

DIRECTORATE-GENERAL FOR INTERNAL POLICIES

**POLICY DEPARTMENT**  
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**Research for PECH Committee -  
Marine recreational and  
semi-subsistence fishing - its value  
and its impact on fish stocks**

**STUDY**





**DIRECTORATE-GENERAL FOR INTERNAL POLICIES**  
**Policy Department for Structural and Cohesion Policies**

**FISHERIES**

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This document was requested by the European Parliament's Committee on Fisheries.

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## **LINGUISTIC VERSIONS**

Original: EN

## **ABOUT THE PUBLISHER**

To contact the Policy Department or to subscribe to updates on our work for the Fisheries Committee please write to: [Poldep-cohesion@ep.europa.eu](mailto:Poldep-cohesion@ep.europa.eu)

Manuscript completed in July 2017  
© European Union, 2017

Print	ISBN 978-92-846-1603-9	doi:10.2861/277908	QA-02-17-965-EN-C
PDF	ISBN 978-92-846-1604-6	doi:10.2861/2728	QA-02-17-965-EN-N

This document is available on the internet at:

[http://www.europarl.europa.eu/RegData/etudes/STUD/2017/601996/IPOL\\_STU\(2017\)6019\\_96\\_EN.pdf](http://www.europarl.europa.eu/RegData/etudes/STUD/2017/601996/IPOL_STU(2017)6019_96_EN.pdf)

### **Please use the following reference to cite this study:**

Hyder, K, Radford, Z, Prellezo, R, Weltersbach, MS, Lewin, WC, Zarauz, L, Ferter, K, Ruiz, J, Townhill, B, Mugerza, E, & Strehlow, HV, 2017, Research for PECH Committee - Marine recreational and semi-subsistence fishing - its value and its impact on fish stocks, European Parliament, Policy Department for Structural and Cohesion Policies, Brussels

### **Please use the following reference for in-text citations:**

Hyder *et al.* (2017)

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**DIRECTORATE-GENERAL FOR INTERNAL POLICIES**  
**Policy Department for Structural and Cohesion Policies**

**FISHERIES**

**Research for PECH Committee -  
Marine recreational and  
semi-subsistence fishing - its value  
and its impact on fish stocks**

**STUDY**

**Abstract**

This study assesses the value and impact on fish stocks of marine recreational and semi-subsistence fisheries in Europe. Total economic impact of marine recreational fishing amounts to 10.5 billion euro, supporting almost 100,000 jobs. The impact varies between fish stocks, representing 2-72% of total catch. The marine recreational fisheries are biologically and economically important, so should be included in stock assessment to ensure sustainability, and considered a sector for development alongside commercial fisheries and aquaculture under the Common Fisheries Policy.

**IP/B/PECH/IC/2016-131**

**July 2017**

**PE 601.996**

**EN**



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## LIST OF ABBREVIATIONS

<b>AZTI</b>	AZTI-Tecnalia, Spain
<b>Cefas</b>	Centre for Environment, Fisheries & Aquaculture Science
<b>CFP</b>	Common Fisheries Policy
<b>CPUE</b>	Catch per unit effort
<b>DCF</b>	Data Collection Framework
<b>DCR</b>	Data Collection Regulation
<b>EAA</b>	European Angler Alliance
<b>EC</b>	European Commission
<b>EP</b>	European Parliament
<b>EU</b>	European Union
<b>EU-MAP</b>	European Union Multi-Annual Programme
<b>FTE</b>	Full time equivalent
<b>GVA</b>	Gross Value Added
<b>I-O</b>	Input-Output
<b>ICES</b>	International Council for Exploration of the Sea
<b>MRF</b>	Marine recreational fisheries
<b>MSFD</b>	Marine strategy Framework Directive
<b>MS</b>	EU Member States
<b>MSY</b>	Maximum Sustainable Yield
<b>NOAA</b>	National Oceanic and Atmospheric Administration, US
<b>SSUBF</b>	Semi-subsistence fisheries
<b>Thünen-OF</b>	Thünen Institute of Baltic Sea Fisheries
<b>WGRFS</b>	ICES Working Group on Recreational Fisheries Surveys



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## EXECUTIVE SUMMARY

### Background

Marine recreational and semi-subsistence fisheries could represent a significant source of fishing mortality, have impact on ecosystems, and interact with commercial fisheries and users of the marine environment. However, the evidence needed to manage these fisheries is often limited and difficult to collect, because of the large numbers of widely-distributed small fishing vessels and individuals on the shore, exploiting highly mixed fisheries using a variety of gear types. These challenges mean that significant components of fishing mortality are not well described, which may affect our ability to manage fisheries to achieve conservation targets. Moreover, marine recreational and semi-subsistence fisheries can have a high economic value, but this is not taken into account in management and allocation decisions within European fisheries management. The European Parliament Committee on Fisheries requested a study to evaluate the value and impact of recreational and semi-subsistence fishing within different regions of the EU and funded a consortium of Cefas, AZTI, and Thünen-OF to deliver the EURecFish project.

EURecFish examined the social benefits, economic value, and environmental impact of marine recreational and semi-subsistence fisheries in six marine regions of Europe. Five questions were addressed:

1. What is recreational and semi-subsistence fishing, and where do they occur?
2. What is the value of recreational and semi-subsistence fishing?
3. How much fish is caught by recreational and semi-subsistence fisheries and how does this compare to commercial fisheries?
4. What other impacts of recreational and semi-subsistence fishing exist?
5. What needs to be done in future to monitor, assess, and manage recreational and semi-subsistence fisheries?

### Key outcomes

Marine Recreational Fisheries (MRF) was defined following the ICES WGRFS definition as *“the capture or attempted capture of living aquatic resources mainly for leisure and/or personal consumption. This covers active fishing methods including line, spear, and hand-gathering and passive fishing methods including nets, traps, pots, and set-lines”*. The main species and gears were categorised for each European country and licence requirements identified. Catches by MRF for certain species should be reported annually under the EU Data Collection Framework, but data are still limited especially in the Mediterranean and important recreational species are excluded. A recent synthesis and meta-analysis demonstrated the importance of MRF in Europe with 9 million people, or 1.6% of the population, participating, spending almost 6 billion euro, and fishing for around 77 million days each year. This excluded tourist fisheries, that could be significant in several countries (e.g. Norway).

It is possible to define semi-subsistence fishing in terms of a threshold of income either from sales of catch or as a proportion of the total income. However, there is no legal definition or an accepted cut-off for semi-subsistence, so this definition is very difficult to use practically. In fact, under EU legislation, any fishery where catches are sold is

considered commercial, so covered under commercial management regimes. Conversely, where catches are not sold, this activity and its impact are generally monitored as recreational fisheries. Hence, it is likely that any data on semi-subsistence fishing will be collected within either commercial or recreational fisheries surveys, and not recorded separately. A literature review was done to identify any potential semi-subsistence fisheries in Europe. It was not possible to separate semi-subsistence from commercial or recreational fisheries, so the focus was to demonstrate this challenge in two case studies. The case studies identified were Croatia and Norway, which had either been captured under existing reporting structures (i.e. Norway) or transitioned to commercial (i.e. Croatia). It is possible that other examples occur, and there are many small vessels that could engage in this type of fishing, so it is important that these are identified by individual countries and included in sampling schemes (either commercial or recreational). Due to the lack of data, it was not possible to estimate the value or the impact of semi-subsistence fisheries.

An Input-Output approach was used to estimate the total economic impact, gross value added (GVA – the amount that contributes to the Gross Domestic Product) and the numbers of jobs (Full Time Equivalents – FTEs) supported by European marine recreational fisheries. The total economic activity supported by marine recreational fisheries was 10.5 billion euro that comprised of 5.1 (direct), 2.3 (indirect) and 3.2 (induced) billion euro expenditure. This supported almost 100,000 FTEs that included 57,000, 18,000 and 24,000 from direct, indirect and induced expenditure, respectively. The amount varied between sea regions with the North Sea being the largest overall contributor, followed by the North-Western Atlantic Waters, Mediterranean, South-Western Atlantic Waters and the Baltic Sea, and the lowest contribution from the Black Sea. On average, 49,000 euros supported one FTE, with the maximum in Denmark (62,909 euros) and the minimum in Estonia (18,979 euro). There were significant gaps in the data identified with an assessment of overall bias indicating that the estimates are likely to be below the real amount especially in the Mediterranean and Black Sea. The size of the economic impact of marine recreational fisheries within Europe is significant enough to consider the development of a common and stable economic data collection program. It should also lead to the implementation of marine recreational fisheries as a sector that is targeted for development alongside commercial fisheries and aquaculture in Europe.

The assessment of the impact of marine recreational fisheries on fish stocks was done using data collected under the European Union Data Collection Framework and surveys from other countries, with extrapolations made to countries or species where no data existed. Large levels of release are common in marine recreational fisheries, so estimates of catches were compiled that included both harvest and release mortality. Comparisons were made at a stock level with commercial catches, where the data allowed, for individual stocks of European sea bass, Atlantic cod, European eel, Atlantic mackerel, pollack, Atlantic salmon, sea trout and Atlantic bluefin tuna. For the stocks where reconstruction was possible, sea bass, salmon and pollack recreational catches represented between 30 and 40% of the total catch, cod and mackerel were less than 21%, and eel catches were variable (13-72% of landings). Hence, catches of fish by MRF can be significant and should be included in stock assessments to ensure sustainable management of fisheries in future. Reconstructions were not possible for the Mediterranean or Black Sea and many stocks in other regions as insufficient data were available. To assess the impact robustly, better data are needed both on catches and post-release mortality by MRF, with regular multispecies surveys proposed.

In addition to the removal of biomass from marine fish stocks, recreational fisheries can have other impacts on the marine environment, particularly in coastal habitats. Impacts



occur at regional, national, and European scale, however, the level of impact as well as the associated effects are often unknown. The separation of marine recreational fisheries-induced impacts from other sources of anthropogenic impacts is difficult. This needs disaggregation into industrial and/or user groups to develop meaningful policies. Impacts originating from marine recreational fisheries are still largely unstudied. To enable evidence-based decision-making, European studies in the marine environment are needed. A sustainable ecosystem-based management of marine recreational fisheries needs to match the temporal and spatial scale of both the marine environment affected and the recreational fishing effort.

## **Recommendations**

Based on the analysis undertaken within the EURecFish project, ten recommendations have been compiled to support the development and understanding of marine recreational and semi-subsistence fisheries in Europe:

1. There is large variation in the understanding of marine recreational fisheries across Europe, generally with less data for Mediterranean and Black Seas countries, and limited time series. This makes any assessment of impact or value difficult, so there is a need for additional regular data collection.
2. A broad range of species are caught by marine recreational fisheries, yet mandatory data collection focusses on a small set of species. Further data collection is needed to develop understanding and should focus on country-specific multispecies surveys.
3. Tourist marine recreational fisheries can be large (e.g. Norway), but there is little knowledge of the benefits or impacts of this sector. More information is required to understand how these fisheries can be managed and developed in future.
4. Semi-subsistence fisheries should not be treated as a separate entity due to the challenges with definition, but individual countries should identify if they have any semi-subsistence fisheries and ensure that the current recreational or commercial fisheries sampling system covers these catches. In some cases, it may be necessary to set up additional sample frames to cover these data and develop approaches for management.
5. The potential total economic impact in Europe is significant, so marine recreational fisheries should become a sector that is targeted for development alongside commercial fisheries and aquaculture under the Common Fisheries Policy. However, data are lacking, so regular economic data collection is needed to monitor development and increase robustness of estimates.
6. The impact of changes in policy and management on the expenditure on marine recreational fisheries is very difficult to quantify and additional studies should be funded to develop these data, including studies of economic value and the human dimension.
7. Only the economic impact of direct expenditure was included in this study, but additional social and wellbeing benefits are provided by marine recreational fisheries that should be accounted for. It is unclear how this can be done, so additional studies should be funded to develop methods.

8. Estimates of discards and post-release mortality make comparison with commercial catches challenging. More information is needed on key marine recreational fisheries species to make more robust comparisons.
9. Where comparisons were possible, marine recreational fisheries catches represented a significant proportion of the total biomass removed for some stocks and could affect sustainability. Marine recreational fisheries catches should be routinely included in stock assessments, as this allows impacts to be properly assessed and appropriate management strategies developed.
10. Marine recreational fisheries can have other impacts on the marine environment, particularly in coastal habitats, but the level of impact as well as the associated effects are unknown. More information is needed to determine marine recreational fisheries-induced impacts and separate them from other anthropogenic impacts.

## 1 BACKGROUND & OBJECTIVES

Marine recreational fishing (MRF) is an important component of fishing mortality across the globe (e.g. Cooke and Cowx, 2006; Lewin *et al.*, 2006), generating significant economic impacts (e.g. Cisneros-Montemayor and Sumaila, 2010) and social benefits (e.g. Parkkila *et al.*, 2010; Griffiths *et al.*, 2016). In many parts of the world, MRF catches (landings and post-release mortality) are included in stock assessments and separate quota allocated to commercial and recreational fisheries (e.g. striped bass in the USA - <http://www.asmfcr.org/>). The economic value of the recreational fishery is recognised in some cases and considered in allocation decisions between the fisheries (Steinback, 1999; Steinback *et al.*, 2004; Lee *et al.*, 2017), and government policies promoting MRF have been developed (e.g. USA – NOAA, 2015). In Europe, a lack of reliable MRF catch estimates has led to it often being ignored in stock assessments and allocations (Pawson *et al.*, 2007). This undermines our ability to manage fish stocks sustainably (Hyder *et al.*, 2014) as required by the Common Fisheries Policy (CFP) (EU, 2013) and Marine Strategy Framework Directive (MSFD) (EU, 2008).

The definition of semi-subsistence fisheries (SSUBF) is more challenging as this applies to fishers that sell a proportion of their catch. However, there is no legal definition at the European level (see response of Maria Damanaki to European Parliamentary Question E-000760-14 surrounding fisheries in Croatia). Semi-subsistence describes unlicensed and unregistered people selling some or all their catch, often without record, and in some cases legally. Within Europe the number of semi-subsistence fishers is likely to be limited, so it is very difficult to separate commercial, recreational, and semi-subsistence fishing. Data on the semi-subsistence fishery is likely to be very limited and any separate estimate of the catches that are sold will be of very poor quality or very specific to local situations.

MRF and SSUBF could represent a significant source of fishing mortality, impact on habitats, and interact with other fishing fleets and users of the marine environment. However, the evidence needed to manage these fisheries is often limited and difficult to collect, because of the large numbers of widely-distributed small fishing vessels and individuals on the shore exploiting highly mixed fisheries using a variety of gear types. These challenges mean that significant components of mortality are not well described, which may affect our ability to manage fisheries to achieve conservation targets (Hyder *et al.*, 2014).

The overall aim of EURecFish was to characterise and quantify the environmental impact and socio-economic benefits of marine recreational and semi-subsistence fishing in Europe. To achieve this, the following objectives were addressed for European marine fisheries:

1. Characterise recreational and semi-subsistence fishing.
2. Estimate the socio-economic impact of recreational and semi-subsistence fishing.
3. Assess the impact of recreational and semi-subsistence fishing on key fish stocks.
4. Review the environmental impact of recreational and semi-subsistence fishing.
5. Provide recommendations for future of recreational and semi-subsistence fishing.

This report summarises literature, methodology, results, and recommendations for the future of marine recreational and semi-subsistence fishing in Europe.



## 2 CHARACTERISING MARINE RECREATIONAL FISHING

### 2.1 Summary

The characterisation of the European MRF was carried out through a literature review. Searches and reviews were done of the scientific literature (peer-reviewed journals, grey literature and internet) and requests to national delegates specialising in MRF from the ICES WGRFS were made. The most appropriate definition of MRF in Europe was identified as the one currently used by ICES (Table 1). Data were collected on the catches and value. Whilst complete data sets are lacking, some countries are developing catch sampling programmes that will be carried out over the next two years and will provide a more complete data set. The expenditure by resident MRF has been estimated to be significant, so the inclusion of MRF as a sector alongside commercial fisheries and aquaculture within Europe should be considered. Catches can be significant, so should be included in stock assessment where appropriate, but data are only collected on limited species. This may impact on the ability to manage fish stocks sustainably, unless multispecies surveys are implemented. In addition, tourist fishers have the potential to bring significant income into remote coastal regions and can catch significant amounts. More data are needed on tourist fisheries to understand both their value and impact, and the potential for development.

### 2.2 Definition of marine recreational fishing

There are many existing definitions of MRF in the literature both from a scientific perspective (e.g. FAO, 2012; ICES, 2013) and legislation (e.g. EU, 2015). A non-exhaustive list of definitions of MRF has been compiled from Europe and the US as examples to develop a single definition for this study (Table 1). Most scientific definitions are based on the reason for the activity (e.g. leisure or sport), description of the gears (e.g. rod and line), may include some statement about consumption (e.g. for personal consumption), and exclude the sale of the majority of the catch, although they acknowledge that small amounts can be sold or traded (e.g. FAO, 2012; ICES, 2013). The legal definition of recreational fisheries covers any non-commercial activities, often excludes the sale of the catch and contains specific gear restrictions (e.g. spearfishing) (e.g. EU, 2001; 2006; 2015). However, in Norway, sale of a proportion of the catch by recreational fishers is legal. Given the large number of definitions that already exist, it was not sensible to develop a separate definition for this study. Instead, the definition of the WGRFS (2013) was used (Table 1) as this was developed for recreational fisheries in Europe, but the FAO (2012) definition would also be appropriate as it attempts to separate subsistence from recreational fisheries.

Most studies of recreational fishing are stratified by gear (e.g. net, lines) and platform (e.g. shore, boat), but do not include categories for sport, leisure and recreation (Hyder *et al.*, in press). In addition, there was no legal definition of sport fishing and leisure fishing was defined in only one piece of legislation in terms of recreation and sport fishing (EU, 2006). Thus, this study will not attempt to separate sport, leisure and recreational fishing as there was no clear definition (i.e. the term recreational fishing also includes sport and leisure fishing) and data were not available for each category. It was clear that the unambiguous demarcation between recreational fisheries and subsistence fisheries was very difficult because many recreational fishers, even in wealthy countries, have strong subsistence-like incentives to harvest fish (Macinko and Schumann, 2007; FAO, 2012).

**Table 1: List of existing definitions of recreational fisheries from both scientific literature and legislation.**

Type	Definition	Source
Recreational (Scientific)	Recreational fishing is the capture or attempted capture of living aquatic resources mainly for leisure and/or personal consumption. This covers active fishing methods including line, spear, and hand-gathering and passive fishing methods including nets, traps, pots, and set-lines	ICES (2013)
	Recreational fishing is defined as fishing of aquatic animals (mainly fish) that do not constitute the individual's primary resource to meet basic nutritional needs and are not generally sold or otherwise traded on export, domestic or black markets.	FAO (2012)
	Recreation fishing is: <ul style="list-style-type: none"> <li>• Not deemed to be commercial fishing, in that recreational fishers do not sell the fish they catch.</li> <li>• Not undertaken for predominantly subsistence purposes.</li> <li>• Not undertaken for primarily cultural or heritage purposes, though these may provide justification for continuance of activities not deemed commercial.</li> <li>• Often synonymous with angling (the activity of catching or attempting to catch fish on hooks, principally by rod and line or hand-held line), but may include the use of small boats equipped with nets, longlines or pots to catch fish or crustaceans, capture of fish by divers with spearguns, and hand-gathering of shellfish from the beach or shore.</li> </ul>	Pawson <i>et al.</i> (2008)
	Due to the elaborate social, cultural and economic aspects of fishing, the Pacific Islands Regional Office is considering expanding the definition of recreational fishing from fishing for sport or pleasure. The following categories of fishermen may better meet the spirit of a recreational definition rather than a commercial: <ul style="list-style-type: none"> <li>• Those who fish for sport or pleasure primarily, but who sell a limited number of fish to assist with trip expenses.</li> <li>• Those who practice the customary exchange of fish.</li> <li>• Those charter fishermen who only sell small amounts of fish.</li> </ul>	NOAA Pacific Islands Regional Office
	"All types of fishing activities including sport fishing activities undertaken by any individual, with or without a boat, for leisure purposes, and does not involve the selling of fish or other aquatic organisms" and discuss the other terms (leisure, sport, amateur, tourism) commonly used in the Mediterranean.	Cacaud (2005)
	Recreational fisheries is separate from subsistence fisheries and refer to the FAO Technical guidelines.	MEDAC (2016)
	Fishing activities exploiting marine living aquatic resources from which it is prohibited to sell or trade the catches obtained.	GFCM (2011)
Recreational (Legal)	Recreational fishing" means fishing for sport or pleasure	MSFCMA (2007)
	"Recreational fisheries" means non-commercial fishing activities exploiting marine living aquatic resources such as for recreation, tourism or sport.	EU (2015)
	Recreational and game fisheries mean all fishing activities not conducted for commercial fishing purposes	EU (2001)
	The applicable regulation has everything included and is legally binding for EU Member States in the Mediterranean. Article 17 of the Council Regulation (EC) No 1967/2006 specifies: <ul style="list-style-type: none"> <li>• The use of towed nets, surrounding nets, purse seines, boat dredges, mechanized dredges, gillnets, trammel nets and combined bottom-set nets shall be prohibited for leisure fisheries*. The use of longlines for highly migratory species shall also be prohibited for leisure fisheries.</li> <li>• Member States shall ensure that leisure fisheries are conducted in a manner compatible with the objectives and rules of this Regulation.</li> <li>• Member States shall ensure that catches of marine organisms resulting from leisure fisheries are not sold. However, the sale of species caught in competitions may be authorized provided that the profits are donated to charity.</li> <li>• Member States shall take measures both to record and to ensure separate collection of data on catches resulting from leisure fisheries in respect of the highly migratory species listed in Annex I to Regulation (EC) 973/2001 and occurring in the Mediterranean.</li> <li>• Member States shall regulate underwater fishing with spearguns in particular to fulfil the obligations set out in Article 8(4).</li> <li>• Member States shall inform the Commission of all measures adopted pursuant to this Article.</li> </ul> <p>* "Leisure fisheries" is defined as fishing activities exploiting living aquatic resources for recreation or sport.</p>	EU (2006)

Definition chosen is highlighted in green.

**Source:** compiled by EURecFish.

**Table 2: Recreational fishing sectors and primary target species by regions for Europe.**

Country	Baltic Sea		North Sea		North Atlantic		Mediterranean & Black Sea	
	Target species	Sector	Target species	Sector	Target species	Sector	Target species	Sector
Albania	---	---	---	---	---	---	E, Sh	S, P, Ch, Pt, Sp
Belgium	---	---	C, B, E	S, P, Ch, L, N	---	---	---	---
Bulgaria	---	---	---	---	---	---	E, Sh	S, P, Ch, Pt, Sp
Croatia	---	---	---	---	---	---	B, E, Sh, T	S, P, Ch, Pt, Sp
Cyprus	---	---	---	---	---	---	B, E, Sh, T	S, P, Ch, Pt, Sp
Denmark	C, E, S	S, P, Ch, N, Pt, Sp	C, B, E, Sh	S, P, Ch, N, Pt, Sp	---	---	---	---
Estonia	C, E, S	S, P, Ch, N, Pt, Sp	---	---	---	---	---	---
Finland	C, E, S	S, P, Ch, N, Pt, Sp	---	---	---	---	---	---
France	---	---	C, B, E, Sh	S, P, Ch, N, Pt, Sp	B, E, Sh, T	S, P, Ch, N, Pt, Sp	B, E, Sh, T	S, P, Ch, N, Pt, Sp
Germany	C, E, S	S, P, Ch, N, Pt	C, B, E, Sh	S, P, Ch, N, Pt	---	---	---	---
Greece	---	---	---	---	---	---	B, Sh, T	S, P, Ch, L, Pt, Sp
Iceland	---	---	---	---	C, Sh	S, P, Ch, N, Pt	---	---
Ireland	---	---	---	---	C, B, E, Sh	S, P, Ch, Pt, Sp	---	---
Italy	---	---	---	---	---	---	B, E, Sh, T	S, P, Ch, Pt, Sp
Latvia	C, E, S	S, P, Ch, N, Pt, Sp	---	---	---	---	---	---
Lithuania	C, E, S	S, P, Ch, Sp	---	---	---	---	---	---
Malta	---	---	---	---	---	---	B, E, Sh, T	S, P, Ch, Pt, Sp
Montenegro	---	---	---	---	---	---	B, E, Sh, T	S, P, Ch, Pt, Sp
Netherlands	---	---	C, B, E, Sh	S, P, Ch, N	---	---	---	---
Norway	---	---	C, B, E, S, Sh	S, P, Ch, L, N, Pt, Sp	---	---	---	---
Poland	C, E, S	S, P, Ch, Sp	---	---	---	---	---	---
Portugal	---	---	---	---	B, E, Sh, T	S, P, Ch, Sp	---	---
Romania	---	---	---	---	---	---	E, S, Sh	S, P, Ch, Pt, Sp
Slovenia	---	---	---	---	---	---	B, E, Sh, T	S, P, Ch, Pt, Sp
Spain	---	---	---	---	B, E, Sh, T	S, P, Ch, N, Pt, Sp	B, E, Sh, T	S, P, Ch, Pt, Sp
Sweden	C, E, S	S, P, Ch, N, Pt	C, B, E, Sh	S, P, Ch, N, Pt	---	---	---	---
UK	---	---	C, B, E, Sh	S, P, Ch, N, Pt, Sp	C, B, E, S, Sh	S, P, Ch, N, Pt, Sp	---	---

Sectors are defined as fishing platforms where S = Shore Angling, P = Private Boat Angling, Ch = Charter Boat Angling, L = Longlining, N = Net fishing, Pt = Pot fishing, Sp = Spearfishing. Potential target species relates to species where data must be collected in Europe, where C = Atlantic cod, B = European sea bass, E = European eel, S = Atlantic salmon, Sh = Elasmobranchs, T = Tuna.

Regions are defined as in the Data Collection Framework (EU, 2008a)

**Source:** reproduced from Hyder *et al.* (in press).

## 2.3 Data collection

The European Commission introduced the Data Collection Framework (DCF) in 2001 to support implementation of the Common Fisheries Policy (CFP) (EU, 2013), that placed a legal requirement for Member States (MS) to collect specified types of data, including estimates of recreational catches and releases for selected species (EU, 2001). Minor changes to the requirements were made in the subsequent DCF regulations (EU, 2008a; 2010; 2016a) that specify the requirements for the collection, management, and use of fisheries data. Under the DCF, annual estimates of catches and releases are required for Atlantic cod (*Gadus morhua*), European sea bass (*Dicentrarchus labrax*), European eel (*Anguilla anguilla*), Atlantic bluefin tuna (*Thunnus thynnus*), Atlantic salmon (*Salmo salar*), and all elasmobranchs, with variation in species requirements across regions (EU, 2010). The EU Multiannual Programme (EU-MAP) is the latest version of the DCF that covers the period 2017-2019, and includes listed species and/or species identified at marine regional scales needed for fisheries management purposes (EU, 2016a). A second mechanism for collection of recreational fishery data in Europe is the Control Regulation (EU, 2009) that specifies recreational fisheries are conducted in a manner compatible with the CFP (EU, 2013) and that recreational catches of stocks subject to recovery plans must be monitored, but only for vessels registered in each country. In some non-EU MS countries, independent surveys are also conducted to collect data on recreational fisheries (e.g. Norway).

## 2.4 Characterisation

A wide variety of gears are used and many different species are targeted in MRF across Europe. This is summarised in Hyder *et al.* (in press) supporting information with respect to the DCF requirements (Table 2). In the Mediterranean, the species targeted are very diverse (Table 3) and a wide variety of gear is used, including: rod-and-line, handlines, pots, traps, nets, spears, and hand-gathering (MEDAC, 2016). The literature was reviewed to identify information on MRF in each country. No information existed for Albania, and Romania, so summaries of each country were based on information gathered on the internet.

### 2.4.1 Albania

MRF exist in Albania as there are several social media sites where pictures of catches are shared. It was unclear what the main target species and catches are as no studies of MRF in Albania were found. However, it is likely that the main target species are the same as in other countries in the Mediterranean (Table 3).

### 2.4.2 Belgium

The recreational fisheries sector in Belgium is relatively small compared to neighbouring countries, but catches of some species can be significant at a national level. MRF include: boat angling (private, charter); beam trawling; otter trawling; longlines; beach fishing with static gear; shore angling; and wading with small towed nets in the surf zone. The use of trammel and gillnets by recreational fishers is banned. The main target species for recreational fishers in Belgium are Atlantic cod, European sea bass, whiting (*Merlangius merlangus*), common dab (*Limanda limanda*), common sole (*Solea solea*), Atlantic mackerel (*Scomber scombrus*) and brown shrimp (*Crangon crangon*) (van den Stein, 2010).



**Table 3: List of main target species in the Mediterranean.**

Shore	Boat	Spearfishing
<i>Argyrosomus regius</i>	<i>Argyrosomus regius</i>	<i>Argyrosomus regius</i>
<i>Belone belone</i>	<i>Auxis thazard</i>	<i>Balistes capriscus</i>
<i>Conger conger</i>	<i>Balistes capriscus</i>	<i>Conger conger</i>
<i>Coriphaena hippurus</i>	<i>Belone belone</i>	<i>Dentex dentex</i>
<i>Dentex dentex</i>	<i>Conger conger</i>	<i>Dicentrarchus labrax</i>
<i>Dicentrarchus labrax</i>	<i>Coriphaena hippurus</i>	<i>Diplodus cervinus</i>
<i>Diplodus species</i>	<i>Dentex dentex</i>	<i>Diplodus puntazzo</i>
<i>Epinephelus aeneus</i>	<i>Dicentrarchus labrax</i>	<i>Diplodus sargus</i>
<i>Epinephelus costae</i>	<i>Diplodus species</i>	<i>Epinephelus aeneus</i>
<i>Euthynnus alletteratus</i>	<i>Epinephelus aeneus</i>	<i>Epinephelus costae</i>
<i>Labrus merula</i>	<i>Epinephelus costae</i>	<i>Epinephelus marginatus</i>
<i>Labrus viridis</i>	<i>Epinephelus marginatus</i>	<i>Labrus merula</i>
<i>Lichia ama</i>	<i>Euthynnus alletteratus</i>	<i>Labrus viridis</i>
<i>Lithognathus mormyrus</i>	<i>Labrus merula</i>	<i>Lichia ama</i>
<i>Loligo vulgaris</i>	<i>Labrus viridis</i>	<i>Lophius piscatorius</i>
<i>Mugilidae species</i>	<i>Lichia ama</i>	<i>Mugilidae species</i>
<i>Mullus surmuletus</i>	<i>Lithognathus mormyrus</i>	<i>Mullus surmuletus</i>
<i>Oblada melanura</i>	<i>Loligo vulgaris</i>	<i>Muraena helena</i>
<i>Octopus vulgaris</i>	<i>Lophius piscatorius</i>	<i>Mycteroperca rubra</i>
<i>Pagrus auriga</i>	<i>Mugilidae species</i>	<i>Octopus vulgaris</i>
<i>Phycis phycis</i>	<i>Mullus surmuletus</i>	<i>Pagrus auriga</i>
<i>Pomatomus saltatrix</i>	<i>Mycteroperca rubra</i>	<i>Phycis phycis</i>
<i>Psetta maxima</i>	<i>Naucrates ductor</i>	<i>Plectorhinchus mediterraneus</i>
<i>Sarda sarda</i>	<i>Oblada melanura</i>	<i>Pomatomus saltatrix</i>
<i>Sarpa salpa</i>	<i>Octopus vulgaris</i>	<i>Sarda sarda</i>
<i>Sciaena umbra</i>	<i>Pagellus acarne</i>	<i>Sarpa salpa</i>
<i>Scomber species</i>	<i>Pagellus bogaraveo</i>	<i>Sciaena umbra</i>
<i>Scorpaena porcus</i>	<i>Pagellus erythrinus</i>	<i>Scorpaena porcus</i>
<i>Seriola dumerili</i>	<i>Pagrus auriga</i>	<i>Scorpaena scrofa</i>
<i>Serranus scriba</i>	<i>Pagrus pagrus</i>	<i>Sepia officinalis</i>
<i>Sparus aurata</i>	<i>Phycis phycis</i>	<i>Seriola dumerili</i>
<i>Sphyaena sphyraena</i>	<i>Plectorhinchus mediterraneus</i>	<i>Serranus scriba</i>
<i>Sphyaena viridiensis</i>	<i>Polyprion americanus</i>	<i>Sparisoma cretense</i>
<i>Symphodus tinca</i>	<i>Pomatomus saltatrix</i>	<i>Sparus aurata</i>
<i>Todarodes sagittatus</i>	<i>Sarda sarda</i>	<i>Sphyaena viridiensis</i>
<i>Trachinotus ovatus</i>	<i>Sarpa salpa</i>	<i>Spondylosoma cantharus</i>
<i>Trachurus species</i>	<i>Sciaena umbra</i>	<i>Symphodus tinca</i>
<i>Umbrina cirrosa</i>	<i>Scomber species</i>	
	<i>Scorpaena porcus</i>	
	<i>Scorpaena scrofa</i>	
	<i>Sepia officinalis</i>	
	<i>Seriola dumerili</i>	
	<i>Serranus scriba</i>	
	<i>Sparisoma cretense</i>	
	<i>Sparus aurata</i>	
	<i>Sphyaena sphyraena</i>	
	<i>Sphyaena viridiensis</i>	
	<i>Spondylosoma cantharus</i>	
	<i>Symphodus tinca</i>	
	<i>Tetrapturus belone</i>	

Source: MEDAC (2016).

### 2.4.3 Bulgaria

MRF occurs primarily within inshore waters and is most popular from April to June, and from September to November (Keskin *et al.*, 2015; 2017). The main target species are gobies (Gobiidae), grey mullets (Mugilidae), Atlantic horse mackerel (*Trachurus trachurus*), bluefish (*Pomatomus saltatrix*), Atlantic bonito (*Sarda sarda*), turbot (*Scophthalmus maximus*), Mediterranean horse mackerel (*Trachurus mediterraneus*) and garfish (*Belone belone*) (Keskin *et al.*, 2015, 2017).

#### **2.4.4 Croatia**

MRF is a popular activity for both residents and tourists in Croatia, with between 25,000 and 80,000 participants (see Matić-Skoko *et al.*, 2014 and references within), but limited data exists for catches (Matić-Skoko *et al.*, 2014). Croatian Adriatic shore-based MRF mainly target seabream species whilst boat-based recreational fishers principally target pandoras, picarels and fish from the Sparidae family, such as white seabream (*Diplodus sargus*) and annular seabream (*Diplodus annularis*). There is also boat-based trolling for European sea bass, common dentex (*Dentex dentex*), and fish from the genus *Seriola*.

#### **2.4.5 Cyprus**

MRF is a popular activity in the north of Cyprus with recreational vessels, spearfishers and shore-based anglers (Ulman *et al.*, 2015). Around 80% of registered vessels fish recreationally (1,425 vessels), and there were 368 licenced spearfishers and an estimated 2,000 shore anglers in 2010. The most commonly caught species include old parrotfish (Scarinae species), porgies, seabreams, dusky grouper (*Epinephelus marginatus*), mottled grouper (*Mycteroperca rubra*), Atlantic bonito and greater amberjack (*Seriola dumerili*) (Ulman *et al.*, 2015). Reconstructions of catches have been made for the south of Cyprus with a focus on the same species (Ulman *et al.*, 2015).

#### **2.4.6 Denmark**

Recreational sea fishing is a popular leisure activity, with three categories of recreational fishery: passive gear fishing using stationary gear (e.g. gill and fyke nets), angling with rod and line, and some spearfishing. Passive gear fishing is typically done from a small boat targeting eel, European flounder (*Platichthys flesus*), cod, and sea trout (*Salmo trutta*) (Sparrevojn *et al.*, 2010). Angling is done from the shore and from boats targeting sea trout, garfish, cod, various flatfish, and salmon (Rasmussen and Geertz-Hansen, 2001; FVM, 2010).

#### **2.4.7 Estonia**

The recreational sea fishery in Estonia comprises of three sectors: (1) a licensed fishery sector with mandatory logbooks that mainly uses passive gear (e.g. gill nets, longlines, crayfish traps); (2) a licensed fishery sector requiring the purchase of fishing rights that comprises of anglers and spearfishers; and (3) a non-licensed fishery with no licence requirement limited to the use of one hand line or rod without a reel. The main species caught by recreational sea fishers in the Baltic Sea are flounder, perch (*Perca fluviatilis*) and pike (*Esox lucius*).

#### **2.4.8 Finland**

Most recreational fishing activity is in inland waters, with around 300,000 marine fishers. The most important species are perch, pike, herring (*Clupea harengus*), roach (*Rutilus rutilus*) and pikeperch (*Sander lucioperca*). MRF is done mostly from small private vessels with most fishers using gill nets, fish traps and trap nets. In general, no licence is needed for angling with hook and line (i.e. bait fishing, ice fishing and herring fishing with a rig). However, a government fishing management fee must be paid for other types of fishing including lure fishing with one rod. A permit from the water body owner is needed for lure fishing with more than one rod and for fishing with other gears (e.g. gillnets).

#### **2.4.9 France**

MRF in France use passive gears, rod and line, and spearguns from the shore and boats (Herfaut *et al.*, 2013; Levrel *et al.*, 2013; Rocklin *et al.*, 2014). The main species caught

are sea bass, Atlantic mackerel, pollack (*Pollachius pollachius*), whiting, pouting (*Trisopterus luscus*), cuttlefish (*Sepia officinalis*), and seabreams. Angling with bait or lures and spearfishing are the main methods used from the shore, with both angling and nets commonly used from boats (Herfaut *et al.*, 2013; Levrel *et al.*, 2013; Rocklin *et al.*, 2014). In 2011, there were 1,319,000 marine recreational fishers in France making around 9,000,000 fishing trips, with roughly 60% and 40% of the effort in Northern and Southern France, respectively. There was an even split of effort between shore and boat fishing, with about 60% of trips resulting in catch (Levrel *et al.*, 2013).

#### **2.4.10 Germany**

MRF in Germany is carried out in the North and Baltic Seas with an outer coastline of 1,096 km and 724 km in length, respectively. There were 174,000 recreational sea fishers in Germany in 2013/2014, with the majority fishing in the Baltic Sea and a smaller fishery in the North Sea. The strong tidal influence and the extensive tidal flats in the North Sea restrict shore angling to harbours and the East Friesian and Friesian Islands, and limit boat angling in this area. The Baltic Sea is popular for shore and boat angling and most charter vessels are stationed in this area (Strehlow *et al.*, 2012). Fishing from the shore and boat-based fishing methods are equally popular with the fishing effort evenly split in the Baltic Sea (Strehlow *et al.*, 2012). Passive fishing gears are also used, but the number of users is limited. The main species targeted by MRF are cod, herring, mackerel, flounder, plaice (*Pleuronectes platessa*), dab, sea trout and salmon. MRF licences are mandatory in all federal states, apart from in the North Sea in the state of Lower Saxony. MRF in the states bordering the Baltic Sea also require a coastal fishing permit (Mecklenburg-Western Pomerania) or a federal fishing licence (Schleswig-Holstein).

#### **2.4.11 Greece**

Greek MRF catch many different species using a variety of different gear types, and reported fisheries landings exclude recreational catches (Tsikliras *et al.*, 2007). MRF is done for leisure and consumption, and consists of boat and shore angling, spearfishing, and shellfish collection. Surveys of MRF are a challenge due to the complex shoreline and variety of different gear types used (Lloret and Font, 2013; Moutopoulos *et al.*, 2013). Shore fishing is a common and represents 8% (3 – 22%) of the total catch (Moutopoulos and Stergiou 2012). It is likely that recreational catches have increased recently due to the issues with the Greek economy, austerity, and unemployment (Moutopoulos *et al.*, 2013). Between 11 and 48 demersal species are targetted, with sea bass and seabreams (Sparidae) representing 40% of the MRF catch (Moutopoulos *et al.*, 2013).

#### **2.4.12 Iceland**

Cod, Atlantic halibut (*Hippoglossus hippoglossus*), Atlantic wolf fish (*Anarhichas lupus*) and haddock (*Melanogrammus aeglefinus*) are the most common target species (Solstrand, 2013). Catch and release is illegal, so all fish must be landed apart from the Atlantic halibut which must be released. Tourists can use only rod and line, and quotas are issued to charter boats that must report all catches.

#### **2.4.13 Ireland**

MRF is generally confined to angling and a limited amount of spearfishing. No licence is required for MRF with the sector comprising of shore, charter boat and private boat angling. Angling activity can be categorised into fishing for pelagic and demersal species and shark angling for larger sharks including blue and porbeagle shark (*Prionace glauca* and *Lamna nasus*) (Wögerbauer *et al.*, 2015). The main species targeted by shore anglers are sea bass, mackerel, cod, pollack, flounder, whiting, common dab and lesser spotted dogfish

(*Scyliorhinus canicula*), with other sharks (e.g. tope - *Galeorhinus galeus*) and rays targeted at specific locations. Shore angling is generally a bait fishery, except for sea bass where artificial lures are used. Boat anglers target species including pollack, cod, ling (*Molva molva*), saithe (*Pollachius virens*), conger eel (*Conger conger*), wrasse (cuckoo - *Labrus mixtus* and ballan - *Labrus bergylta*), sea bass, gurnards (Triglidae), blue shark, spurdog (*Squalus acanthias*), rays, and tope.

#### **2.4.14 Italy**

Italy has 600,000 to 1,000,000 people participating in MRF. The majority fish from shore, but one third use private or charter boats. The most common gears are lines (rod and line 50%, longline 18%) and pots (7%), and spearfishing accounts for 12% of the effort. Targeted species include seabream, tuna species, sea bass, common dolphin fish (*Coryphaena hippurus*), little tunny (*Euthynnus alletteratus*), cuttlefish, common squid (*Loligo vulgaris*), and shark bycatch when targeting tuna (Cingolani *et al.*, 2005; Pranovi *et al.*, 2016).

#### **2.4.15 Latvia**

MRF is carried out in the Baltic Sea and comprises two sectors: personal consumption fishers using passive gears that are not permitted to sell their catches (e.g. gillnets, fyke nets, longlines) and active methods including angling and spearfishing. Catches using passive gears must be reported and are included in the national catch statistics. No catch reporting is required by anglers, but those between 16 and 65 years must have a licence (Latvijas Nacionālās, 2013) and are not allowed to sell their catch (Anonymous, 2007). Angling is more common from the shore than from boats, and ice fishing occurs in the Gulf of Riga. Flounder, perch, cod, garfish, herring and round goby (*Neogobius melanostomus*) are the main target species.

#### **2.4.16 Lithuania**

MRF is mainly practiced in shallow water less than 20 meters deep, and a variety of species are targeted including flounder, herring, cod, turbot, and salmonids (Lithuanian Fishing Services, 2016). Cod seasons are March to May and October to December with a bag limit of 15 kg per day. Angling is the only legal method.

#### **2.4.17 Malta**

A reconstruction of Maltese MRF landings has been conducted for the years 1950 to 2014 (Khalfallah *et al.*, 2017). The main target species were frigate tuna (*Auxis thazard*), common octopus (*Octopus vulgaris*), argonauts, common dolphinfish, Atlantic bonito, greater amberjack, swordfish (*Xiphias gladius*), comber (*Serranus cabrilla*), little tunny (*Euthynnus alletteratus*), black scorpionfish (*Scorpaena porcus*), smoothhound (*Mustelus mustelus*), sea bass, groupers (Serranidae), hinds, common cuttlefish, and white skate (*Rostroraja alba*).

#### **2.4.18 Montenegro**

MRF is a popular activity in Montenegro with increasing participation and an estimated 1,500 registered sport fishers organised into 12 clubs (Keskin *et al.*, 2014). Tunas such as Atlantic bluefin tuna, albacore (*Thunnus alalunga*), and little tunny are common fishing targets in the Adriatic, but game fish also include sharks, swordfish, amberjacks, and common dolphin fish (Keskin *et al.*, 2014). Keskin *et al.*, (2014) reconstructed the MRF landings in Montenegro, based on data from Croatia (Matić-Skoko *et al.*, 2014), so it is likely that recreational fishers also target similar species to Croatia.

#### 2.4.19 Netherlands

In 2013, 3.2% of the population participated in MRF and generally made one to five trips each year. The main species caught were flatfish (plaice, flounder, dab), mackerel, cod and sea bass. Most fishing was conducted with rod and line, and occurred from the shore, charter boats, and private boats. Gillnets are used in marine waters mainly to target sea bass and a license is required, but no license is required for sea angling. Some species, such as European eel, must be released.

#### 2.4.20 Norway

Recreational sea fishing is a popular activity in Norway with around 33% of the population involved (Vaage, 2015). Domestic recreational fishers can fish with rod and line, jigging machines, traps, pots, gillnets, and longlines (Anonymous, 2006) meaning that a high diversity of species are caught. The main target species are cod, ling, tusk (*Brosme brosme*), saithe, haddock and mackerel, with cod in northern Norway and saithe and mackerel in western and southern Norway representing the main interactions with commercial fishing (ICES, 2010). Fishing tourism is important (Borch *et al.*, 2011; Vølstad *et al.*, 2011) with foreign tourists limited to using hand-held tackle and exporting 15 kg of fillet and one trophy fish. Cod and saithe are the main targets for tourists (Vølstad *et al.*, 2011) and a large proportion of fish are released (Ferber *et al.*, 2013a; 2013b). Boat fishing is the main platform, with 63% of private boats used for MRF (KNBF and NORBOAT, 2012). Shore fishing is also popular because of the high quality of fishing. Many charter fishing companies operate, but the magnitude of the activity is unknown. Spearfishing and hand gathering using SCUBA is allowed for most species. Neither resident nor tourist MRF require a license.

#### 2.4.21 Poland

The Polish MRF sector includes two main fishing techniques: angling and spearfishing, conducted from both shore and boats. Shore angling mainly targets European flounder, common bream (*Abramis brama*, Cyprinidae), sea trout, garfish, Atlantic herring and European eel. Trolling for Atlantic salmon has increased in Polish waters recently. Anglers in estuaries and lagoons often target freshwater species, including Eurasian perch, pikeperch, roach and common bream. Around 37,000 fishing licences were issued in 2014, but no data are available on spearfishers.

#### 2.4.22 Portugal

MRF is a very popular leisure activity in Portugal. The number of licences issued suggest that the total number of fishers to be between 170,000 and 200,000 (DGRM, 2015; Regional Fisheries Department of Azores, unpublished data). Shore angling is the most common method, followed by boat fishing, and spearfishing. Charter boat angling can be economically important in some regions including southern Portugal, Azores and Madeira. Methods used are limited to rod and line for shore and boat angling, spearfishing, and specific hand collection of shellfish and bait. Restrictions including fishing licences, bag limits, minimum landing sizes, and closed areas and seasons have been in place on the mainland and the Azores since 2006 and 2008, respectively (Veiga *et al.*, 2012; Diogo and Pereira, 2014). Spearfishing is the only regulated activity in Madeira and subject to fishing licences. Important target species on the mainland are seabreams and sea bass. Intertidal collectors target mainly common octopus, velvet swimming crab (*Necora puber*), bivalves and stalked barnacle (Cruz *et al.*, 2015). Important targeted species in the Azores, include seabream, parrotfish, wrasse, grouper, jack and mackerel. Intertidal collectors target limpets, common octopus, and crabs (Diogo and Pereira, 2013a; 2013b; 2014).

#### **2.4.23 Romania**

MRF exist in Romania as there are social media sites where pictures of catches are shared, but no studies of MRF in Romania were found. The main target species are black spotted goby (*Neogobius melanostomus*), flat-head goby (*Mesogobius batracephalus*), Mediterranean horse mackerel (*Trachurus mediterraneus ponticus*), bluefish (*Pomatomus saltatrix*) and golden grey mullet (*Liza aurata*) (Romanian National Work Plan 2017-2019 for the EU-MAP).

#### **2.4.24 Slovenia**

MRF is carried out in Slovenia, but no licence is required for shore fishing (Gaudin and De Young, 2007). An annual fishing licence is required for boat fishing, with gear restrictions and bag limits in place (Ministry of Agriculture, Fisheries and Food, 2016). The main species targeted from the shore are seabream and from boats are picarels, seabream, sea bass and squid (Gaudin and De Young, 2007).

#### **2.4.25 Spain**

In Spain, the management of MRF is carried out by the Spanish Autonomous Regions for inshore areas and the Ministry of Fisheries for offshore areas. A fishing licence is required that is issued by the Autonomous Regions. Restrictions to recreational practices exist including gear and tackle restrictions and bag limits. The fisheries in Spain differ considerably between the Atlantic and the Mediterranean. The main targeted species in the Atlantic are albacore, ballan wrasse, conger eel, Atlantic horse mackerel, Atlantic mackerel, common octopus, sea bass, seabream and common squid. In the Mediterranean, the main targets are amberjack, sea bass and diverse species of Scianenidae, Sparidae, and Serranidae. All methods and platforms are used including shore, private boat, and charter boat angling, spearfishing, and hand gathering.

#### **2.4.26 Sweden**

Sweden has a range of methods including passive and active gear fished from shore, private boats, and charter boats. Recreational fishing is a popular activity, with around 1.7 million participants in marine and freshwater (Svergies Officiella Statistik, 2013). The main marine target species include cod, mackerel, flatfish species, herring, sea trout, crabs, and European lobster (*Homarus gammarus*) (Karlsson *et al.*, 2014). MRF includes passive gears such as gillnets and fyke nets, and active methods like angling. Fishing licenses are not required for most recreational sea fishing, with some exceptions (e.g. coastal trolling, net fishing).

#### **2.4.27 Turkey**

Several studies have been performed to characterise MRF in different parts of Turkey (Aydin *et al.*, 2013, Tunca 2015, Tunca *et al.*, 2012, Tunca *et al.*, 2016, Ünal *et al.*, 2010). In the Mediterranean, the main target species are from the families Sparidae, Mullidae, Caragnidae and Scombridae (Aydin *et al.*, 2013, Tunca *et al.*, 2016). In the Black Sea, six species dominated the catch composition in all provinces: Atlantic horse mackerel, flathead grey mullet, Atlantic bonito, bluefish, and garfish (Tunca *et al.*, 2017).

#### **2.4.28 UK**

The MRF sector in the UK is diverse with most effort by anglers on the shore and boats (private and charter) (Armstrong *et al.*, 2013). There were 1,080,000 sea anglers in Great Britain, with 2% of all adults going sea angling (Armstrong *et al.*, 2013) and an additional 64,800 in Northern Ireland (McMinn, 2013). Shore fishing is the most common method



accounting for almost 3 million angler-days each year, with 1 million days on private or rented boats and 0.1 million days on charter boats. Anglers had most success on charter boats, catching 10 fish per day, compared with around 5 from private boats and only 2 from the shore. The most common species caught, by number, were mackerel and whiting, and anglers actively target mackerel, sea bass, cod, and elasmobranchs (Armstrong *et al.*, 2013).

## 2.5 Numbers, participation rates, and effort

Although reporting of recreational catches has been a European legislative requirement since 2002, robust estimates of MRF are only available for some MS. In countries outside of the EU, some surveys have been done, but reporting is not mandatory (e.g. Norway). A synthesis of these studies was done by Hyder *et al.* (in press) to produce robust estimates of MRF in Europe. The numbers of fishers, participation rates, days fished, expenditures, and catches of two key species were synthesised to provide estimates of MRF in Europe. National data were collated by local experts and, where surveys did not exist, combined with extrapolations from donor countries corrected for population size or GDP. There was an estimated 8.7 million European MRFs and a participation rate of 1.6% (Figure 1A; Table 4). Each year, 77.6 million days were fished and direct expenditure was 5.9 billion euro including taxes (Figure 1B; Table 4). The Atlantic region had higher participation, numbers of fishers, days fished, effort, and expenditure than the Mediterranean region (Hyder *et al.*, in press).

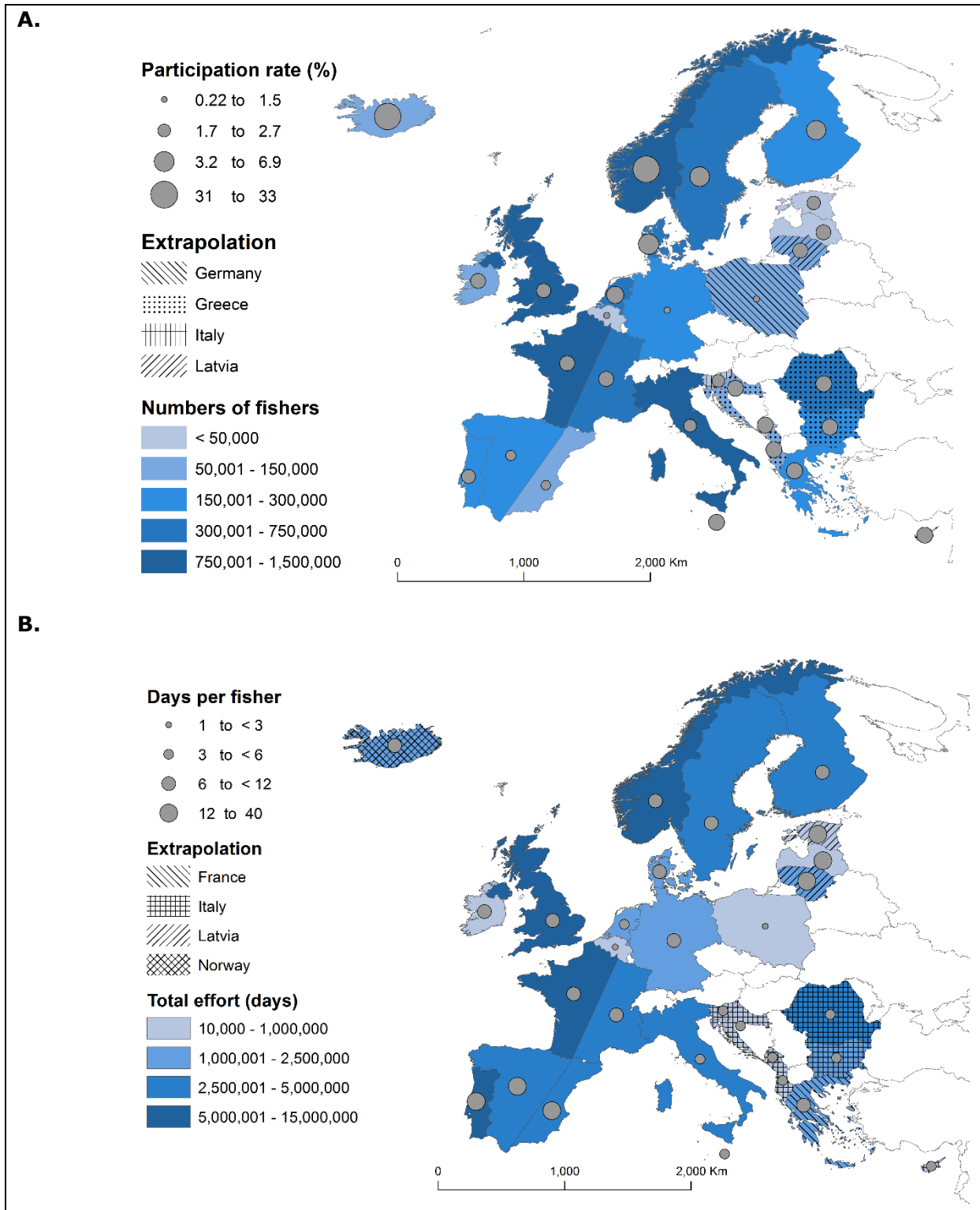
A request was sent to the ICES WGRFS for any additional studies. No studies have been conducted and reported since the compilation of data by the WGRFS in 2016 (Hyder *et al.*, in press). Correspondence with members of the WGRFS indicated that multispecies pilot studies are planned for 2017 to cover requirements under the EU-MAP (EU, 2016a; 2016b) for several countries including Belgium, with pilot studies required for countries with no MRF data within 2 years. In addition, Norway is conducting a national survey of MRF. Thus, it would be prudent to update the analysis of MRF in 2019 once the new survey data have been reported, as it is likely that many of the uncertainties in the analyses will be substantially reduced.

**Table 4: Estimates of numbers, participation, expenditure, and activity by anglers in Europe, split into the Atlantic and Mediterranean regions.**

Category	Total	Atlantic	Med
Numbers (millions)	8.67	5.90	2.77
Participation (%)	1.60	1.70	1.41
Expenditure (billion €)	5.89	4.97	0.92
Spend per angler (€)	679	842	342
Activity (million days)	77.6	56.8	20.9

**Source:** reproduced from Hyder *et al.* (in press).

**Figure 1: Estimated numbers and participation rate (A) and effort (B) of marine recreational fishers in Europe where cross-hatching indicates the country used to extrapolate where no data exist.**



Source: reproduced from Hyder *et al.* (in press).



## 2.6 Tourist fisheries

Many tourists undertake sea fishing while visiting European countries. It is difficult to capture the data on their catches since they will not be included in many traditional household or diary off-site survey techniques. The only way to estimate tourist MRF is through compilation of onsite surveys, recruiting tourist diarists, or through specific surveys of tourists entering the country. Fishing tourism can bring a lot of economic benefit to some areas. In Ireland, it was estimated that in 2011, sea anglers from overseas totalled 113,000 angling trips and spent a total of between 56 and 67 million euro, with each spending around 858 euro during their visit (Failte Ireland, 2015). The study estimated that Ireland had between 132,000 and 144,400 tourist anglers from overseas in 2014. Across Europe, countries take several different approaches to manage marine angling tourism. Some countries do not require licences, while others provide licences for free or for a charge. Countries with licences rely on tourist awareness of the restrictions and ability to enforce licences. In some countries, the situation is more complex, with individual regions issuing licences with their own requirements (e.g. Spain). Tourists using charter boats in countries with licences are often covered by the boat owners licence. In Spain, a licence is required for tourists MRF that are issued by Autonomous Regions (Spain Info, 2017). In Portugal, a licence is required by all those fishing in the sea, including tourists (DGRM, 2017). A licence is required in Germany that varies depending on the state and a coastal fishing permit is needed in the Baltic (Fishing Licence, 2017). Even in countries which require a licence to fish, it is not clear whether tourists must submit a catch return, and so tourism fishing is one area where data is severely lacking.

As with many countries, the lack of licences in Norway makes it challenging to identify tourists for surveys. A study in 2011 aimed to assess the economic impact of tourism, and made assessments based on tourism spend on package fishing holidays (Borch *et al.*, 2011). A total of 104 million euro was estimated to be spent each year, with a total of 434 enterprises belonging to the Industrialised Fishing Tourism Sector. Another study estimated catches by tourists by recruiting tourist-fishing businesses to complete catch and effort diaries (Vølstad *et al.*, 2011). They found that the total catch of all species was 3,335 tonnes (including 1,613 tonnes of cod) and the catch of coastal cod within the Industrialised Fishing Tourism Sector was insignificant compared to commercial and resident MRF. The Norwegian studies showed that a large amount of effort is required to estimate catches by recruiting diarists from the tourism sector. The studies also demonstrate that, even for a country which has a large number of tourist fishers, the catches are insignificant in comparison to commercial and resident recreational catches. Therefore, this sector is not included further in the assessment here. However, economically and in terms of total catch from certain stocks on a regional scale, the contribution of angling tourism may be significant and more information is required to understand how these fisheries can be managed and developed in the future. The use of on-site creel surveys and better collaboration between countries at a regional level would help to close this data gap.

## 2.7 Licensing & regulation

The licensing and regulation of MRF is not consistent across Europe, and varies considerably between countries. It is beyond the scope of this study to provide a comprehensive review of governance structures across Europe, so only a short summary is provided. A review of the national definitions and management of MRF by Pawson *et al.* (2008) found that 14 countries in Europe required some sort of licence, 10 had bag limits and all had gear restrictions. Updating this analysis demonstrated that inconsistencies in licencing both within and between countries across Europe remain (Table 5). No licence is

required in some countries (e.g. Slovenia), others, like the UK, have licences for freshwater, but not for MRF. In other countries, licences are required for specific platforms (e.g. vessels in France and Greece), or are required for all MRF (e.g. Germany, Portugal). The licencing requirements can even vary by region within a country (e.g. Spain) (Table 5). Estimating recreational catches is inevitably more straightforward in countries with a list of licensed fishers, from which governments can recruit diarists or ask for catch returns. Regulations can be implemented by Europe for specific stocks, but can also vary both within and between countries within Europe. For example, bag limits and closed seasons have been implemented for sea bass and western Baltic cod (EU, 2016b) in response to declining levels of stocks and significant recreational catches (ICES, 2016d; 2016f). European MS also have minimum conservation reference sizes (MCRS) where fish below a size threshold must be released. Gear restrictions, seasonal closures, catch limits and bag limits vary both within and between countries and stocks, with limits applied to both resident and tourist MRF (Table 5). Estimating recreational catches is inevitably more straightforward in countries with a list of licensed fishers, from which governments can recruit diarists or ask for catch returns.

**Table 5: The licensing, bag limits and gear restrictions for MRF in selected European countries.**

Country	Licence or permit	Bag limits	Gear restrictions
Albania	Yes	Unknown	Yes
Belgium	No	Yes	Yes
Bulgaria	Unknown	Unknown	Unknown
Croatia	Yes (shore)	Yes	Yes
Cyprus	Yes (boat)	Yes	Yes
Denmark	Yes	Yes	Yes
Estonia	Yes (fishing card)	No	Yes
Finland	Yes (part)	Unknown	Yes
France	Yes (vessel)	Yes	Yes
Germany	Yes (ticket)	Yes	Yes
Greece	Yes (boat only)	Yes	Yes
Iceland	No	Unknown	Unknown
Italy	No	Yes	Yes
Latvia	Yes (part)	Yes	Yes
Lithuania	Yes (permit)	Yes	Yes
Malta	No	Yes	Yes
Montenegro	Yes	Unknown	Unknown
Netherlands	No	Yes	Yes
Norway	No	Yes	Yes
Poland	Yes	Yes	No
Portugal	Yes	Yes	Yes
Republic of Ireland	No	Yes	Yes
Romania	Unknown	Unknown	Unknown
Slovenia	Yes (sports)	Yes	Yes
Spain	Yes (regional)	Yes	Yes
Sweden	Yes (part)	Yes	Yes
Turkey	Yes	Unknown	Unknown
UK	No	Yes	Yes

**Source:** Updated following Pawson *et al.* (2008).

## 2.8 Recommendations

Based on this review of MRF, the following recommendations are appropriate:

- There are many definitions of MRF, with the ICES WGRFS (ICES, 2013) definition capturing the situation in Europe best and should therefore be used. It is: *“the capture or attempted capture of living aquatic resources mainly for leisure and/or personal consumption. This covers active fishing methods including line, spear, and hand-gathering and passive fishing methods including nets, traps, pots, and set-lines”* (WGRFS, 2013).
- An existing study has estimated that each year almost 9 million people or 1.6% of the European population participated in MRF spending almost 6 billion euro and fishing for around 77 million days. This is an important activity within Europe, so should be treated as a discrete sector alongside commercial fishing and aquaculture.
- There is large variation in the understanding of MRF across Europe varying from countries with good understanding (e.g. Germany) to countries with little data (e.g. Montenegro). There were less data for Mediterranean and Black Seas countries making any assessment of impact or value difficult, and time series data are limited for most countries. Hence, there is a need for regular data collection on MRF across Europe as mandated in the EU-MAP (EU 2016a) that should be implemented.
- A broad range of species are caught by MRF, yet mandatory MRF data collection within Europe only focusses on a very small set of species outlined in the EU-MAP (cod, sea trout, salmon, eel, sea bass, elasmobranchs, pollack and tuna). Hence, it would be prudent to develop multispecies surveys that cover all MRF species, as there is little difference in cost.
- Tourist fisheries can be large in a number of countries (e.g. Norway), but there is little knowledge of the benefits or impacts of this sector. More information is required to understand how these fisheries can be managed and developed in the future.
- MRF licensing varies between countries and is inconsistently applied. Licensing would help to develop surveys and understanding of the sector, as it would provide a list of known individuals practicing in MRF to sample, making data collection much simpler. However, this requires that licensing is agreed at a country level, supported by MRF, and enforced.



### 3 CHARACTERISATION OF SEMI-SUBSISTENCE FISHERIES

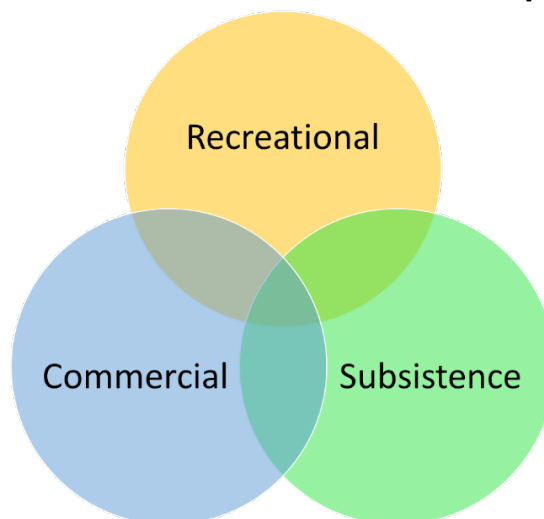
#### 3.1 Summary

The literature was searched to develop a definition and characterise SSUBF. Defining SSUBF is difficult due to the requirement for a threshold to separate it from subsistence and commercial fishing. Even with a definition including a threshold, it does not help to separate SSUBF in reality, as information does not exist either on the earnings from sales or total earnings of the fishers involved. This makes any meaningful separation from recreational and commercial fisheries at best difficult, and at worst impossible. As a result, it was not possible to estimate the value or impact of SSUBF in Europe. Investigation of two case studies showed that the catches and value are captured under recreational or commercial fisheries sampling programmes. Hence, the most appropriate action for SSUBF in Europe is for individual countries to identify where these fisheries exist and ensure that they are captured either in the recreational or commercial catch sampling programmes. This is a pragmatic approach that will capture value and impact without having to develop a legal definition and new legislation. Given the lack of data on value and impact of SSUBF, only MRF will be considered in the subsequent chapters.

#### 3.2 Definition

Consideration of subsistence fishing activities is particularly relevant to coastal fisheries policy, yet formal recognition of subsistence fishing is often absent from policy frameworks (Schumann and Macinko, 2007). A critical problem is the meaning of the term subsistence. This is in part because subsistence, compared with other types of fishing, is diffuse, sporadic, and difficult to monitor. Semi-subsistence fishing is even more challenging as there is no legal definition and is likely to be included under either commercial or MRF surveys (Figure 2). The demarcation between recreational and commercial fisheries in Europe is reasonably clear and the fisheries legislation of many countries ban sales of recreational catches, although exceptions exist (i.e. Norway). However, the demarcation between subsistence fisheries and the recreational and commercial fisheries sectors is difficult (Macinko and Schumann, 2007, EIFAC, 2008).

**Figure 2: Semi-subsistence fisheries is at the boundary between subsistence, commercial and recreational fisheries, so is likely to be covered under either commercial or recreational fisheries sampling.**



Source: EURecFish.

To develop a definition of SSUBF it is first necessary to define subsistence fishing and then apply a threshold above which SSUBF becomes commercial fishing. Many definitions of subsistence fishing are available, some of which have been compiled in Table 6 and generally relate to meeting nutritional needs. However, a formal recognition of (semi-) subsistence fishing is often absent from legislation, because it has been included in the more general legal definition of recreational fisheries as any non-commercial fishery (Table 1). While the demarcation between recreational and commercial fisheries is reasonably clear within Europe (Section 2.2), the demarcation between subsistence and recreational fisheries is absent. This is because separating recreational fisheries and subsistence fisheries is very difficult, as many recreational fishers have strong subsistence-like incentives to harvest fish (Macinko and Schumann, 2007; EIFAC, 2008). For the purpose of this study, subsistence is defined as fishing for livelihood rather than profit, regardless of whether the catch is eaten or sold.

An extensive review of literature has been done to identify a definition for SSUBF, as it is not possible to assess either the value or impact without simple clear demarcation that can be applied to existing data. The literature on SSUBF is dominated by North American research focussing on Arctic indigenous populations. To our knowledge, only three fisheries are recognised as semi-subsistence and have specific management regimes: Alaska's subsistence halibut fishery (Macinko and Schumann, 2007; Schumann and Macinko, 2007); the International Whaling Commission's special treatment of aboriginal subsistence whaling; and South Africa's Marine Living Resources Act, (South African Government, 1998; Hauck *et al.*, 2002; Harris *et al.*, 2002). South Africa has created a Subsistence Fisheries Task Group (SFTG) to explore implementation of a subsistence management regime based on the motivations of fishers.

It is possible to define SSUBF using a threshold of income, either as a limit to the value of catch sold or as a proportion of the total income comprised of sales of the catch. However, there is no legal definition or an accepted cut off for semi-subsistence, so this definition is very difficult to use practically. In fact, under EU legislation any fishery where catches are sold is considered commercial, so covered under commercial fisheries data collection and management regimes. Conversely, where catches are not sold, this activity and its impact are generally monitored as recreational fisheries. Hence, it is likely that any data on SSUBF will be collected within either commercial or recreational fisheries surveys, and not recorded separately. This makes partitioning of data between recreational or commercial and SSUBF very difficult and renders any threshold definition unusable. This does not mean that these fisheries do not exist in Europe, in fact there have been questions in the European Parliament on specific fisheries in Croatia (Parliamentary Question, 27 January 2014, E-000760-14).

Despite noting that *a priori* clarity in the definition of SSUBF would be useful in some respects, the precise definition of SSUBF could emerge from the management process itself. For example, in South Africa fishing sectors (recreational and subsistence) are distinguished by means of characteristics such as the main use of resource, income level, needs met by the resource, locality of harvest, who does the harvesting, gear used, origin of the fishery, and value of the resource (South African Government, 1998; Hauck *et al.*, 2002; Harris *et al.*, 2002). This approach could be followed in future in Europe, but it is not possible to separate (semi-)subsistence from commercial or recreational fisheries at present. As a result, the aim of this work area will be to identify and analyse recognised SSUBF as case studies. This will help to identify further SSUBF in Europe and provide recommendations to ensure data collection in the future and guidance on further research priorities. SSUBF estimates of value or impact will be based on review of existing studies where catch estimates have been provided. The criteria to define SSUBF will be

investigated rather than a comprehensive analysis of the quality of the data. In addition, a more comprehensive analysis will be provided for specific case-studies

**Table 6: List of existing definitions of subsistence fisheries.**

Definition	Source
Fishing for aquatic animals that contribute substantially to meeting an individual's nutritional needs. In pure subsistence fisheries, fishing products are not traded on formal domestic or export markets but are consumed personally or within a close network of family and friends. Pure subsistence fisheries sustain a basic level of livelihood and constitute a culturally significant food-producing and distributing activity.	FAO (2008)
Local, non-commercial fisheries, oriented not primarily towards recreation but for the procurement of fish for consumption of the fishers, their families and community.	Berkes (1988)
Subsistence fisher: a person who regularly catches fish for personal consumption of his or her dependents, including one who engages from time to time in local sale or barter of excess catch, but does not include a person who engages on a substantial scale in the sale of fish on a commercial basis.	South Africa's Marine Living Resources Act (South African Government, 1998)

**Source:** compiled by EURRecFish.

### 3.3 Characterisation

The main objective here is to demonstrate how SSUBF is mostly covered either under recreational fisheries or the commercial fleet (usually within the small-scale fleet) and varies between countries. The gears used by the SSUBF are more similar to those commonly used by commercial fleets rather than the recreational fisheries. Common gears used by subsistence fishermen include gillnets, longlines and traps. Recreational fisheries in some countries are also allowed to use these gears, generally with more limitations than the commercial fisheries (Section 2.4). The most extreme case is Norway where resident recreational fishers are allowed to sell their catch up to a threshold of value. In these cases, it is even more difficult to distinguish between the recreational, semi-subsistence, and commercial small-scale fleets. Furthermore, in these specific cases the regulation is more similar to commercial than recreational fisheries, so it is common for these fishermen to have to complete logbooks and sale notes. In addition, limitations are placed on effort and catches, and control is often stricter. The case studies discussed in Sections 3.3.2 and 3.3.3 are evidence of the different approaches found among different countries in Europe.

#### 3.3.1 Semi-subsistence catch estimates

In this section, SSUBF catch estimates from several countries are provided. These estimates are based on the Sea Around Us project (<http://www.searoundus.org/>), where the main objective was to reconstruct the estimates of unreported catches within individual country Exclusive Economic Zones (EEZs). A critique of the methods used to compile or reconstruct data is not the aim of this study, solely to identify likely levels of SSUBF catches. SSUBF catch estimates were obtained from different studies (Table 7), but comparisons between estimates are not valid as the criteria used to define these fisheries differed. This highlights challenges with definition and distinguishing these fisheries from commercial and recreational fisheries (Section 3.2). Hence, the main aim of this section is to analyse the criteria used to define SSUBF and assess if it is currently possible to develop estimates of impact.

Semi-subsistence catch estimates were not available within any official statistics provided by Europe. However, these catches have been estimated within academic studies that aim

to provide long-term reconstructions of fisheries over time (<http://www.seaaroundus.org/>). These have been made for several countries, but often it was assumed that the SSUBF did not exist (Table 7). For some countries, these were obtained by extrapolating recreational and small-scale fisheries data to country level and assuming that a percentage of the total catch was from subsistence catches (e.g. Romania, Bulgaria). In other cases, a percentage of the recreational or small-scale total catches was considered to be harvested and taken to represent subsistence catches (e.g. Cyprus, Malta). In the case of Spain (Atlantic area), these estimates were based on extrapolation from a small number of interviews conducted at a regional level. Whilst these estimates serve a purpose for estimating trends in fisheries over time, they do not provide a robust and consistent estimate that can be easily compared between countries or summed to a European level.

**Table 7: 2010-2014 Semi-subsistence fisheries catch estimates (tonnes).**

Country	2010	2011	2012	2013	2014	Source
Bulgaria	37	36	34	32		Keskin et al. (2015)
Cyprus	246	246	246	246	246	Ulman et al. (2015)
Croatia	6,000	6,000	6,000	6,000	6,000	Matić-Skoko et al. (2014)
Slovenia	56	58	58	58	58	Bolje et al. (2015)
France	9,010	9,053	9,097	9,143	9,189	Bultel et al. (2015)
Greece	284	284	284	284	284	Moutopoulos et al. (2015)
Italy	4,000	3,000	3,000	3,000	4,000	Piroddi et al. (2014)
Malta	36	37	46	51	54	Khalfallah et al. (2015)
Portugal (mainland)	120	122	113	108	88	Leitão et al. (2014)
Romania	628	615	602	589	577	Bănaru et al. (2015)
Spain	8,009	8,021	8,021	8,021	8,021	Atlantic - Villasante et al. (2015) Mediterranean - Coll et al. (2015)
Sweden	542	518	522	526	531	Persson (2014)

**Source:** compiled by EUREcFish.

As a large proportion of any SSUBF is likely to be done by small-scale fisheries, a larger proportion of small boats within a fishery may indicate that the biological and socio-economic impact of SSUBF could be important. Information on catches taken by these vessels is collected under the Control Regulation (EU, 2009) as they are registered as commercial fishing vessels (<http://ec.europa.eu/fisheries/fleet/index.cfm>). For each of the countries where subsistence catches were estimated (Table 7), the proportion of small fishing vessels of under 10 m in length that could engage in SSUBF was calculated (Table 8). If semi-subsistence activity occurs, it is most likely in the smallest vessels (0-6 and 6-10 m segments) and these make a large proportion of the total numbers of vessels in many countries (above 85% in most cases) (Table 8). Furthermore, the number of vessels in Greece, Portugal, Spain, Italy, and Croatia of less than 6 m in length is large (Table 8).



**Table 8: Number of vessels by fleet segment by EU member state with semi-subsistence estimates.**

Country	0<6m	6<10m	10<12m	>12m	Total	% vessels
Bulgaria	656	1,095	64	95	1,910	92
Cyprus	377	335	48	38	798	89
Croatia	3,960	2,564	369	602	7,495	87
Slovenia	82	66	11	16	175	85
France	785	4,103	937	892	6,717	73
Greece	5,317	8,485	493	871	15,166	91
Italy	2,500	5,206	969	3,619	12,294	63
Malta	453	339	51	63	906	87
Portugal	3,579	3,327	312	757	7,975	87
Romania	14	83	23	18	138	70
Spain	2,953	3,195	659	2,494	9,301	66
Sweden	250	651	200	157	1258	71

% vessels represent the percentage of the total vessels that are under 10 m in length.

**Source:** EU fleet register 2017 data.

Unlike large-scale fisheries, official statistics for small-scale fisheries are often limited, and are lacking for MRF. Several exemptions and conditions in the Control Regulation (EU, 2009) may result in incomplete landings and sales notes for small-scale fisheries. There is no obligation to register the catch (and discards) where the catch is less than 50 kg live weight for each species (Article 14). Similarly, vessels of under 10 m in length are exempt from requiring sales notices for any commercial landings of 50 kg or less of fish or fish products in cases where the countries have a sampling system in place (Article 65). Catch, effort, and socio-economic data are therefore dependent on fisheries sampling. This has traditionally hampered the understanding of these fisheries and may have underestimated their impacts.

### 3.3.2 Croatia

The baseline regulation governing fisheries in Croatia is the Sea Fisheries Act (Matic-Skoko *et al.*, 2011). Four types of fisheries are defined: commercial, subsistence, recreational, and sport fishing. Commercial fishing is a profit-making activity, while fish and other marine organisms caught during subsistence, recreational or sport fishing are not allowed to be sold and are intended for fishers and their families' own consumption. Use of limited types and quantity of nets primarily differentiates subsistence fishing practices, which are trap and line fishing orientated, from recreational and sport fishing. SSUBF are still considered a socio-cultural activity for most of the island and shoreline inhabitants. The key distinguishing factor between commercial and subsistence fisheries are the intention of the activity, type and quantity of fishing gear allowed, and daily catch limits (daily catch limit of 5 kg).

For the period 2009-2010, 12,000 vessels were registered in Croatia as subsistence vessels and 3,360 under the commercial small-scale fleet category. The interactions between these two fisheries created conflicts, with commercial small-scale fishermen arguing that the catches of subsistence fishers were relevant due to the number of fishermen, and that part of the catches were sold in unofficial or black markets, with a resulting negative impact on price. Limited institutional capability to conduct surveillance and monitoring of fishing activities of subsistence fishers who did not comply with regulations (gear type and limits, daily bag limits) were considered as real management problems.

When Croatia became part of the EU, the category of subsistence fisheries that existed in Croatia was ultimately a hybrid model between leisure and commercial fisheries which did not exist in the EU legislation. Article 17 of the Mediterranean Regulation (Council Regulation (EC) No 1967/2006 of 21 December 2006) requested Member States to ensure that leisure fisheries are conducted in a manner compatible with the sustainable exploitation of living aquatic resources and not significantly interfere with commercial fishing. Countries must ensure that catches taken in leisure fisheries are not marketed (with limited exceptions). In addition, the use of certain fishing gears such as nets is prohibited. Leisure fisheries imply fishing activities for recreation and sport (Table 1), and are widespread in the Mediterranean and can have a significant impact. As a result, the use of nets in leisure fisheries is prohibited in Croatia.

During the accession negotiations for the EU membership, Croatia was granted a transitional period where 2,000 fishermen continued to operate in this category and to use nets until 31st of December 2014. In parallel, subsistence fishermen had the opportunity to register under the category of commercial fisheries. By January 2015, the transition was effective and subsistence fishermen in Croatia are now part of the commercial fleet. However, under the national fisheries management perspective, these semi-subsistence fishermen are regulated similarly to before (fishing gear characteristics, amount, variety, and allowed catch - Alen Aldo pers. comm.) despite the legal categorisations as commercial.

Croatian subsistence catches have never been accounted for in landings and are generally very scarce in the literature. Matic-Skoko *et al.* (2014) made a reconstruction of recreational subsistence catches in Croatia from 1950-2010. The methodology used was based on the number of subsistence fishers with data obtained from the different reports available. The average impact in the last decade from 2000-2010, was close to 6,000 tonnes per year. It is relevant to comment that there is not any analysis of the quality and accuracy of the data and these estimates should be taken with care.

### 3.3.3 Norway

In this case study, Norwegian resident recreational fishermen are analysed because they can use commercial gears and sell catches up to 50,000 NOK (about 5,300 euro) each year via registered fish landing sites. The gears that can be used are:

- Handheld tackle (e.g. fishing rod, handline).
- 1 jigging machine.
- Gillnets up to 210 m. When targeting cod, the max length is 165 m.
- Longline up to 300 hooks.
- Up to 20 pots or fyke nets.

Here, the catches sold via registered landing sites are analysed. Additional catches not sold by recreational fishermen in Norway are excluded because survey data are lacking, although the number is expected to be much higher. Catches collected from the registered landing sites sales notes were 1,781 tonnes and 2,335 tonnes in 2015 and 2016 respectively (Table 9). Nearly 50 species were caught, but two were the main target species: cod (*Gadus morhua*) with near 50% of the total catches and saithe (*Pollachius virens*) with 25%. These catches were fished in coastal waters and may be relevant for certain stocks e.g. Norwegian coastal cod.

**Table 9: Tonnage of catches of all species sold by Norwegian recreational fishers in 2015 and 2016.**

Catches (tonnes)	2015	2016
Total	1,781	2,335

**Source:** Selected by IMR for 2015 and 2016 from Norwegian Directorate of Fisheries data.

### 3.4 Recommendations

Based on the review of SSUBF, the following recommendations are appropriate:

- The monitoring and surveys of SSUBF are very limited, and specific data are lacking. Without more data, it is not possible to assess the impact of SSUBF further, so only MRF is considered in the subsequent chapters.
- Under the current European regulation, SSUBF are not regulated and have no legal definition. Only two examples of SSUBF in Europe could be identified, and these are either captured under existing reporting structures (e.g. Norway) or have been transitioned to commercial (e.g. Croatia). This, alongside the challenges with defining SSUBF, indicated that a new category should not be produced and the status quo maintained (recreational and commercial fisheries).
- SSUBF catches can easily be captured either under the commercial or recreational fisheries sampling programmes. This already happens in most cases, but it is important that individual countries should identify if they have any SSUBF and ensure that the sampling system covers these catches. In some cases, it may be necessary to set up additional sample frames to cover these data.
- Seventy four percent of the total EU fishing fleet is small scale (vessels <12 m) and it is possible that the smallest vessels could be similar to SSUBF. The biological and socio-economic impact of these vessels is unknown and may be relevant in the coastal areas and communities.



## 4 SOCIO-ECONOMIC IMPACT OF MARINE RECREATIONAL FISHING

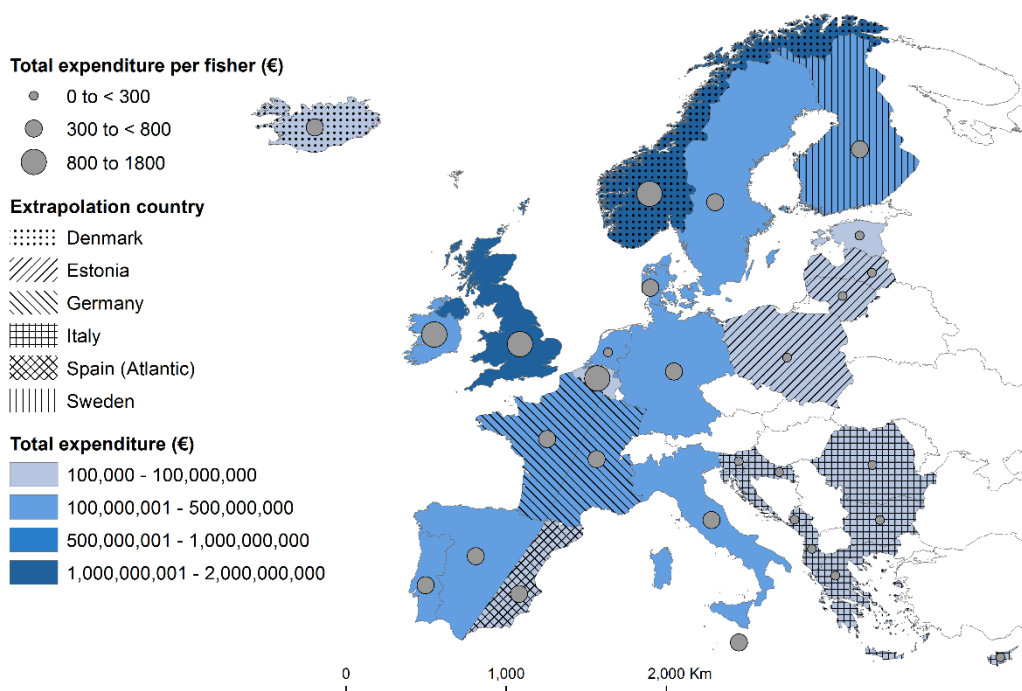
### 4.1 Summary

An analysis of the economic impact of MRF within the European sea regions was conducted, using an Input-Output methodology. Raw expenditure by anglers was collected using a meta-analysis review, and total economic impact, GVA, and number of FTEs were estimated. The total production contribution of MRF was estimated to be 10.5 billion euro and supporting 100,000 jobs (FTEs). It was concluded that the economic contribution of MRF within the European sea regions is high enough to consider implementing common and stable economic data collection for MRF. Given these estimates, MRF in Europe should be considered a discrete sector for development alongside commercial fisheries and aquaculture under the CFP.

### 4.2 Introduction

MRF has been shown to have important economic and social benefits in other parts of the world, making a significant contribution to the economy and employment (e.g. US – NMFS, 2016). A recent study of MRF in Europe showed that direct expenditure was 5.9 billion euro annually (Table 4) and varied between countries (Figure 3) (Hyder *et al.*, in press). Recreational fisheries have been recognized as an important source of income for national economies (Haab *et al.*, 2001), but can also impact on marine biodiversity (Lewin *et al.*, 2006). There is often a conflict between recreational and commercial fisheries (Arlinghaus, 2005) as they target the same stocks, but have different management objectives (Hyder *et al.*, 2014). Hence, recreational fisheries should be included as an integral part in the ecosystem, social and economic considerations of fisheries management in Europe.

**Figure 3: Estimated direct expenditure of marine recreational fishers (expenditure per fisher and total expenditures) in Europe where cross-hatching indicates the country used to extrapolate where no data exist.**



Source: reproduced from Hyder *et al.* (in press).

There are many approaches to estimate the economic value of an activity or sector with good summaries available elsewhere (e.g. Parkilla *et al.*, 2010; EFTEC, 2015). Most studies of recreational fisheries focus on either willingness to pay (e.g. Toivonen *et al.*, 2004) or estimates of expenditure (e.g. Armstrong *et al.*, 2013).

Willingness to pay is estimated using revealed preference or stated preference valuation methods. Revealed preference methods use information on observed behaviour to infer the demand for, and value of, goods and services like recreational fishing experiences. In these cases, there are no explicit markets and therefore no explicit prices available to quantify the value of experiences. Stated preference methods, on the other hand, utilise responses to questions, to provide information about people's preferences and values. In both cases, the methods attempt to capture the non-use value of recreational fishing. Outside Europe, travel cost methods (revealed preference) have been used for economic valuation of recreational fisheries where the distance travelled is related to attributes of the site fished (e.g. catch, numbers of other anglers, beauty etc.). For example, this method has been used to compare the economic value of recreational fisheries with the cost associated with the protection of the Great Barrier Reef in Australia (Prayaga *et al.*, 2010). The most common stated preference approach historically has been contingent valuation (e.g. Lew and Larson 2014), which asks people questions to reveal their willingness to pay for specified programmes and can be used to assess management policies. However, these approaches are not suitable to address questions around social (employment) and economic (expenditure) impacts of recreational fisheries.

An alternate approach is to estimate the economic activity supported by MRF. This can be done using an Input-Output modelling approach (e.g. Armstrong *et al.*, 2013), a quantitative approach that represents the interdependencies between different branches of a given economy. However, it is difficult to use this approach to judge the impact on the economy of a change in policy or management as these are usually incremental changes. In addition, in the absence of recreational fishing, this expenditure could go to other activities (e.g. other hobbies, cars, etc.), so would still contribute to the economy. Nevertheless, recreational fisheries exist in Europe and the past economic activity supported can be estimated, even where there is a lack of national data. The Input-Output methodology is traditionally used to produce multipliers for direct expenses (increase in demand) to estimate the indirect and induced effects, and derive the Gross Value Added (GVA) and the employment based on the structure of the economy. The direct effect relates to the increase in output as producers react to meet the increased demand. Indirect effects capture the increase in demand on suppliers down the supply chain due to the increased production. The direct and indirect effects impact on the level of household income throughout the economy with a proportion spent on final goods and services (induced effect). The total economic impact is the sum of direct, indirect and induced effects and measures the overall impact of an activity on the economy.

Here, the economic and social impacts of MRF were estimated in six regions of Europe. Building on data compilation and analysis from a previous study (Hyder *et al.*, in press), the direct, indirect, and induced expenditure was estimated using an Input-Output methodology. The GVA and number of employed Full Time Equivalents (FTEs) supported by MRF expenditure were assessed. The results are considered in the context of other industries across Europe.

## 4.3 Methods

### 4.3.1 Expenditure data

To obtain the direct expenses, a review of the existing MRF literature was conducted that included papers, reports, and data sources (both national and regional). Much of the MRF expenditure data had already been compiled (Hyder *et al.*, in press), but was limited especially in the Mediterranean and provided only the total direct expenditure including taxes (Table 10). Additional data on expenditure was identified from the literature where available and used to fill in gaps for countries in the Black Sea that were not covered by the existing analysis (Table 10).

**Table 10: Sources of information and extrapolations used for distribution and employment.**

Country	Direct expenses	Distribution	Source and year ( ) of I-O matrix	Employment
Albania	Hyder <i>et al.</i> (in press)	Italy	Italy	Italy
Belgium	Hyder <i>et al.</i> (in press)	Coussement (2012)	Eurostat;2010 (2017)	Eurostat;2010 (2017)
Bulgaria	Hyder <i>et al.</i> (in press)	Aydin <i>et al.</i> (2013)	Eurostat;2010 (2017)	Italy
Croatia	Hyder <i>et al.</i> (in press)	Tragsatec (2004)	Italy	Italy
Cyprus	Hyder <i>et al.</i> (in press)	Tragsatec (2004)	Eurostat;2009 (2017)	Eurostat;2009 (2017)
Denmark	FVM (2010)	FVM (2010)	Eurostat;2010 (2017)	Eurostat;2010 (2017)
Estonia	Hyder <i>et al.</i> (in press)	Austria	Eurostat;2010 (2017)	Lithuania
Finland	Hyder <i>et al.</i> (in press)	Toivonen <i>et al.</i> (2004)	Eurostat;2011 (2017)	Denmark
France	Levrel <i>et al.</i> (2013) Herfaut <i>et al.</i> (2013)	Levrel <i>et al.</i> (2013) Herfaut <i>et al.</i> (2013)	Eurostat;2010 (2017)	Spain
Germany	Hyder <i>et al.</i> (in press)	Austria	Eurostat;2010 (2017)	Eurostat;2010 (2017)
Greece	Hyder <i>et al.</i> (in press)	Tragsatec (2004)	Eurostat;2011 (2017)	Italy
Ireland	Hyder <i>et al.</i> (in press)	UK	Eurostat;2010 (2017)	UK
Italy	Hyder <i>et al.</i> (in press)	Tragsatec (2004)	Eurostat;2010 (2017)	Eurostat;2010 (2017)
Latvia	Hyder <i>et al.</i> (in press)	Austria	Eurostat;2010 (2017)	Lithuania
Lithuania	Hyder <i>et al.</i> (in press)	Austria	Eurostat;2010 (2017)	Eurostat;2010 (2017)
Malta	Hyder <i>et al.</i> (in press)	Italy	Eurostat;2008 (2017)	Italy
Montenegro	Hyder <i>et al.</i> (in press)	Italy	Eurostat;2009 (2017)	Italy
Netherlands	Smit <i>et al.</i> (2004)	Smit <i>et al.</i> (2004)	Eurostat;2010 (2017)	Eurostat;2010 (2017)
Norway	Hyder <i>et al.</i> (in press)	Toivonen <i>et al.</i> (2004)	Eurostat;2011 (2017)	Eurostat;2011 (2017)
Poland	Hyder <i>et al.</i> (in press)	Estonia	Eurostat;2009 (2017)	Lithuania
Portugal	Veiga (2012)	Veiga (2012)	Eurostat;2010 (2017)	Spain
Romania	Hyder <i>et al.</i> (in press)	Aydin <i>et al.</i> (2013)	Eurostat;2010 (2017)	Italy
Slovenia	Sullivan <i>et al.</i> (2003)	Sullivan <i>et al.</i> (2003)	Eurostat;2010 (2017)	Italy
Spain	Hyder <i>et al.</i> (In press) Zarauz <i>et al.</i> (2013) Morales-Nin <i>et al.</i> (2015)	Tragsatec (2004) Zarauz <i>et al.</i> (2013) Morales-Nin <i>et al.</i> (2015)	Eurostat;2009 (2017)	Eurostat;2009 (2017)
Sweden	SCB (2015)	SCB (2015)	Eurostat;2010 (2017)	Eurostat;2010 (2017)
United Kingdom	Armstrong <i>et al.</i> (2013) McMinn (2013) Monkman <i>et al.</i> (2015) Radford <i>et al.</i> (2009)	Armstrong <i>et al.</i> (2013) McMinn (2013) Radford <i>et al.</i> (2009)	Eurostat;2010 (2017) Armstrong <i>et al.</i> (2013)	Armstrong <i>et al.</i> (2013)

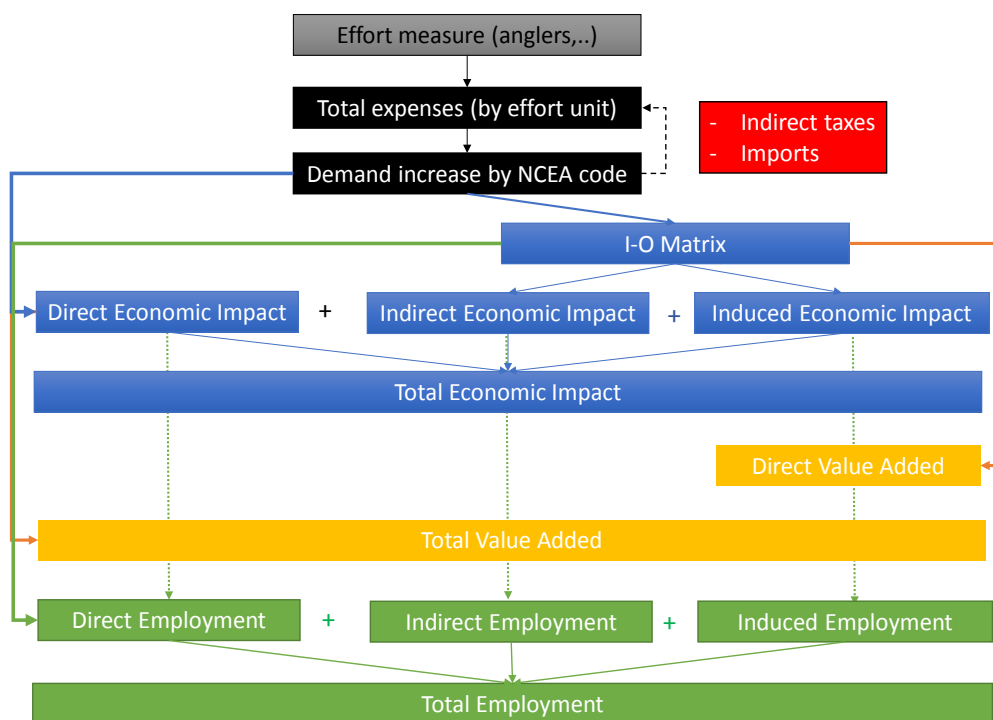
**Source:** compiled by EUREcFish.

### 4.3.2 Input-Output methodology

To estimate the economic activity supported by MRF, an Input-Output modelling approach was used. This is a quantitative economic technique that represents the interdependencies

between branches of an economy and is a commonly used and well-established methodology (see Kowalewski, 2009 for a general review). This approach is based on the principle that if there is an increase in the final demand for a product, there will be an increase in the output of that product, as producers react to meet the increased demand. To understand the data requirements and limitations of the results, some aspects of the methodology should be highlighted including the requirement for Input-Output tables for each country-specific economy and estimates of MRF expenditure. Gross expenditure only provides a snapshot of current levels of expenditure in each country and supports regional household income and employment. Here, gross expenditure is the basis for estimating the economic activity supported by MRF expenditure. This method produces multipliers for the direct expenses (increased demand) for production, which can be used to estimate the economic impacts of incremental spending in an economy, which varies depending on the structure of the economy considered (the Input-Output matrix of each European country) (see Figure 4).

**Figure 4: Process for calculation of the total economic impact of the European MRF from the total expenditure to the calculation of the direct, indirect and induced effects on production, value added, and employment.**



Source: EURecFish.

In this study, the balance of these effects (the net economic impact) was not considered due to the lack of data available to estimate the impact of substitution on the net change in expenditure. A good example of a substitution event is where fishers in a country substitute the loss of their angling by switching to another economic branch or activity (e.g. cars, other hobbies). In this case, the net loss in regional expenditure, income, and jobs could be relatively minor, as it will gain from increased expenditure on other activities, but flow through industries and the multipliers will be different. In addition, MRF expenditure is often a source of income in vulnerable coastal communities, so any changes could direct this income to other areas. For this analysis, the symmetric Input-Output tables at basic prices of the economies were obtained from EUROSTAT, apart from the UK from where values are already provided in the literature (Table 10).



### 4.3.3 Estimation of economic and social value

An estimation of the economic activity supported by European countries with a coastline was provided for the following sea regions: Mediterranean Sea, Black Sea, Baltic Sea, North Sea, North-Western Atlantic waters, and South-Western Atlantic waters. The approach taken was to assign totally or partially the sea area of each country to a region. In total, 27 European countries with a coastline and 23 of the 28 EU Member States were included. These were: Albania, Belgium, Bulgaria, Croatia, Cyprus, Denmark, Estonia, Finland, France, Germany, Greece, Iceland, Ireland, Italy, Latvia, Lithuania, Malta, Montenegro, Netherlands, Norway, Poland, Portugal, Romania, Slovenia, Spain, Sweden and United Kingdom. It should be noted that the remaining five MS have inland (freshwater) recreational fisheries, but the economic impact calculations of these fisheries was not included. The following economic and social indicators were calculated:

- Direct expenditure: spend on goods and services, either capital or operating expenditures as outputs from the model are static.
- Gross Value Added (GVA): measure of the value of economic activity in a reference economy that is comprised of wages, salaries, and profits of businesses.
- Employment: number of jobs supported, reported as the Full-Time Equivalents (FTEs) that represents full and part time jobs in common currency.

Direct expenditure was not available for all countries, so extrapolation from another country was required. A 'donor' country was selected as the most representative of the 'recipient' country (Table 10). Numbers of recreational fishers were taken from Hyder *et al.* (in press) and multiplied by the average expense from the 'donor' country (Table 10). The expenditures were converted to 2015 prices using Harmonised Consumer Price Index obtained from EUROSTAT, and the prices are in 2015 euro (Table 11). Finally, these expenses were partitioned among different branches of the economy (see below). Where these did not exist, a donor country was selected that was most comparable to the recipient country (Table 10) for the cases where no raw (partition among branches) data were available. The expenditure estimates provided by Hyder *et al.* (in press) included taxes and imports, so these were removed to estimate the direct effects. Taxes are removed as they go directly to the government and are redistributed by it, so have no impact on the supply chain. Likewise, imports are removed because their production is not part of the economy of the country. This was done using information of taxes and imports in different countries (UK, Ireland, France, Spain, Norway Sweden) using the same source as the "distribution" (Table 10). Where this information was not available, a flat rate of 25% (approximately the average of those countries for which data was available) was deducted from the direct expenses.

It was necessary to partition total direct expenditure into the different branches of the economy. There were several studies where this has been done for different countries (e.g. Armstrong *et al.*, 2013; Zarauz *et al.*, 2013). However, the difficulty was to obtain a common set of values for all regions, as published studies are only available for some regions (e.g. North Atlantic), but not in other regions (e.g. EU Black Sea). This meant that it was necessary to estimate these values from other data (e.g. GDP, coastline) and effort (e.g. number of MRF or boats). GVA and employment cannot be directly calculated due to the lack of a full value added based Input-Output symmetric matrix. To overcome this limitation, the approach used in Zarauz *et al.* (2013) was taken. The proportion of each branch over the total production was used (obtained from the Input-Output tables) and applied to the overall gross value added and employment (obtained from EUROSTAT - <http://ec.europa.eu/eurostat/>). Using this approach, 'dummy' value added and employment symmetric matrixes were computed, and used to derive the indirect and

induced effect. It should be noted that the approach assumed that the total GVA was equal to the direct expenses, so all the inputs required were produced within the Europe (see Zarauz *et al.*, 2013). This was also consistent with subtracting the imports from the direct expending's explained above. The regional contributions were estimated by partitioning countries into regions and making assumptions about the proportion of expenditure in each region where countries spanned regions (Table 11).

To assess the potential bias in the regional estimates, the level of bias in each estimate of direct expenditure was assessed using a simple semi-quantitative approach (Hyder *et al.*, in press). Each individual country value was assessed for bias which was rated on a seven-point scale, ranging between highly overestimated (+3), negligible bias (0) and highly underestimated (-3). It was necessary to weight the contribution of the bias in a country, so that, for example, a large error in a small estimate does not have as much influence on the overall bias as a small bias in a large value for a country. The relative bias in the overall estimates were ratios, so were categorised by sign on categorical logarithmic scale (negligible < 0.2; 0.2 ≤ minimal < 0.4; 0.4 ≤ small < 0.8; 0.8 ≤ moderate < 1.6; and 1.6 ≤ large).

**Table 11: Partitioning of country expenditure into regions.**

Region	Country	Notes
Baltic Sea	Denmark (Baltic)	Use half total Denmark
	Estonia	
	Finland	
	Germany (Baltic)	Use 87.5% as about 7/8 anglers
	Latvia	
	Lithuania	
	Poland	
	Sweden	
North Sea	Belgium	
	Denmark (North Sea)	Use half total Denmark
	Germany (North Sea)	Use 12.5% as about 1/8 anglers
	Netherlands	
	Norway	
North-Western Atlantic Waters	UK (North Sea)	Use half total UK
	France (Channel)	Use split from Hyder <i>et al.</i> (in press)
	Ireland	Use split from Hyder <i>et al.</i> (in press)
South-Western Atlantic Waters	UK (NWW)	Use half total UK
	France (Biscay)	Use split from Hyder <i>et al.</i> (in press)
	Portugal	
Mediterranean Sea	Spain (Atlantic)	Use split from Hyder <i>et al.</i> (in press)
	Albania	
	Bosnia & Herzegovina	
	Croatia	
	Cyprus	
	France (Mediterranean)	Use split from Hyder <i>et al.</i> (in press)
	Greece	
	Italy	
	Malta	
	Montenegro	
	Slovenia	
	Spain (Mediterranean)	
	Turkey (Mediterranean)	
Black Sea	Bulgaria	
	Georgia	
	Romania	
	Russia	
	Turkey (Black Sea)	
	Ukraine	

Source: EURecFish.

## 4.4 Results

The total economic activity by European MRF was 10.5 billion euro that comprised of 5.1, 2.3 and 3.2 billion euro of direct (after taxes removed), indirect and induced expenditure, respectively (Table 12). This supported almost 100,000 FTEs that included 57,000 from direct, 18,000 from indirect, and 24,000 from induced expenditure respective (Table 12). The amount varied between countries, with the main contributors being the UK and France (Table 12). When these numbers were divided into sea regions, the North Sea (35%) was the largest overall contributor, followed by the North-Western Atlantic Waters, Mediterranean, South-Western Atlantic Waters, and Baltic Seas (15% each), with the lowest contribution from the Black Sea (2.5%) (Table 13; Figure 5). A similar pattern was seen for employment (Table 12; Table 13; Figure 5). It was estimated that every euro spent on MRF had an average effect on the economy of 2.2 euro, with the maximum multiplier in Bulgaria (2.5) and the minimum in the UK (1.7). Results showed the difference in the economic impacts of incremental spending in an economy by sea area with the Black Sea estimated to be the highest (2.4) with North-Western Atlantic waters the lowest (1.8).

The percentage of GVA created from the direct expenditure of marine recreational fishers was estimated to be on average 46%. This implies that 46% of the direct expenses are converted into contribution to the GDP of the European countries. The country with the highest value estimate was Germany (55%), while the lowest was Ireland (33%). The multipliers explain the relationship between the increase in demand and their cumulative total economic impact (TEI). The expenditure required to support one FTE varies between countries (Figure 6B). The maximum spend required was for Denmark (62,909 euro) and the minimum for Estonia (18,979 euro). On average, the result is that an annual expenditure of 49,084 euro is required to support one FTE (47,408 euro for the EU). These differences can also be interpreted by sea area. In this case, the lowest expenditure is required in the Black Sea (42,526 euro) and the highest one in the North Sea (59,403 euro).

Many different biases may be included in the survey data (e.g. recall, avidity, non-response). A full statistical error analysis of the estimates was not done as estimation of the errors was not available. Instead, a semi-quantitative estimate of the magnitude and direction of the bias in the total estimate was assessed (Hyder *et al.*, in press). The overall bias was -0.34 indicating that there was a minimal underestimation of the results. This varied by sea region, with negligible bias in the Atlantic, but significant underestimation found in the Black Sea (-2) (Figure 7).

**Table 12: Estimation of direct, indirect and induced effects, for production and employment for European countries with marine recreational fisheries.**

Country	Production (million euro)				Employment (FTEs)			
	Direct	Indirect	Induced	Total	Direct	Indirect	Induced	Total
Albania	6	3	3	12	85	22	21	128
Belgium	25	15	20	60	209	79	118	407
Bulgaria	24	13	24	61	335	102	156	593
Croatia	16	9	14	39	230	67	89	385
Cyprus	5	2	2	9	110	29	22	161
Denmark	118	56	75	249	1313	243	322	1877
Estonia	4	2	3	10	102	52	59	213
Finland	79	41	60	180	875	179	258	1311
France	1052	515	756	2324	13649	4729	6150	24527
Germany	88	40	48	176	1279	337	341	1957
Greece	53	25	28	106	744	188	185	1116
Iceland	45	25	34	104	380	143	210	733
Ireland	95	43	57	195	988	444	598	2029
Italy	194	103	162	460	2721	790	1052	4562
Latvia	7	4	5	16	188	83	87	358
Lithuania	12	6	7	26	315	140	131	586
Malta	3	2	2	6	37	12	13	62
Montenegro	2	1	1	4	24	7	8	38
Netherlands	113	66	91	270	941	356	538	1835
Norway	866	479	647	1992	7312	2741	4027	14079
Poland	16	9	15	39	229	72	104	433
Portugal	104	51	85	240	1351	468	693	2513
Romania	73	38	57	168	1018	292	369	1679
Slovenia	17	8	11	36	238	62	73	374
Spain	163	87	124	374	2115	797	1009	3921
Sweden	495	228	288	1010	6050	1120	1752	8921
UK	1394	390	585	2370	14489	4057	6085	24632
<b>Total</b>	<b>5070</b>	<b>2258</b>	<b>3206</b>	<b>10534</b>	<b>57325</b>	<b>17609</b>	<b>24469</b>	<b>99431</b>
<b>Total EU</b>	<b>4151</b>	<b>1751</b>	<b>2521</b>	<b>8422</b>	<b>49524</b>	<b>14697</b>	<b>20204</b>	<b>84452</b>

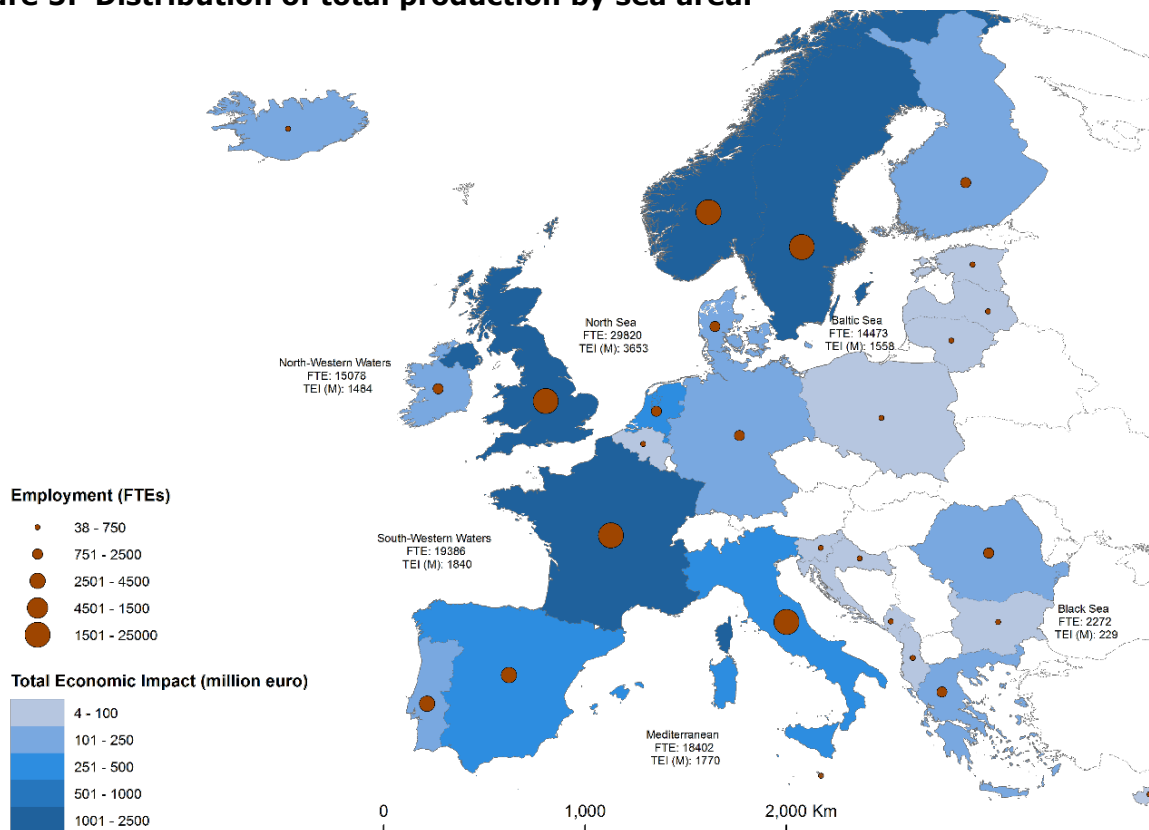
Source: EUREcFish.

**Table 13: Total production (A), GVA (B) and employment (C) by region.**

<b>A. Production (million euro)</b>	<b>Direct</b>	<b>Indirect</b>	<b>Induced</b>	<b>Total</b>
Baltic Sea	749	351	458	1558
Black Sea	97	51	81	229
Mediterranean	791	397	582	1770
North Sea	1771	788	1094	3653
North-Western Atlantic Waters	837	263	384	1484
South-Western Atlantic Waters	825	408	607	1840
<b>Total</b>	<b>5070</b>	<b>2258</b>	<b>3206</b>	<b>10534</b>
<b>B. GVA (million euro)</b>	<b>Direct</b>	<b>Indirect</b>	<b>Induced</b>	<b>Total</b>
Baltic Sea	395	158	197	749
Black Sea	43	21	33	97
Mediterranean	392	163	236	791
North Sea	781	318	673	1771
North-Western Atlantic Waters	378	120	340	837
South-Western Atlantic Waters	417	165	243	825
<b>Total</b>	<b>2406</b>	<b>944</b>	<b>1721</b>	<b>5070</b>
<b>C. Employment (FTEs)</b>	<b>Direct</b>	<b>Indirect</b>	<b>Induced</b>	<b>Total</b>
Baltic Sea	9533	2062	2850	14473
Black Sea	1352	394	526	2272
Mediterranean	10600	3426	4376	18402
North Sea	16522	5368	7930	29820
North-Western Waters	8613	2615	3850	15078
South-Western Waters	10704	3744	4938	19386
<b>Total</b>	<b>57325</b>	<b>17609</b>	<b>24469</b>	<b>99431</b>

Source: EURecFish.

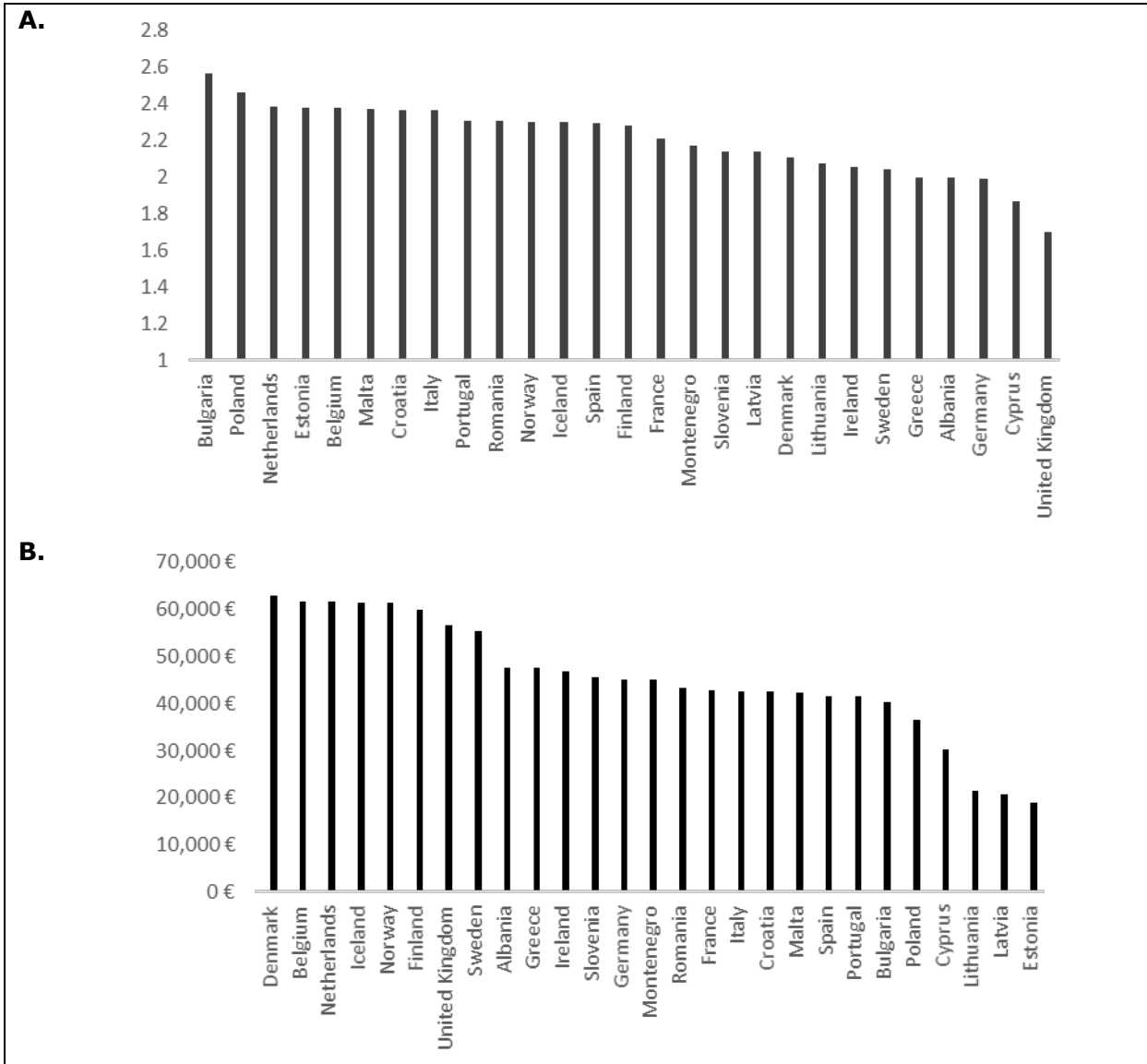
**Figure 5: Distribution of total production by sea area.**



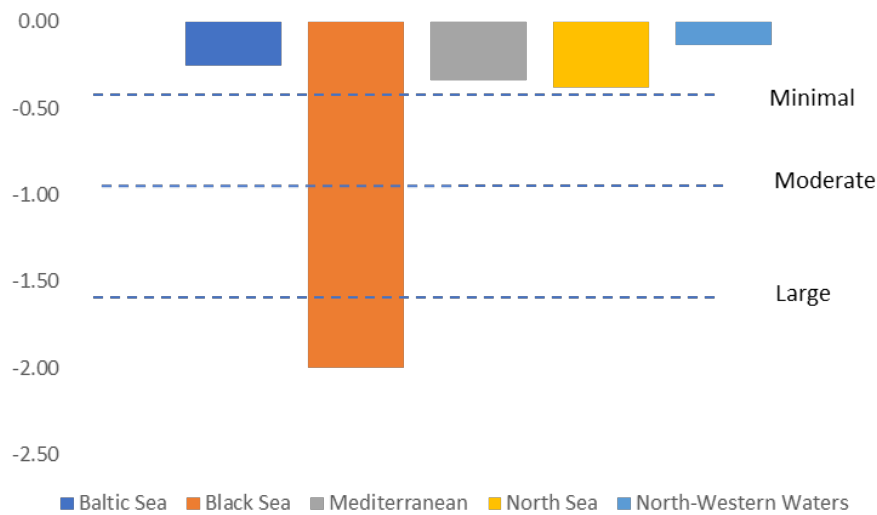
TEI: total economic impact (million euro); FTE: employed Full Time Equivalent.

Source: EURecFish.

**Figure 6: Incremental spending in an economy (multiplicative effect) by country (A) and expenditure required to support one employed Full Time Equivalent (B).**



Source: EUREcFish.

**Figure 7: Assessment of potential bias by sea area.**

Source: EURecFish.

## 4.5 Discussion

The total economic impact of MRF was estimated to be 10.5 billion euro creating almost 100,000 jobs and is significant at a European scale. To put this figure into context, it is equivalent to 7% of the annual EU budget of 145 billion euro or 0.07% of the EU economy. It is roughly equivalent to the cost of the London 2012 Olympic Games (SMSI, 2013) and is similar to the contribution from golf (15 billion euro). The 100,000 FTEs supported is almost four times the staff of the EU Commission. If this was a single company, it would be in the top 10 in Europe, in terms of number of employees, and the top 100 in the world. However, this is not the only benefit that may arise from recreational fisheries, with significant social benefits in terms of human welfare (Arlinghaus and Mehner, 2003; Armstrong *et al.*, 2013, Monkman *et al.*, 2015) and health (Griffiths *et al.*, 2016). It would be useful to have a good understanding of how the expenditure by MRF varied over time. However, very limited time series exist, and those that do either have variable objectives and survey intensities, or include both freshwater and marine fishers. Hence, only a point estimate has been provided and assessing the impact of recreational fishing in response to changing regulations or development is not possible without regular surveys.

The availability of data was the main issue that could affect the robustness of the estimates produced. For example, many countries did not have data leading to extrapolation from donor countries correcting for GDP. This had the underlying assumption that fishers in the recipient country spend the same proportion of their wealth as in the donor country. In some areas (e.g. Black Sea), there was no data to estimate any economic effect, making the assumptions underlying extrapolation less robust. In addition, many different studies have been used with different survey methodologies and of variable quality (e.g. missing sectors, avidity etc.). The impact of these factors is reflected in the robustness of values estimated using the semi-quantitative assessment of bias. For all regions, the estimates were likely to be less than reality mainly due to missing sectors in the surveys. However, North-Western Atlantic waters were generally the most robust, with minimal bias for all regions apart from the Black Sea where the bias was very large, reflecting the lack of data in this region. An additional limitation is our knowledge of the structure of the national economies (the symmetric Input-Output matrixes). This was obtained from EUROSTAT in all cases except the UK. More regional approaches to generate these matrices could improve estimations, but this is case specific and could introduce other sources of variation, making comparisons between countries difficult. Additional variation could be generated by

the difference in years that the Input-Output matrix was generated and the availability of the cost of each FTE in a particular country. Even with these limitations, the estimates of expenditure and FTEs demonstrated the importance of MRF in Europe. The sustainability of MRF is important to society due to the large total economic impact and jobs supported. Hence, there is a need to promote initiatives that take account of the strengths, weakness, opportunities and constraints of MRF for the benefit of future generations.

There are several regional estimates of MRF globally that include Europe. For example, Cisneros-Montemayor and Sumaila (2010) estimated annual expenditure of US\$413 per angler in 2003 for Europe. The European Anglers Alliance has estimated that there are 8-10 million recreational sea anglers spending 8-10 billion euro per year (Pawson *et al.*, 2008). Annual direct expenditure excluding taxes was estimated to be 5 billion euro in this study. There are several potential reasons for this difference. Firstly, the focal system is the coastal European countries and only MRF. Secondly, both the number of fishers and their spend can vary depending on the estimation method, and inter-annual variation can be high. For example, there are around 300,000 habitual recreational fishers in Denmark, but more than 600,000 that fish sporadically (FVM, 2010). In Sweden, the expenditure was estimated to be SEK 5.3 billion in 2013 and SEK 11.3 billion in 2015, suggesting a large inter-annual variation in expenditure.

The magnitude of MRF within the EU is significant enough to consider the development of a common and stable economic data collection program. It should also lead to the implementation of MRF as a discrete sector that is targeted for development alongside commercial fisheries and aquaculture under the CFP. This would help to measure the contribution, but also to design management in the context of the EU blue growth strategy. MRF is often in conflict with commercial fisheries and conservation, for example, spatial interactions in inshore areas or conflicts with conservation policies such as for endangered species. A time series is required to allow analysis of the main drivers of economic impact, but regular data collection is unlikely for the EU member states without inclusion in the DCF. The net change in expenditure by MRF due to changes in management depends on substitution effects that are very difficult to estimate. For example, if MRF was banned in Europe then expenditure may simply be diverted into a different activity (e.g. golf) or people may just spend less. However, it is very difficult to estimate this change or its impact, as the net loss in regional expenditure, income and jobs could be relatively minor if all the expenditure is diverted to a different activity. Obviously, this will result in the loss of income and jobs supported by MRF, but there may be a corresponding gain from increased expenditure in other areas. This may also affect the spatial distribution of income within a country with expenditure diverted away from coastal communities. Assessment of this was not possible using the approach in this study, and would only be possible with additional surveys of willingness to pay using revealed or stated preference methods. This information would be of great help to decision-makers as it would allow assessment of the cost and benefits of MRF in the context of other interests. The nature and magnitude of the impact of changing MRF regulations will depend on the magnitude of the resource, the level of exploitation, and the level and method of allocation (commercial / non-commercial). To fully assess the implications of changing MRF regulations, decision-makers need an understanding of the values of fishing in its different forms. In particular, an understanding of the interactions between regulations, participation, and effort and economic value would be of great use, but this is unlikely to be collected by European MS without inclusion in the DCF.



## 4.6 Recommendations

Based on the estimates of socio-economic impact of MRF, the following recommendations are appropriate:

- The potential economic impact within Europe is significant, so MRF should become a sector that is targeted for development alongside commercial fisheries and aquaculture.
- The main limitation in the accurate estimation of the total economic impact of MRF is the lack of data and the variable quality of it, so a common and stable MRF economic data collection programme should be implemented in Europe.
- A change in policy and management of MRF will alter the economic value of MRF and affect the economic impact. Additional studies on economic value should be funded that assess the economic consequences of changes in the management of MRF.
- Only the economic impact of direct expenditures was included in this study, but additional social and wellbeing benefits are provided by MRF that should be accounted for. Additional studies should be funded to develop these methods.



## 5 IMPACT OF MARINE RECREATIONAL FISHING ON FISH STOCKS

### 5.1 Summary

Recent studies indicate that MRF can contribute substantially to fishing mortality for marine fish stocks. In Europe, MRF assessments have been conducted for a limited number of stocks, which led to the inclusion of MRF data in some stock assessment and implementation of management measures (e.g. sea bass, western Baltic cod). MRF exploits a wide range of fish species, so it is possible that catches could be considerable and exclusion of MRF from stock assessments may impact on the effectiveness of stock management. Here, the impact of MRF was quantified for species listed under the DCF by comparing the percentage contribution to total catch. A literature search revealed a lack of MRF data from many EU member states for several DCF species. MRF catches were reconstructed where estimates were not available using average release proportions, average fish weights, or extrapolating from catch per fisher in similar countries. Furthermore, as the robustness of MRF survey methodologies can be variable, a semi-quantitative estimate of bias was calculated for each stock. Due to the lack of data, it was not possible to provide estimates for all tuna species and several salmon, pollack, and mackerel stocks. Percentage contribution to total catches by MRF ranged between 1.8% for mackerel (ICES divisions 3 and 4) and 72% for the European eel (ICES divisions 3a, 4, and 7). The biomass removed ranged between 88 tonnes (Baltic European eel sub-stock) and 4,089 tonnes (mackerel in ICES divisions 3 and 4). The findings of this study indicate that MRF has the potential to contribute to total catches of some marine fish stocks, so should be routinely included in stock assessments and management. To achieve this, regular surveys of MRF are needed that include all species caught to provide the time series of catch and size distributions for inclusion in stock assessment.

### 5.2 Introduction

A decline in the biomass of several commercially important marine fish stocks has been observed in recent decades that has been particularly apparent in some species. For example, the Atlantic cod (Cook *et al.*, 1997; Myers *et al.*, 1997) has displayed two thirds decline in spawning biomass in North Atlantic stocks (Christensen *et al.*, 2003). Traditionally, the consensus amongst policy makers and the scientific community has been to attribute the observed declines to commercial fishing operations and climate change (Jackson *et al.*, 2001; Hilborn *et al.*, 2003; Pauly *et al.*, 2003; Watson *et al.*, 2003). However, in many parts of the world the importance of MRF in terms of exploiting marine fish stocks has been recognised leading to recreational catches being included in stock assessments and separate quota allocations are made for commercial and recreational fisheries for some stocks (Ryan *et al.*, 2016). In Europe, a lack of reliable estimates of recreational catches has resulted in recreational sea fishing being excluded from stock assessments and allocations for many years (Pawson *et al.*, 2007). This undermines our ability to manage fish stocks to Maximum Sustainable Yield (Hyder *et al.*, 2014). Recent studies have demonstrated the impact of MRF in Europe (e.g. Morales-Nin *et al.*, 2005; Veiga *et al.*, 2010; Vølstad *et al.*, 2011; Sparrevohn and Storr-Paulsen 2012; Strehlow *et al.*, 2012; Armstrong *et al.*, 2013; Ferter *et al.*, 2013a; Herfaut *et al.*, 2013; van den Hammen *et al.*, 2016), yet only commercial catches are included in most stock assessments.

Under the DCF (EU, 2001) member states must collect data on recreational landings and releases of Atlantic cod, Atlantic mackerel, Atlantic pollack, Atlantic salmon, European sea bass, European eel, sea trout, elasmobranchs, and tuna, with variation in the requirements

in different management regions. Despite reporting of recreational catches being a requirement since 2002, limited data exist, with few countries carrying out regular surveys until recently. There are a number of potential reasons for this, including: cost, challenges associated with survey design and implementation, and the belief that MRF has limited impact on fish stocks. However, this will change, as pilot studies are required within two years, under the latest implementation of the DCF, before any derogation will be granted (EU, 2016a).

Hyder *et al.* (in press) found that the participation rate and effort in recreational fisheries can be substantial in both the Atlantic Ocean and Mediterranean Sea (Table 4). However, until recently MRF was excluded from stock assessments in Europe, possibly due to lack of recreational catch data. Recent publications relating to the MRF harvest of sea bass, cod, sea trout, and salmon stocks have, however, led to MRF being incorporated in stock assessments in Europe. The stock assessments that currently include MRF are: sea bass (ICES divisions 4b&c, 7a,d-h - bss-47 - ICES, 2016f; 2017c), Atlantic salmon in the Baltic sea (sal-bal - ICES, 2017d), sea trout in the Baltic Sea (trt-bal - ICES, 2017d) and western Baltic cod (cod-2224 - ICES, 2016d; 2017e).

Typically, the literature concerning harvest of marine organisms by MRF focuses on single countries catches of one or more species of fish (e.g. Strehlow *et al.*, 2012; Herfaut *et al.*, 2013; van der Hammen and de Graaf, 2015; van der Hammen *et al.*, 2016). Whilst these data are useful for determining a single nations impact on a stock by MRF, the total impact of MRF by all countries on commercially important fish stocks remains unknown. Due to the high MRF effort and catches, the potential impact of MRF may be high for some stocks and could introduce additional uncertainty not currently captured in the stock assessments. In this chapter, the impact of MRF by all countries exploiting marine fish stocks in the Northeast Atlantic, Mediterranean Sea and Black Sea, collected under DCF and other surveys, was estimated. Where data were not available, extrapolation of catch per unit effort (CPUE) from a similar country was used. The quality of estimates was assessed using the proportion reconstructed and bias investigated using a simple semi-quantitative estimate. The results are discussed in the context of fisheries assessment and future monitoring needs.

## 5.3 Methods

### 5.3.1 Data collection

A review of the literature was conducted to obtain MRF data from existing studies for the Baltic Sea, North Sea, North-Western Atlantic Waters and South-Western Atlantic Waters (Table 11). As a considerable proportion of fish caught by MRF can be released (Policansky, 2002; Bartholomew and Bohnsack, 2005; Ferter *et al.*, 2013a), post-release mortalities were collected to determine the total MRF catch (kept and post-release mortality). Where suitable post-release mortalities could not be identified, a precautionary value of 100% was used. Data were compiled by stock for the key recreational species in the DCF for each of the six marine regions. This involved building on the approach developed by Hyder *et al.* (in press) and use of the landings and releases by MRF in each country collated annually by the WGRFS (e.g. ICES, 2017b).

Assessing the impact of MRF on fish stocks proved challenging as not all countries conduct surveys and, where surveys were done, the focus was often on one or two key species. Thus, extrapolations were conducted to determine the impact of MRF in countries where data were not available (Section 5.3.2). In the interests of reporting MRF impact at a regional scale, stocks with large boundaries (such as mackerel in the Northeast Atlantic) were split into smaller regions that are already defined in the ICES stock assessments.

Commercial landings data for each country were taken from ICES stock assessments for comparisons of commercial and recreational catches. This was not possible for some countries as catches were grouped (e.g. 'other' where minimal, or English, Welsh and Northern Irish grouped under the 'UK'). Where countries catches were not reported in the stock assessment, the ICES catch statistics database (ICES, 2017a) or Marine Management Organisation (MMO) landings database (MMO, 2012) were used to determine commercial landings. The use of the ICES and MMO databases was a potential source of error as the landings in these databases often differ from those reported in ICES stock assessments, so the source of commercial catch was recorded. When comparing recreational and commercial catch at stock level, commercial landings and discards data were taken from ICES stock assessments, therefore, the potential error from using the ICES and MMO databases was mitigated. As most MRF data were from 2012, this was used as a reference year and commercial data also selected from 2012. The latest recreational and commercial figures were used where the ICES stock assessment included estimates for MRF catches.

In the Mediterranean and Black Seas, MRF data were limited and a wide range of species targeted, so analyses of catches were conducted separately. A literature review of the MRF in the Mediterranean and Black Seas identified several studies that had already attempted catch reconstruction, but no additional data were found. Therefore, the results of pre-existing reconstructions of species landed by MRF in the Mediterranean and Black Seas were presented. Furthermore, commercial landings in the Mediterranean and Black Seas were also limited, so no comparisons between recreational and commercial catches were made.

### 5.3.2 Reconstructions of marine recreational catches

MRF catches were not available for all countries that exploited each stock, so it was necessary to reconstruct catches in order to assess the total biomass removed and make a comparison with commercial fishing. As each country had different data available, a number of different approaches were needed that are described in detail below.

To calculate the total MRF catch for each stock ( $C_R$ ) it was necessary to add the tonnages kept ( $K_i$ ) and released tonnage that died ( $\partial R_i$ ) for each country ( $i$ ), where  $m$  countries exploit the stock. This was done using the following equation:

$$C_R = \sum_{i=1}^m (K_i + \partial R_i) \quad (1)$$

where  $\partial$  is the post-release mortality rate and  $R_i$  is the weight of released fish in tonnes. Post-release mortality rates would usually be applied to the numbers of fish released, but this made no difference as the average weight was used to derive numbers, so was constant across all countries within a stock.

There were three ways of obtaining kept weight for each country depending on the data available and were applied in order from the top of Equation 2 down, as this represented less robust solutions and increased uncertainty. Where kept weight ( $K_i$ ) was available it was used directly, but in many cases only numbers kept were available and kept weight was derived from numbers kept ( $k_i$ ) multiplied by the average weight of a kept fish across the whole stock ( $\bar{w}_k$ ). Where no data were available, it was necessary to extrapolate using the number of fish kept per fisher from a donor country ( $\theta_j$ ) along with the number of fishers from the recipient country ( $n_i$ ). This calculation was done using the following equation:

$$K_i = \begin{cases} K_i & (\text{weight kept}) \\ k_i \bar{w}_k & (\text{number kept}) \\ \theta_j n_i \bar{w}_k & (\text{no data}) \end{cases} \quad (2)$$

The calculation of released weight was similar to kept weight, but four calculations were required. Again, these were applied in order from the top of Equation 3 down as this represented a less robust solution and increased uncertainty. Where the weight of released fish ( $R_i$ ) was available it was used in the calculation. If the number released ( $r_i$ ) was available for the country ( $i$ ), then this was multiplied by the average weight of a released fish for the whole stock ( $\bar{w}_r$ ). If only kept weight existed, then released weight was derived from the average proportion released across the whole stock ( $\rho$ ), the numbers kept ( $K_i$ ), and the average weight ( $\bar{w}_r$ ). Finally, where there were no data, it was necessary to extrapolate using the number of fish released per fisher from a donor country ( $\Psi_j$ ) along with the number of fishers from the recipient country ( $n_i$ ) and the average weight ( $\bar{w}_r$ ). This calculation was conducted using the following equation:

$$R_i = \begin{cases} R_i & \text{(weight released)} \\ r_i \bar{w}_r & \text{(number released)} \\ K_i \rho \bar{w}_r & \text{(number kept)} \\ \Psi_j n_i \bar{w}_r & \text{(no data)} \end{cases} \quad (3)$$

To estimate the tonnage of fish released that die it was necessary to assume that recreational post-release proportion and mortality, and commercial discard proportion and mortality were the same across countries and stocks. This is unlikely to be the case as fishing gears and practices vary between countries, but limited experimental data existed, so this was the most reasonable approach. As different recreational gears are often employed by fishers, each of which will have a different post-release mortality, but the catch by individual gear type were not available, the post-release mortality used was that associated with the highest yielding gear type, which was rod and line in every circumstance. Where extrapolations of catch per fisher were made between countries, the implicit assumptions were that fishers in the recipient country fish in the same way and catch the same amount as in the donor country. In addition, average weights of individual fish were assumed to be constant within each stock. For countries that exploited multiple stocks of the same species (e.g. French exploitation of sea bass), it was necessary to assume that CPUE was uniform across the country as no other information existed.

As reconstructions introduce additional uncertainty into the catch estimates, a threshold level was set above which reconstructions were not valid and were not used. This threshold was set at 50% for both total catch and landed weights. Thus, if the percent reconstructed of both total catch and landings were below 50% then total catches were valid, but where the total catch reconstructed was over 50% only landings were used. If the total and retained weights reconstructed were both over 50%, the data were considered too limited to be reliable and were excluded from further analysis.

### 5.3.3 Bias estimation

The scope and methods used by recreational fisheries surveys can differ (e.g. some studies do not sample all platforms), so an assessment of the potential bias in the estimates was conducted. Expert judgement was used to develop a semi-quantitative measure of bias following a simple approach similar to that used in other fields (EFSA Scientific Committee, 2015). The method is briefly described here, but see Hyder *et al.* (in press) for details. A seven-point scale (ranging from -3, denoting highly underestimated, to +3, denoting highly overestimated) was used to determine the magnitude and direction of bias in each study ( $b_i$ ). All potential sources of bias were considered when assessing the bias in the projected landings and discards including recall, avidity biases, coverage, and non-response (see Pollock *et al.*, 1994 and ICES, 2010 for reviews). It was necessary to weight the contribution of the bias from each country ( $w_i$ ), so that, for example, a large error in a small estimate does not have as much influence on the overall bias as a small bias in a

large value for a country. Hence, to calculate the relative bias in each stock  $s$  ( $B_s$ ) the following equation was used:

$$B_s = \frac{\sum_{i=1}^m b_i w_i}{\sum_{i=1}^m w_i} \quad (4)$$

where  $w_i$  was the individual country weight and  $b_i$  was assumed to be the same for the donor and recipient countries. The relative biases in the overall estimates were ratios, so were categorised by sign to indicate direction of bias (positive – overestimates, negative – underestimate) and on a categorical logarithmic scale ( $0.2 \leq \text{minimal} < 0.4$ ;  $0.8 \leq \text{moderate} < 1.6$ ; and  $1.6 \leq \text{large}$ ).

### 5.3.4 Comparison of recreational and commercial catches

Usually an assessment of impact would be done through a full analytical stock assessment that included both recreational and commercial catches, but that was not possible with the data or number of species, and was not within the scope of the study. Instead, a simple approach was used to quantify the potential impact of MRF that compared the relative contribution of commercial and recreational exploitation of the stock. Comparisons were generally made in terms of biomass removed in terms of total catch of commercial (landings plus dead discards) and recreational (retained fish plus post-release mortality). However, where more than 50% of the recreational catch was reconstructed only landings were compared. The comparison of salmon and sea trout were based on number of fish caught and released.

## 5.4 Results

### 5.4.1 Data compilation and selection

#### 5.4.1.1 Atlantic and Baltic seas

Data were compiled for cod, eel, mackerel, pollack, salmon, sea trout and tuna stocks. Out of the 22 stocks assessed, sufficient data for full reliable estimates of MRF impact were only available for 10 stocks and partial comparisons were available for 2 stocks (Table 14). Although the percentage of total weight reconstructed was above 50% in sea bass in ICES divisions 8c & 9a (bss-8c9a) (Table 14), the data were raised from small spatial scale national surveys rather than from another nation's catch per fisher; therefore, it was decided that a full comparison between MRF and commercial harvest for bss-8c9a could be made as the data were likely to be representative of the entire population.

The average proportions were used to calculate the number of fish released by MRF (Table 15), where estimates of the number and weight of fish released by MRF were not provided by national sampling. Recreational release proportions, ranging between 0.09 and 0.67, were greater than commercial discards, ranging between negligible and 0.23 (Table 15). However, recreational fishing exhibited a much lower post-release mortality (Table 16) than commercial discard mortality, although, this may have been exacerbated by the large number of precautionary 100% commercial post-release mortality values used. Experiment 2007B from Huse and Vold (2010) was chosen for commercially discarded mackerel mortality as the conditions were the most comparable to those of commercial fishing.

**Table 14: Comparisons of recreational and commercial landings conducted in addition to the percentage of the total weights reconstructed for each stock.**

Species	Stock	Comparison	Reconstruction %		Total
			Retained	Released	
Cod	cod-347d	Full	16.6	39.8	17.7
	cod-7e-k	Full	5.4	69.6	7.9
	cod-2224	Full	0	23.2	2.3
	<b>cod-2532</b>	<b>None</b>	<b>53.0</b>	<b>NA</b>	<b>NA</b>
Eel*	ele-balti	Retained	0.8	99.9	<b>50.6</b>
	ele-3a,4,7	Retained	6.4	99.7	<b>65.4</b>
Mackerel*	<b>mac-1,2,5,14</b>	<b>None</b>	<b>NA</b>	<b>NA</b>	<b>NA</b>
	mac-34	Full	32.1	36.1	32.4
	<b>mac-6</b>	<b>None</b>	<b>NA</b>	<b>NA</b>	<b>NA</b>
	mac-7,8abde	Full	0.7	93.7	9.4
	<b>mac-8c9a</b>	<b>None</b>	<b>65.6</b>	<b>NA</b>	<b>NA</b>
Pollack	pol.27.67	Full	2.6	81.1	18.4
	<b>pol-89a</b>	<b>None</b>	<b>77.1</b>	<b>NA</b>	<b>NA</b>
	pol-nsea	None	NA	NA	NA
Salmon	sal-bal	Full	5.9	80.5	43.3
	<b>sal-7</b>	<b>None</b>	<b>NA</b>	<b>NA</b>	<b>NA</b>
	<b>sal-4</b>	<b>None</b>	<b>NA</b>	<b>NA</b>	<b>NA</b>
Sea bass	bss-47	Full	2.0	26.4	7.9
	bss-8c9a**	Full	63.2	67.3	64.8
	bss-8ab	Full	0	0	0
Sea trout	<b>trt-bal</b>	<b>None</b>	<b>96.1</b>	<b>NA</b>	<b>NA</b>
Tuna	<b>Northeast Atlantic</b>	<b>None</b>	<b>NA</b>	<b>NA</b>	<b>NA</b>

A full comparison refers to both recreational and commercial landings and discards being compared whereas a retained comparison refers to where only the retained proportion of the catch was compared. NA values indicate there are no data to reconstruct landings. Bold values are above the 50% reconstructed weight cut off.

\* = ICES stock assessment based on entire Northeast Atlantic stock and so have been split into sub stocks for this report.

\*\* = Despite the reconstruction percentage being over 50%, data were considered robust enough estimates by expert judgement to be included as the data were raised from small scale national sampling.

**Source:** EURecFish.



**Table 15: The recreational release and commercial discard proportions for each stock analysed.**

Species	Area	Recreational releases	Commercial discards
Sea bass	Celtic Seas and English Channel	0.27	0.05
	Bay of Biscay	0.35	0
	South Bay of Biscay & Atlantic Iberian Waters	0.57	0
Cod	North Sea, Eastern English Channel, Skagerrak	0.34	0.23
	Western English Channel and Southern Celtic Seas	0.11	0.12
	Eastern Baltic sea	NA	0.11
	Western Baltic sea	0.47	0.04
Eel	North Sea, English Channel, Skagerrak	0.67	NA
	Baltic Sea	0.39	NA
Mackerel	Eastern Arctic,	NA	Negligible
	North Sea, Skagerrak	0.09	0.01
	West of Scotland	NA	0.01
	Celtic sea, Bay of Biscay	0.18	0.15
	South Bay of Biscay & Atlantic Iberian Waters	NA	0.14
Pollack	Celtic Seas and the English Channel	0.65	Negligible
	Bay of Biscay & Atlantic Iberian Waters	NA	Negligible
	North Sea	NA	Negligible
Salmon	North Sea	NA	NA
	Celtic Sea	NA	NA
	<b>Baltic Sea</b>	<b>0.43</b>	<b>0.13</b>
Seatrout	<b>Baltic Sea</b>	<b>NA</b>	<b>&lt;0.01</b>
Tuna	Northeast Atlantic	NA	NA

Proportions were calculated by dividing the discards by the landings. Commercial data were sourced from ICES stock assessments; recreational proportions were an average of all studies. Bolded rows indicate 2015 rather than 2012 values.

Source: EURecFish.

**Table 16: The recreational post-release mortality and commercial discard mortality percentages used to estimate the quantity of dead discards.**

Species	Areas	Recreational		Commercial	
		%	Source	%	Source
Sea bass	All	5.0	Lewin <i>et al.</i> (submitted)	100	Precautionary value
Cod	Atlantic	16.5	Capizzano <i>et al.</i> (2016)	68.0	Depestele <i>et al.</i> (2014)
Cod	Baltic	11.2	Weltersbach and Strehlow (2013)	100	Precautionary value
Eel	All	24.7	Weltersbach <i>et al.</i> (in prep.)	100	Precautionary value
Mackerel	All	100	Precautionary value	83.5	Huse and Vold (2010) - Experiment 2007B
Pollack	All	100	Precautionary value	100	Precautionary value
Salmon	All	25.0	ICES (2017c)	100	Precautionary value
Seatrout	All	100	Precautionary value	100	Precautionary value
Tuna	All	5.6	Stokesbury <i>et al.</i> (2011)	100	Precautionary value

Source: EURecFish.

Table 17 presents the MRF retained and released weights and numbers for stocks in which the reconstruction percentage was considered appropriate. The data presented in Table 17 were identified or calculated during the data collection and reconstruction phases of this study. In total, 51% (38) of retained weights and 68% (52) of retained numbers were reconstructed whilst 80% (40) of released weights and 69% (40) of released numbers were reconstructed in all countries and stocks (Table 17).

**Table 17: The weight (tonnes) and number of thousand fish retained and released through MRF.**

Stock	Country	Retained		Released		Source/notes
		Weight (t)	Number	Weight (t)	Number	
bss-47	Belgium	60	75*	8.4*	20▲	ICES (2016f)
	France	1,699	2,125	332	567*	Rocklin et al. (2014)
	Netherlands	168	248	135*	321	van der Hammen and de Graaf (2015); van der Hammen et al. (2016)
	England	285	243	197	467	Armstrong et al. (2013)
	Wales	17*	21▼	17*	40▼	NA
	Channel Islands	0.9*	1.1▼	0.9*	2.2▼	NA
	Isle of Man	0.3*	0.4▼	0.4*	0.83▼	NA
	Scotland	28*	34▼	28*	66▼	NA
bss-8ab	France	1,405	1,168	496	1,190	Rocklin et al. (2014)
bss-8c9a	Spain	411‡	491‡	249‡	655‡	DCF Sampling (2015; 2016 unpublished data)
	Portugal	21‡	61‡	13‡	34‡	Veiga et al (2010)
cod-2224	Denmark	1,272	1,019*	111*	657	Sparrevojn and Storr-Paulsen (2012)
	Germany	3,032	2,430	410	1,139	Strehlow et al. (2012)
	Sweden	215	458*	13*	76▲	Karlsson et al. (2016)
cod-347d	Belgium	265	149*	8*	51▲	Persoon (2015)
	Denmark	523	349	18 *	118▲	Sparrevojn and Storr-Paulsen (2012)
	England	280	227	43	85	Armstrong et al. (2013)
	France	190	107*	5.6*	36▲	Herfaut et al. (2013)
	Germany	30	17*	0.9*	5.7▲	Strehlow et al. (2012)
	Netherlands	1,145	527*	73*	170▲	van der Hammen and de Graaf (2015)
	Norway	585*	330▼	19*	124▲	NA
	Scotland	57*	32▼	1.9*	12▼	NA
Sweden	795	449*	23*	152▲	Karlsson et al. (2016)	
cod-7e-k	Channel Islands	0.4*	0.4▼	0.02*	0.04▼	NA
	England	90	86	3.6	9.1	Armstrong et al. (2013)
	France	190	182*	7.7*	19▲	Herfaut et al. (2013)
	Ireland	7.8*	7.5▼	0.3*	0.8▼	NA
	Wales	7.7*	7.4▼	0.3*	0.8▼	NA

Stock	Country	Retained		Released		Source/notes
		Weight (t)	Number	Weight (t)	Number	
ele-3a,4,7	Belgium	0.9*	4.4♦	Data limited		NA
	Denmark	43	210*			Sparrevohn and Storr-Paulsen (2012)
	England	1.1*	5.3			Armstrong et al. (2013)
	France	1.8*	8.8▼			Illegal to fish
	Germany	4.0	17			ICES (2017b)
	Ireland	0.2*	0.9▼			NA
	Netherlands	18	91			van der Hammen and de Graaf (2015); van der Hammen et al. (2016)
	Norway	0	0			Illegal to fish
	Scotland	0.3*	1.4▼			NA
	Wales	0.2*	0.8▼			NA
ele-balti	Denmark	73	196*	Data limited		Sparrevohn and Storr-Paulsen (2012)
	Estonia	0.01	27*			ICES (2017b)
	Finland	8	22*			ICES (2017b)
	Germany	1.5	4.0			ICES (2017b)
	Latvia	0.1	269*			ICES (2017b)
	Lithuania	4.9	13*			ICES (2017b)
	Poland	0.7*	1.9†			NA
	Sweden	0	0			Illegal to fish
mac-3,4	Belgium	68*	185♦	4.01*	20♦	NA
	Denmark	134■	366*	7.1*	355	NA
	England	463	1,726	50	518	Armstrong et al. (2013)
	Germany	481*	1,317♦	29*	141♦	NA
	Netherlands	1,564	3,815	141	408	van der Hammen and de Graaf, (2013)
	Norway	445*	1,219■	24*	115▲	NA
	Scotland	89*	244▼	15*	73▼	NA
	Sweden	546	1,493*	29*	141▲	Karlsson et al. (2016)

Stock	Country	Retained		Released		Source/notes
		Weight (t)	Number	Weight (t)	Number	
mac-7,8abde	Channel Islands	0.4*	1.3▼	0.1*	0.4▼	NA
	England	84	280	15	90	Armstrong et al. (2013)
	France	2,205	7,336*	220*	1,319▲	Herfaut et al. (2013)
	Ireland	7.3*	24▼	1.3*	7.8▼	NA
	Isle of Man	0.2*	0.5▼	0.03*	160▼	NA
	Wales	7.2*	24▼	1.3*	7.8▼	NA
pol.27.67	Channel Islands	0.9*	0.6▼	0.6*	1.1▼	NA
	England	195	123	126	250	Armstrong et al. (2013)
	France	2,398	1,511*	497*	981▲	Herfaut et al. (2013)
	Ireland	17*	11▼	11*	21▼	NA
	Isle of Man	0.3*	0.2▼	0.2*	0.4▼	NA
	Northern Ireland	7.1*	4.5▼	4.6*	9.2▼	NA
	Scotland	28*	17▼	18*	35▼	NA
	Wales	17*	11▼	11*	21▼	NA
sal-bal**	Denmark	Weights not provided	11	Weights not provided	6	ICES (2017b)
	Estonia		1		0.45▲	ICES (2017d)
	Finland		11		4.9▲	ICES (2017d)
	Germany		4		0.32	ICES (2017d)
	Latvia		0.98		0.42▲	ICES (2017d)
	Lithuania		0.62		0.27▲	ICES (2017d)
	Poland		1.9†		0.81▲	NA
	Sweden		1.6		0.69▲	ICES (2017d)

Symbols indicate the reconstruction source of the data. NA sources indicate where no MRF surveys were conducted.

Key: \* = Average weight, ▲ = Average released proportion, ▼ = English CPUE, † = Danish CPUE, ◆ = Dutch CPUE, ■ = Swedish CPUE, ‡ = Raised from study on small area of country, # = German CPUE.

**Source:** EURRecFish.

### 5.4.1.2 Mediterranean Sea

The characterisation of MRF in individual countries around the Mediterranean and Black Seas is provided in Section 2.4 and includes the main species targeted, so this section focuses on compilation of catch estimates. Comparisons between recreational and commercial fishing in the Mediterranean and Black Seas could not be made as there are very little data on either MRF or commercial fisheries. However, several studies (e.g. Keskin *et al.*, 2014; Matić-Skoko *et al.*, 2014; Ulman *et al.*, 2015; Khalfallah *et al.*, 2017) have attempted to quantify recreational catch by Mediterranean and Black Sea countries. This was often done as part of the 'SeasAroundUs' project (<http://www.seaaroundus.org/>), where limited data were used to reconstruct likely catches over time (Pauly and Zeller, 2016). A summary of the outcomes is presented below.

No studies could be identified of the catches by MRF in Albania, Italy, and Romania. There were no DCF species caught in Bulgaria (Cervera *et al.*, 2009), but catch rates were available for the Turkish Black Sea (Ulman *et al.*, 2013). These were used to reconstruct catches of the most important MRF species in Bulgaria that were less than 30 tonnes per species (Table 18) (Keskin *et al.*, 2015). Reconstructions of MRF catches in Croatia have been conducted that highlighted some catches of small pelagics, but catches of all other individual species were less than 30 tonnes (Table 19) (Matić-Skoko *et al.*, 2014). Ulman *et al.* (2015) estimated the catches by recreational vessels, spearfishers and shore-based anglers in Northern Cyprus (Table 20) and South Cyprus (Table 21). In Northern Cyprus, at least 2,000 people were engaged in shore-based angling catching 120 tonnes and a total MRF catch of 614 tonnes, with a maximum of 55 tonnes for a single species (Table 20). A reconstruction of catches in South Cyprus showed that all MRF catches were less than 30 tonnes for all individual species (Table 21).

**Table 18: The species most commonly caught by MRF in Bulgaria during 2014.**

Common name	Scientific name	Landings (t)
Mediterranean horse mackerel	<i>Trachurus mediterraneus</i>	29
Mulletts, grey mullets	Mugilidae	24
Gobies	Gobiidae	24
Bluefish	<i>Pomatomus saltatrix</i>	2
Atlantic bonito	<i>Sarda sarda</i>	2
Turbot	<i>Scophthalmus maximus</i>	1

Source: Keskin *et al.* (2015).

**Table 19: The species most commonly caught by MRF in Croatia.**

Common name	Scientific name	Landings (t)
Herrings, sardines, menhadens	Clupeidae	1,059
Anchovies, round herrings	Engraulidae	177
Marine fishes not identified elsewhere	Marine fishes not identified elsewhere	142
Squids, cuttlefishes, octopuses	Cephalopoda	27
Cods	Gadiformes	23
Goatfishes	Mullidae	22
Mackerels, tunas, bonitos	Scombridae	22
Jacks, pompanos	Carangidae	9
Clawed lobsters	Nephropidae	6
Soles	Soleidae	5
Porgies, seabreams	Sparidae	5
Mulletts, grey mullets	Mugilidae	2
Scorpionfishes, rockfishes	Scorpaenidae	1

Source: Matić-Skoko *et al.* (2014).

**Table 20: The species most commonly caught by MRF in Northern Cyprus.**

Common name	Scientific name	Landings (t)
Marine fishes not identified elsewhere	Marine fishes not identified elsewhere	114
Old parrotfish	<i>Sparisoma cretense</i>	55
Porgies, seabreams	Sparidae	52
Dusky grouper	<i>Epinephelus marginatus</i>	44
Mottled grouper	<i>Mycteroperca rubra</i>	37
Atlantic bonito	<i>Sarda sarda</i>	30
Greater amberjack	<i>Seriola dumerili</i>	22
Leerfish	<i>Lichia amia</i>	15

Source: Ulman *et al.* (2015).**Table 21: The species most commonly caught by MRF in Southern Cyprus between 1950 to 2014.**

Common name	Scientific name	Landings (t)
Marine fishes not identified elsewhere	Marine fishes not identified elsewhere	35
Old parrotfish	<i>Sparisoma cretense</i>	17
Porgies, seabreams	Sparidae	16
Dusky grouper	<i>Epinephelus marginatus</i>	14
Mottled grouper	<i>Mycteroperca rubra</i>	11
Atlantic bonito	<i>Sarda sarda</i>	9
Greater amberjack	<i>Seriola dumerili</i>	7
Leerfish	<i>Lichia amia</i>	5

Source: Ulman *et al.* (2015).

A reconstruction of Maltese MRF landings has been done and showed that octopus and frigate tuna landings were around 160 tonnes with most species catches of less than 50 tonnes (Table 22) (Khalfallah *et al.*, 2017). Landings were generally low in Montenegro with reconstructions indicating landings of less than 50 tonnes per species (Table 23) (Keskin *et al.*, 2014). Several studies have been performed to characterise MRF in different parts of Turkey (Ünal *et al.*, 2010; Aydin *et al.*, 2013; Tunca *et al.*, 2012; Tunca, 2015; Tunca *et al.*, 2016). In the Mediterranean, the main target species are from the families Sparidae, Mullidae, Caragnidae and Scombridae (Aydin *et al.*, 2013, Tunca *et al.*, 2016). In the Black Sea, six species dominated the catch composition in all provinces: *T. trachurus*, *M. cephalus*, *S. sarda*, *P. saltatrix*, *S. sarda*, *B. belone*. The low number of species in the catch is due to the Black Sea having a low number of species compared to the Mediterranean Sea (Tunca *et al.*, 2017). Aydin *et al.* (2013) identified the most commonly caught species by Turkish MRF, however, the catch calculated in this study was not raised to the entire Turkish fishing population. Nevertheless, Ünal *et al.* (2010) reconstructed Turkish MRF data, a summary of which is provided in Table 24Table 24.

**Table 22: The species most commonly caught by MRF in Malta between 1950 to 2014.**

Common name	Scientific name	Landings (t)
Frigate tuna	<i>Auxis thazard</i>	166
Octopuses, argonauts	Octopoda	161
Marine fishes not identified elsewhere	Marine fishes not identified elsewhere	61
Common dolphinfish	<i>Coryphaena hippurus</i>	58
Atlantic bonito	<i>Sarda sarda</i>	45
Greater amberjack	<i>Seriola dumerili</i>	23
Swordfish	<i>Xiphias gladius</i>	22
Comber	<i>Serranus cabrilla</i>	19
Little tunny	<i>Euthynnus alletteratus</i>	18
Black scorpionfish	<i>Scorpaena porcus</i>	12
Smooth-hound	<i>Mustelus mustelus</i>	12
Basses, groupers, hinds	Serranidae	12
Common cuttlefish	<i>Sepia officinalis</i>	12
White skate	<i>Rostroraja alba</i>	12
Other species	Other species	86

Source: Khalfallah *et al.* (2017).

**Table 23: The species most commonly caught by MRF in Montenegro.**

Common name	Scientific name	Landings (t)
Herrings, sardines, menhadens	Clupeidae	46
Anchovies, round herrings	Engraulidae	23
Marine fishes not identified elsewhere	Marine fishes not identified elsewhere	15
Mackerels, tunas, bonitos	Scombridae	2
Squids, cuttlefishes, octopuses	Cephalopoda	2
Goatfishes	Mullidae	1
Jacks, pompanos	Carangidae	1
Clawed lobsters	Nephropidae	1

Source: Keskin *et al.* (2014), which was based on Croatian data from Matic-Skoko *et al.* (2014).

**Table 24: Catch of some of the species most commonly caught in Turkey.**

Common name	Scientific name	Catch (t)
Atlantic horse mackerel	<i>Trachurus trachurus</i>	256
Gilt-head bream	<i>Sparus aurata</i>	186
<i>Diplodus vulgaris</i>	<i>Diplodus vulgaris</i>	158
Atlantic bonito	<i>Sarda sarda</i>	86
European sea bass	<i>Dicentrarchus labrax</i>	57
Turbot	<i>Scophthalmus maximus</i>	4

Source: Aydin *et al.* (2013), reconstruction data provided by Ünal *et al.* (2010).

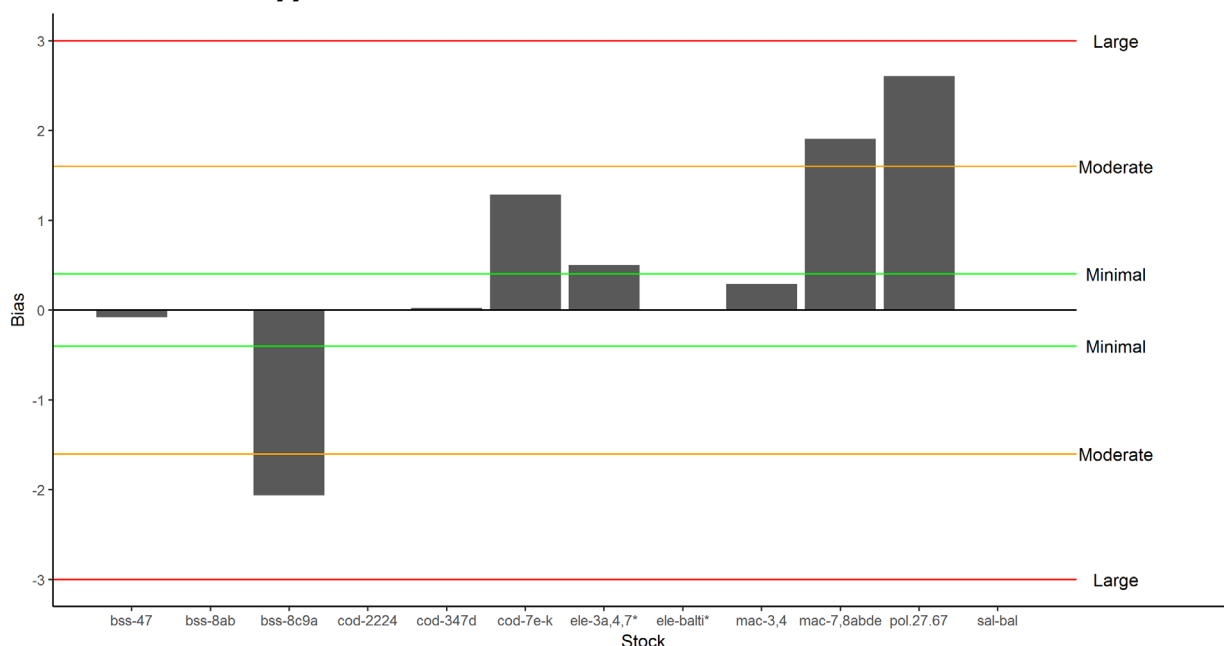
### 5.4.2 Bias estimation

As the strata sampled by MRF sampling schemes can vary, a weighted semi-quantitative bias of MRF catches was calculated. Overall, the level of bias in all the studies used was low (Figure 8); though, it is worth noting that 43% (35) of the weighted study biases were extrapolated from other studies. MRF catches in six of the 12 stocks assessed were considered overestimated, whilst two were underestimated and four did not have any discernible bias (Figure 8). The level of bias was generally very low, with seven stocks having minimal to no bias in the catch estimates (Figure 8). Sea bass in the Atlantic Iberian waters (bss-8c9a) and the central and southern North Sea, Irish Sea, English Channel, Bristol Channel, and Celtic Sea (bss-47) were the only stocks considered to be underestimated (Figure 8). For bss-8c9a this was due to both the total Portuguese and Spanish catch estimates being raised from small scale national studies (Veiga *et al.*, 2010; DCF sampling, 2015; 2016, unpublished data). Whilst no bias in MRF catch estimates was detected on a stock level for cod in the western Baltic (cod-2224), eel in the Baltic (ele-balti) and salmon in the Baltic (sal-bal) (Figure 8), bias within individual country’s MRF catch estimates as well as that introduced by reconstructing data almost certainly occurred.

### 5.4.3 Comparison of recreational and commercial catches

The contribution to total catch of a stock by all countries MRF catch was higher than the commercial catch in 25 instances (38% of total - Figure 9; Table 25). However, in terms of total impact on the stock, the impact by MRF was only greater than the commercial impact in the ele-3a,4,7 and almost equal in pol.27.67 (Figure 9; Table 25). The total landings by MRF were considerable in the pol.27.67 stock, cod and both sea bass stocks (Figure 9; Table 25). Despite having the largest biomass removed by MRF, MRF catches of mackerel in all stocks were generally small in comparison to commercial landings (Figure 9; Table 25).

**Figure 8: The estimated bias in the total MRF catches (landings + post-release mortality) for each stock.**

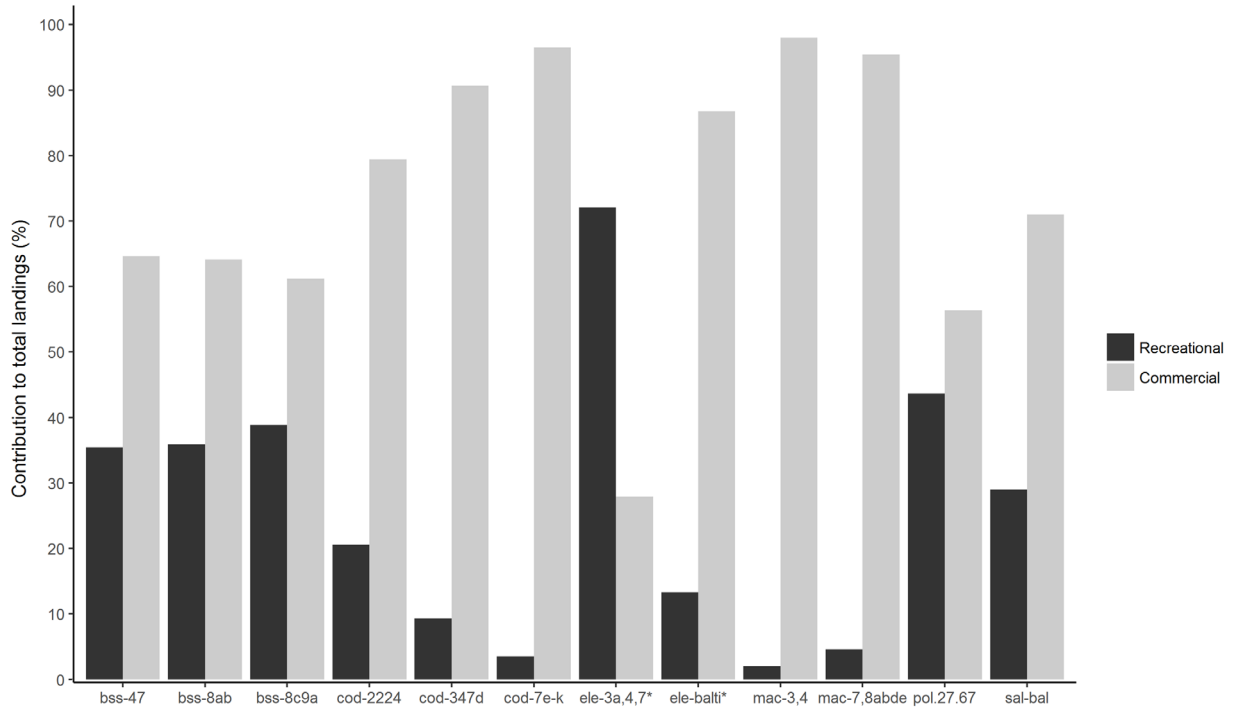


Stocks with a \* did not include post-release mortalities as the percent of the released weight reconstructed was too high.

Source: EURRecFish.



**Figure 9: The percentage contribution to total catch by recreational and commercial fishing for each stock assessed in this report.**



\* = only comparisons between the retained portion of the catch (i.e. no discards) were made.

**Source:** EURecFish.

**Table 25: The total recreational and commercial catches in addition to the percentage contribution to the total catches by both recreational and commercial landings by each country and the total for each stock.**

Stock	Country	Catches (t)		Percentage of total landings		Commercial landings source
		Recreational	Commercial	Recreational	Commercial	
bss-47	Belgium	60	162	27	73	ICES (2016f)
	Channel Islands	0.95	58	2	98	MMO, 2012
	England	295	818	27	73	MMO, 2012
	France	1716	2519	41	59	ICES (2016f)
	Isle of Man	0.36	0	100	0	MMO, 2012
	Netherlands	175	395	31	69	ICES (2016f)
	Scotland	29	54	35	65	MMO, 2012
	Wales	18	64	21	79	ICES (2016f)
	<b>Total</b>	<b>2294</b>	<b>4186</b>	<b>35</b>	<b>65</b>	<b>ICES (2016f)</b>
bss-8ab	France	1430	2325	38	62	ICES (2015c)
	<b>Total</b>	<b>1430</b>	<b>2554</b>	<b>36</b>	<b>64</b>	<b>ICES (2015c)</b>
bss-8c9a	Portugal	22	271	7	93	ICES (2017c)
	Spain	423	317	57	42	ICES (2017c)
	<b>Total</b>	<b>472</b>	<b>701</b>	<b>40</b>	<b>60</b>	<b>ICES (2017c)</b>
cod-2224	Denmark	1290	8716	13	87	ICES (2016d)
	Germany	3100	4703	40	60	ICES (2016d)
	Sweden	217	1932	10	90	ICES (2016d)
	<b>Total</b>	<b>4607</b>	<b>17755</b>	<b>21</b>	<b>79</b>	<b>ICES (2016d)</b>
cod-347d	Belgium	266	1045	20	80	ICES (2016c)
	Denmark	526	9247	5	95	ICES (2016c)
	England	288	1714	14	16	MMO, 2012
	France	190	1452	12	88	ICES (2016c)
	Germany	30	2854	1	99	ICES (2016c)
	Netherlands	1157	2311	33	67	ICES (2016c)
	Norway	588	6043	8.9	91.1	ICES (2016c)
	Scotland	57	12240	0.5	99.5	MMO, 2012
	Sweden	799	1149	41	59	ICES (2016c)
	<b>Total</b>	<b>3901</b>	<b>37918</b>	<b>9</b>	<b>91</b>	<b>ICES (2016c)</b>
cod-7e-k	Channel Islands	0.42	0.64	40	60	MMO, 2012
	England	90	605	13	87	MMO, 2012
	France	191	5601	3	97	ICES (2016b)
	Ireland	7.8	1665	0	100	ICES (2016b)
	Wales	7.8	2	79	21	MMO, 2012
	<b>Total</b>	<b>297</b>	<b>8339</b>	<b>3</b>	<b>97</b>	<b>ICES (2016b)</b>

Stock	Country	Catches (t)		Percentage of total landings		Commercial landings source
		Recreational	Commercial	Recreational	Commercial	
ele-3a,4,7*	Belgium	7.3	0	100	0	ICES (2016e)
	Denmark	48	37	56	44	ICES (2016e)
	England	6.4	0.32	95	5	MMO, 2012
	France	2.6	0	100	0	ICES (2016e)
	Germany	4.1	0	100	0	ICES (2016e)
	Ireland	0.67	0	100	0	ICES (2016e)
	Netherlands	19	1	95	5	ICES (2016e)
	Norway	Illegal to fish	ICES (2016c)			
	Scotland	11	0.25	98	2	MMO, 2012
	Wales	0.67	0	100	0	MMO, 2012
	<b>Total</b>	<b>100</b>	<b>39</b>	<b>72</b>	<b>28</b>	<b>ICES (2016e)</b>
ele-balti*	Denmark	73	267	21	79	ICES (2016e)
	Estonia	0.01	0	100	0	ICES (2016e)
	Finland	8	0	100	0	ICES (2016e)
	Germany	1.5	41	4	96	ICES (2016e)
	Latvia	22	0	100	0	ICES (2016e)
	Lithuania	5	0	100	0	ICES (2016e)
	Poland	0.72	31	2	98	ICES (2016e)
	Sweden	0	237	0	100	ICES (2016e)
	<b>Total</b>	<b>110</b>	<b>576</b>	<b>16</b>	<b>84</b>	<b>ICES (2016e)</b>
	mac-3,4	Belgium	72	43	62	38
Denmark		141	36331	0	100	ICES (2016g)
England		513	11039	4	96	MMO, 2012
Germany		510	4578	10	90	ICES (2016g)
Netherlands		1705	3651	32	68	ICES (2016g)
Norway		469	64449	1	99	ICES (2016g)
Scotland		104	57521	0	100	MMO, 2012
Sweden		575	4579	11	89	ICES (2016g)
<b>Total</b>		<b>4089</b>	<b>219489</b>	<b>2</b>	<b>98</b>	<b>ICES (2016g)</b>
mac-7,8abde	Channel Islands	0.47	0.31	60	40	MMO, 2012
	England	99	1423	7	93	MMO, 2012
	France	2425	15755	13	87	ICES (2017f)
	Ireland	8.6	14917	0	100	ICES (2017f)
	Isle of Man	0.18	12	1	99	ICES (2017f)
	Wales	8.5	3	74	26	MMO, 2012
	<b>Total</b>	<b>2542</b>	<b>73828</b>	<b>3</b>	<b>97</b>	<b>ICES (2016f)</b>

Stock	Country	Catches (t)		Percentage of total landings		Commercial landings source
		Recreational	Commercial	Recreational	Commercial	
pol.27.67	Channel Islands	1.5	0.82	64	36	MMO, 2012
	England	321	1436	18	82	MMO, 2012
	France	2895	1423	67	33	ICES (2016h)
	Ireland	28	1175	2	98	ICES (2016h)
	Isle of Man	0.56	8.2	6	94	MMO, 2012
	Northern Ireland	12	8.1	59	41	MMO, 2012
	Scotland	45	32	59	41	MMO, 2012
	Wales	28	1.4	95	5	MMO, 2012
	<b>Total</b>	<b>3331</b>	<b>4524</b>	<b>42</b>	<b>58</b>	<b>ICES (2016h)</b>
sal-bal**	Denmark	12	18	40	60	ICES (2017d)
	Estonia	1.1	0.82	58	42	ICES (2017d)
	Finland	13	28	31	69	ICES (2017d)
	Germany	4	1.8	69	31	ICES (2017d)
	Latvia	1.1	0.82	57	43	ICES (2017d)
	Lithuania	0.69	0.33	68	32	ICES (2017d)
	Poland	2.1	4.1	33	67	ICES (2017d)
	Sweden	1.8	33	5	95	ICES (2017d)
	<b>Total</b>	<b>35</b>	<b>87</b>	<b>29</b>	<b>71</b>	<b>ICES (2017d)</b>

The values provided for sal-bal are in 1000's of fish rather than weight.

\* = Comparisons only made in terms of the retained recreational and commercial catch as the percentage of the released weight reconstructed was considered too high to be representative of the actual value. \*\* = Numbers (in 1000s) caught rather than weight used.

**Source:** EURecFish.

## 5.5 Discussion

The total impact of MRF, as illustrated by the percentage contribution to total removal of fish biomass, on fish stocks was found to be very high in several stocks. Even in those stocks in which the percentage contribution to total landings were low, the total biomass removal by MRF was greater than some national catches by commercial fishing (e.g. mac-3,4 recreational catches were greater than Dutch and Belgian commercial landings despite only contributing to 1.83% of the total catches). The large catches by MRF in many stocks could be impacting on the ability to manage fish stocks sustainably as this biomass removal is not accounted for when providing advice regarding the total allowable catch (TAC) of a stock. MRF catches were examined in an international context by Hyder *et al.* (in press), which found that the MRF harvest of western Baltic cod (ICES subdivision 22-24) and European sea bass in ICES division areas IVb-c and VIIa, d-h (bss-47) were high (4,679 tonnes and 1,468 tonnes, respectively). The MRF catches reported by Hyder *et al.* (in press) were similar to those found in this study, providing further evidence for the potential for MRF to remove a significant biomass from a stock.

In general, the percentage removal by MRF was lower in stocks than the catches by commercial fishing. The stocks bss-47 (recreational: 36%, 2,369 tonnes; commercial: 64%, 4,186 tonnes), and pol.27.67 (recreational: 42%, 3,331 tonnes; commercial: 58%, 4,524 tonnes) were found to have the highest percentage contribution to total landings by MRF. However, the high MRF catches calculated for pol.27.67 were due to high landings in France (Herfaut *et al.*, 2013) that may represent an overestimate. Nevertheless, the findings of this study and several others (e.g. Armstrong *et al.*, 2013; Rocklin *et al.*, 2014; ICES, 2016f; 2017d) have illustrated that MRF's impact on sea bass stocks is significant, which caused the EU's Scientific, Technical and Economic Committee for Fisheries to call for an 80% reduction in sea bass landings (Scientific, Technical and Economic Committee for Fisheries, 2014). Cooke and Cowx (2006) also compared the percentage contribution to total harvest by recreational and commercial fishing within the U.S.A. and found significant proportions (17%) of Atlantic cod catches were taken by MRF. Whilst the proportion of cod removed by MRF in the U.S.A. is greater than that found in this study (cod-347d: 9%, cod-7e-k: 3%), this is probably due to the larger number of recreational fishers exploiting the stock (U.S. Fish and Wildlife Service, 2006; Arlinghaus *et al.*, 2015).

This study also identified a large variation in the proportion of total catches by MRF between species. This variation may be attributed to the accessibility to a stock and the perceived enjoyment of catching a certain species (e.g. due to its fighting abilities or a good taste when eating) by angling, which is the predominant method of MRF (Armstrong *et al.*, 2013; Sparrevohn and Storr-Paulsen, 2012). Therefore, inshore stocks of popular species may be at greater risk of high exploitation by MRF than offshore stocks, although, further studies are required to test the factors driving the species targeted by recreational fishers.

All the data used here were taken from other studies, so the reliability of the estimates needs to be evaluated prior to interpreting the results. Several assumptions were made when extrapolating the data and applying post-release mortalities, introducing additional uncertainty into the estimates. However, extrapolation percentages were full and partial comparisons were made were generally very low except for bss-8c9a. In this case, the MRF catches were raised from small scale national studies (Veiga *et al.*, 2010; DCF sampling, 2015; 2016 unpublished data), which reduced the risk of bias of MRF harvest and was therefore likely to be representative of the true catch.

Another potential source of error in the results was the post-release mortalities used for both recreational and commercial discards/releases as in most cases country-specific

differences in fishing practices (e.g. gear type used) could not be taken into account. Additionally, where post-release mortalities could not be determined, a precautionary 100% mortality rate was used. A full description of the factors affecting post-release mortality is not provided here, but see Bartholomew and Bohnsack (2005) for examples of recreational angling post-release mortalities between species, and Alverson *et al.* (1994) for examples of commercial fishing discard mortality. Whilst commercial discard mortality and recreational post-release mortality can be very high for some species, it is unlikely that a 100% post-release mortality rate occurs, so the precautionary mortality values used may have induced overestimation of catches by both MRF and commercial fishing. Conversely, using post-release mortality values derived from studies conducted under certain circumstances for discards by commercial fishing or post-release mortality by MRF may result in underestimation of catches as country-specific differences in fishing practices will cause different associated post-release mortalities (e.g. Alverson *et al.*, 1994). The impact of not taking into account gear type in MRF post-release mortality was likely to be larger in Scandinavian countries. This was because static gears (e.g. nets) are permitted for recreational use, such as in Sweden (Karlsson *et al.*, 2016), and these gears are likely to have significantly higher mortality than angling. However, as angling was the predominant method of MRF found in all studies that quantified percentage contribution to total MRF landings (e.g. Armstrong *et al.*, 2013; Karlsson *et al.*, 2016), the impact of applying post-release mortality estimates from angling is minimal.

Whilst an attempt to estimate bias in the MRF catches estimates produced in this study was made, the method used may not have been sensitive or robust enough to detect all forms of bias in the catch estimates; furthermore, the bias estimation method used did not take into account the impacts of reconstructing data on the reliability of the results. The issues with the estimation of bias in this study were due to the subjectivity in quantifying the bias of an individual study, in addition to the difficulty in measuring bias introduced by reconstructing data. However, the issues with the estimation of bias in this study are inconsequential as the biases calculated are only designed to indicate the potential scale and direction of the biases in the results, rather than a full characterisation of the bias.

Recreational fisheries data pertaining to tuna species were lacking, with the only data found in the ICCAT statistical database (ICCAT, 2017). This only provided recreational tuna landings data post-2002 in the Northeast Atlantic from Senegal, a relatively small West-African nation (ICCAT, 2017). In addition to the ICCAT statistical database, a study on recreational tuna landings by the Scientific, Technical and Economic Committee for Fisheries (2016) examined recreational tuna fisheries, but did not provide catch estimates. As a consequence of the lack of recreational tuna catch data, the impact of MRF on tuna species could not be determined.

The results of this study are likely to be relatively representative of the true catches by MRF despite the issues with the dataset. However, due to the potential for uncertainty and bias in the data, the results of this study should only be used as an indicator of the impact of MRF in European marine fish stocks and should not be used as a replacement for national recreational fisheries sampling schemes. Furthermore, time series of MRF catches show large variation in catch-per-unit-effort and catches between years (Strehlow *et al.*, 2012). These variations underline the importance to collect annual estimates of catches for inclusion in stock assessments. Otherwise assumptions are required to generate time series from data from either a single year (e.g. sea bass) or to deal with intermittent data.

Despite the potential impact of MRF on marine fish stocks, as revealed by this study and several others (e.g. Armstrong *et al.*, 2013; Strehlow *et al.*, 2012; Hyder *et al.*, 2014; Karlsson *et al.*, 2016; van der Hammen and de Graaf, 2015; van der Hammen *et al.*, 2016) and the requirement by the DCF to sample MRF (EU, 2016a), a large number of countries

do not produce estimates for MRF catches for use in stock assessments, which can severely impact the advice provided by stock assessments and may induce overfishing. Furthermore, national MRF sampling schemes often vary in methodology and scope, whilst some variation in MRF sampling methodology is inevitable due to differences between angler behaviour and fishing practices. A standardised method in which MRF surveys are conducted is required to ensure that over/underestimation of catches does not occur.

Although an attempt to characterise fisheries within the Mediterranean and Black Seas has been made (FAO, 2016), the landings provided were an average for 2010-2013 and were not split by country. Furthermore, the recreational landings attained were reconstructed for almost all countries (Keskin *et al.*, 2014; Ulman *et al.*, 2015; Khalfallah *et al.*, 2017) and further reconstructions would introduce a large amount of uncertainty, so were unlikely to be representative of the true catch. Consequently, the impact of MRF in the Mediterranean and Black Seas could not be determined. It is the recommendation of this study that further funding should be provided for MRF sampling schemes to be conducted by Mediterranean and Black Sea countries; in addition, commercial fisheries landings in the Mediterranean and Black Seas should be made more readily available.

The comparison made here between the levels of commercial and recreational catches gives an idea of the potential impact of MRF, but inclusion in a full analytical assessment is needed to assess impact properly. This is because the catches by MRF and commercial fisheries will have different selectivity, meaning that it was possible that MRF may not have an equivalent impact to the share of catches if, for example, MRF was selecting more small fish that have a higher mortality. Full analytical stock assessments were not possible within the scope of this work because these would require more data, some of which was not available (e.g. gear selectivity, time series of catches etc.), and significant understanding of the methods and assumptions of each individual stock assessment to incorporate MRF catches. Hence, it would be sensible to assess the data available for inclusion of MRF in individual stock assessments at the next benchmark for the stock and carry out an assessment that includes MRF to assess the impact.

## 5.6 Recommendations

Based on the comparison of recreational and commercial catches, the following recommendations are appropriate:

- MRF catches are lacking for many key species especially in the Mediterranean and Black Seas. Further data collection is needed to develop understanding and should focus on country specific multispecies surveys that include diadromous and highly migratory species (e.g. tuna).
- The lack of estimates of discards and post-release mortality make comparison with commercial catches challenging. More information is needed on key species in order to make more robust comparisons.
- Where comparisons were possible, MRF catches represented a significant proportion of the total biomass removed by fishing for some stocks and could affect sustainability, so MRF catches should be routinely included in stock assessment.
- A stock assessment is needed to assess the impact of MRF properly, as comparisons of total catches do not account for the different sizes of fish caught. MRF should be included in the next benchmark assessment for sea bass, cod, eel, salmon, pollack, and sea trout, and other species where recreational catches could be significant.





## 6 ENVIRONMENTAL IMPACT OF MARINE RECREATIONAL FISHING

### 6.1 Summary

MRF has the potential to have impacts on the marine environment beyond the removal of biomass from fish stocks (e.g. introduction of non-native species, spread of disease, littering, lead emission). The impacts can occur at local, regional, national, or international scales with reviews and, in some cases quantification, of the environmental impacts of freshwater angling available. In contrast, the pressures, magnitude, and associated impacts from MRF are largely unknown and are difficult to separate from other anthropogenic sources. However, a better understanding is vital for policy development to mitigate impacts. Here, a review was done of the potential MRF pressures and a ranking developed to identify the most important sources of environmental impacts. A review of the literature was done that identified the following pressures: use of natural and live bait organisms, lead input from tackle loss, bycatch of threatened and protected species, commercial bait harvest, private bait collection, boating and noise, disturbance, litter, loss of gear, walking and driving, small-scale trawling, and environmental impacts of spearfishing. The associated risks of these activities were identified and ranked using a risk assessment approach. The majority of impacts were considered to be of minor importance due to their local scale, reversibility, and ease of management. The introduction of disease, pathogens, and non-native species and lead were the highest risk due to the severity, difficulty to reverse, and challenge to manage. However, there was a lack of studies to evaluate the impact, meaning that other factors may also be important, so new studies are required to support policy and decision-making. MRF pressure varies significantly in space and time, so the scale of the impact compared to other anthropogenic impacts also needs to be accounted for by decision-makers.

### 6.2 Introduction

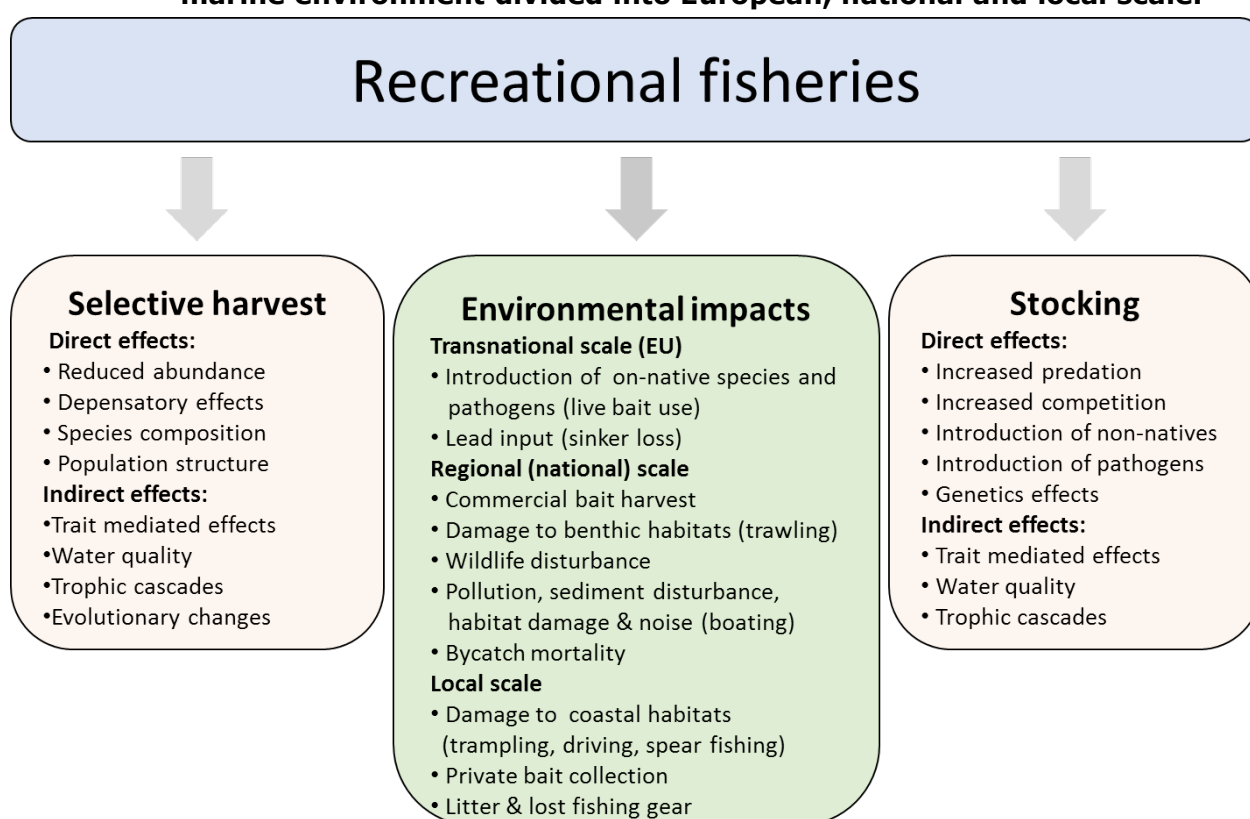
Most studies concerning impacts of MRF on the aquatic environment deal with the impact on fish stocks (e.g. Ryan *et al.*, 2016). However, there are other pressures of recreational fishing that can impact on ecosystems (e.g. loss of fishing gear or lead, litter, bycatch of seabirds and marine mammals, spread of disease, benthic impact) and a recent review is available for freshwater (Lewin *et al.*, 2006). The potential impacts of recreational fishing can be split into selective harvest (direct and indirect effects), environmental effects (at varying scales), and stocking impacts (direct and indirect effects) (Figure 10). There are a number of effects in each of the categories, but the environmental and selective harvest impacts are likely to be most important for MRF, as stocking is limited to a small number of species (e.g. salmon, sea trout, eels) (Figure 10). Despite the potential impacts of MRF on the environment, few studies exist and no reviews of the environmental impact of MRF exist.

MRF is carried out from two different platforms (shore and boats) and mainly occurs in coastal areas. Consequently, the impacts associated with MRF primarily concentrate on the littoral and nearshore zones. Although MRF can have impacts on marine ecosystems, relative little effort has been undertaken in the past to investigate the effects of scale and the associated processes (Parnell *et al.*, 2010). However, recreational fishing effort is not evenly distributed, rather there are areas of high and low fishing pressure (e.g. Hunt *et al.*, 2011; Cabanellas-Reboredo *et al.*, 2014; McPhee, 2017) and the impacts from recreational fishing activity vary according to country-specific fishing practices (fishing gear/methods, season etc). Moreover, coastal habitats are more sensitive to MRF due to the proximity to centres of population and the cumulative effects of many different pressures. In addition,

small areas can be particularly sensitive (e.g. fish nursery areas or nesting sites) so local pressures can have a disproportionately large impact.

Here, the environmental impacts of MRF and the scale and location that they occur were reviewed. The pressures included: lead, litter, seabird bycatch, spread of disease, non-native species, and Protected, Endangered and Threatened Species (PETS). The impacts were identified and a simple risk-based approach used to rank the impacts as high (H), moderate (M), and low (L). Assessment criteria were the risk severity concerning genetic and biological diversity, the spatial scale and potential reversibility of the effects, and the complexity and spatial scale of appropriate management measures for the mitigation of adverse effects. This ranking of the potential risks was used to identify any key areas to prioritise and to highlight data gaps. The results were discussed in the context of the requirements for new studies and the impact on management of MRF.

**Figure 10: Impacts resulting from recreational fisheries, including impacts on the marine environment divided into European, national and local scale.**



The activities and associated risks are ranked according to the severity of the risk of their potential impact on the marine environment in descending order.

**Source:** EURRecFish.

## 6.3 Methods

### 6.3.1 Review of literature

A review was undertaken of existing literature on the environmental impacts of recreational fishing on the marine environment. A literature search was carried out using Google Scholar (<http://scholar.google.co.uk/>). Google Scholar was chosen as the outputs have been shown to be comparable to Web of Science for peer-reviewed literature and may be superior due to the range of grey literature (e.g. dissertations, academic books, and technical reports) included (Beckmann and von Wehrden, 2012). Search results are ranked and weighted based on the level of agreement with search terms of the full text of each

document and the number of citations (Beel and Gipp, 2009). To reduce the risk of missing articles by authors presenting different views to the majority (Beel and Gipp, 2009), the databases Science Direct, Scientific Research, PubMed, and Researchgate were also searched. Initially, general keywords were searched for alone or in combination with keywords such as “recreational fishery” or “coastal habitats”. For detailed search queries, general keywords were combined with more specific terms (Table 26). To limit the results to European waters where possible, geographic designations indicated by the terms North Sea, Baltic Sea, English Channel, Atlantic, and Mediterranean Sea were added to the search.

**Table 26: Individually and in combination used search terms for the literature search.**

Area	Search terms
General search terms	Marine, recreational fishery, fishing, angling, hook and line, coastal, rod and line, gillnet, fish, marine mammal, seabird, cetaceans, elasmobranchs, benthic invertebrate, trophic cascade, food web, trophic cascade, European waters, North Sea, Baltic Sea, English Channel, Atlantic, Mediterranean Sea
Coastal habitats	Coastal, habitat, seagrass, intertidal, rocky, sandy, shore*, mudflat, coral reef
Use of natural (live) bait organisms	Live bait, live bait trait, introduction, disease, parasites, non-native species, non-indigenous species, earthworm, fish
Lost lead sinkers	Lead, lead sinkers, lead loss
Bycatch of marine mammals, birds, marine turtles, and elasmobranchs	Bycatch, turtles
Commercial and private bait collection	Bait, bait collection, bait digging, benthic invertebrates, live bait, benthic invertebrate, <i>Arenicola marina</i>
Litter and discarded or lost fishing gear	Angling litter, debris, lost fishing gear, plastic litter, ghost nets, ghost fishing, plastic lures
Litter and discarded or lost fishing gear	Angling litter, debris, lost fishing gear, plastic litter, ghost nets, ghost fishing, plastic lures
Boating	Recreational boating, anti-fouling, two stroke, four stroke, nutrient input, waves, propeller scars, sediment, ballast water, non-native, non indigenous species, noise, macrophytes
Noise and wildlife disturbance	Wildlife disturbance, anthropogenic noise, hearing, hearing loss, stress, flight distance, disturbance distance
Walking, removal of vegetation, and driving on sensitive habitats	Trampling, walking, vegetation cover, foot path, four-wheel-drive, littoral vegetation
Disturbance of benthic habitats by trawling	Trawling, otter trawl, beam trawl, sediment, benthic community, fauna
Spearfishing	Spearfishing, SCUBA diving

Source: EURecFish.

### 6.3.2 Assessment of impact & prioritisation

The rating of the impacts as high (H), moderate (M), and low (L) was based on scientific literature and expert opinion. Assessment criteria were: the severity of the risk in terms of the potential impact on the marine environment, communities, and genetic diversity, the spatial scale of the potential impact and management measures, the reversibility of the effects, and the complexity of management measures for the mitigation of adverse effects. For example, the use of live bait was classified as high impact due to the risk of introducing non-indigenous species or disease. Non-indigenous species and disease do not necessarily cause impact, but where they do they usually have severe impacts on the receiving ecosystem that are not confined to specific habitats, and are difficult to control (Lodge, 1993; Simberloff *et al.*, 2005). In particular the genetic effects of hybridization that affect

the adaptedness of native species to the local environment are long lasting and difficult to reverse (Rhymer and Simberloff, 1996). Consequently, the management measures should address risk on a broad, European Union wide scale. In contrast, walking on sensitive coastal habitats is rated as low impact. The effects of trampling may be severe but locally confined, usually reversible, and comparably easy to manage on a local scale. Activities that occur on larger spatial scales and/or affect threatened populations or species are rated as moderate if the impacts are considered reversible. Boating may affect large areas and management measures at regional or national scale may be required if, for example, transboundary MPAs are concerned.

## **6.4 Results**

### **6.4.1 Review of literature**

The review led to the identification of peer-review publications and items of grey literature that were categorised by subject and reviewed for content. A further rationalisation for marine relevant studies was done leading to peer-review publications and items of grey literature for an in-depth review (Table 27). A summary of the marine relevant literature relating to each search category (Table 26) is captured below (see Table 27 for a synthesis).

#### **6.4.1.1 Use of live bait**

The use of live bait (e.g. worms, fish) is very common in recreational fishing (Fidalgo e Costa *et al.*, 2006; Font and Lloret, 2011) and the trade of live bait organisms is economically important (Watson *et al.*, 2016). While there is extensive literature on the impact of live bait in freshwater fisheries, few studies exist that focus on the marine environment. Generally, it is not possible to extrapolate from effects in freshwater to marine systems and even where the effects are plausible, it is likely that the effects are smaller in the marine environment. Many anglers release live bait organisms regularly into places where they are not native and are released in a different area to collection, often because they were unaware of the consequences or because of the (mis)perception that released bait organisms provide an ecological benefit to natural resources (Kilian *et al.*, 2012; Micael *et al.*, 2016). Transfer and release of live bait organisms resulted in several introductions of non-indigenous fish, invertebrate and plant species in Europe and the U.S.A., some of which were able to establish self-sustaining populations and to invade new habitats (Randall, 1987; Carlton, 1992; Cohen *et al.*, 1995; Ludwig and Leitch, 1996; Tiunov *et al.*, 2006; Hendrix *et al.*, 2008; Pernet *et al.*, 2008; Winfield *et al.*, 2011; Cohen, 2012; Arias *et al.*, 2013; Sa *et al.*, 2017). An additional risk associated with the marine live bait trade results from the common practice to store live bait in packagings containing seaweed or sediments which can harbour various small live organisms or pathogens and are often discarded into the environment (Haska *et al.*, 2011, Fowler *et al.*, 2016).

The introduction of non-indigenous species contributed to the homogenization of the world's fauna, impacted aquatic ecosystems and caused high economic damage (Baltz, 1991; Litvak and Mandrak, 1993; Rahel, 2002; Olden *et al.*, 2004; Keller and Lodge 2007; Molnar *et al.*, 2008; Williams and Grosholz, 2008; Hixon *et al.*, 2016). Even the distribution of species within their native range, but across biogeographical borders, can impair their local adaptation. Whereas a small gene flow between populations may increase their adaptation capability (Krueger and May, 1991), large gene flows can lead to the loss of the locally adapted gene complexes and ultimately the fitness of local populations (Stockwell and Leberg, 2002; Gilk *et al.*, 2004). An additional risk associated with the introduction of live or even frozen bait is the spread of diseases and parasites (Hedrick *et al.*, 2001; Goodwin *et al.*, 2004; Gozlan *et al.*, 2005; Blakeslee *et al.*, 2012; Phelps *et al.*, 2014).

**Table 27: Recreational fisheries related activity, documented impacts, spatial scale of the impact and management measures, risks ranking, and references.**

Activity	Documented impacts	Potential risks	Spatial scale	Impact strength	References (selection)
Live bait, use of non native bait organisms	Introduction of non native species, negative impacts on local fish populations and invertebrates due to inter- and intraspecific interactions, introduction of diseases and pathogens.	Spread of non-native species, decrease or loss of native species following inter- or intraspecific interactions, changes in native fish and invertebrate communities, hybridization, impacts on soil and fauna, spread of diseases and pathogens.	International	H	<i>Angler behaviour</i> : Kilian <i>et al.</i> , 2012; Micael <i>et al.</i> , 2016.  <i>Marine</i> : Randall, 1987; Carlton, 1992; Cohen <i>et al.</i> , 1995; Pernet <i>et al.</i> , 2008; Cohen, 2012; Goodwin <i>et al.</i> , 2004; Gozlan <i>et al.</i> , 2005; Haska <i>et al.</i> , 2011; Blakeslee <i>et al.</i> , 2012; 2016; Arias <i>et al.</i> , 2013; Fowler <i>et al.</i> , 2016; Sa <i>et al.</i> , 2017.  <i>Freshwater</i> : Litvak and Mandrak, 1993; Ludwig and Leitch, 1996; Kircheis, 1998; Bohlen <i>et al.</i> , 2004a, b; Tiunov <i>et al.</i> , 2006; Keller and Lodge 2007; Keller <i>et al.</i> , 2007; Cucherousset and Olden, 2008; Hendrix <i>et al.</i> , 2008; DiStefano <i>et al.</i> , 2009; Winfield <i>et al.</i> , 2011; Drake and Mandrak, 2014; Phelps <i>et al.</i> , 2014.
Lead loss	Lead emission Damage to wildlife.	Water and sediment pollution, increased mortality and fitness reduction in ducks and piscivorous birds, risk for human health.	International	H	<i>General</i> : Eisler, 1988; Scheuhammer and Norris, 1996; Scheuhammer <i>et al.</i> , 2003; Mowad <i>et al.</i> , 1998; Clark and Scheuhammer, 2003; Franson <i>et al.</i> , 2003; Gustavsson and Gerhardsson, 2005; Rattner <i>et al.</i> , 2008; Goddard <i>et al.</i> , 2008; O'Connell <i>et al.</i> , 2009; Mathee <i>et al.</i> , 2013; Haig <i>et al.</i> , 2014; Micael <i>et al.</i> , 2016.  <i>Marine</i> : Zabka <i>et al.</i> , 2006.  <i>Freshwater</i> : Mudge, 1983; Birkhead and Perrins, 1985; Forbes, 1986; Cryer <i>et al.</i> , 1987b; Sears, 1988; Blus, 1994; Twiss and Thomas, 1998; Jacks <i>et al.</i> , 2001; Cao <i>et al.</i> , 2003; Perrins <i>et al.</i> , 2003; Warner <i>et al.</i> , 2016.
Bycatch, catch of protected species	Stress and injuries in target and non-target species.	Immediate or delayed post-release mortality, sublethal effects, negative impacts on fitness.	Regional	M	Davis <i>et al.</i> , 2001; Borucinska <i>et al.</i> , 2002; Campana <i>et al.</i> , 2006; Skomal, 2007; Abraham <i>et al.</i> , 2010; Heberer <i>et al.</i> , 2010; Molina and Cooke. 2012; McCallum <i>et al.</i> , 2013; Morizur <i>et al.</i> , 2013; Reeves <i>et al.</i> , 2013; Robbins <i>et al.</i> , 2013; Zydellis <i>et al.</i> , 2013; Carapetis <i>et al.</i> , 2014; Danylchuk <i>et al.</i> , 2014; Gallagher <i>et al.</i> , 2014; 2017; Lyle and Tracey, 2016; McClellan Press <i>et al.</i> , 2016; Sheen and Robinson, 2017.
Commercial bait collection	Disturbance of benthic fauna, flora and sediment, Wildlife disturbance.	Decrease in abundance and mean size, fitness reduction in some invertebrate species, impact on seabird prey availability.	Regional	M	Van den Heiligenberg, 1987; Beukema, 1995; Currie and Parry, 1996; Shepherd and Boates, 1999; Ferns <i>et al.</i> , 2000; Watson <i>et al.</i> , 2016.

Activity	Documented impacts	Potential risks	Spatial scale	Impact strength	References (selection)
Private bait collection	Disturbance of benthic fauna, flora and sediment, wildlife disturbance.	Decrease in abundance and mean size, fitness reduction in some invertebrate species.	Local	L	McLusky <i>et al.</i> , 1983; Howell., 1985; Cryer <i>et al.</i> , 1987a; Keough <i>et al.</i> , 1993; Townshend and O'Connor, 1993; Chapman and Underwood, 1996; Ambrose <i>et al.</i> 1998; Roy <i>et al.</i> , 2003; Contessa and Bird, 2004; Logan, 2005; Griffith <i>et al.</i> , 2006; Prescott, 2006; Watson <i>et al.</i> , 2007; Carvalho <i>et al.</i> , 2013; Fearnley <i>et al.</i> , 2013; Watson, 2014.
General litter	Decreasing visual quality Damage to wildlife (ingestion and entanglement).	Water pollution, increased mortality and fitness reduction in various marine taxa, negative impacts on tourism.	Local	L	<i>Marine</i> : Baird and Hooker, 2000; Derraik, 2002; Andrady, 2011; Blight and Burger, 1997; Boerger <i>et al.</i> , 2010; Possatto <i>et al.</i> , 2011; Baulch and Perry, 2014; Jang <i>et al.</i> , 2014; Setälä <i>et al.</i> , 2014; De Sa <i>et al.</i> , 2015; Van Cauwenbergh <i>et al.</i> , 2015. <i>Freshwater</i> : Bell <i>et al.</i> , 1985; Forbes, 1986; Cryer <i>et al.</i> , 1987b.
Loss of other fishing gear	Damage to wildlife (entanglement).	Increased mortality and fitness reduction in marine wildlife.	Local	L	<i>Marine</i> : Gorzelany, 1998; Wells <i>et al.</i> , 1998; Adimey <i>et al.</i> , 2010; Asoh <i>et al.</i> , 2004; Brown and Macfadyen, 2007; Stelfox <i>et al.</i> , 2016 <i>Freshwater</i> : Radomski <i>et al.</i> , 2006; Danner <i>et al.</i> , 2009; Raison <i>et al.</i> , 2014
Boating	Wave action, resuspension of sediments, emission of inorganic and organic compounds, wildlife disturbance, accidental introduction of non native species.	Reduction or loss of littoral macrophytes and seagrass, sediment stirring, water pollution, Stress, injuries, fitness decrease, recruitment failure in some species.	Local	L	<i>General</i> : Jackivicz Jr. and Kuzminski, 1973; Jüttner <i>et al.</i> , 1995; Tjärnlund <i>et al.</i> , 1995; 1996; Simpson <i>et al.</i> , 2015. <i>Marine</i> : Davis, 1977; Kocan <i>et al.</i> , 1987; Turner <i>et al.</i> , 1997; Scarlett <i>et al.</i> , 1999a; b; Biselli <i>et al.</i> , 2000; Thomas <i>et al.</i> , 2001; Bauer <i>et al.</i> , 2002; Bell <i>et al.</i> , 2002; Baden <i>et al.</i> , 2003; Uhrin and Holmquist, 2003; Chesworth <i>et al.</i> , 2004; Eriksson <i>et al.</i> , 2004; Schiff <i>et al.</i> , 2004; Warnken <i>et al.</i> , 2004; Bishop, 2005; 2008; Sandström <i>et al.</i> , 2005; Sara <i>et al.</i> , 2007; Verney <i>et al.</i> , 2007; Leon and Warnken, 2008; Lloret <i>et al.</i> , 2008a; Bellefleur <i>et al.</i> , 2009; Bickel <i>et al.</i> , 2011; Papale <i>et al.</i> , 2012; Brine <i>et al.</i> , 2013; Ros <i>et al.</i> , 2013; Zabin <i>et al.</i> , 2014; La Manna <i>et al.</i> , 2015; Berthe and Lecchini, 2016. <i>Freshwater</i> : Morgan <i>et al.</i> , 1976; Yousef <i>et al.</i> , 1980; Murphy and Eaton, 1983; Nanson <i>et al.</i> , 1994; Killgore <i>et al.</i> , 2001; Arlinghaus <i>et al.</i> , 2002; Haviland-Howell <i>et al.</i> , 2007; Alexander and Wigart, 2013; Lorenz <i>et al.</i> , 2013.



Activity	Documented impacts	Potential risks	Spatial scale	Impact strength	References (selection)
Noise (boating)	Stress, loss of hearing abilities, wildlife disturbance.	Stress, fitness decrease in seabirds, fish, invertebrates, and marine mammals.	Local	L	<p><i>General:</i> Aaden Yim-Hol Chan <i>et al.</i>, 2010; Popper and Fay, 2011; Bruintjes and Radford, 2013.</p> <p><i>Marine:</i> Cox <i>et al.</i>, 2006; Clark <i>et al.</i>, 2009; Codarin <i>et al.</i>, 2009; Jensen <i>et al.</i>, 2009; Picciulin <i>et al.</i>, 2010; Buscaino <i>et al.</i>, 2010; Andre <i>et al.</i>, 2011; Bracciali <i>et al.</i>, 2012; Dow Piniak <i>et al.</i>, 2012; Rako <i>et al.</i>, 2013; Wale <i>et al.</i>, 2013a; b; Celi <i>et al.</i>, 2014; Morley <i>et al.</i>, 2014; Nedelec <i>et al.</i>, 2014; 2015; 2017; Nichols <i>et al.</i>, 2015; Peng <i>et al.</i>, 2015; 2016; Williams <i>et al.</i>, 2015; Neo <i>et al.</i>, 2016.</p> <p><i>Freshwater:</i> Scholik and Yan, 2002a; 2002b; Amoser and Ladich, 2003; McCauley <i>et al.</i>, 2003; Holt and Johnston, 2011.</p>
Disturbance of wildlife	Disturbance of waterbirds and other marine wildlife.	Stress, fitness decrease in seabirds and marine mammals.	Local	L	<p><i>General:</i> Madsen, 1985; Hockin <i>et al.</i>, 1992; Platteeuw and Henkens, 1997; Frid and Dill, 2003; Beale and Monaghan, 2004; Borgmann, 2011.</p> <p><i>Marine:</i> Gillett <i>et al.</i>, 1975; Pfister <i>et al.</i>, 1992; Fitzpatrick and Bouchez, 1998; Carney and Sydeman, 1999; Leseberg <i>et al.</i>, 2000; Ronconi and Cassady St. Clair, 2002; Stolen, 2003; Thomas <i>et al.</i>, 2003; ; Bejder <i>et al.</i>, 2006; Stillmann <i>et al.</i>, 2007; Glover <i>et al.</i>, 2011; Velando and Munilla, 2011; McLeod <i>et al.</i>, 2013; Schlacher <i>et al.</i>, 2013; Martin <i>et al.</i>, 2015.</p> <p><i>Freshwater:</i> Bell and Austin, 1985; Pierce <i>et al.</i>, 1993; Mori <i>et al.</i>, 2001.</p>
Walking on coastal habitats	Damage to littoral fauna, flora, and soil (sediment), disturbance of wildlife.	Changes of macrophyte assemblages, species loss, soil compaction, habitat changes, erosion.	Local	L	<p><i>General:</i> Ros <i>et al.</i>, 2004; Pescott and Steart, 2014.</p> <p><i>Marine:</i> Hylgaard and Liddle, 1981; Beauchamp and Gowing, 1982; Addessi, 1994; Brosnan and Crumrine, 1994; Chandrasekara and Frid, 1996; Schiel and Taylor, 1998; Milazzo <i>et al.</i>, 2002; Smith <i>et al.</i>, 2008; Farris <i>et al.</i>, 2013; Purvis <i>et al.</i>, 2015.</p> <p><i>Freshwater:</i> Liddle and Scorgie, 1980; Bar, 2017.</p>
Four wheel drive	Impact on littoral fauna, flora, and soil (sediment), disturbance of wildlife.	Changes of macrophyte assemblages, species loss, soil compaction, habitat changes, erosion.	Local	L	<p>Moss and McPhee, 2006; Schlacher and Thompson, 2008; Davies <i>et al.</i>, 2016.</p>

Activity	Documented impacts	Potential risks	Spatial scale	Impact strength	References (selection)
Trawling	Impact on marine sediment and damage to benthic flora and invertebrates, high bycatch rate.	Disturbance of sediments and benthic flora, increased mortality and fitness reduction of benthic invertebrate and other bycatch species.	Local <sup>1)</sup>	L	Auster <i>et al.</i> , 1996; Kaiser <i>et al.</i> , 2000; 2002; Tillin <i>et al.</i> , 2006; Asch and Collie, 2008; Olsgard <i>et al.</i> , 2008; Althaus <i>et al.</i> , 2009; Hinz <i>et al.</i> , 2009; Shephard <i>et al.</i> , 2010; Bergman <i>et al.</i> , 2015; Sciberras <i>et al.</i> , 2016.
Spear fishing	Impact on marine sediment and damage to benthic flora and invertebrates, low bycatch.	Disturbance of sediments and benthic flora, increased mortality and fitness reduction of benthic invertebrate and bycatch species.	Local	L	Rouphael and Inglis, 1997; Hawkins <i>et al.</i> 1999; Zakai and Chadwick-Furman, 2002; Barker and Roberts, 2004; Gotanda <i>et al.</i> , 2009; Luna <i>et al.</i> , 2009; Feary <i>et al.</i> , 2010; Guzner <i>et al.</i> 2010; Font <i>et al.</i> , 2012; Font and Lloret, 2014; Lamb <i>et al.</i> , 2014; Lindfield <i>et al.</i> , 2014; Lyons <i>et al.</i> , 2015; Tran <i>et al.</i> , 2016; Giglio <i>et al.</i> , 2017.

The activities and associated impacts were ranked as H (high, severe impact, difficult to reverse and to manage, management measures on a broad spatial scale, international), M (moderate, medium severe impacts on regional or national scale), and L (low, impact occurs locally, is reversible and comparably easy to manage on local scale).

<sup>1)</sup>: only few boats, locally restricted (Belgium), comparatively small fishing gear

**Source:** EURecFish.



#### **6.4.1.2 Loss of lead sinkers and lures**

Anglers use terminal tackle and fishing lures (e.g. sinkers, pilks, shads and spoons) that consist of, or contain, lead. Sinkers and lures are regularly lost if the hook or line becomes entangled and attempts are made to free the hook. Lead is a toxic heavy metal whose widespread use causes environmental contamination globally and may cause mortality in various organisms including mammals and humans (Zabka *et al.*, 2006; UNEP, 2010; Mathee *et al.*, 2013). The total consumption of lead for angling tackle in EU15 and EU25 is estimated at 1,900-5,600 and 2,000-6,000 tonnes each year respectively, of which about 50% is used in freshwater fisheries (EU, 2004). Given high numbers of anglers, the amount of lead introduced into water can be substantial (Forbes, 1986; Cryer *et al.*, 1987b; Bell *et al.*, 1985; Jacks *et al.*, 2001; Scheuhammer *et al.*, 2003; Radomski *et al.*, 2006; Rattner *et al.*, 2008). Scheuhammer *et al.* (2003) estimated that 500 tonnes of lead from lost and discarded lead sinkers and jigs were deposited annually representing 14% of all lead releases in Canada. While there are no studies on the ingestion of lead shots or sinkers by fish, many studies indicated that lead sinkers (split shot weights) as well as spent ammunition from hunting were directly (waterbirds) or indirectly (raptorial birds and scavengers) ingested by birds (Mudge, 1983; Blus, 1994; Haig *et al.*, 2014), and the mortality following lead intoxication can be substantial (Birkhead and Perrins, 1985; Sidor *et al.*, 2003). Increased lead concentrations in blood and internal organs due to the ingestion of lead sinkers have also been found in harbour seals (*Phoca vitulina richardsi*) (Zabka *et al.*, 2006) and crocodiles (*Crocodylus niloticus*) (Warner *et al.*, 2016).

#### **6.4.1.3 Recreational bycatch**

MRF accidentally captures unwanted, undersized, or protected species. Commercial bycatch is viewed as a serious environmental impact of modern commercial fisheries (Alverson *et al.*, 1994; Lloret *et al.*, 2016), but information about the extent of recreational bycatch is scarce. With regards to a single angler, bycatch may be a rare event. However, due to the high participation in MRF, the overall recreational bycatch may be substantial and impacts on the populations of non-target species cannot be ruled out. The legislation of several European countries (e.g., France, Denmark, Finland) allows the restricted use of gillnets in recreational fishing (Pawson *et al.*, 2008). Due to their indiscriminate nature, recreational gillnets may incidentally capture seabirds, protected fish species, elasmobranchs, and mammals (Morizur *et al.*, 2013; Zydalis *et al.*, 2013). However, the number of gillnets per fisher is restricted and recreational gillnets are much shorter than commercial nets (Sparrevojn and Stoff-Paulsen, 2012; Lyle and Tracey, 2016).

Several studies indicated that recreational rod and line fishers occasionally catch seabirds, in particular if they use natural baits (Ferris and Ferris 2004; U.S. Fish and Wildlife Service, 2007; Bardtrum *et al.*, 2009). However, recreational fishing gear is light compared to commercial fisheries and most birds can be released unharmed (Abraham *et al.*, 2010). Although marine angling is considerably selective, anglers may occasionally catch protected fish species. However, protected species can often be released after catch unharmed, although some post-release mortality cannot be ruled out (Muoneke and Childress, 1994; Bartholomew and Bohnsack, 2005). Sharks and rays were regularly caught as bycatch, but some shark species are also an important target species in catch and release recreational sea angling (Gallagher *et al.*, 2017). Many sharks and rays are released whether they were deliberately or incidentally caught (McClellan Press *et al.*, 2016), but sublethal effects and some degree of post-release mortality are possible (Skomal, 2007; Robbins *et al.*, 2013).

#### **6.4.1.4 Bait collection**

Many anglers prefer wild caught invertebrates as bait (Watson, 2014) and the demand for live bait is increasing leading to both commercial and recreational bait collection in

intertidal zones and mud flats (Watson *et al.*, 2016). Intensive bait collection and, in particular, the mechanical harvest may reduce species number and abundance of benthic faunal and floral organisms by direct (removal and damage) and indirect (habitat destruction) mechanisms (Cryer *et al.*, 1987a; Beukema, 1995; Prescott, 2006). Invertebrate communities respond resiliently to low intensity bait collection, but the recolonization depends on invertebrate species and abundance, harvest depth and frequency, volume of sediment excavated, substrate type, and tidal height (Cowie *et al.*, 2000; Griffiths *et al.*, 2006; Johnson *et al.*, 2007; Carvalho *et al.*, 2013). In addition, intensive bait collection may also lead to the truncation of the population size structure (Roy *et al.*, 2003; Watson *et al.*, 2007). Those changes in abundance or size distributions potentially alter the composition of shoreline communities if the target species are key components of the coastal food web (Underwood, 1993; Lindberg *et al.*, 1998; Shepherd and Boates, 1999; Ferns *et al.*, 2000; Fearnley *et al.*, 2013). Bait digging may also directly and indirectly influence the physical and chemical sediment properties (Contessa and Bird, 2004; Watson *et al.*, 2007; Birchenough, 2013). However, such bait digging events cause temporary disturbances rather than permanent damage (McLusky *et al.*, 1983) and may only impact on a small proportion of the total habitat. The impacts of bait digging on heavy metal circulation may be more serious. Consistent bait digging has been shown to increase the amount of bioavailable lead and cadmium from the sediment and enhanced the uptake of these metals by benthic polychaetes (Howell, 1985).

#### **6.4.1.5 Litter and lost fishing gear**

Litter in the form of lost recreational fishing gear can cause habitat degradation and ecological damage. Studies from freshwater and marine environments quantifying angling litter showed locally high levels of both lost fishing tackle and general litter at high activity angling sites (Bell *et al.*, 1985; Forbes, 1986; Cryer *et al.*, 1987b). In particular, plastic litter has been identified as major threat to aquatic ecosystems and tourism revenue (Derraik, 2002; Andrady, 2011; Jang *et al.*, 2014). The sources of marine plastic litter are numerous, but some studies demonstrated that a substantial amount of traceable plastic debris originated from recreational activities such as boating, tourism, and fishing (Galgani *et al.*, 2013, Moriarty *et al.*, 2016; Nelms *et al.*, 2017). Nearly all marine taxa are threatened through entanglement in, and ingestion of, plastic litter (Beck and Barros, 1991; Blight and Burger, 1997; Possatto *et al.*, 2011; Farrell and Nelson, 2013; Baulch and Perry, 2014; Setälä *et al.*, 2014; Van Cauwenberghe *et al.*, 2015). Entanglement causes drowning and lacerations, and decreases the ability to feed and to avoid predation. The ingestion of plastic items may lead to starvation due to gut obstruction and can also reduce individual fitness. Moreover, plastic litter is potentially organ toxic because of the adsorbed chemical pollutants (Derraik, 2002; de Sa *et al.*, 2015).

Soft plastic lures are commonly used in recreational fisheries. A lake survey revealed that the deposition rate of soft plastic lures was about 80 pieces per kilometre shoreline each year (Raison *et al.*, 2014), but this is likely to be much lower in the sea. Most soft plastic lures swell, so may have an obstructive effect on the gastrointestinal tract of fish or other organisms when a lure has been ingested (Danner *et al.*, 2009; Raison *et al.*, 2014). Lost lures form an additional source of plastic pollution in aquatic ecosystems, although the quantity of lost lures is negligible compared to the amount of other plastic litter. Lost or discarded fishing gear remains in the environment for years, continuously entangling aquatic wildlife ("ghost fishing") (Brown and Macfadyen, 2007; Anderson and Alford, 2014; Stelfox *et al.*, 2016).

The loss of gillnets by recreational fishers is rarely investigated, but are likely to affect marine wildlife in a similar way to commercial gillnets. However, the number and size of lost recreational gillnets is likely to be low compared to lost commercial gear. In contrast,

the amount of lost fishing lines and hooks can be substantial (Forbes, 1986; O'Hara *et al.*, 1988; Bell *et al.*, 1985; Yorio *et al.*, 2014). Lost gear can impair health and survival of sessile invertebrates due to entanglement and covering (Asoh *et al.*, 2004; Angiolillo *et al.*, 2015; Oliveira *et al.*, 2015) and have occasionally been shown to entangle seabirds, marine mammals, crustaceans, squid, and turtles (Laist, 1997; Gorzelany, 1998; Nemoz *et al.*, 2004; Abraham *et al.*, 2010; Carapetis *et al.*, 2014). Although angling associated litter may not substantially impact wildlife populations, vulnerable species may need protection in areas with a high recreational angling activity and where protected species are likely to be present.

#### **6.4.1.6 Boating**

Recreational fishing is often carried out from boats, so it is sensible to consider boating and fishing as related activities (Farr *et al.*, 2014). However, for policy development it is important to disaggregate the recreational boating and fishing experiences (Farr *et al.*, 2014). A high density of recreational boats has environmental impacts that can be roughly categorised into chemical, physical, and biological impacts (Liddle and Scorgie, 1980; Davenport and Davenport, 2006). Chemical impacts result from the introduction of chemical components from fuel and combustion products (Tjärnlund *et al.*, 1995; 1996; Kempinger *et al.*, 1998; Mastran *et al.*, 1994). In addition, antifouling paints that are toxic to marine flora and fauna (Kocan *et al.*, 1987; Hardy *et al.*, 1987; Jüttner *et al.*, 1995; Biselli *et al.*, 2000; Macinnis-Ng and Ralph, 2002; 2003; Valkirs *et al.*, 2003; Chesworth *et al.*, 2004; Warnken *et al.*, 2004; Labieniec *et al.*, 2009; Abdel-Shafy *et al.*, 2016) and can accumulate in marine food webs (Baumard *et al.*, 1999; Berto *et al.*, 2007; Antizar-Ladislao, 2008; Le Croizier *et al.*, 2016). Physical impacts result primarily from boat wakes and propeller action that occur in addition to natural turbulence. Boat wakes can increase sediment and nutrient resuspension, affect aquatic fauna and flora, and increase bank erosion (Yousef *et al.*, 1980; Murphy and Eaton, 1983; Nanson *et al.*, 1994; Bauer *et al.*, 2002; Verney *et al.*, 2007; Koehl and Hadfield, 2010; Alexander and Wigart, 2013).

Seagrass is an important component of coastal ecosystems and provides structural complexity on sediments that is used by invertebrates and fish for shelter and feeding (Orth *et al.*, 1984; 2006; Beck *et al.*, 2001; Heck *et al.*, 2003; 2008; Duffy, 2006; Fourqurean *et al.*, 2012; Liley and Unsworth, 2014). Fragmentation of seagrass habitat reduces the abundance of species (Bell *et al.*, 2002; Uhrin and Holmquist, 2003). Propeller action, waves, and mooring impair submerged macrophytes and seagrass (Sargent *et al.*, 1995; Short and Wyllie-Echeverria, 1996; Baden *et al.*, 2003; Eriksson *et al.*, 2004; Madley *et al.*, 2004) and recreational boating is listed among the most serious threats to seagrass (Cullen-Unsworth and Unsworth, 2016; Hotaling-Hagan *et al.*, 2017). Coral reefs are also threatened by boat waves and anchoring (Davis, 1977; La Manna *et al.*, 2015).

Boating can indirectly impact coastal fish assemblages by reducing prey availability and habitat quality (Robertson, 1984; Sandström *et al.*, 2005; Bishop, 2008). Resuspended sediment particles can reduce the survival of fish eggs, larvae, and juveniles (Morgan II *et al.*, 1983; Wilber and Clarke, 2001) and induce gill trauma (Berg and Northcote, 1985). The increases of turbidity beyond natural levels can also affect competition and predator-prey interactions (Redding *et al.*, 1987; Mackenzie and Leggett, 1991; Barrett *et al.*, 1992; Visser and Stips, 2002; Horppila *et al.*, 2004; Zamor and Grossman, 2007). Furthermore, shear stress can kill fish eggs and larvae by causing rotation and deformation, and displace individuals into unsuitable habitats (Morgan II *et al.*, 1976; Killgore *et al.*, 2001; Huckstorf *et al.*, 2011; Becker *et al.*, 2013).

Ground nesting shorebirds are threatened by boat wakes impacting their nests (Söhngen *et al.*, 2008). Although large animals will be able to escape, collisions between fast vessels,

boats and unwary cetaceans have been recognized as a source of mortality and injury (Van Waerebeek *et al.*, 2006).

Impacts of boat wakes on sediment and shores are only important in sheltered waters of lagoons, estuaries, and harbour mouths that do not have heavy navigation traffic, and the impact of recreational boating is directly related to boat size and speed (Arlinghaus *et al.*, 2002b; Maynard, 2005; Sandström *et al.*, 2005; Huckstorf *et al.*, 2011). The impacts of wave and propeller action is likely to be less for angler boats compared to other motorcraft (e.g. powerboats) if anglers reduce boat speed and avoid sensitive areas. However, the effect of waves due to boating is probably minor in the marine environment compared to freshwater systems, as there is much more water movement and wave action.

Recreational boaters can contribute to the introduction of aquatic plants, invertebrates, and pathogens into other habitats through entanglement, biofouling and live-well water, especially where the species can survive periods of air exposure (Sant *et al.*, 1996; Johnson *et al.*, 2006; West *et al.*, 2009; Ros *et al.*, 2013; Klatt *et al.*, 2014; Zabin *et al.*, 2014).

#### **6.4.1.7 Anthropogenic noise**

In general, recreational fishing activity is not linked to noise. However, marine recreational fishers frequently use private or charter boats that can be a dominant source of noise in aquatic environments (Haviland-Howell *et al.*, 2007). Underwater noise has deleterious effects on many fish and other marine taxa that use sound for orientation, navigation, communication, and the detection of predators, competitors, prey, and potential mates (Slabbekoorn *et al.*, 2010; Peng *et al.*, 2015). Noise can cause physical damage to the hearing system, including temporary or permanent loss of hearing abilities (Amoser and Ladich, 2003; McCauley *et al.*, 2003; Smith *et al.*, 2004; Andre *et al.*, 2011; Sole *et al.*, 2016), threshold shifts (Wysocki and Ladich, 2005), and stress and avoidance reactions (Vabø *et al.*, 2002; Radford *et al.*, 2014; Wysocki *et al.*, 2006; Buscaino *et al.*, 2009; Nichols *et al.*, 2015; Berthe and Lecchini, 2016; Neo *et al.*, 2016). The collective noise from many sources can impede the ability to perceive, recognise or decode sounds of interest (Popper *et al.*, 2004; Vasconcelos *et al.*, 2007; Clark *et al.*, 2009) and cause attention shifts (Purser and Radford, 2015; Spiga *et al.*, 2017) that impact foraging, reproduction, schooling and predator avoidance (Simpson *et al.*, 2014; Nedelec *et al.*, 2015; 2017). Negative impacts of noise have been also observed in invertebrates (Wale *et al.*, 2013a; 2013b; Celi *et al.* 2014; Nedelec *et al.*, 2014; Peng *et al.*, 2016) and marine mammals avoid human noise (Bejder *et al.*, 2006; Papale *et al.*, 2012; Rako *et al.*, 2013). However, it is impossible to separate recreational fishing-induced noise from other anthropogenic noise sources which are likely to be much greater (e.g. wind farms, underwater construction, marine traffic).

#### **6.4.1.8 Human disturbance of wildlife**

Human disturbance can be viewed as a form of predation risk, with predator avoidance behaviour in response to a threshold level of threatening stimuli found in many species. This behaviour creates a trade-off between the avoidance of predation and other fitness-related activities (Crowder *et al.*, 1997; Frid and Dill, 2002). In addition, the avoidance of certain areas may influence local distributions of both predators and prey organisms (Dill *et al.*, 2003) and contribute to lower prey abundance in refugia (Lenihan *et al.*, 2001). Recreational activities carried out on the shores generate conflicts between humans and coastal wildlife. Depending on location and season, human disturbance can cause stress and disturb overwintering, resting, feeding, and reproduction of resident and migratory seabirds that use shoreline habitats (Gillett *et al.*, 1975; Madsen, 1985; Mitchell *et al.*, 1989; Goss-Custard and Verboven, 1993; Verhulst *et al.*, 2001; Weimerskirch *et al.*, 2002;

Robinson and Cranswick, 2003; Velando and Munilla, 2011; Schlacher *et al.*, 2013; Martin *et al.*, 2015). The disturbance caused by MRF originates mainly from direct contact, boating, and sight. Slow walking causes less disturbance compared to human activities with rapid and unpredictable movements or fast-moving boats (Bellefleur *et al.*, 2009). Sitting humans cause only low disturbances (Fitzpatrick and Bouchez, 1998). Anglers, however, may show long periods of inactivity interspersed with short periods of rapid movements (Bell and Austin, 1985).

Recreational boats can induce flight reactions in fish, marine mammals and seabirds in deeper water (Rodgers and Schwikert, 2002; Ronconi and Cassidy St. Clair, 2002; Nowacek *et al.*, 2004; Haviland-Howell *et al.*, 2007; Bellefleur *et al.*, 2009; Curtin *et al.*, 2009; Glover *et al.*, 2011; Bracciali *et al.*, 2012; Papale *et al.*, 2012). No existing studies separate recreational fishing-induced impacts from other anthropogenic sources, so the impact of MRF is difficult to assess.

#### **6.4.1.9 Walking and driving on sensitive habitats**

Walking in sensitive habitats can affect terrestrial, semi-aquatic or aquatic plants, algal assemblages, and soil properties and the associated invertebrate fauna (Rees and Tivy, 1978; Liddle and Scorgie, 1980; Hylgaard and Liddle, 1981; Milazzo *et al.*, 2002; Farris *et al.*, 2013; Chandrasekara and Frid, 1996; Milazzo *et al.*, 2002; Cunha Escarpinati *et al.*, 2014). Corals are extremely susceptible to trampling and this can cause substantial damage to corals primarily due to breakage (Woodland and Hooper, 1977). Anglers impact littoral habitats by making paths to access water, walking parallel to the shoreline, cutting bank vegetation, and removing submerged vegetation (Rees, 1978; Williams and Moss, 2001; O'Toole *et al.*, 2009). Generally, the removal of littoral vegetation enhances erosion and decreases the habitat quality for many species that rely on the shelter provided by aquatic vegetation (Liddle and Scorgie, 1980; Smokorowski and Pratt, 2007; McCloskey and Unsworth, 2015). Driving on the shore is not generally associated with recreational fishing. However, in areas without marinas or boat ramps anglers may use cars to launch boats. Driving impacts on shoreline habitats through visual degradation, noise, air pollution, soil structure, crushing fauna and flora, disturbance, and nest damage (Burger and Gochfield, 1990; Moss and McPhee, 2006; Hardiman and Burgin, 2010; Davies *et al.*, 2016). No existing studies separate recreational fishing-induced impacts from other anthropogenic sources, so the impact of MRF is difficult to assess.

#### **6.4.1.10 Trawling**

Trawling is not generally a common practice in MRF, but there are localities where trawling can be high (e.g. Belgium allows small recreational beam and otter trawlers). Generally, bottom trawling is a destructive fishing gear (Freiwald *et al.*, 2004; Murray Roberts and Freiwald, 2005; Althaus *et al.*, 2009) that can impair water quality, sediment structure, and benthic flora and fauna (Auster *et al.*, 1996; Rumohr and Kujawski, 2000; Duplisea *et al.*, 2002; Kaiser *et al.*, 2000; 2002; 2006; Warken *et al.*, 2003; Hily *et al.*, 2008; Ramey *et al.*, 2009; Pusceddu *et al.*, 2014). Several studies indicated that trawling can impact demersal fish species where the available prey declines (Shephard *et al.*, 2010; Hiddink *et al.*, 2016). However, to the best of our knowledge recreational trawling is restricted to some areas in Belgium with few participants and the impact is most likely negligible compared to commercial trawling activities.

#### **6.4.1.11 Spearfishing**

Spearfishing is actively practiced as sport or for leisure (e.g. in the Baltic and Mediterranean Sea). Spearfishers remove a substantial proportion of preferred species and actively select the large, long lived fish species and large individuals with high reproductive potential (Coll *et al.*, 2004; Lloret *et al.*, 2008b; Lindfield *et al.*, 2014). Other impacts are



comparable to those resulting from activities such as SCUBA diving. For example, many spearfishers use boats (Font *et al.*, 2012; Font and Lloret, 2014), and the impacts of boating are similar independent of the fishing gear used. Spearfishers may further damage benthic organisms and in particular corals by touching through fin kicks, spearfisher's bodies and spearguns (e.g. Roupheal and Inglis, 1997; Hawkins *et al.* 1999; Zakai and Chadwick-Furman, 2002; Barker and Roberts, 2004; Luna *et al.*, 2009; Guzner *et al.* 2010; Lamb *et al.*, 2014; Lyons *et al.*, 2015; Roche *et al.*, 2016; Giglio *et al.*, 2017).

#### 6.4.2 Assessment of impact & prioritisation

Activities and associated risks were identified, ranked as high (H), moderate (M), and low (L) by using a risk assessment matrix that is based on scientific literature and expert opinion (e.g., Cooke and Cowx, 2006; Pawson *et al.*, 2007; 2008; Ihde *et al.*, 2011; Pranovi *et al.*, 2016). A full description of the risk categorisation is provided (see Table 27; Table 28 for details), and are listed in descending order of importance as:

- Use of natural and live bait organisms.
- Lead input in water bodies due to lead containing fishing tackle loss.
- Bycatch of threatened and protected elasmobranchs, marine mammals and seabirds.
- Commercial bait harvest.
- Private collection of natural bait organisms.
- Boating and noise.
- Wildlife disturbance.
- Angling-related littering.
- Discarding or loss of fishing gear.
- Walking and driving on shorelines.
- Small-scale trawling.
- Environmental impacts of spearfishing.

Following a description of the potential impacts and receiving habitats, key areas to address first, and potential approaches to mitigate the risks, were identified and included in the discussion.

**Table 28: Severity of environmental consequences of activities associated with recreational fishing.**

Criteria	Scale	Reversibility	Impact	Management complexity	Ecosystem	Rating
Non-native species	H	H	H	H	H	H
Lead fishing weights	M	H	M	H	H	H
Bycatch	M	M	M	M	M	M
Commerc. bait collection	M	M	M	M	M	M
Private bait collection	L	L	L	L	L	L
General litter	L	L	L	L	L	L
Lost fishing gear	L	L	L	L	L	L
Boating	L	L	M	L	L	L
Noise	L	L	L	L	L	L
Wildlife disturbance	L	L	M	L	L	L
Habitat damage	L	L	L	L	L	L
Driving	L	L	M	L	L	L
Benthic disturbance	L	L	L	L	L	L
Spearfishing	L	L	L	L	L	L

According to the ranking criteria, the impacts are classified as low (L), moderate (M), and high (H).

Source: EURecFish.

## 6.5 Discussion

Based on this review and others (Blaber *et al.*, 2000; Cooke and Cowx, 2006; Coleman *et al.*, 2004), it can be concluded that the MRF can have impacts on coastal ecosystems that occur in addition to those resulting from the removal of fish biomass. However, marine ecosystems are influenced by many user groups (e.g. commercial fishing, wind farms, aggregate industries, marine traffic, recreational boating, coastal tourism, etc.), which all have impacts on the state of the ecosystem. In many cases it is difficult to quantify and separate the contribution of different actors and activities as they may interact and act cumulatively. While there are extensive studies of impacts of recreational fishing on freshwater ecosystems, few studies focus on MRF. Hence, further studies are needed as the impacts and associated effects cannot be extrapolated from freshwater to marine systems and effects are likely to vary considerably in magnitude, with less pronounced effects in the marine environment.

### 6.5.1 High impacts/risks and management measures on a broad spatial scale

The use of live bait was classified as high risk activity because the released or lost bait organisms can impair the biological diversity, introduce pathogens or disease, and genetic effects are difficult to reverse. The EIFAC Code of Practice for Recreational Fisheries (EIFAC, 2008) and the FAO Technical Guidelines for responsible recreational fisheries (FAO, 2012) recommend the use of live bait organisms in agreement with local or national regulation and in the water body from which they were collected. However, the use of imported bait organisms in marine coastal waters is largely unmanaged (Font and Lloret, 2011). Currently, there is a lack of knowledge about the impacts of live bait, so there is a need for research to build an evidence base for decisions.

Significant quantities of lead could be deposited in the water from recreational fishing activities, so the risk associated with the loss of lead containing terminal tackle and fishing lures was classified as high. However, the degree of bioavailability of lead compounds in soil and sediments is different (Cao *et al.*, 2003; Rattner *et al.*, 2008). The solution rate of lead in water is low and depends on water properties (30 µg/L in hard, basic water, ≤ 500 µg/L in soft, acidic water), surface to mass ratio, sediment structure, and water velocity. Jacks *et al.* (2001) measured a dissolution rate of lead sinkers in running and backwater environments of 20 mg per cm<sup>2</sup> and year. Under anoxic conditions lead will become adsorbed onto sediment particles and the dissolution of elemental lead and the probability of ingestion by organisms will become reduced (Jacks *et al.*, 2001; EU, 2004). Population effects resulting from the ingestion of lead sinkers have rarely been documented (but see Birkhead and Perrins, 1985). Furthermore, the loss of lead tackle is influenced by the intensity of fishing effort, the type of fishing, angler skills and the characteristics (vegetation, bottom structure) of the water body, and varies spatially and seasonally (Rattner *et al.*, 2008). At present, neither the amount nor the fate of lead sinkers deposited in marine environments have been investigated. However, according to the European Commission, the total consumption of lead for angling tackle in EU15 and EU25 is estimated at 1,900-5,600 and 2,000-6,000 tonnes each year respectively, of which about 50% is used in MRF. The consumption of lead by commercial fishing is similar, estimated to be 1,900-8,700 (EU15) and 2,000-9,000 (EU25) tonnes each year (European Commission 2004). As some European countries have banned or restricted the use of lead for fishing tackle, and alternative tackle made from non-toxic material are already available, it would be possible to assess the potential for reduction in lead and the economic effects on the tackle industry.

### **6.5.2 Moderate risks/impacts and management measures on a regional spatial scale**

Potential post-release mortality, high numbers of recreational fishers, and number of (protected) species affected led to the impact of bycatch being ranked as moderate. The risks can be mitigated by reducing bycatch through an appropriate choice of fishing gear and location, and by minimizing the extent of stress and injuries due to handling, hooking or entanglement (Bartholomew and Bohnsack, 2005; Uhlmann and Broadhurst, 2015).

The risk associated with bait collection was also rated as moderate, primarily because of the large spatial dimension and the use of mechanical gear in the commercial bait harvest. The private bait collection for MRF occurs on local scales only and can be ranked as low risk activity, because coastal ecosystems may be resilient to bait harvest, although local depletion may occur in areas near major population centres (Watson *et al.*, 2016). The recovery period can be shortened if holes and trenches are back filled and boulders carefully replaced after the collection. Where impacts on benthic communities occur, short term restrictions or rotating closures may stimulate the recovery of benthic communities.

### **6.5.3 Low risks/impacts and management measures on local spatial scale**

The risks associated with MRF litter were considered to be low. Litter problems are spatially restricted and management measures are easy to develop. Smaller "ghost nets" and hook and lines originating from MRF are similar features to commercial fishing gear, so may therefore endanger aquatic wildlife in a similar way. However, the size of recreational gear is small in comparison and used by a very small proportion of MRF, so the impact on wildlife can be assumed to be local and low. Even if it is not possible to quantify the effects of disturbance, boat traffic, and noise exclusively linked to MRF, impacts on marine wildlife and environment are possible, given a substantial level of boating and angling activity. However, the corresponding risks were ranked as low because they occur locally, can easily be managed, and are likely to be small in comparison with other sources. Management options include speed limits in nearshore areas and regulating boat (engine) type and density, as well as the implementation of buffer zones according to species specific traits and environmental conditions (Pierce *et al.*, 1993; Ruddock and Whitfield, 2007; Pape Møller, 2015; Piratelli *et al.*, 2015; Mallory, 2016), and the establishment of seasonal or temporary closures of high priority sites. The risks associated with walking on the shore were also rated as low because the effects occur on local scales only and are easily to reverse. Furthermore, many other users (walkers, water sports etc.) use the shore and recreational fishers are unlikely to be the main source of disturbances apart from in specific areas or seasons. Seasonal or access restrictions can be established to protect sensitive coastal zones of high ecological value. The impacts of recreational trawling on benthic habitats were considered to be low, because of the restriction of this activity to few countries, limited number of boats, low boat speed, and small size of fishing gear. Spearfishing affects fish stocks primarily due to the selective removal of large fish. The environmental impacts of spearfishing occur locally and were of low importance. Closures to diving and spearfishing or quotas may be appropriate management measures in sensitive areas.

Important coastal habitats often consist of small patches distributed along the coastline. Management measures to protect those patches have to work on a fine spatial scale. Moreover, recreational fishing also varies locally depending on the local environmental, biological and fishery related conditions. Nonetheless, MRF management usually focuses on a broad spatial scale which can lead to wide-scale mismatches in management regimes (Jordan *et al.*, 2012). Consequently, fisheries management should be tailored to specific local or regional features of both the marine environment and the recreational fishing practice to assure an adequate protection. However, in most cases, present management is



confronted with a lack of comprehensive data with sufficient high spatial and temporal resolution at country yet alone European scale. The development of appropriate management measures is further complicated by the lack of detailed information on regional fishing practices such as, for example, bait collection, proportion of shore and boat angling, and the use of lead containing lures and sinkers. In addition, some biological impacts of recreational fishing and associated activities in marine waters are not well investigated and in particular quantitative long-term data are rare. The present review indicates a strong geographical focus on North America and Australia in the research on both the potential impacts and the management of recreational fisheries. Therefore, we advocate similar research in Europe because information on the temporal and spatial patterns of MRF, as well as on the responses of marine taxa and coastal ecosystems to both recreational fishing and management measures, is widely lacking.

## **6.6 Recommendations**

Based on the assessment of environmental risk, the following recommendations are appropriate:

- MRF can have other impacts on the marine environment in particularly coastal habitats, but the level of impact as well as the associated effects are unknown. More information is needed to determine MRF-induced impacts and separate them from other anthropogenic impacts.
- Management of MRF needs to match the temporal and spatial scales of both the marine environment affected and the recreational fishing effort.



## **7 CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE OF MARINE RECREATIONAL AND SEMI-SUBSISTENCE FISHING**

MRF and SSUBF fisheries could represent a significant source of fishing mortality, have impact on ecosystems, and interact with other fisheries and users of the marine environment. However, the evidence needed to manage these fisheries is often limited and difficult to collate due to the large numbers of studies that vary in design and quality. The EURecFish project was commissioned to bring together this information with the aim of assessing the social benefits, economic value, and environmental impact of marine recreational and semi-subsistence fisheries in six marine regions of Europe. MRF and SSUBF in Europe were defined and characterised. While this was simple for MRF, it was very challenging for SSUBF due to the lack of differentiation from commercial and recreational fisheries and the lack of studies. Hence, it was not possible to estimate the social or economic impact of SSUBF or the impacts, with only two case studies identified. MRF is important in Europe with almost 9 million individuals or 1.6% of the European population taking part in MRF, generating a total economic impact of 10.5 billion euro, and supporting 100,000 jobs. It is also important for the management of fish stocks as between 2-72% of the biomass removed was by MRF for some stocks, so needs to be included in stock assessment routinely. MRF also had the potential to impact on ecosystems in ways beyond the removal of fish, but quantification and separation from other anthropogenic factors was very difficult. Despite being able to estimate value and impact of recreational fisheries, there is still a significant lack of data and knowledge across this area, so more studies are needed to provide a robust scientific evidence base that can be used to underpin decision-making and policy.

Based on these conclusions and combining individual recommendations from each chapter, 10 key recommendations have been made to support the development and understanding of MRF and SSUBF in Europe:

1. There is large variation in the understanding of MRF across Europe, generally with less data for Mediterranean and Black Seas countries, and limited time series. This makes any assessment of impact or value difficult, so there is a need for additional regular data collection.
2. A broad range of species are caught by MRF, yet mandatory data collection focusses on a small set of species. Further data collection is needed to develop understanding and should focus on country specific multispecies surveys.
3. Tourist MRF can be large (e.g. Norway), but there is little knowledge of the benefits or impacts of this sector. More information is required to understand how these fisheries can be managed and developed in the future.
4. Semi-subsistence fisheries should not be treated as a separate entity due to the challenges with definition, but individual countries should identify if they have any semi-subsistence fisheries and ensure that the current recreational or commercial fisheries sampling system covers these catches. In some cases, it may be necessary to set up additional sample frames to cover these data and develop approaches for management.
5. The potential total economic impact in Europe is significant, so MRF should become a sector that is targeted for development alongside commercial fisheries and aquaculture under the Common Fisheries Policy. However, data are lacking, so regular economic data collection is needed to monitor development and increase robustness of estimates.

6. The impact of changes in policy and management on the expenditure on MRF is very difficult to quantify and additional studies should be funded to develop these data, including studies of economic value and the human dimension.
7. Only the economic impact of direct expenditure was included in this study, but additional social and wellbeing benefits are provided by MRF that should be accounted for. It is unclear how this can be done, so additional studies should be funded to develop methods.
8. Estimates of discards and post-release mortality make comparison with commercial catches challenging. More information is needed on key MRF species to make more robust comparisons.
9. Where comparisons were possible, marine recreational fisheries catches represented a significant proportion of the total biomass removed for some stocks and could affect sustainability. Marine recreational fisheries catches should be routinely included in stock assessments, as this allows impacts to be properly assessed and appropriate management strategies developed.
10. MRF can have other impacts on the marine environment, in particularly coastal habitats, but the level of impact as well as the associated effects are unknown. More information is needed to determine MRF-induced impacts and separate them from other anthropogenic impacts before any policy is developed.

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ISBN 978-92-846-1603-9 (paper)

ISBN 978-92-846-1604-6 (pdf)

doi:10.2861/277908 (paper)

doi:10.2861/2728 (pdf)