict fish traps on fish populations is considered to be low or insufficient despite the great number of fish traps lost annually. However, this is univocal only for traps with incorporated biodegradable escape panel, while fish mortality in traps without any escape mechanisms may reach 100% for some species (Scarsbrook et al., 1988).

The results obtained in this study demonstrate that generally most trap fisheries lack sustainable management: gear restrictions and other regulations applied are insufficient in preventing capture of fish juveniles, which result in growth and recruitment overfishing of target species and subsequent reduction of catches, currently observed almost ubiquitously (Hawkins & Roberts, 2004; Rudershausen et al., 2016; Hicks & McClanahan, 2012). Temporal and spatial closures are important conservational tools; however, they are successful only if well enforced (Friedlander & Beets, 2008; Brown et al., 2017). Changes from top-down management approach to decentralization and involvement of stakeholders from local communities to co-management may provide more flexible and adequate reaction towards changing environment and thus is essential for mitigation of impact of fishing (McClanahan et al., 2016).

ACKNOWLEDGEMENTS

I would like to express my deepest gratitude to my supervisor Dr. Rodrigo Riera for sharing not only vast knowledge and experience, but also great passion for work, which was always inspiring me. Without his encouragement, patience and advice, this Master´s Thesis would be a completely different story.

I am very grateful to Prof. Alberto Brito Hernández for giving me an opportunity to enter the master program despite the language gap. I would also like to thank professors of the ULL as well as my classmates for their understanding and assistance during studies.

Special thanks to my family and close friends for their continuous support, encouragement and understanding throughout my study period.

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